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# 1 Spatialized freshwater ecosystem Life Cycle Impact 2 Assessment of water consumption based on instream 3 habitat change modeling

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## 10 **Abstract**

11 In this article a new characterization model and factors are proposed for the life cycle impact  
12 assessment (LCIA) of water consumption on instream freshwater ecosystems. Impact pathways  
13 of freshwater consumption leading to ecosystem damage are described and the alteration of  
14 instream physical habitat is identified as a critical midpoint for ecosystem quality. The LCIA  
15 characterization model aims to assess the change in habitat quantity due to consumptive water  
16 use. It is based on statistical, physical habitat simulation for benthic invertebrates, fish species  
17 and their size classes, and guilds of fish sharing common habitat preferences. A habitat change

18 potential (HCP) midpoint, mechanistic indicator, is developed and computed on the French river  
19 network at the river reach scale (the river segment with variable length between the upstream and  
20 downstream nodes in the hydrographic network), for median annual discharges and dry seasons.  
21 Aggregated, multi-species HCPs at a river reach are proposed using various aggregation  
22 approaches. Subsequently, the characterization factors are spatially aggregated at watershed and  
23 sub-watershed scales. HCP is highly correlated with median and low flow discharges, which  
24 determine hydraulic characteristics of reaches. Aggregation of individual HCPs at reach scale is  
25 driven by the species most sensitive to water consumption. In spatially aggregated HCPs,  
26 consistently with their reduced smaller average discharge rate, small stream habitats determine  
27 the overall watershed characterization. The study is aimed primarily at life cycle assessment  
28 (LCA) practitioners and LCIA modelers. However, since it is the result of a productive cross-  
29 fertilization between the ecohydrology and LCA domains, it could be potentially useful for  
30 watershed management and risk assessment as well. At the moment, the proposed model is  
31 applicable in France. For a broader implementation, the development of global, high resolution  
32 river databases or the generalization of the model are needed. Our new factor represents  
33 nevertheless an advancement in freshwater ecosystems LCIA laying the basis for new metrics for  
34 biodiversity assessment.

## 35 **Keywords**

36 Life Cycle Assessment, water consumption, watershed ecology, hydraulic habitat,  
37 environmental flows.

## 38 **Abbreviations**

39 LCA, life cycle assessment; LCIA, life cycle impact assessment; CF, characterization factor;  
40 FF, fate factor; XF, exposure factor; EF, effect factor; SAR, species-area relationship; SDR,  
41 species-discharge relationship; HCP, habitat change potential; Q, river discharge; CWU,  
42 consumptive water use; HS, habitat suitability; WUA, weighted usable area; Re, Reynolds  
43 number; W, river width.

## 44 **1. Introduction**

45 Inland waters are a habitat for rich species diversity. Approximately 126 000 inland aquatic  
46 species have been described according to IUCN (2009), representing the 9.5% of all currently  
47 identified species and circumscribed in a living environment which is equal to just 0.01% of the  
48 total terrestrial surface (Balian et al., 2008). Nevertheless, 65% of continental waters are  
49 moderately or highly threatened by anthropogenic disturbance and climate change. With the  
50 prospect of an intensification of anthropic pressure on ecosystems and an increase in freshwater  
51 needs driven by population growth, a lot of efforts have been dedicated to the development of  
52 more sustainable water management strategies (Davis et al., 2015; Lapointe et al., 2014). While  
53 these efforts have been capable of ensuring substantial improvement of water security for  
54 humans, there is still a mismatch with what has been achieved in terms of biodiversity  
55 conservation, partly because of the likelihood that the ways to meet water needs of humans and  
56 ecosystems can be substantially antagonistic (Vörösmarty et al., 2010).

57 In this context, several life cycle impact assessment (LCIA) models have been proposed to link  
58 freshwater consumptive use to biodiversity loss (Núñez et al., 2016) and addressing specifically

59 wetlands and surface water-dependent ecosystems (Amores et al., 2013; Hanafiah et al., 2011;  
60 Tendall et al., 2014; Verones et al., 2013b). These approaches provide endpoint indicators built  
61 on cause-effect pathways in which the impact characterization factor (CF, eq. 1) results from the  
62 combination of three sub-factors (Núñez et al., 2018). The fate factor (FF) represents the  
63 environmental change (e.g. change in m<sup>3</sup>/y river discharge or m<sup>2</sup> wetland area) due to water  
64 consumption, defined as the withdrawn water that is not returned to the original drainage basin  
65 (International Organization for Standardization ISO/TC 207/SC 5, 2014). The exposure factor  
66 (XF) indicates how far this alteration can be offset, e.g. in a river it can be approximated to 1  
67 since most freshwater species have less mobility than terrestrial species to compensate the lack  
68 of water. Finally, the effect factor (EF) describes the consequence on the ecosystem (e.g.  
69 potentially disappeared fraction of species, PDF).

$$CF = FF \cdot XF \cdot EF \quad (1)$$

70 In the literature, impact scores related to volumetric change in water availability and, as a  
71 consequence, indirectly linked to water quantity needs of affected taxa, have been calculated  
72 based on species-area relationships in wetlands (SAR) (Verones et al., 2017, 2013a, 2013b) or  
73 species-discharge relationships (SDR) in rivers (Hanafiah et al., 2011; Tendall et al., 2014;  
74 Xenopoulos et al., 2005). Despite the relative ease of applying SAR and SDR to LCIA, such  
75 empirical approaches to relate species richness to water quantity involve some underlying,  
76 necessary assumptions making these models less suitable to be used in certain circumstances.

77 Regarding, for instance, the completeness of covered taxa, SAR-based methods do not  
78 consider instream species, namely fish species and invertebrates. On the other hand, SDR aim at  
79 quantifying species occurrence related to river discharge and therefore estimating species  
80 mortality of fish and macroinvertebrates derived from flow reduction. Tendall et al. (2014)

81 addressed some limits of SDR, for instance, by better regionalizing species-discharge curves.  
82 However, building a mechanistic LCIA model on SDR implies considering equal responses to  
83 stress for highly differentiated taxa, concealing the complexity of the relationships between  
84 living organisms and their habitat. It could also lead to interpreting discharge (calculated at the  
85 catchment outlet) as the direct cause of species richness in the catchment, which remains  
86 unproven. In particular, more species can be found in large catchments due to larger available  
87 space and not only greater discharge, and this is interestingly what an approach based on SAR  
88 would suggest (Iwasaki et al., 2012; McGarvey and Terra, 2016). Moreover, at the local scale  
89 species traits and specific flow preferences, which can be essential for shaping community  
90 structures, are not taken into account. These considerations are important to evaluate ecosystem  
91 response occurring below the extinction threshold, which is determined usually by extreme or  
92 prolonged events (Lytle and Poff, 2004), rather than marginal water flow change (what most of  
93 LCIA models appraise). In addition, when evaluating long-term effects of flow reduction on  
94 species, such as in modeling global climate change scenarios, fish communities' background  
95 extinction rates (natural extinction rates in undisturbed conditions) and extinction time horizons  
96 (when species committed to extinction go actually extinct) are not defined, leading to a potential  
97 overestimation of species loss induced by flow reduction (Tedesco et al., 2013; Xenopoulos et  
98 al., 2005).

99 While LCIA methods based on SAR and SDR have the advantage of taking into account  
100 biological aspects compared to stress-based indicators (Berger and Finkbeiner, 2013; Boulay et  
101 al., 2017), exploring complementary options to model aspects of biological communities other  
102 than species richness (number of species), such as species abundance (number of individuals per  
103 species) or diversity (Tuomisto, 2010), would therefore support a more comprehensive analysis

104 of ecosystem quality (Curran et al., 2011; Damiani et al., 2017). This raises the question on  
105 whether it would be possible to improve current LCIA models to appropriate recent, available  
106 knowledge concerning freshwater ecology and especially environmental flow management  
107 (Angus Webb et al., 2013; Damiani et al., 2017; Poff and Zimmerman, 2010). The aim of this  
108 study is first to trace, in detail, the potential water consumption impact pathways on freshwater  
109 habitat and ecosystems. The purpose is to identify the main, relevant environmental mechanisms  
110 and develop a local, bottom-up, mechanistic model addressing the limitations of current LCIA  
111 models. Since lotic habitats are currently the most vulnerable to freshwater consumptive use  
112 (Vörösmarty et al., 2010), a habitat-based characterization factor linking marginal hydrological  
113 alteration to instream ecosystems effects is proposed, based on existing literature on the  
114 applicability of ecohydrological methods in LCIA (Damiani et al., 2017). The model is  
115 subsequently applied on French streams and the feasibility, ecological relevance and limitations  
116 of upscaling from the river to the watershed scale is discussed.

## 117 **2. Impact pathway analysis**

118 The simplified diagram in Fig. 1 outlines the impact pathways of consumptive freshwater use  
119 linked to water-dependent ecosystems damage. The picture is built on existing environmental  
120 flow management and ecohydrology literature (Angus Webb et al., 2013; Gillespie et al., 2015;  
121 Poff and Zimmerman, 2010). Ecological, bibliographical sources in Appendix A (section 1)  
122 provide additional details on the environmental mechanisms represented that would otherwise be  
123 difficult to include in the chart. Freshwater habitats are multifaceted and it is essential for  
124 mechanistic LCIA to identify relevant impact pathways to be modeled separately, particularly

125 when these are linked to specific environmental interventions (e.g. water withdrawal, river  
126 damming).

127 As for the abiotic components of habitat (e.g. morphodynamic and physicochemical aspects of  
128 groundwater and surface water bodies), the whole ecosystem may undergo cascading effects  
129 triggered by water balance changes. In fact, while an ecological system remains relatively stable  
130 over time, a human induced alteration may promote a directional shift to a new dynamic  
131 equilibrium resulting from the ecological response of the first affected ecosystem compartments  
132 and of all those subsequently connected. These mechanisms involve all inter- and intra-specific  
133 relationships between different biota and are only generically represented in Fig. 1. As an  
134 example, species loss and reduced riparian vegetation cover limits shading on rivers which in  
135 turn influences food and shelter availability for reproduction and juvenile growth in fish and  
136 macroinvertebrate species (Li and Dudgeon, 2008; Mokany et al., 2008; Riley et al., 2009).  
137 Moreover, under environmental stress, trophic webs play an important role in determining the  
138 stability and evolution of an ecological community (Downing and Leibold, 2010; Thompson et  
139 al., 2012).

140 It is also necessary to bear in mind that habitat characteristics can be influenced by multiple  
141 drivers, for instance nutrient availability in water can be determined by flow regime, thermal  
142 regime and the presence or absence of forest buffers (Kennen et al., 2008; Kløve et al., 2014).  
143 This draws attention to the fact that habitats not only drive the establishment of a specific type of  
144 ecosystem, but can also be shaped themselves by the biota they harbor. These ecosystem  
145 mechanisms involve all abiotic and biotic changes induced or limited by the presence of certain  
146 ecological communities (Berke, 2010; Jones et al., 2010). In addition, habitat modifications can

147 be further amplified by the proliferation of non-native species (Crooks, 2002; Ward and  
148 Ricciardi, 2010).

149 For these reasons, LCIA modeling would require describing ecological mechanisms at  
150 different scales: from species response to community composition and with short to long term  
151 time horizons. With this respect, hydrologic alteration of flow regimes should be characterized in  
152 all its components (magnitude, timing, duration, frequency and rate of change) as well as flow-  
153 dependent habitat characteristics (The Brisbane Declaration, 2007). In the present article we  
154 analyze, for the first time in LCIA modeling, the midpoint effect of flow magnitude alteration  
155 (i.e. river discharge) on physical habitat (i.e. hydraulics), for fish species and stream  
156 invertebrates. This represents a bottom-up approach aimed at determining the relations between  
157 freshwater species and their habitat at an early stage of the mechanistic impact pathway. The  
158 alteration of river flow conditions and hydraulics can result from surface water consumption,  
159 groundwater consumption, and from water infrastructures building and operation. The study  
160 focuses mainly on the development of an effect factor for marginal discharge alteration. In the  
161 following section a simplified fate factor is adopted (water balance between groundwater and  
162 surface water is not modeled) and non-marginal change in flow regime is not considered (e.g.  
163 from river damming). For these reasons, at present the proposed model would perform better for  
164 assessing the impact of direct surface water withdrawal and release.

165 [Figure 1]

### 166 **3. Materials and methods**

#### 167 *3.1. Freshwater habitat modeling in LCIA*

168 Water consumption may result in the alteration of river discharge and other related physical  
169 variables such as reach hydraulics (velocities, depths, bed forces, turbulence). Depending on the  
170 reach morphology and on their different habitat preferences, species can be favored or disfavored  
171 by these changes. In ecohydrology, habitat preferences are modeled by means of habitat  
172 suitability equations for physical habitat variables such as microhabitat hydraulics (width, depth,  
173 velocity), the substrate composition and the turbulence (Bovee, 1982; Payne and Jowett, 2013).  
174 Among ecohydrological habitat models, generalized statistical habitat models have been  
175 developed based on the findings that species habitat suitability in reaches strongly depends on  
176 reach-scale hydraulic geometries, i.e. variations in reach average width and water depth with  
177 discharge (Lamouroux and Capra, 2002; Lamouroux and Jowett, 2005; Lamouroux and  
178 Souchon, 2002). This approach to freshwater habitat modeling is well suited for mechanistic  
179 LCIA indicators evaluating habitat alteration effects on instream assemblages (Damiani et al.,  
180 2017), in particular because reach hydraulic geometries can be modeled over the whole  
181 hydrographic networks (Lamouroux, 2008; Miguel et al., 2016; Snelder et al., 2011). On this  
182 basis, a habitat-based midpoint characterization factor is proposed in eq. 2 to quantify the change  
183 in habitat availability for freshwater fish species and stream invertebrates according to river  
184 discharge alteration.

$$CF_i = FF_i \cdot EF_i = \frac{dQ_i}{dCWU_i} \cdot HCP_i \quad (2)$$

185  $CF_i$  is the characterization factor for the river reach  $i$ , the fate factor ( $FF_i$ ) represents the  
186 marginal change in discharge  $dQ_i$  ( $m^3/s$ ) for marginal change in consumptive water use  $dCWU_i$   
187 ( $m^3/s$ ). In the present study, the fate factor is considered equal to 1 as done by Hanafiah et al.  
188 (2011) and Tendall et al. (2014) LCIA models, meaning that 1  $m^3/s$  of water withdrawn or  
189 released in the environment causes 1  $m^3/s$  discharge alteration. This could be modeled more

190 precisely using a mass-balance, multimedia fate modeling approach as proposed by Núñez et al.  
191 (2018). The effect factor (EF<sub>i</sub>) is calculated as the habitat change potential HCP<sub>i</sub> in m<sup>2</sup> · s/m<sup>3</sup> of  
192 habitat surface derived from marginal discharge alteration (eq. 3). In the proposed approach, the  
193 value of the characterization factor corresponds therefore to the value of the effect factor and will  
194 be indicated indifferently as CF or HCP hereafter.

$$HCP_{ij} = \frac{HS_{ij}}{\sum_{j=1}^n HS_{ij}} \cdot \frac{dWUA_{ij}}{dQ_i} \quad (3)$$

195 HCP<sub>ij</sub> is calculated from seventeen multivariate microhabitat suitability (*HS*) equations *j*,  
196 developed empirically based on the abundance of eight fish species at different ontogenetic  
197 stages, four fish guilds of species with similar habitat preference and the production of  
198 invertebrates biomass (Table 1). The definitions of the four guilds are adopted from Lamouroux  
199 et al. (2002). The *Pool* guild includes species or size classes preferring deep and slow-flowing  
200 habitats with fine substrate sediment. The *Bank* guild individuals are adapted to shallow and  
201 slow-flowing waters with fine sediment. Shallow microhabitats harbor also *riffle* species, if  
202 velocities are intermediate to high and with intermediate particle size. *Midstream* guild species  
203 are instead adapted to fast-flowing and deep waters with coarse substrate composition. More  
204 details on guilds composition are given in Appendix A (Table A.2). Most Habitat suitability  
205 equations considered here are included in the generalized statistical habitat simulation model  
206 "Estimhab" (Lamouroux and Capra, 2002; Lamouroux and Souchon, 2002; Souchon et al., 2003)  
207 which constitutes the habitat modeling module of the modeling platform "Estimkart"  
208 (Lamouroux et al., 2010). The first term of the HCP equation therefore weights the species, guild  
209 or invertebrates habitat suitability at a given reach against the overall habitat suitability of all fish  
210 species, guilds or invertebrates respectively, in the same river. In short, the weighted *HS*

211 represents the reference habitat condition for the chosen species, guild or for benthic  
212 macroinvertebrates.

213 In the second term of equation 3, *WUA* is the weighted usable area in  $m^2$  calculated as  
214  $HS_{ij} \cdot W_i \cdot 100$  where  $W_i$  is the width of the river reach *i* in meters, which is multiplied by 100 m  
215 length. *WUA* represents therefore the surface of suitable habitat in a reach of a given width  
216 (Bovee, 1982), and it is quantified on a constant 100 m length to allow comparability between  
217 river segments of different length (e.g. 100 m of 0.4 *HS* and 50 m of 0.8 *HS* would give the same  
218 *WUA* as a result of two completely different ecological conditions). The derivative of *WUA* in  
219 the second term is the change in habitat area to discharge change calculated through two  
220 different models (Lamouroux and Capra, 2002; Lamouroux and Jowett, 2005; Lamouroux and  
221 Souchon, 2002) depending on the *WUA* equation attributed in literature to the different taxa  
222 based on the best fit to observed abundance data (Table 1, eq. 4 and 5). The different models  
223 correspond to different types of species response to flow.

224 *Model 1:*

$$WUA_i = A_i \cdot [Re_i^C \cdot \exp(-K \cdot Re_i)] \cdot W_i \cdot 100 \quad (4)$$

225 *Model 2:*

$$WUA_i = A_i \cdot [1 - C \cdot \exp(-K \cdot Re_i)] \cdot W_i \cdot 100 \quad (5)$$

226 [Table 1]

227 In both models,  $Re_i$  is the Reynolds number, representing specific river discharge and  
228 turbulence in the river reach  $i$ .  $Re_i$  is defined as  $Q_i / \nu \cdot W_i$  where  $\nu$  is the kinematic viscosity of  
229 water, considered equal to  $10^{-6} \text{ m}^2 \text{ s}^{-1}$  (Lamouroux et al., 1999). The dimensionless parameter  $A_i$   
230 is a distinctive, static descriptor of the reach. It is based on its average characteristics at a median  
231 discharge level, i.e. it is independent of discharge and its alteration (Appendix A, section 3).  
232 Conversely, the constants  $C$  and  $K$ , shared by all river segments, determine the rate of change of  
233  $WUA_i$  with  $Re_i$ , within reaches. As in Estimhab, viscosity is multiplied by  $10^7$  to run the  
234 calculation with low  $Re_i$  numbers. Since river width varies with discharge,  $W_i$  can in turn be  
235 written as  $a_i Q_i^{b_i}$  where  $a_i$  and  $b_i$  are the hydraulic geometry coefficient and exponent of the width-  
236 discharge power relation (Leopold and Maddock, 1953; Miguel et al., 2016). The analytical  
237 derivative of  $WUA_i$  on discharge  $Q_i$  can be calculated as:

238 *Model 1:*

$$\frac{dWUA_i}{dQ_i} = A_i \cdot \left( \frac{Q_i}{\nu \cdot a_i Q_i^{b_i}} \right)^C \cdot \exp\left(-K \cdot \frac{Q_i}{\nu \cdot a_i Q_i^{b_i}}\right) \cdot \left[ a_i Q_i^{b_i-1} \cdot (C - C b_i + b_i) - \frac{K}{\nu} \cdot (1 - b_i) \right] \cdot 100 \quad (6)$$

239 *Model 2:*

$$\frac{dWUA_i}{dQ_i} = A_i \cdot \left[ -C \cdot \exp\left(-K \cdot \frac{Q_i}{\nu \cdot a_i Q_i^{b_i}}\right) \cdot \left(-\frac{K}{\nu} \cdot (1 - b_i) + b_i a_i Q_i^{b_i-1}\right) + b_i a_i Q_i^{b_i-1} \right] \cdot 100 \quad (7)$$

240 The values for  $K$  and  $C$ , as well as the models used for the parameter  $A$ , are included in  
241 Appendix A, Table A.3.  $K$ ,  $C$  and  $A$  are applied indifferently to *HS* empirical equations and to

242 the related *WUA* derivatives. Since the parameter *A* represents the average river characteristics,  
243 values lower or equal to 0 mean that the habitat is not suitable for a given species or group. In  
244 such cases, the terms of the HCP equation are therefore set to 0 in the calculation of the  
245 characterization factor. The comparison of *WUA* analytical and numerical derivatives confirmed  
246 the consistency of calculations (see Appendix A, Fig. A.1).

247 To provide the reader with an overall interpretation of the characterization factor, the proposed  
248 indicator (HCP) represents therefore the change in  $\text{m}^2$  habitat quantity (*WUA*) from baseline  
249 habitat suitability conditions (*HS*), induced by river discharge alteration ( $\text{m}^3/\text{s}$ ). For instance,  
250 consuming water in a river with HCP equal to  $100 \text{ m}^2 \cdot \text{s}/\text{m}^3$  means altering ten times more  
251 usable habitat surface than in a river with HCP equal to  $10 \text{ m}^2 \cdot \text{s}/\text{m}^3$ .

### 252 *3.2. Application of the characterization model and aggregation*

253 The characterization model based on HCP has been implemented in the software suite R (R  
254 Core Team, 2016; RStudio Team, 2016) using some elements developed by Miguel et al. (2016)  
255 and applied to the French hydrographic network (RHT, Pella et al., 2012) which includes 114  
256 332 river segments with the associated discharges and other topographical information (e.g.  
257 altitude, river length, Strahler order). The mean river reach length is 24.7 km (20.4 km standard  
258 deviation). Reach length is generally sufficient for including the diversity of available aquatic  
259 habitats. In order to appraise the sensitivity of habitat conditions to dry seasons, the CF has been  
260 calculated for RHT *Q50* (median) and *Q90* (low) flows, which are the water discharges in cubic  
261 meters per second equaling or exceeding respectively the 50 and 90 percent of the time in the  
262 year. *Q90* applies to dry seasons and *Q50* is the median discharge that is assumed to be  
263 characteristic for the rest of the year. The reason of this choice is that most water abstraction

264 works (except reservoirs) do not alter flows much higher than  $Q_{50}$ , and low flow quantiles ( $Q_{90}$ )  
265 are good predictors of aquatic community characteristics (Lamouroux et al., 1999). The  
266 ecological consequences of high flow pulses and temporal variability of flow events are out of  
267 the scope of the present study, and the preference models used here are relevant for low to  
268 intermediate discharge rates only. Where  $Q_{50}$  and  $Q_{90}$  data were not available (3 635 river  
269 segments, which represent 3% of the overall RHT river network database), median and low  
270 flows have been calculated from the given inter-annual average discharge ( $QM$ ) based on the  
271 coefficients of the linear model fitted to available  $QM$ ,  $Q_{50}$  and  $Q_{90}$  data. The regression has  
272 been carried out on RHT reaches with  $QM$  between 0.001 and 950 m<sup>3</sup>/s since all missing values  
273 were for rivers within this range of discharge (Appendix A, Fig. A.2). In order to calculate the  
274 HCP model input variables that were not included in the RHT, namely the parameters of the  
275 width-discharge relation and hydraulic geometry dependent variables, the hydraulic geometry  
276 and the habitat simulation (Estimhab) modules of Estimkart have been used. In this way, it has  
277 been possible to calculate all model's input variables from the hydrological and topographical  
278 information provided by the RHT database (specifically discharges, Strahler order, drainage area  
279 and river slope).

280 For each river segment, the HCPs indicated in Table 1 have been calculated separately.  
281 Aggregated characterization factors are also provided to enable the applicability of habitat  
282 models in Life Cycle Assessment (LCA), which requires aggregation into coarser spatial  
283 resolutions in order to align with the resolution of life cycle inventory data quantifying  
284 elementary flows such as a water abstraction or discharge. Multi-species aggregated indicators at  
285 the reach scale have been calculated: one for species and one including guilds and invertebrates'  
286 biomass production. Species and guilds HCPs cannot be combined because some species are

287 already counted in one or more guilds depending on the size class (Appendix A, Table A.2). *HS*  
288 functions are not monotonic and can increase or decrease depending on discharge. Therefore,  
289 *WUA* derivatives and HCP values can be positive or negative, meaning that discharge alteration  
290 may respectively lead to habitat loss or habitat gain. In order to test the results' sensitivity to  
291 positive and negative HCPs, multi-species aggregation has been carried out respectively with the  
292 individualist, hierarchist and egalitarian cultural perspectives (Thompson et al., 1990). The  
293 choice between different aggregated LCIA indicators is based on the consideration of different  
294 perceptions of nature as previously addressed in environmental risk assessment (Steg and  
295 Sievers, 2000) and life cycle assessment (Goedkoop and Spriensma, 2001; Huijbregts et al.,  
296 2016).

297 According to the *individualist* perspective (eq. 8), nature is in equilibrium and able to  
298 compensate for anthropogenic environmental alterations. Positive and negative HCPs are  
299 therefore not weighted and habitat gain counterbalances habitat loss among species.

$$iHCP_i = \sum_{j=1}^n HCP_{ij} \quad (8)$$

300 The *hierarchist* approach (eq. 9) assumes that nature can offset an impact within certain  
301 acceptable limits that can be defined and controlled by expert judgement. In such a regulation-  
302 oriented perspective, only the most vulnerable species are considered and therefore habitat loss  
303 corresponding to positive HCP. This approach is based on the same logic adopted in Miguel et  
304 al. (2016) where maximum percent habitat alteration is calculated.

$$hHCP_i = \sum_{j=1}^n HCP_{ij} > 0 \quad (9)$$

305 Under the *egalitarian* perspective (eq. 10) nature is ephemeral. Every perturbation of its  
306 equilibrium is equally weighted and judged negatively according to the precautionary principle.

$$eHCP_i = \sum_{j=1}^n |HCP_{ij}| \quad (10)$$

307 To allow for regionalized water consumption LCIA of instream habitats, each individual and  
 308 multi-species aggregated characterization factor for a given reach has been upscaled to the sub-  
 309 watershed and watershed scale. The spatial aggregation has been performed based on the length  
 310 of each river segment and thus on the related habitat quantity against the total habitat availability  
 311 in the watershed (eq. 11), the latter being identified according to four HydroBASINS Pfafstetter  
 312 levels (Lehner and Grill, 2013). Pfafstetter codes have been merged to RHT attributes using the  
 313 Quantum GIS geographic information system (Quantum GIS Development Team, 2017).

$$CF_w = FF_w \cdot EF_w = \frac{dQ_w}{dCWU_w} \cdot \sum_{i=1}^n HCP_i \cdot \frac{l_i}{\sum_{i=1}^n l_i} \quad (11)$$

314 In the above equation CF, FF and EF are calculated at watershed  $w$ , and  $l$  is the river reach  
 315 length weighted to total length of river segments in the watershed.

#### 316 **4. Results**

317 Characterization results in RHT river segments are highly variable, as represented in Fig. 2 for  
 318 riffle species and in Fig. 3 for all four guilds. Depending on the observed biological group, river  
 319 reach HCPs fall within three or four order of magnitude ranges. The detail of riffle species HCP  
 320 density distribution confirms that habitat sensitivity to water consumption is predictably greater  
 321 in low flow periods ( $Q_{90}$ ) than in normal conditions ( $Q_{50}$ ). Leptokurtic, heavy tailed and right-  
 322 skewed distributions are highlighted in the graphs, indicating a non-normal distribution of the  
 323 data sample, confirmed by the Q-Q plots in Appendix A (Fig. A.3), suggesting a gamma  
 324 distribution. For this reason, robust statistical measures have been used to analyze the  
 325 characterization results, namely the Median Absolute Deviation (MAD) for statistical dispersion  
 326 and the Medcouple measure of skewness to identify outliers (Hubert and Vandervieren, 2008), in

327 order to limit the influence of extreme values. In Fig. 2, compared to median yearly flow  
328 conditions, the dry season moves the medians (M) toward higher HCPs ( $M_{Q50} = 54.3 \text{ m}^2 \cdot \text{s/m}^3$ ;  
329  $M_{Q90} = 137.7 \text{ m}^2 \cdot \text{s/m}^3$ ), increases the number of extreme values ( $\text{Max}_{Q50} = 2\,519.3 \text{ m}^2 \cdot \text{s/m}^3$ ;  
330  $\text{Max}_{Q90} = 4\,958.8 \text{ m}^2 \cdot \text{s/m}^3$ ) and increments the statistical dispersion ( $\text{MAD}_{Q50} = 62.7 \text{ m}^2 \cdot \text{s/m}^3$ ;  
331  $\text{MAD}_{Q90} = 139.2 \text{ m}^2 \cdot \text{s/m}^3$ ). Because of the large size of the sample and the nature of data  
332 distribution, a relevant number of outliers were identified (Fig. 3). The corresponding river  
333 segments were however kept unmodified in the resulting HCP database. A check of outlying  
334 reaches was performed and no artifact was detected. For this reason, it was assumed the general  
335 validity of the modeled extreme habitat conditions, based essentially on their morphological and  
336 hydrological characteristics (low order streams, small size and low discharge).

337 [Figure 2]

#### 338 *4.1. HCP multi-species aggregation at reach scale*

339 HCP frequency distributions of the other taxa included in this study follow a tendency akin to  
340 the one discussed above for riffle species. However, results demonstrate that for the same  
341 amount of water consumed, some species are more sensitive to habitat change than others. Fig. 3  
342 shows that fish guilds adapted to shallow water habitats are in fact the most vulnerable to habitat  
343 loss from water consumption (see Appendix A, figures A.4 and A.5 for all species and  
344 invertebrates HCPs), which is most likely accentuated by the adoption of the weighting factor in  
345 eq. 3. It is also evident that positive effect factors, meaning habitat loss, are more frequent than  
346 negative ones (Fig. 3), this is due to the fact that the *WUA* derivatives are generally positive for  
347 the considered flows. For these reasons, the multi-species aggregation of HCPs at the reach scale  
348 is driven by a limited number of taxa most likely subject to habitat loss from water consumption.  
349 The adopted *individualist*, *hierarchist* and *egalitarian* aggregation approaches therefore do not

350 show substantial differences and, following the parsimony principle, the *individualist* perspective  
351 should be used for aggregating LCIA habitat indicators at the reach scale (see example in  
352 Appendix A, Fig. A.8, for an application of the three perspectives at watershed scale).

353 [Figure 3]

354 As an illustrative example, the application of habitat indicators, aggregated at reach scale, to  
355 the Durance-Verdon river basin in France (Fig. 4) shows the distribution of habitat sensitivity to  
356 water consumption in that watershed. Fish guilds and benthic invertebrates HCPs result in a  
357 cumulative characterization factor to which riffle and bank species have a major contribution.  
358 Individualist HCP (iHCP<sub>gi</sub>) for guilds and invertebrates were chosen for the representation.

359 [Figure 4]

360 In Fig. 4, iHCP classes are defined by percentiles (p10 – p100) indicating the river segments  
361 amount that falls below or is equal to the upper bounds of each class. For instance, in 90% of  
362 river reaches  $iHCP_{gi} \leq 1\,711 \text{ m}^2 \cdot \text{s/m}^3$ . As a consequence of a gamma-like distribution of HCP  
363 values, 10% of rivers (between p90 and p100) present the highest scores, varying by almost a  
364 factor of five from the lower to the upper boundary of the same class.

365 Physical habitat indicators at the river reach scale are highly correlated to the Reynolds  
366 number, as it is the discharge-dependent input parameter of the habitat model. Spearman's  
367  $\rho = -0.99$  indicates a non-linear, negative, and monotonic correlation between iHCP<sub>gi</sub> and  
368 Reynolds number. Following the definition of the Reynolds number given in materials and  
369 methods, it is straightforward that habitat change potentials depend largely on discharge ( $Q_{90}$   
370  $\rho = -0.97$ ) and river size (width  $\rho = -0.81$ ; depth  $\rho = -0.91$ ; velocity  $\rho = -0.86$ ). These variables  
371 are in fact at the root of the differences represented in Fig. 4 between the north-eastern and the  
372 south-western area of the watershed. It is also interesting to note that altitude is not significant

373 for the habitat change potential definition ( $\rho = -0.05$ ) while stream order (Strahler, 1957) is a  
374 necessary condition, but not sufficient, to determine habitat sensitivity (negative correlation  $\rho = -$   
375 0.67 with Strahler order). In other terms, high Strahler stream orders are generally less sensitive  
376 to water consumption while low order tributaries show high HCPs unless certain conditions of  
377 discharge and size are satisfied.

378 Fig. A.6 in Appendix A shows the difference between guilds and invertebrates  $iHCP_{gi}$  in  
379 normal condition and dry season at the national scale. Median and average values at  $Q50$   
380 increase in  $Q90$  periods (median from 175.5 to 442.8  $m^2 \cdot s/m^3$ ; average from 254.9 to 764.5  $m^2 \cdot$   
381  $s/m^3$ ). Results are consistent with species-specific, aggregated habitat change potentials in  
382 Appendix A, Fig. A.7, where aggregated HCP are driven by individual habitat indicators of  
383 brown trout juvenile, gudgeon, stone loach and minnow.

#### 384 *4.2.HCP spatial aggregation at watershed scale*

385 In addition to the reach scale, characterization factors have been aggregated at four different  
386 spatial scales based on the HydroBASINS data base watershed boundaries (Fig. 5). HCPs at  
387 reach scale are weighted by the relative river length against the total length of watershed river  
388 segments. Weighted habitat surface represents therefore the habitat frequency in the watershed. It  
389 is positively correlated to the probability of habitat alteration at watershed scale due to water  
390 consumption, should site specific information is not available.

391 In Fig. 5,  $iHCP_{gi}$  values are progressively averaged as the spatial resolution decreases.  
392 Maximum HCP is 2 625.8  $m^2 \cdot s/m^3$  and 759.2  $m^2 \cdot s/m^3$  at HydroBASINS levels 6 and 3  
393 respectively, for 1  $m^3$  of water consumed in dry season. The picture highlights how sub-  
394 watersheds characterized by high habitat change potentials determine the overall watershed CF at

395 each step of the spatial aggregation. A valid alternative to the spatial aggregation formula  
396 proposed in the present study implies using weighted medians to limit the influence of extremes  
397 in aggregated CFs. However, since HCP distribution is the same in all watersheds, the change in  
398 CF values would not be significant for comparative LCA. Moreover, moving the CF closer to the  
399 median value of the data sample would imply higher risk of underestimating the habitat change  
400 potential of the watershed considering potential model uncertainties. For this reason and  
401 following a precautionary principle, the proposed aggregation method should be used (see  
402 Appendix A, section 8 for an example of the application of the weighted median spatial  
403 aggregation method).

404 [Figure 5]

## 405 **5. Discussion**

406 River reach characterization factors may represent a useful instrument for site-specific LCA,  
407 complementary to Environmental Risk Assessment (ERA) and environmental impact assessment  
408 (EIA) (Larrey-Lassalle et al., 2017). Multi-species aggregation at the reach scale and spatial  
409 aggregation at the watershed scale represent a parsimonious approach to modeling habitat change  
410 potential, which is necessary to respond to the need for large-scale spatialized water management  
411 and LCIA approaches advocated by ISO 14044 (International Organization for Standardization  
412 ISO/TC 207/SC 5, 2006) and increasingly applied in LCA (Loiseau et al., 2014; Nitschelm et al.,  
413 2016; Patouillard et al., 2018).

### 414 *5.1. Model uncertainty*

415 Optimizing the spatial resolution is crucial to reduce the uncertainty of the impact assessment  
416 coming from neglecting spatial variability. In principle, the higher the spatial resolution of the  
417 CF, the smaller the uncertainty contribution due to spatial variability. However, the choice of the  
418 most appropriate scale depends largely on the availability of regionalized inventory data  
419 (Henderson et al., 2017; Mutel et al., 2012). In addition, the characterization factor presented in  
420 this study is built on the HCP model which is essentially an effect factor. The development of a  
421 regionalized fate factor at comparable spatial resolutions is therefore necessary for the  
422 optimization of the resolution of the characterization model.

423 Section 8 of Appendix A discusses another potential source of spatial uncertainty deriving  
424 from the aggregation formula of watershed CF. The weighted average approach implies  
425 overestimating impact assessment results for certain river segments compared to the weighted  
426 median method which, on the contrary, shows higher risk of underestimation. The choice of the  
427 best aggregation method can thus be subject to the need of more or less conservative approaches  
428 depending on the specific application scenario (e.g. LCA in critical regions where small stream  
429 habitats are endangered or exposed to multiple stressors).

430 The uncertainty deriving from model parameters should also be considered in the  
431 interpretation of characterization results. For instance, since discharge data for ungauged river  
432 reaches are hardly available, in the RHT database flow duration curves are modeled for the  
433 whole river network, and therefore  $Q_{50}$  and  $Q_{90}$  values. The same goes for hydraulic geometry  
434 variables modeled through Estimkart. Uncertainty plays a relevant role especially for small  
435 catchments, low discharges and mountainous areas. On the contrary, parameters estimates are  
436 more accurate in downstream river segments and bigger catchments, where the boundaries are  
437 defined more precisely (Lamouroux et al., 2014). Contrary to spatial uncertainty, model

438 parameters uncertainty is therefore potentially higher at the river reach scale than it is for habitat  
439 alteration quantification over large areas and thus spatially aggregated CFs taking into account  
440 average watershed conditions. In support of these considerations, the uncertainty analysis by  
441 Miguel et al. (2016) demonstrated that habitat changes at the regional scale are generally robust  
442 despite high uncertainties at the reach scale. For this reason, a reliable CF for LCIA should aim  
443 at minimizing spatial variability of inventory data, FF and EF, although ensuring reasonable  
444 modeling of average watershed characteristics.

## 445 *5.2. Operationalization and model extension*

446 For a complete operationalization of the CF, associating a multimedia fate factor to the HCP  
447 model would be needed (Núñez et al., 2018). This implies including the consideration of  
448 different water sources (groundwater and surface water), the water flow transport between  
449 different compartments in the hydrological cycle (e.g. lateral flow from irrigated land, returning  
450 to the original water basin), along with water withdrawal and discharge areas and therefore the  
451 spatialization of hydrologic alteration induced by water consumption (e.g. direct and indirect  
452 alteration, potential longitudinal cascade effects on surface waters as in Loubet et al., 2013). In  
453 the same way, the characterization of different water uses in the life cycle inventory should  
454 justify the adoption of an effect factor for marginal water consumption (i.e. life cycle impact of  
455 river damming would imply non-marginal hydrologic alteration in river basins).

456 Data availability could be the major constraint for refining and extending the model outside of  
457 France. At present, comprehensive databases including all necessary input data (i.e. discharge,  
458 substrate composition, hydraulic geometry) are not available globally, although useful data  
459 sources are currently being developed (e.g. Lehner, 2018). The model can be therefore applied

460 using local data. For an application on the larger scale the input variables should be modeled if  
461 not available, such as in the French RHT hydrographic network, or the model should be  
462 simplified retaining the factors that most determine characterization results (see section 4.1). For  
463 improving taxa coverage, more habitat equations should be included when possible (other  
464 freshwater fish species could be assigned to the four guilds based on habitat preference). In  
465 addition, using Q50 and Q90 at monthly resolution would improve the precision and temporal  
466 relevance of the CF, since river flow regimes can be different in relation to topographical and  
467 climatic conditions.

## 468 **6. Conclusions**

469 In this study a new, mechanistic characterization factor based on freshwater physical habitat  
470 modeling has been presented. The effect factor model has been applied at the river and watershed  
471 scale to assess marginal impact of water consumption on instream ecosystems. HCP has been  
472 calculated at  $Q_{50}$  and  $Q_{90}$  as representative of the median discharge condition and dry season  
473 respectively.

474 A simplified fate factor has been associated to the HCP model, limiting the CF applicability to  
475 surface water consumption LCIA. In addition, the appropriate impact assessment spatial scale  
476 has to be evaluated considering spatialized life cycle inventory data availability and the  
477 resolution of the fate factor, to limit the risk of hyper-regionalization (Heijungs, 2012) and the  
478 uncertainty of the model's parameters. However, modeling the potential midpoint impact on  
479 freshwater physical habitat availability can be considered as a first breakthrough from empirical  
480 towards mechanistic quantification of instream ecosystems damage due to marginal water  
481 consumption.

482 Notwithstanding the environmental relevance of HCP as a proxy for ecosystem damage, the  
483 development of large scale endpoint indicators for LCIA against this background should consider  
484 actual community structures through species distribution data or probability of presence/absence  
485 (Miguel et al., 2016). This would allow moving from the characterization of potential habitat  
486 alteration to species damage, developing, for example, vulnerability metrics for endangered  
487 specialists (Woods et al., 2017). Ultimately, species density and abundance (number of  
488 individuals per species) linked to habitat change could be used to derive LCIA density and  
489 abundance indicators contributing to a more comprehensive assessment of freshwater ecosystems  
490 quality through different, complementary biodiversity indicators.

#### 491 **Associated content**

492 Appendix A and the HCP modeling dataset file are available with this article. The RHT  
493 database and the R script are available upon request to the authors.

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729

730 **Tables**

731 **Table 1.** Fish species, guilds and invertebrates models used for HCP calculation (Lamouroux  
732 and Capra, 2002; Lamouroux and Souchon, 2002)

### 733 **Figures**

734 **Fig. 1.** Simplified impact pathways of water consumption and ecosystem responses for  
735 freshwater-dependent biological communities. Codes in brackets indicate additional references  
736 used to construct the flow diagram (Appendix A, Table A.1)

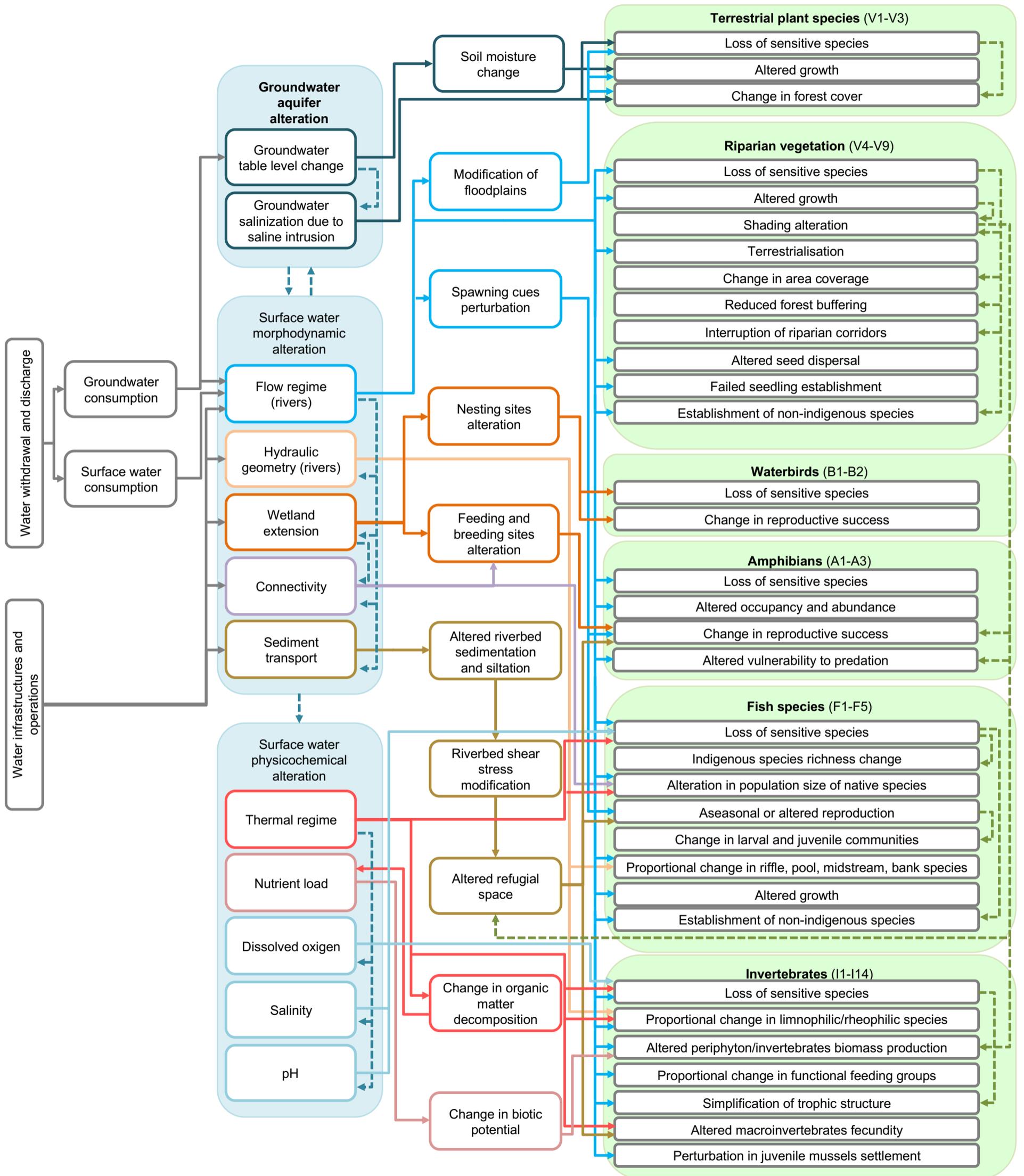
737 **Fig. 2.** Density distribution of the HCP for riffle fish species in French river reaches ( $m^2 \cdot s / m^3$ )  
738 with Min: minimum value; Max: maximum value; M: median; MAD: Median Absolute  
739 Deviation

740 **Fig. 3.** Spread of habitat change potentials HCP for pool, bank, riffle and midstream fish guilds  
741 ( $m^2 \cdot s / m^3$ )

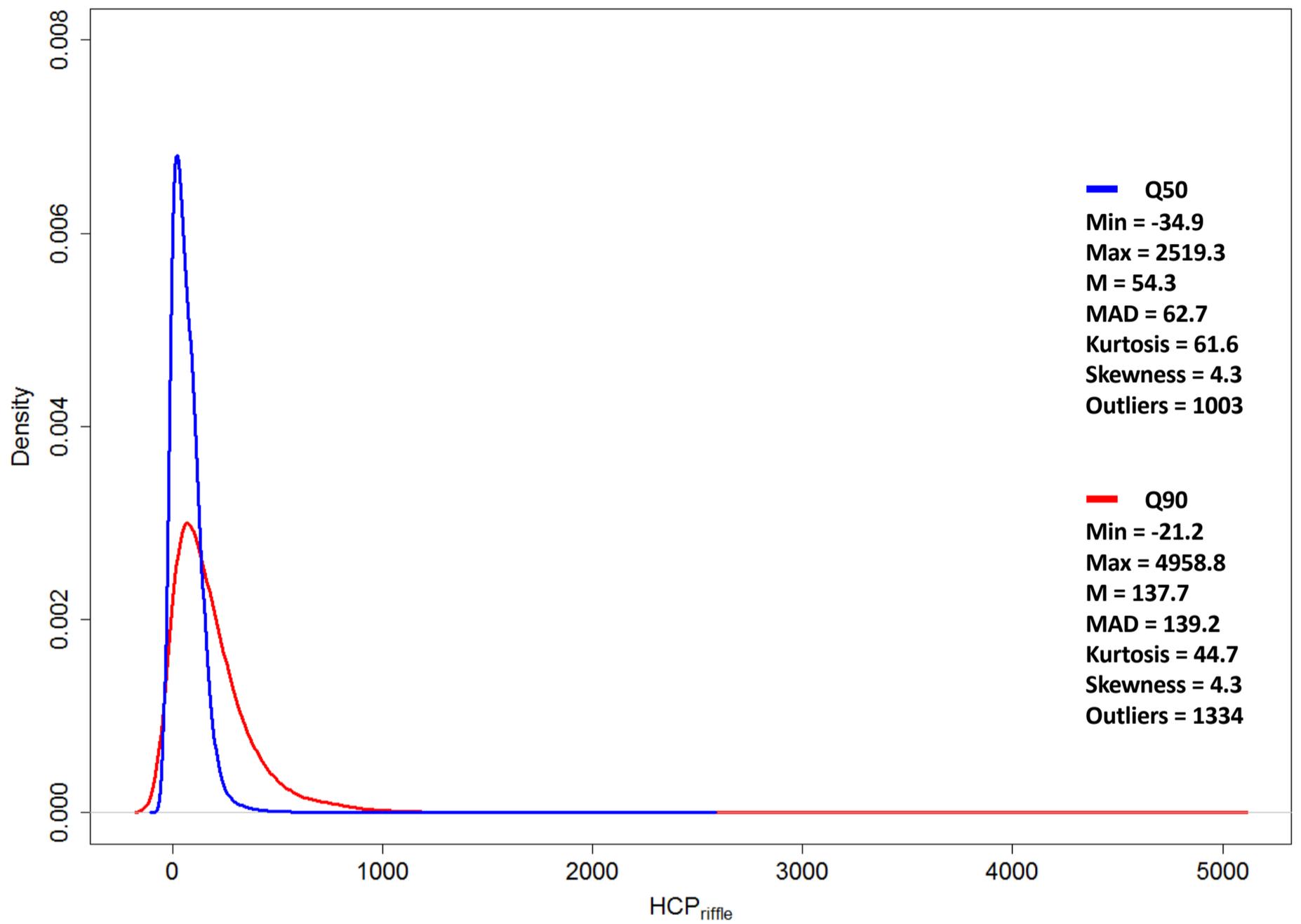
742 **Fig. 4.** Percentiles (p10 - p100) of individualist HCP ( $iHCP_{gi}$ ) values aggregated at reach scale  
743 for fish guilds and invertebrates in  $m^2 \cdot s / m^3$ . Application of the characterization model to the  
744 Durance-Verdon river basin in France at  $Q_{90}$

745 **Fig. 5.** Percentiles (p10 - p100) of aggregated  $iHCP_{gi}$  at HydroBasins levels 6 (a), 5 (b), 4 (c) and  
746 3 (d), values in  $m^2 \cdot s / m^3$  at  $Q_{90}$

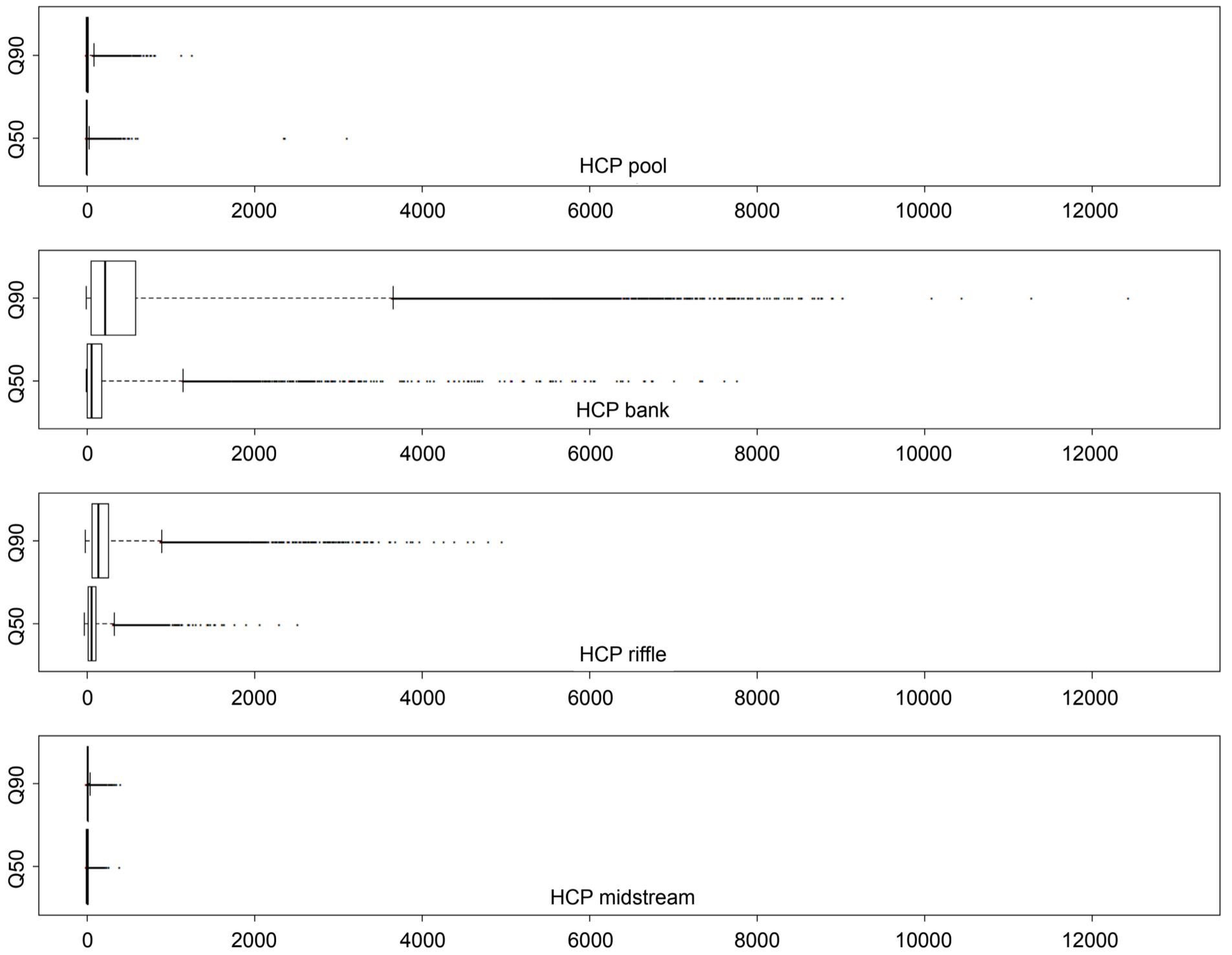
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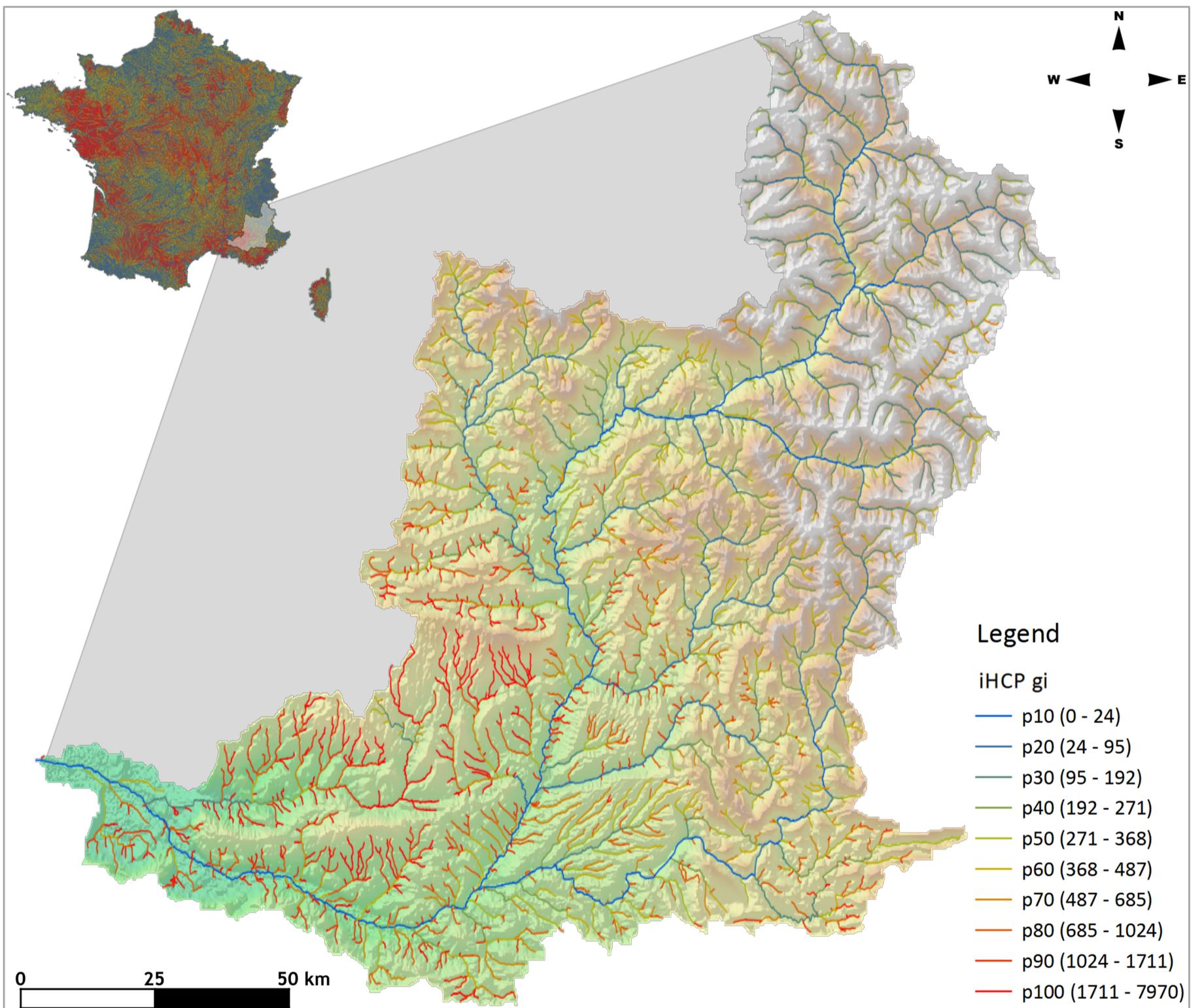
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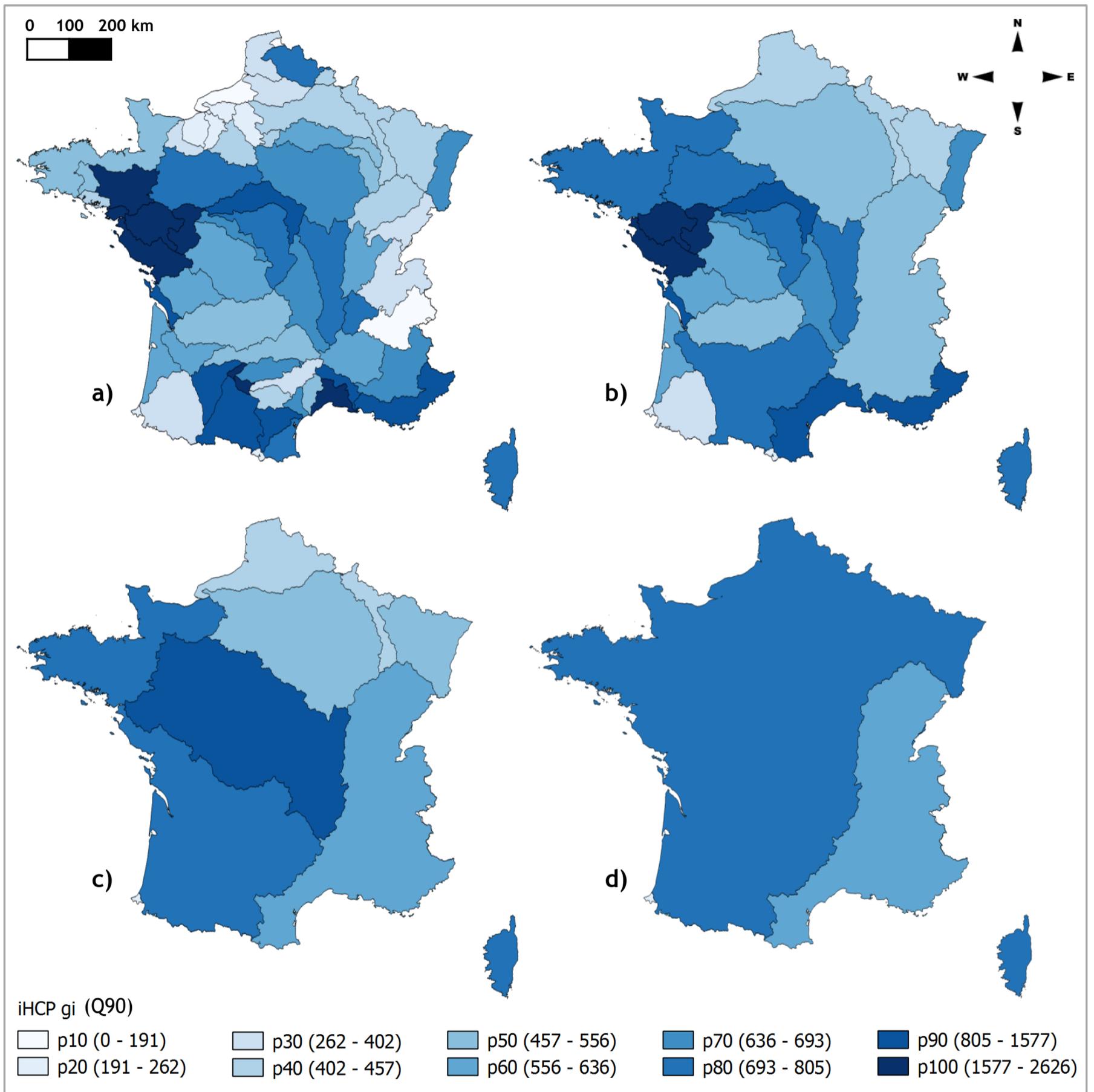
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**Fig. 3.** Spread of habitat change potentials HCP for pool, bank, riffle and midstream fish guilds ( $m^2 \cdot s / m^3$ )



**Fig. 4.** Percentiles (p10 - p100) of individualist HCP (iHCP<sub>gi</sub>) values aggregated at reach scale for fish guilds and invertebrates in  $\text{m}^2 \cdot \text{s}/\text{m}^3$ . Application of the characterization model to the Durance-Verdon river basin in France at  $Q_{90}$



**Fig. 5.** Percentiles (p10 - p100) of aggregated  $iHCP_{gi}$  at HydroBasins levels 6 (a), 5 (b), 4 (c) and 3 (d), values in  $m^2 \cdot s/m^3$  at  $Q90$

1 **Table 1**

2 Fish species, guilds and invertebrates models used for HCP calculation (Lamouroux and Capra,  
3 2002; Lamouroux and Souchon, 2002)

| <b>Fish species</b>                      |                                       |                    |                   |
|--|---------------------------------------|--------------------|-------------------|
| <b>Family</b>                            | <b>Scientific name</b>                | <b>Common name</b> | <b>Model</b>      |
|  |                                       |                    | Adult: 1          |
| <i>Salmonidae</i>                        | <i>Salmo trutta</i> (L., 1758)        | Brown trout        | Juvenile & fry: 1 |
| <i>Cyprinidae</i>                        | <i>Barbus barbus</i> (L., 1758)       | Barbel             | 2                 |
| <i>Cottidae</i>                          | <i>Cottus gobio</i> (L., 1758)        | Sculpin            | 1                 |
| <i>Cyprinidae</i>                        | <i>Gobio gobio</i> (L., 1758)         | Gudgeon            | 1                 |
| <i>Cobitidae</i>                         | <i>Barbatula barbatula</i> (L., 1758) | Stone loach        | 1                 |
| <i>Cyprinidae</i>                        | <i>Phoxinus phoxinus</i> (L., 1758)   | Minnow             | 1                 |
|  |                                       |                    | Fry: 1            |
| <i>Salmonidae</i>                        | <i>Salmo salar</i> (L., 1758)         | Atlantic salmon    | Juvenile: 1       |
|  |                                       |                    | Fry: 1            |
|  |                                       |                    | Juvenile: 1       |
| <i>Salmonidae</i>                        | <i>Thymallus thymallus</i> (L., 1758) | European grayling  | Adult: 1          |
| <b>Fish guilds</b>                       |                                       |                    |                   |
|  |                                       |                    | 2                 |
| Pool                                     |                                       |                    |                   |
| Bank                                     |                                       |                    | 2                 |
| Riffle                                   |                                       |                    | 1                 |
| Midstream                                |                                       |                    | 1                 |
| <b>Invertebrates</b>                     |                                       |                    |                   |
|  |                                       |                    | 1                 |
| Benthic invertebrates biomass production |                                       |                    |                   |

4

