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Abstract

In this article a new characterization model and factors are proposed for the life cycle impact assessment (LCIA) of water consumption on instream freshwater ecosystems. Impact pathways of freshwater consumption leading to ecosystem damage are described and the alteration of instream physical habitat is identified as a critical midpoint for ecosystem quality. The LCIA characterization model aims to assess the change in habitat quantity due to consumptive water use. It is based on statistical, physical habitat simulation for benthic invertebrates, fish species and their size classes, and guilds of fish sharing common habitat preferences. A habitat change potential (HCP) midpoint, mechanistic indicator, is developed and computed on the French river network at the river reach scale (the river segment with variable length between the upstream and downstream nodes in the hydrographic network), for median annual discharges and dry seasons. Aggregated, multi-species HCPs at a river reach are proposed using various aggregation approaches. Subsequently, the characterization factors are spatially aggregated at watershed and sub-watershed scales. HCP is highly correlated with median and low flow discharges, which determine hydraulic characteristics of reaches. Aggregation of individual HCPs at reach scale is driven by the species most sensitive to water consumption. In spatially aggregated HCPs, consistently with their reduced smaller average discharge rate, small stream habitats determine the overall watershed characterization. The study is aimed primarily at life cycle assessment (LCA) practitioners and LCIA modelers. However, since it is the result of a productive cross-fertilization between the ecohydrology and LCA domains, it could be potentially useful for watershed management and risk assessment as well. At the moment, the proposed model is applicable in France. For a broader implementation, the development of global, high resolution river databases or the generalization of the model are needed. Our new factor represents nevertheless an advancement in freshwater ecosystems LCIA laying the basis for new metrics for biodiversity assessment.

Keywords

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

Abbreviations

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

1. Introduction

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

In this context, several life cycle impact assessment (LCIA) models have been proposed to link freshwater consumptive use to biodiversity loss (Núñez et al., 2016) and addressing specifically wetlands and surface water-dependent ecosystems (Amores et al., 2013; Hanafiah et al., 2011; Tendall et al., 2014; Verones et al., 2013b). These approaches provide endpoint indicators built on cause-effect pathways in which the impact characterization factor (CF, eq. 1) results from the combination of three sub-factors (Núñez et al., 2018). The fate factor (FF) represents the environmental change (e.g. change in m^3/y river discharge or m^2 wetland area) due to water consumption, defined as the withdrawn water that is not returned to the original drainage basin (International Organization for Standardization ISO/TC 207/SC 5, 2014). The exposure factor (XF) indicates how far this alteration can be offset, e.g. in a river it can be approximated to 1 since most freshwater species have less mobility than terrestrial species to compensate the lack of water. Finally, the effect factor (EF) describes the consequence on the ecosystem (e.g. potentially disappeared fraction of species, PDF).

$$\mathrm{CF} = \mathrm{FF} \cdot \mathrm{XF} \cdot \mathrm{EF} \tag{1}$$

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

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

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

While LCIA methods based on SAR and SDR have the advantage of taking into account biological aspects compared to stress-based indicators (Berger and Finkbeiner, 2013; Boulay et al., 2017), exploring complementary options to model aspects of biological communities other than species richness (number of species), such as species abundance (number of individuals per species) or diversity (Tuomisto, 2010), would therefore support a more comprehensive analysis of ecosystem quality (Curran et al., 2011; Damiani et al., 2017). This raises the question on whether it would be possible to improve current LCIA models to appropriate recent, available knowledge concerning freshwater ecology and especially environmental flow management (Angus Webb et al., 2013; Damiani et al., 2017; Poff and Zimmerman, 2010). The aim of this study is first to trace, in detail, the potential water consumption impact pathways on freshwater habitat and ecosystems. The purpose is to identify the main, relevant environmental mechanisms and develop a local, bottom-up, mechanistic model addressing the limitations of current LCIA models. Since lotic habitats are currently the most vulnerable to freshwater consumptive use (Vörösmarty et al., 2010), a habitat-based characterization factor linking marginal hydrological alteration to instream ecosystems effects is proposed, based on existing literature on the applicability of ecohydrological methods in LCIA (Damiani et al., 2017). The model is subsequently applied on French streams and the feasibility, ecological relevance and limitations of upscaling from the river to the watershed scale is discussed.

2. Impact pathway analysis

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

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

It is also necessary to bear in mind that habitat characteristics can be influenced by multiple drivers, for instance nutrient availability in water can be determined by flow regime, thermal regime and the presence or absence of forest buffers (Kennen et al., 2008; Kløve et al., 2014). This draws attention to the fact that habitats not only drive the establishment of a specific type of ecosystem, but can also be shaped themselves by the biota they harbor. These ecosystem mechanisms involve all abiotic and biotic changes induced or limited by the presence of certain ecological communities (Berke, 2010; Jones et al., 2010). In addition, habitat modifications can be further amplified by the proliferation of non-native species (Crooks, 2002; Ward and Ricciardi, 2010).

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

[Figure 1]

3. Materials and methods

3.1. Freshwater habitat modeling in LCIA

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

$$\mathrm{CF_i} = \mathrm{FF_i} \cdot \mathrm{EF_i} = \frac{\mathrm{d}Q_\mathrm{i}}{\mathrm{d}CWU_\mathrm{i}} \cdot \mathrm{HCP_i} \tag{2}$$

$\mathrm{CF_i}$ is the characterization factor for the river reach i, the fate factor ($\mathrm{FF_i}$) represents the marginal change in discharge $\mathrm{d}Q_\mathrm{i}$ (m^3/s) for marginal change in consumptive water use $\mathrm{d}CWU_\mathrm{i}$ (m^3/s). In the present study, the fate factor is considered equal to 1 as done by Hanafiah et al. (2011) and Tendall et al. (2014) LCIA models, meaning that 1 m^3/s of water withdrawn or released in the environment causes 1 m^3/s discharge alteration. This could be modeled more precisely using a mass-balance, multimedia fate modeling approach as proposed by Núñez et al. (2018). The effect factor (EF_i) is calculated as the habitat change potential HCP_i in $m^2 \cdot s/m^3$ of habitat surface derived from marginal discharge alteration (eq. 3). In the proposed approach, the value of the characterization factor corresponds therefore to the value of the effect factor and will be indicated indifferently as CF or HCP hereafter.

$$HCP_{\mathrm{ij}} = \frac{HS_{\mathrm{ij}}}{\sum_{\mathrm{j=1}}^{\mathrm{n}} HS_{\mathrm{ij}}} \cdot \frac{\mathrm{d}WUA_{\mathrm{ij}}}{\mathrm{d}Q_{\mathrm{i}}} \qquad (3)$$

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

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

Model 1:

$$WUA_{\mathrm{i}} = A_{\mathrm{i}} \cdot \left[Re_{\mathrm{i}}{}^{C} \cdot \exp(-K \cdot Re_{\mathrm{i}})\right] \cdot W_{\mathrm{i}} \cdot 100 \tag{4}$$

Model 2:

$$WUA_{\mathrm{i}} = A_{\mathrm{i}} \cdot [1 - C \cdot \exp(-K \cdot Re_{\mathrm{i}})] \cdot W_{\mathrm{i}} \cdot 100 \tag{5}$$

[Table 1]

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

Model 1:

$$\frac{\mathrm{d}WUA_{\mathrm{i}}}{\mathrm{d}Q_{\mathrm{i}}} = A_{\mathrm{i}} \cdot \left(\frac{Q_{\mathrm{i}}}{v \cdot a_i Q_{\mathrm{i}}{}^{b_i}}\right)^{C} \cdot \exp\left(-K \cdot \frac{Q_{\mathrm{i}}}{v \cdot a_i Q_{\mathrm{i}}{}^{b_i}}\right) \cdot \left[a_i Q_{\mathrm{i}}{}^{b_i - 1} \cdot (C - Cb_i + b_i) - \frac{K}{v} \cdot (1 - b_i)\right] \cdot 100 \tag{6}$$

Model 2:

$$\frac{\mathrm{d}WUA_{\mathrm{i}}}{\mathrm{d}Q_{\mathrm{i}}} = A_{\mathrm{i}} \cdot \left[-C \cdot \exp\left(-K \cdot \frac{Q_{\mathrm{i}}}{v \cdot a_i Q_{\mathrm{i}}{}^{b_i}}\right) \cdot \left(-\frac{K}{v} \cdot (1 - b_i) + b_i a_i Q_{\mathrm{i}}{}^{b_i - 1}\right) + b_i a_i Q_{\mathrm{i}}{}^{b_i - 1}\right] \cdot 100 \tag{7}$$

The values for K and C, as well as the models used for the parameter A, are included in Appendix A, Table A.3. K, C and A are applied indifferently to *HS* empirical equations and to the related *WUA* derivatives. Since the parameter *A* represents the average river characteristics, values lower or equal to 0 mean that the habitat is not suitable for a given species or group. In such cases, the terms of the HCP equation are therefore set to 0 in the calculation of the characterization factor. The comparison of *WUA* analytical and numerical derivatives confirmed the consistency of calculations (see Appendix A, Fig. A.1).

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

3.2. Application of the characterization model and aggregation

The characterization model based on HCP has been implemented in the software suite R (R Core Team, 2016; RStudio Team, 2016) using some elements developed by Miguel et al. (2016) and applied to the French hydrographic network (RHT, Pella et al., 2012) which includes 114 332 river segments with the associated discharges and other topographical information (e.g. altitude, river length, Strahler order). The mean river reach length is 24.7 km (20.4 km standard deviation). Reach length is generally sufficient for including the diversity of available aquatic habitats. In order to appraise the sensitivity of habitat conditions to dry seasons, the CF has been calculated for RHT *Q50* (median) and *Q90* (low) flows, which are the water discharges in cubic meters per second equaling or exceeding respectively the 50 and 90 percent of the time in the year. *Q90* applies to dry seasons and *Q50* is the median discharge that is assumed to be characteristic for the rest of the year. The reason of this choice is that most water abstraction works (except reservoirs) do not alter flows much higher than *Q50*, and low flow quantiles (*Q90*) are good predictors of aquatic community characteristics (Lamouroux et al., 1999). The ecological consequences of high flow pulses and temporal variability of flow events are out of the scope of the present study, and the preference models used here are relevant for low to intermediate discharge rates only. Where *Q50* and *Q90* data were not available (3 635 river segments, which represent 3% of the overall RHT river network database), median and low flows have been calculated from the given inter-annual average discharge (*QM*) based on the coefficients of the linear model fitted to available *QM*, *Q50* and *Q90* data. The regression has been carried out on RHT reaches with *QM* between 0.001 and 950 m^3/s since all missing values were for rivers within this range of discharge (Appendix A, Fig. A.2). In order to calculate the HCP model input variables that were not included in the RHT, namely the parameters of the width-discharge relation and hydraulic geometry dependent variables, the hydraulic geometry and the habitat simulation (Estimhab) modules of Estimkart have been used. In this way, it has been possible to calculate all model's input variables from the hydrological and topographical information provided by the RHT database (specifically discharges, Strahler order, drainage area and river slope).

For each river segment, the HCPs indicated in Table 1 have been calculated separately. Aggregated characterization factors are also provided to enable the applicability of habitat models in Life Cycle Assessment (LCA), which requires aggregation into coarser spatial resolutions in order to align with the resolution of life cycle inventory data quantifying elementary flows such as a water abstraction or discharge. Multi-species aggregated indicators at the reach scale have been calculated: one for species and one including guilds and invertebrates' biomass production. Species and guilds HCPs cannot be combined because some species are already counted in one or more guilds depending on the size class (Appendix A, Table A.2). *HS* functions are not monotonic and can increase or decrease depending on discharge. Therefore, *WUA* derivatives and HCP values can be positive or negative, meaning that discharge alteration may respectively lead to habitat loss or habitat gain. In order to test the results' sensitivity to positive and negative HCPs, multi-species aggregation has been carried out respectively with the individualist, hierarchist and egalitarian cultural perspectives (Thompson et al., 1990). The choice between different aggregated LCIA indicators is based on the consideration of different perceptions of nature as previously addressed in environmental risk assessment (Steg and Sievers, 2000) and life cycle assessment (Goedkoop and Spriensma, 2001; Huijbregts et al., 2016).

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

$$iHCP_{\mathrm{i}} = \sum_{\mathrm{j}=1}^{\mathrm{n}} HCP_{\mathrm{ij}} \tag{8}$$

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

$$hHCP_{\mathrm{i}} = \sum_{\mathrm{j}=1}^{\mathrm{n}} HCP_{\mathrm{ij}} > 0 \tag{9}$$

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

$$eHCP_{\mathrm{i}} = \sum_{\mathrm{j}=1}^{\mathrm{n}} \left|HCP_{\mathrm{ij}}\right| \tag{10}$$

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

$$CF_{\mathrm{w}} = FF_{\mathrm{w}} \cdot EF_{\mathrm{w}} = \frac{dQ_{\mathrm{w}}}{dCWU_{\mathrm{w}}} \cdot \sum_{\mathrm{i}=1}^{\mathrm{n}} HCP_{\mathrm{i}} \cdot \frac{l_{\mathrm{i}}}{\sum_{\mathrm{i}=1}^{\mathrm{n}} l_{\mathrm{i}}} \tag{11}$$

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

4. Results

Characterization results in RHT river segments are highly variable, as represented in Fig. 2 for riffle species and in Fig. 3 for all four guilds. Depending on the observed biological group, river reach HCPs fall within three or four order of magnitude ranges. The detail of riffle species HCP density distribution confirms that habitat sensitivity to water consumption is predictably greater in low flow periods (*Q90*) than in normal conditions (*Q50*). Leptokurtic, heavy tailed and right-skewed distributions are highlighted in the graphs, indicating a non-normal distribution of the data sample, confirmed by the Q-Q plots in Appendix A (Fig. A.3), suggesting a gamma distribution. For this reason, robust statistical measures have been used to analyze the characterization results, namely the Median Absolute Deviation (MAD) for statistical dispersion and the Medcouple measure of skewness to identify outliers (Hubert and Vandervieren, 2008), in order to limit the influence of extreme values. In Fig. 2, compared to median yearly flow conditions, the dry season moves the medians (M) toward higher HCPs ($M_{Q50} = 54.3\ m^2 \cdot s/m^3$; $M_{Q90} = 137.7\ m^2 \cdot s/m^3$), increases the number of extreme values ($Max_{Q50} = 2\,519.3\ m^2 \cdot s/m^3$; $Max_{Q90} = 4\,958.8\ m^2 \cdot s/m^3$) and increments the statistical dispersion ($MAD_{Q50} = 62.7\ m^2 \cdot s/m^3$; $MAD_{Q90} = 139.2\ m^2 \cdot s/m^3$). Because of the large size of the sample and the nature of data distribution, a relevant number of outliers were identified (Fig. 3). The corresponding river segments were however kept unmodified in the resulting HCP database. A check of outlying reaches was performed and no artifact was detected. For this reason, it was assumed the general validity of the modeled extreme habitat conditions, based essentially on their morphological and hydrological characteristics (low order streams, small size and low discharge).

[Figure 2]

4.1.HCP multi-species aggregation at reach scale

HCP frequency distributions of the other taxa included in this study follow a tendency akin to the one discussed above for riffle species. However, results demonstrate that for the same amount of water consumed, some species are more sensitive to habitat change than others. Fig. 3 shows that fish guilds adapted to shallow water habitats are in fact the most vulnerable to habitat loss from water consumption (see Appendix A, figures A.4 and A.5 for all species and invertebrates HCPs), which is most likely accentuated by the adoption of the weighting factor in eq. 3. It is also evident that positive effect factors, meaning habitat loss, are more frequent than negative ones (Fig. 3), this is due to the fact that the *WUA* derivatives are generally positive for the considered flows. For these reasons, the multi-species aggregation of HCPs at the reach scale is driven by a limited number of taxa most likely subject to habitat loss from water consumption. The adopted *individualist*, *hierarchist* and *egalitarian* aggregation approaches therefore do not show substantial differences and, following the parsimony principle, the *individualist* perspective should be used for aggregating LCIA habitat indicators at the reach scale (see example in Appendix A, Fig. A.8, for an application of the three perspectives at watershed scale).

[Figure 3]

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

[Figure 4]

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

Physical habitat indicators at the river reach scale are highly correlated to the Reynolds number, as it is the discharge-dependent input parameter of the habitat model. Spearman‘s $\rho = -0.99$ indicates a non-linear, negative, and monotonic correlation between $iHCP_{gi}$ and Reynolds number. Following the definition of the Reynolds number given in materials and methods, it is straightforward that habitat change potentials depend largely on discharge (*Q90* $\rho = -0.97$) and river size (width $\rho = -0.81$; depth $\rho = -0.91$; velocity $\rho = -0.86$). These variables are in fact at the root of the differences represented in Fig. 4 between the north-eastern and the south-western area of the watershed. It is also interesting to note that altitude is not significant for the habitat change potential definition (ρ = -0.05) while stream order (Strahler, 1957) is a necessary condition, but not sufficient, to determine habitat sensitivity (negative correlation ρ = -0.67 with Strahler order). In other terms, high Strahler stream orders are generally less sensitive to water consumption while low order tributaries show high HCPs unless certain conditions of discharge and size are satisfied.

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

4.2. HCP spatial aggregation at watershed scale

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

In Fig. 5, $iHCP_{gi}$ values are progressively averaged as the spatial resolution decreases. 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 respectively, for 1 m^3 of water consumed in dry season. The picture highlights how sub-watersheds characterized by high habitat change potentials determine the overall watershed CF at each step of the spatial aggregation. A valid alternative to the spatial aggregation formula proposed in the present study implies using weighted medians to limit the influence of extremes in aggregated CFs. However, since HCP distribution is the same in all watersheds, the change in CF values would not be significant for comparative LCA. Moreover, moving the CF closer to the median value of the data sample would imply higher risk of underestimating the habitat change potential of the watershed considering potential model uncertainties. For this reason and following a precautionary principle, the proposed aggregation method should be used (see Appendix A, section 8 for an example of the application of the weighted median spatial aggregation method).

[Figure 5]

5. Discussion

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

5.1. Model uncertainty

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

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

The uncertainty deriving from model parameters should also be considered in the interpretation of characterization results. For instance, since discharge data for ungauged river reaches are hardly available, in the RHT database flow duration curves are modeled for the whole river network, and therefore *Q50* and *Q90* values. The same goes for hydraulic geometry variables modeled through Estimkart. Uncertainty plays a relevant role especially for small catchments, low discharges and mountainous areas. On the contrary, parameters estimates are more accurate in downstream river segments and bigger catchments, where the boundaries are defined more precisely (Lamouroux et al., 2014). Contrary to spatial uncertainty, model parameters uncertainty is therefore potentially higher at the river reach scale than it is for habitat alteration quantification over large areas and thus spatially aggregated CFs taking into account average watershed conditions. In support of these considerations, the uncertainty analysis by Miguel et al. (2016) demonstrated that habitat changes at the regional scale are generally robust despite high uncertainties at the reach scale. For this reason, a reliable CF for LCIA should aim at minimizing spatial variability of inventory data, FF and EF, although ensuring reasonable modeling of average watershed characteristics.

5.2. Operationalization and model extension

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

Data availability could be the major constraint for refining and extending the model outside of France. At present, comprehensive databases including all necessary input data (i.e. discharge, substrate composition, hydraulic geometry) are not available globally, although useful data sources are currently being developed (e.g. Lehner, 2018). The model can be therefore applied using local data. For an application on the larger scale the input variables should be modeled if not available, such as in the French RHT hydrographic network, or the model should be simplified retaining the factors that most determine characterization results (see section 4.1). For improving taxa coverage, more habitat equations should be included when possible (other freshwater fish species could be assigned to the four guilds based on habitat preference). In addition, using Q50 and Q90 at monthly resolution would improve the precision and temporal relevance of the CF, since river flow regimes can be different in relation to topographical and climatic conditions.

6. Conclusions

In this study a new, mechanistic characterization factor based on freshwater physical habitat modeling has been presented. The effect factor model has been applied at the river and watershed scale to assess marginal impact of water consumption on instream ecosystems. HCP has been calculated at *Q50* and *Q90* as representative of the median discharge condition and dry season respectively.

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

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

Associated content

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

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Tables

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

Figures

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

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

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 ($iHCP_{gi}$) values aggregated at reach scale for fish guilds and invertebrates in $m^2 \cdot s/m^3$. Application of the characterization model to the Durance-Verdon river basin in France at *Q90*

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*

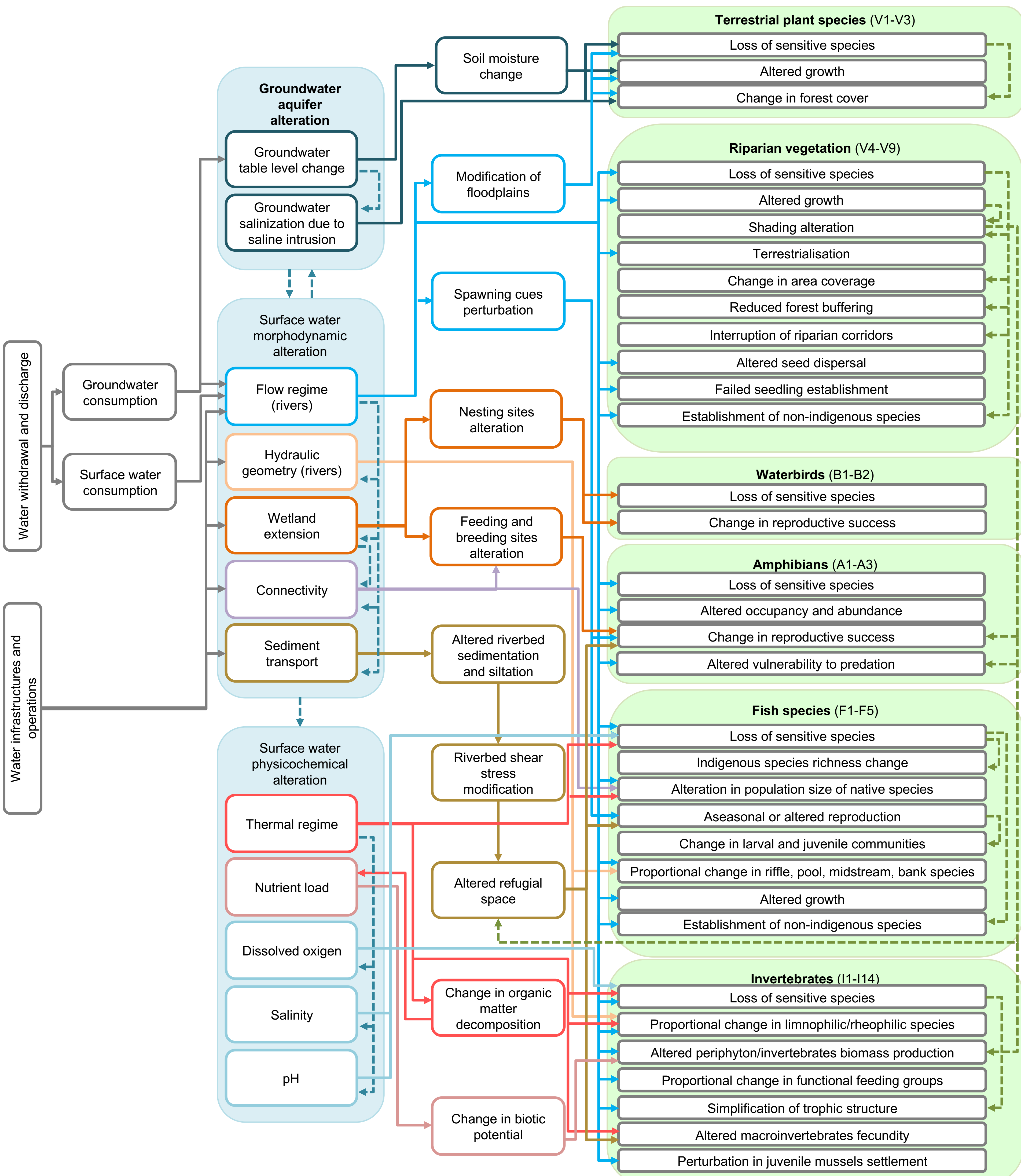


Fig. 1. Simplified impact pathways of water consumption and ecosystem responses for freshwater-dependent biological communities. Codes in brackets indicate additional references used to construct the flow diagram (Appendix A, Table A.1)

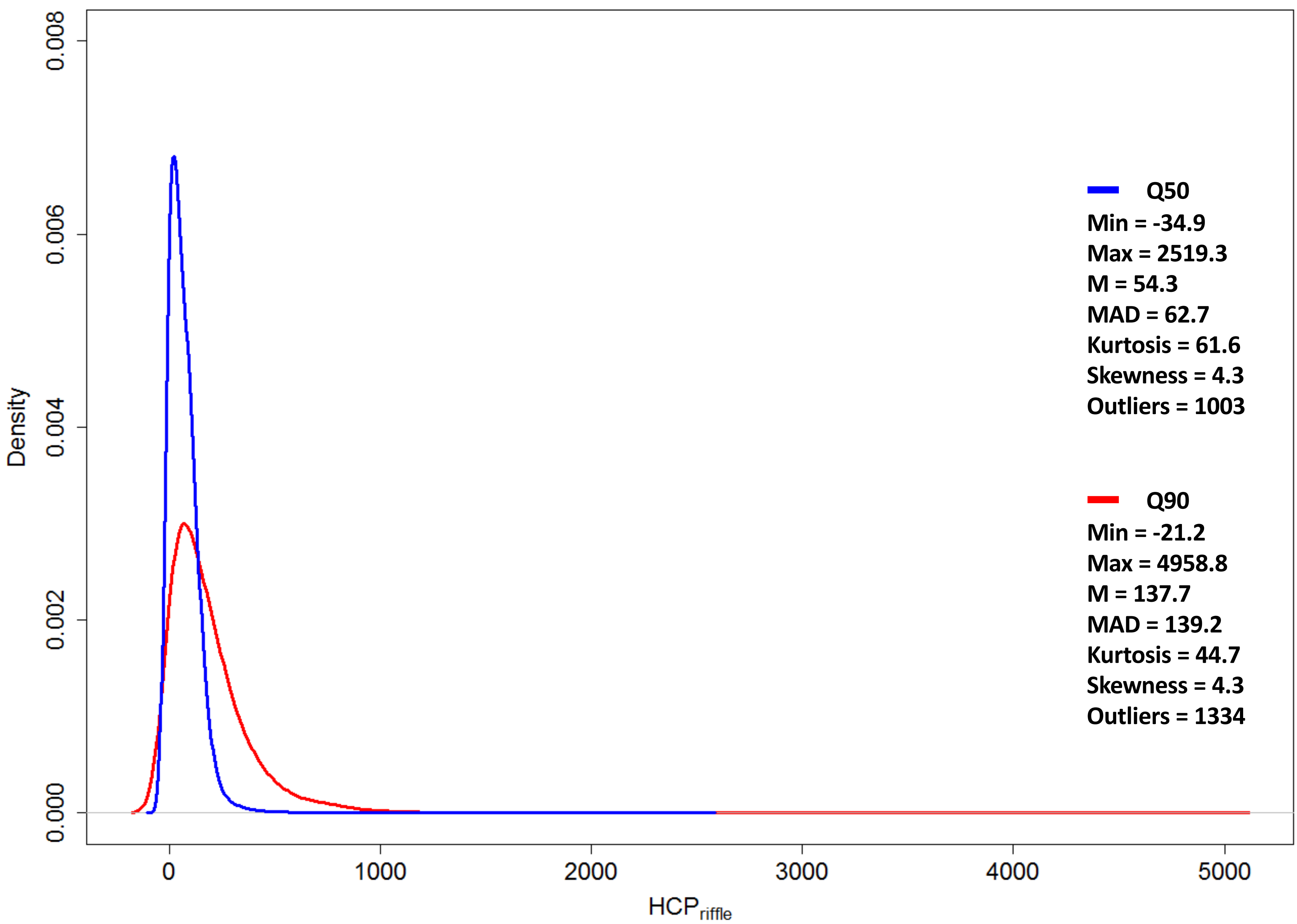


Fig. 2. Density distribution of the HCP for riffle fish species in French river reaches ($m^2 \cdot s / m^3$) with Min: minimum value; Max: maximum value; M: median; MAD: Median Absolute Deviation

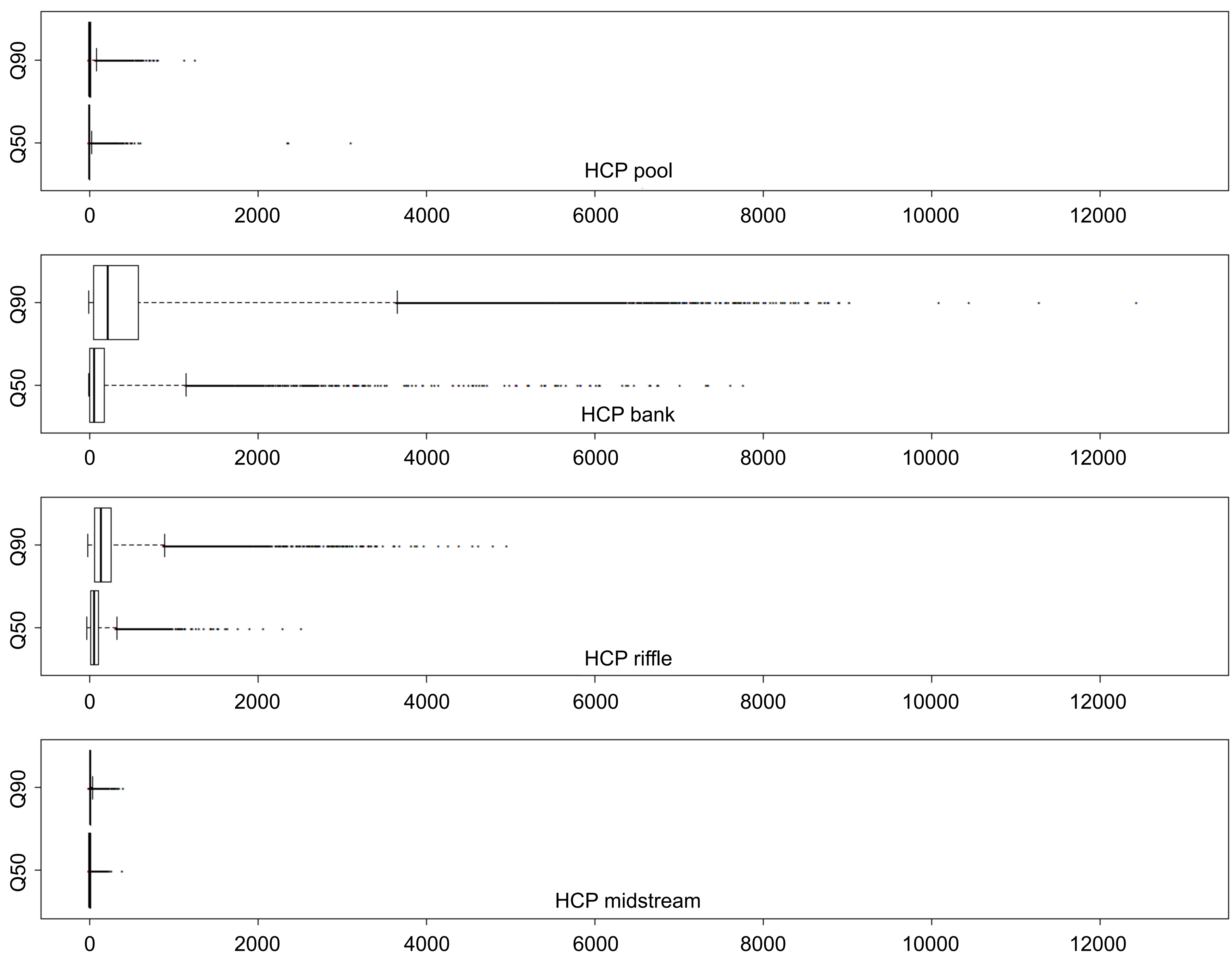


Fig. 3. Spread of habitat change potentials HCP for pool, bank, riffle and midstream fish guilds ($m^2 \cdot s/m^3$)

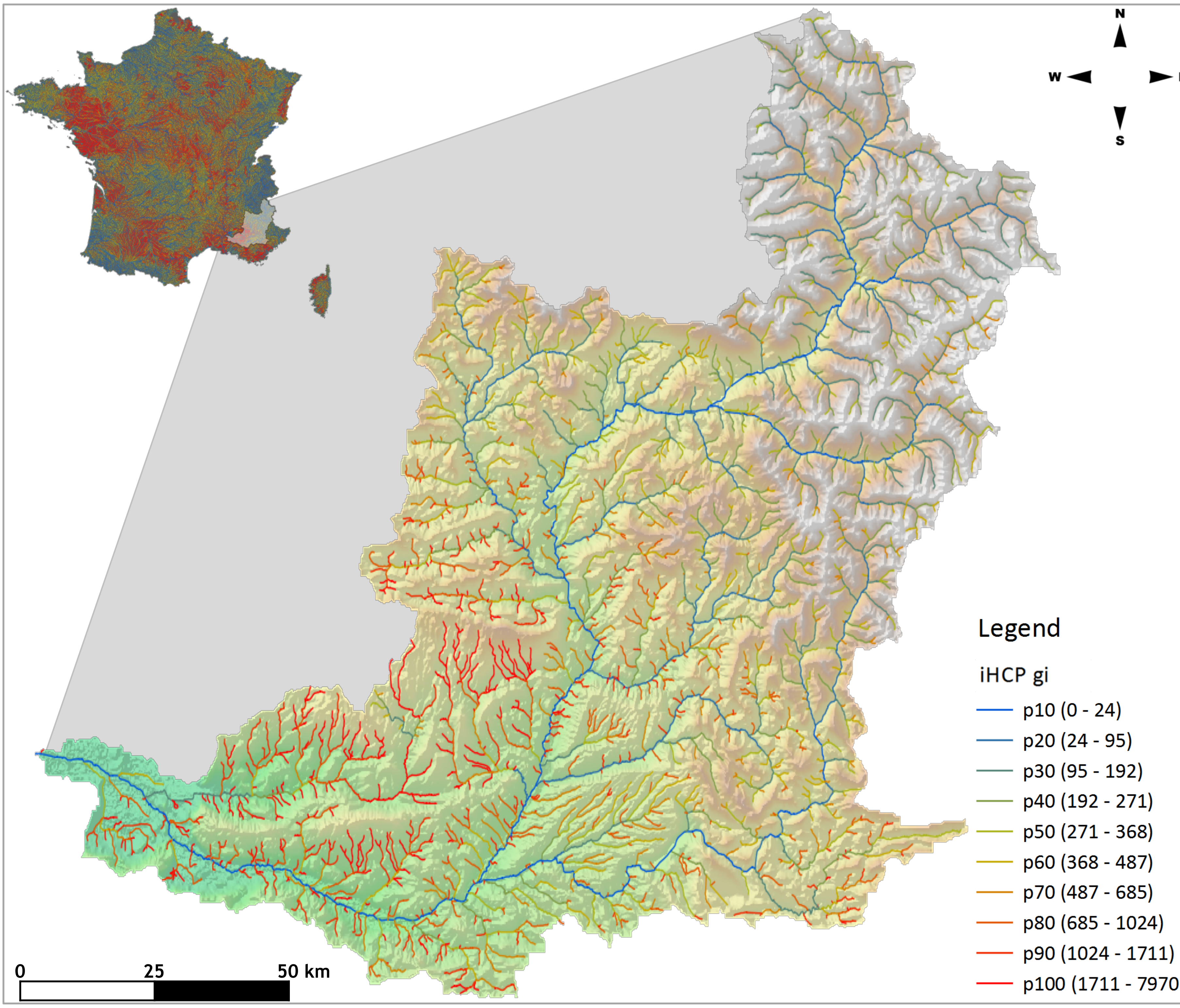


Fig. 4. Percentiles (p10 - p100) of individualist HCP ($iHCP_{gi}$) values aggregated at reach scale for fish guilds and invertebrates in $m^2 \cdot s/m^3$. Application of the characterization model to the Durance-Verdon river basin in France at *Q90*

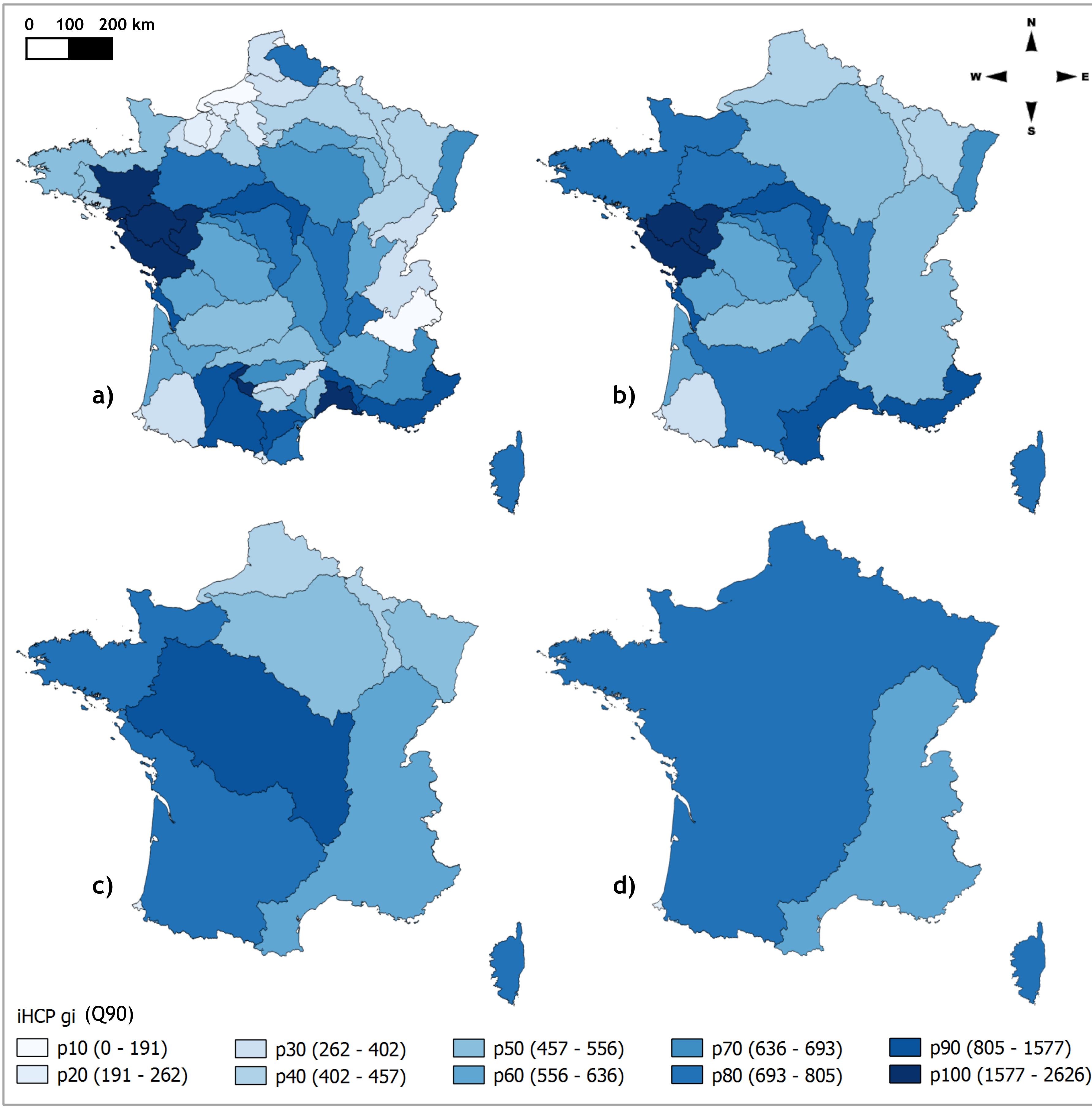
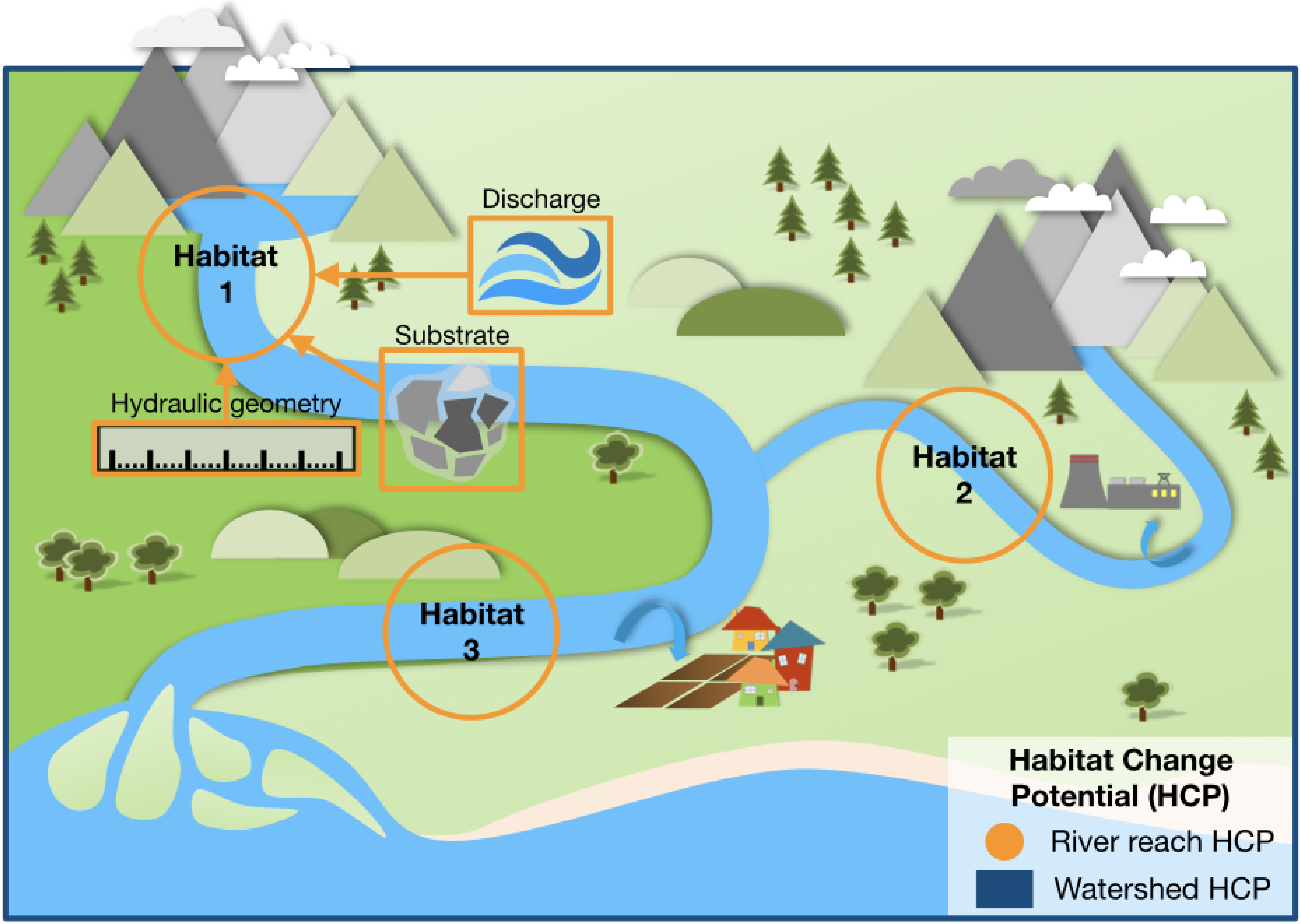


Fig. 5. Percentiles (p10 - p100) of aggregated iHCP$_{gi}$ at HydroBasins levels 6 (a), 5 (b), 4 (c) and 3 (d), values in $m^2 \cdot s/m^3$ at *Q90*

Table 1

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

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


Discharge
Habitat 1
Substrate
Hydraulic geometry
Habitat 2
Habitat 3
Habitat Change Potential (HCP)
River reach HCP
Watershed HCP