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Dorber, M.; Scherer, L.A.; Verones, F.

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# Midpoint characterization factors to assess impacts of turbine water use from hydropower production

Martin Dorber<sup>1</sup> · Laura Scherer<sup>2</sup> · Francesca Veronesi<sup>1</sup>

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## Abstract

**Purpose** Life cycle assessment (LCA) distinguishes three types of water use: (1) consumptive water use, (2) degradative water use, and (3) in-stream water use. When it comes to assessing the impact of turbine water use (TWU, major source of in-stream water use) in LCA, so far, no method exists to quantify the related environmental impacts. Here, we developed the first midpoint characterization factors (CFs) with global coverage for turbine water use of storage and pumped storage hydropower power plants.

**Methods** The midpoint CF at the basin scale describes the hydropower regulation potential (HRP) [HDOR·y] per TWU [m<sup>3</sup>]. The HRP indicates the probability of how strongly the natural flow regime of a river is potentially affected by all upstream reservoir operation, calculated as the quotient between reservoir volume [m<sup>3</sup>] and the annual river discharge [m<sup>3</sup>/y]. The hydropower degree of regulation (HDOR) thereby equals the unitless m<sup>3</sup>/m<sup>3</sup> fraction. The TWU depends on the electricity production [kWh] and the turbine efficiency [m<sup>3</sup>/kWh]. We tested the sensitivity of the input data on the calculated CFs for four parameters (discharge, turbine efficiency, multipurpose allocation, and plant type). Furthermore, we performed a case study to analyze if consumptive and TWU impacts of producing 1 kWh are correlated or not.

**Results and discussion** The calculated CFs for the 342 basins vary from 1.13E-13 HDOR·y/m<sup>3</sup> to 3.28E10-7 HDOR·y/m<sup>3</sup>. The HDOR values range from 0.0015 to 16.66, and the TWU varies between 0.0030 km<sup>3</sup> and 2824 km<sup>3</sup>. A HDOR ≥ 0.02 can be interpreted as affected basin, and only 23 out of 342 basins have a HDOR below this threshold. This confirms that TWU of hydropower production can have important environmental impacts. The sensitivity analyses revealed that discharge and turbine efficiency are the most sensitive parameters because they are influencing almost all basins. The results of the case study showed that a high consumptive water-use impact does not automatically lead to a high TWU impact and vice versa ( $R^2$  values of 0.0081 and 0.003).

**Conclusion** Our study highlights that it is important to account for the environmental impacts of in-stream water use in LCA, as otherwise, the environmental impact can be underestimated, which could lead to wrong conclusions. However, the CFs are not meant to replace a local risk assessment of hydropower reservoir operation and should only be used for relative comparison between basins. The CF application in LCA will represent a step forward towards more sustainable hydropower development.

**Keywords** Life cycle impact assessment · Flow regulation · Biodiversity · In-stream water use · Reservoir

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✉ Martin Dorber  
martin.dorber@ntnu.no

<sup>1</sup> Department of Energy and Process Engineering,  
Norwegian University of Science and Technology (NTNU),  
Høgskoleringen 5, 7491 Trondheim, Norway

<sup>2</sup> Institute of Environmental Sciences (CML), Leiden  
University, Leiden, the Netherlands

## 1 Introduction

We are currently facing a triple crisis: climate change, pollution, and biodiversity loss (UNFCCC 2022). To mitigate climate change, a drastic increase in renewable energy production is required (Bogdanov et al. 2019). This is exemplified by the mitigation pathways limiting global warming to 1.5 °C (with no or limited overshoot) of the IPCC, where up to 85% of the total electricity demand has to be produced from renewable energy sources (Intergovernmental

Panel on Climate Change 2018). Especially, reservoir-based hydropower plays an important role in future renewable energy supply because it can provide affordable, flexible, and reliable electricity with a comparably low carbon footprint (UNEP 2016). Further hydropower development can thus contribute to fulfilling the United Nations' sustainable development goal (SDG) 7 (Affordable and clean energy) and SDG 13 (Climate action) (United Nations 2015).

Notwithstanding the climate and other benefits of hydropower development (Almeida et al. 2019; Fuso Nerini et al. 2018; Intergovernmental Panel on Climate Change 2011; Muller 2019; Pehl et al. 2017; UNEP 2016), reservoir creation and operation cause ecological impacts (Turgeon et al. 2019; Wu et al. 2019) that may interfere with SDG 6 (Clean water and sanitation) and SDG 15 (Life on land) (Hermoso et al. 2019) and also collides with the second element of the triple planetary crisis.

To avoid a trade-off between mitigating the climate crisis on the one hand and increasing the biodiversity crisis on the other hand, a quantitative assessment of potential biodiversity impacts from new hydropower projects is needed (Liu et al. 2018). It has been highlighted that life cycle assessment (LCA) (ISO 2006; Koehler 2008) can be used to give decision support related to the triple planetary crisis (Hellweg et al. 2023). When it comes to life cycle impact assessment (LCIA) from hydropower, Gracey and Verones (2016) identified (1) freshwater habitat alteration, (2) water quality degradation, and (3) land use change as the main impact pathways that should be covered.

Here we focus on freshwater habitat alteration and, more specifically, on hydrologic alteration of the natural flow regime, which has been identified as a key variable for river ecosystems (Poff et al. 1997) and is considered the second main environmental impact pathway of dams and reservoirs (Grill et al. 2014). Several studies (e.g., Gillespie et al. 2015; Gracey & Verones 2016; Liew et al. 2016; Poff & Zimmerman 2010; Scherer & Pfister 2016; Taylor et al. 2014; Turgeon et al. 2019; Wu et al. 2019) have highlighted the adverse freshwater biodiversity impacts of flow regime changes, which among others may lead to local species extinctions of fish, macroinvertebrates, riparian vegetation, or even terrestrial flora and fauna. In addition, Grill et al. (2015) estimated that, currently, 48% of the global river volume is impacted by dams.

From a hydrological point of view, the natural flow regime can be split into five components: (1) magnitude, (2) frequency, (3) duration, (4) timing, and (5) rate of change of the natural flow regime.

From an LCA point of view, these components are affected by two different types of water use from hydropower (ISO 2014): (1) consumptive water use and (2) in-stream water use (in-situ use of freshwater) (Bayart et al. 2010; Berger & Finkbeiner 2010; Pfister et al. 2009, 2015).

Consumptive water use affects the magnitude, while in-stream water use can affect all flow components.

Consumptive water use is caused by evaporation from the reservoirs, when their construction increases the water surface area compared to the natural state (Dorber et al. 2019). This water represents a water loss in the related basin, which can inter alia lead to a loss of annual discharge (Pierrat, et al. 2023b) and consequently to a change in flow magnitude. To assess this, several characterization factors (CF) exist: On the midpoint level, for example, models assessing the global impacts of consumptive water use based on available water remaining (Boulay et al. 2018) or the potential for habitat change in Europe exist (Damiani et al. 2021). On the damage-level, within the area of protection “ecosystem quality,” for example, models quantifying the potential loss of freshwater fish species resulting from reduced river discharge (Pierrat et al. 2023a), the potential loss of terrestrial plant species resulting from changes in the groundwater table in the Netherlands (Zelm et al. 2011), the potential loss of species in wetlands due to water consumption (Verones et al. 2016), and impacts on vascular plants due to surface water consumption (Pfister et al. 2009) could be applied.

In-stream water use of hydropower comes from water run through a turbine to produce electricity. In for example, Ecoinvent this water is called “Water, turbine use, unspecified natural origin” (Ecoinvent 2020). For simplicity, we will refer to this water as “turbine water use (TWU)” from now on. Hydropower turbines do not consume the water that drives the turbines (in contrast to evaporation from the reservoir (Dorber et al. 2019)) and do not change the water quality (Kumar et al. 2011). In other words, the sum of water to turbine and water from turbine is 0, and hence there is no consumptive water use occurring.

The impact itself is caused by the flexible release of water from reservoirs into storage and pumped storage hydropower plants (Egré & Milewski 2002). The related reservoirs are used to store water in times of surplus (e.g., snowmelt, heavy rainfall) and to produce electricity with a release of water, with short reaction times, during peak energy demand (e.g., heating in winter) (Ashraf et al. 2018). As this operation regime is usually not in accordance with the natural flow regime, it commonly produces a stabilizing effect on a river's annual discharge by removing flow peaks. For example, in rivers characterized by snowmelt, TWU for flexible electricity production can lead to a reduction in spring floods due to reservoir filling and increased winter flows during peak demand (Renöfält et al. 2010). Hence, TWU affects all five components of the natural flow regime, depending on the operation scheme of the reservoirs.

When it comes to assessing the impact of TWU in LCA, currently, only the pure amount of turbine water ( $\text{m}^3/\text{kWh}$ ) (Flury & Frischknecht 2012) or an environmental water stress index (Scherer & Pfister 2016) is considered.

The water stress index assumes that, globally, 20% of flow alterations regarding the magnitude of river discharge as the boundary between low and moderate stress and does not allow for an extension of the impact pathway to the damage-level. However, so far, no method exists to quantify the environmental impacts of TWU in LCA, as already pointed out in 2016 (Gracey & Verones 2016).

In this study, we aim to diminish this research gap by developing new midpoint CFs at the global scale to quantify the flow alteration impacts caused by TWU of storage and pumped-storage hydropower plants.

## 2 Methods

### 2.1 Midpoint scope

Flexible in-stream water use for hydropower production can alter five components of the natural flow regime: (1) magnitude, (2) frequency, (3) duration, (4) timing, and (5) rate of change of the natural flow regime (Gracey & Verones 2016). For each of the five components, various ecologically relevant parameters have been developed, adding up to a total of 171 parameters (Olden & Poff 2003). Consequently, one of the biggest challenges for ecological research is to identify the most relevant indicators out of this large number of competing hydrologic indicators to reduce data requirements and index redundancy prior to analyses (Gao et al. 2009; Olden & Poff 2003). A commonly used set of indicators to assess natural flow regime alterations (Zhou et al. 2020) are those of hydrologic alteration (IHAs) (Richter et al. 1996). The IHAs are a set of 32 indicators to compare the flow regime characteristics between pre- and post-dam conditions. The outcome is the % change in indicator between post- and pre-dam conditions. Even if the IHAs are not directly used, the classification approach of attributing a change of a certain indicator to a descriptive impact class is widely used (e.g., Bakken et al. 2021; Grill et al. 2015; Phelan et al. 2017). This is done because there is a *mechanistic* or *process-based* relationship between an ecological response and a particular flow regime component (Poff et al. 2010). The higher the score, the greater the impact on the flow regime and the higher the risk of damage to the ecosystem. This information is helpful for developing midpoint CFs.

However, the IHAs do not provide information on the magnitude of freshwater biodiversity impacts for a certain magnitude of hydrologic alteration (Mathews & Richter 2007). This is because there is generally little support for either simple linear relationships or threshold relationships between ecological responses and flow alteration (Harper et al. 2022; Poff & Zimmerman 2010). The main reasons for this are data gaps (Mathews & Richter 2007; Poff & Zimmerman 2010), for example, resulting from ecological study

design, variation in the reporting of river flow alteration, inability to examine species-specific responses, and lack of river discharge data.

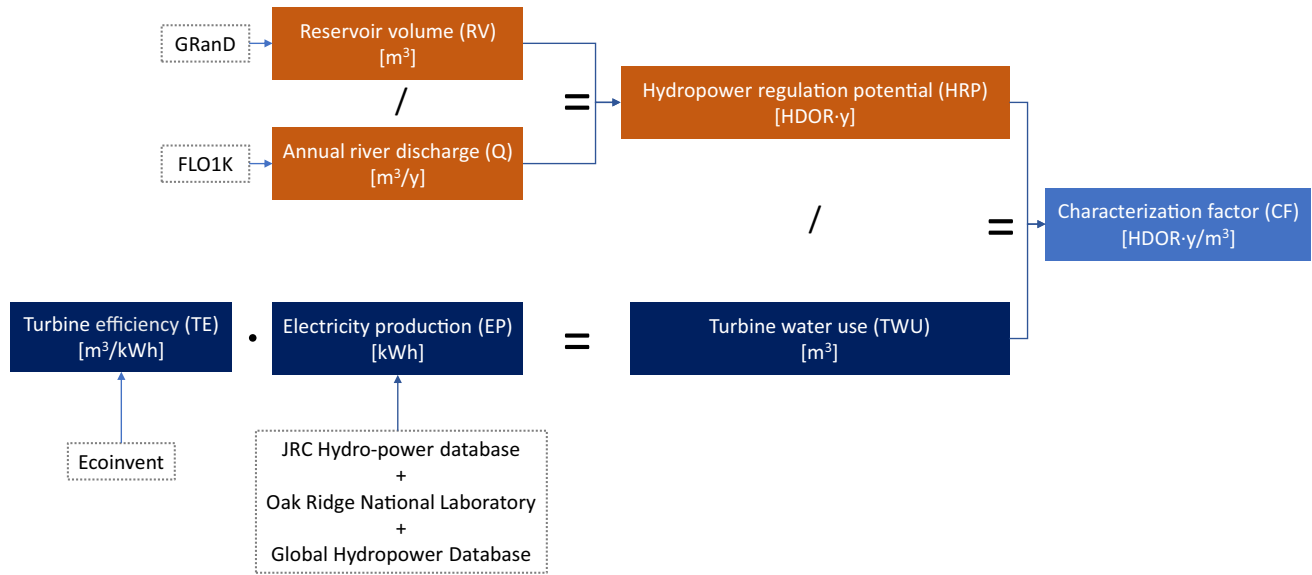
However, for developing a damage-level indicator for TWU in LCIA, a relationship between biodiversity impact and specific flow alteration indicators is required, similar to the species-discharge relationship used for biodiversity impacts of consumptive water-use (Pierrat et al. 2023a), because the indicator must be able to quantify freshwater species losses inside different taxonomic groups (for example, fish and macroinvertebrates) as the Potentially Disappeared Fraction of Species (PDF) (Verones et al. 2017). Hence, given the current data situation, it is, to our knowledge, currently not possible to develop a damage-level indicator for TWU, and we limit our scope therefore to midpoint CFs.

### 2.2 Framework for midpoint characterization factors

Our midpoint CF at the basin scale describes the hydropower regulation potential (HRP) [HDOR·y] per m<sup>3</sup> turbine water use (TWU) (Fig. 1). The CFs quantify the impact per m<sup>3</sup> turbine water use because this is the inventory unit used in Ecoinvent, i.e., “Water, turbine use, unspecified natural origin” (Pfister et al. 2015). The HRP depends on the reservoir volume [m<sup>3</sup>] and the annual river discharge [m<sup>3</sup>/y] (Eq. 1). The hydropower degree of regulation (HDOR) thereby equals the unitless m<sup>3</sup>/m<sup>3</sup> fraction (see Sect. 2.3). The underlying assumption for the HRP is that a large reservoir volume in a basin with low annual river flow rates will have a higher probability of a regulatory effect on the natural flow regime than a small reservoir volume in a basin with high annual river flow rates (Grill et al. 2015, 2019). Hence, the larger the CF value, the larger the probability of a regulatory effect of TWU on the natural flow regime. The TWU [m<sup>3</sup>] depends on the electricity production [kWh] and the turbine efficiency [m<sup>3</sup>/kWh].

### 2.3 Hydropower regulation potential (HRP)

The best available indicator to start development from is the Degree of Regulation (DOR) (Grill et al. 2015, 2019, 2014; Lehner et al. 2011; Nilsson et al. 2005), which also has been identified by Gracey & Verones (2016) as a potential input variable for LCIA models. The concept of the unitless DOR index is based on the relationship between the storage volume of a reservoir and the annual river discharge at the dam's location, expressed as the quotient between reservoir volume [m<sup>3</sup>] and annual river discharge [m<sup>3</sup>]. It is important to note that the DOR does not specify if the regulatory effect leads to an increase or decrease in the flow regime in comparison to the natural flow regime. Furthermore, it assumes



**Fig. 1** Overview of midpoint characterization factor (CF) framework. Colored box, model variable; dotted box, data source for model variable. GRanD, Global Reservoir and Dam Database (GRanD) v1.3 from Lehner et al. (2011); FLO1K from Barbarossa et al. (2018); Oak

Ridge National Laboratory from Johnson et al. (2020), Global Hydropower Database (GHD) from Wan et al. (2021), and JRC Hydropower database from the Joint Research Centre (JRC) (2019)

that the entire reservoir volume is currently usable, independent of the water inflow to the reservoir. Consequently, the DOR provides a quantitative proxy of how strongly the natural flow regime of a river may be affected by upstream reservoir operations but is not an absolute indicator. The higher the DOR, the higher the probability of a regulatory effect on the natural flow regime.

The DOR itself is calculated for each river section in a basin. However, for water use in LCA, CFs are commonly calculated at the basin scale (e.g., Boulay et al. 2018; Damiani et al. 2021; Pierrat et al. 2023a) since this is the native spatial resolution of the impact (Mutel et al. 2019). Furthermore, for CFs, it is important to consider the time aspect of the flow (e.g., Boulay et al. 2018; Damiani et al. 2021; Pierrat et al. 2023a).

As a result, we directly calculated the hydropower regulation potential (HRP) for the entire basin with Eq. 1:

$$HRP_i = \frac{\sum_{x=1}^n RV_{x,i}}{Q_i} = \left[ \frac{m^3}{m^3/y} \right] = \left[ \frac{m^3 \cdot y}{m^3} \right] = [HDOR \cdot y] \quad (1)$$

where  $n$  is the number of reservoirs in basin  $i$ ,  $RV$  is the storage capacity of hydropower reservoir  $x$  in  $m^3$ , and  $Q$  is the annual river discharge of basin  $i$  in  $m^3/y$ . In line with the unitless DOR, we call  $m^3/m^3$  the hydropower degree of regulation (HDOR). While the DOR uses  $RV$  of all reservoirs, the HDOR only accounts for  $RV$  of hydropower reservoirs.

The basin delineation  $i$  was obtained from Lin et al. (2021), who provide a vector-based global hydrography dataset using the latest digital elevation model.

The number of reservoirs and the related storage capacity ( $RV$  as representative maximum storage capacity) was obtained from the Global Reservoir and Dam Database (GRanD) v1.3 (Lehner et al. 2011). We included all reservoirs that are used for hydropower electricity production (Main major, and secondary user) in our study (2461, see Fig. S1, S11).

The annual river discharge ( $Q$ ) was derived from Barbarossa et al. (2018). The FLO1K dataset provides interpolated mean, maximum, and minimum annual flow at a 1-km resolution for each year in the period 1960–2015. The foundation for this data set is measured discharge from 6600 monitoring stations worldwide, therefore representing measured rather than modeled discharge. This is important because measured discharge already discounts for water consumption in the basin. For  $Q_i$ , we calculated the mean annual discharge [ $m^3/s$ ] over the years 2005–2015. The annual flow rate  $Q_i$  was then obtained by selecting the max  $Q$  inside each basin  $i$  and multiplying it by the number of seconds in a year (i.e., 31,536,000).

## 2.4 Turbine water use (TWU)

The turbine water use (TWU) [ $m^3$ ] per basin  $i$  was calculated with Eq. 2:

$$TWU_i = \sum_{y=1}^m TE_{y(c,t)} \cdot EP_y \quad (2)$$

where  $m$  is the number of power plants in basin  $i$ ,  $TE$  is the turbine efficiency of hydropower power plant  $y$  [ $m^3/kWh$ ]



depending on its country  $c$  and its type  $t$ . EP is the electricity production [kWh] of power plant  $y$ , during the same time horizon as  $Q$ . In our case, EP equals the average annual electricity production. Country delineation was obtained from the Word Bank (2021).

For type  $t$ , we only considered storage and pumped storage hydropower plants and excluded run-of-river plants since their electricity production does not depend on hydropower reservoirs (Egré & Milewski 2002).

To calculate TWU, information on the location, the type of hydropower plant, and the average annual electricity production (EP) are needed. However, there is a lack of global comprehensive hydropower plant data sets, which contain all this basic information at the same time (Wan et al. 2021). We combine, therefore, three hydropower plant datasets (i.e., Oak Ridge National Laboratory (Johnson et al. 2020), Global Hydropower Database (GHD) (Wan et al. 2021), and JRC Hydro-power database (Joint Research Centre (JRC), 2019)). From all databases, we excluded run-of-the-river power plants and deleted records with missing annual electricity production information. In case of missing plant type information, we assumed that all plants with a capacity above 50 MW are storage hydropower reservoirs (S1, S2). This was done due to the comparably low number of pumped-storage plants (International Hydropower Association 2022) and because it is reported that small-scale power plants with an installed capacity between 1 and 50 MW (Singh & Singal 2017; Yildiz & Vrugt 2019) are in most cases “run-of-the-river” power plants (Sachdev et al. 2015).

In total, we obtained 3340 hydropower storage and pumped storage power plants that fit our requirements (Figure S2). In total, these power plants produce 3180 TWh per year, which equals 72% of the global hydropower electricity production in the year 2020 (IEA 2021).

We obtained the turbine efficiency (TE) (“Water, turbine use, unspecified natural origin” [ $\text{m}^3/\text{kWh}$ ]) for each country  $c$ , for storage (“electricity production, hydro, reservoir”) and pumped (“electricity production, hydro, pumped storage”) from Ecoinvent 3.7.1 (Ecoinvent 2020). If several values were reported for one country, we averaged the values to obtain one country value. If no country-specific value for country  $c$  was available, we used the “Global (GLO)” value from Ecoinvent 3.7.1. The used turbine efficiency values are provided in SI2.

## 2.5 Sensitivity analysis

We tested the sensitivity of the input data and our assumptions on the calculated CFs for four different variables  $X$ : (1) discharge, (2) multipurpose use of the reservoirs, (3) power plants with “no type” information, and (4) turbine efficiency. Case Xa always results in a lower CF, while Case Xb results in higher CFs, compared to the original input data.

Case 1a: We used the maximum flow value provided by the FLO1K dataset (Barbarossa et al. 2018) between 1960–2015 to calculate the HRP.

Case 1b: We used the minimum flow value provided by the FLO1K dataset (Barbarossa et al. 2018) between 1960–2015 to calculate the HRP.

Case 2a: If electricity production is the only user, all impacts are allocated to it. However, in multipurpose reservoirs, the flow regulation impact should be allocated between the different users (Bakken et al. 2016), as part of the reservoir volume could, for example, also be used for irrigation. Here, we used the allocation method from Scherer & Pfister (2016) to allocate RV, which is based on the ranking of hydropower production among all users of the reservoir. The GRanD v1.3 database (Lehner et al. 2011) specifies the main user and secondary users, without distinguishing a ranking between multiple secondary purposes. When hydropower was listed as one of several secondary purposes, we followed Scherer and Pfister (2016) and assumed that electricity production ranks second. For example, in cases where hydropower is the main user among three total users, 50.0% ( $3 / (3 + 2 + 1)$ ) of the impact is allocated to it. In contrast, when hydropower is the secondary user among three total users, it receives 33.3% ( $2 / (3 + 2 + 1)$ ) of the impact.

There is no case 2b because we already allocated the entire regulation impact to hydropower in the CF.

Case 3a: We included all power plants, independent of the capacity, with “no type” as “storage plants” to calculate TWU.

Case 3b: We excluded all power plants, independent of the capacity, with “no type” information to calculate TWU.

Case 4a: We used the largest TE value provided by Ecoinvent for storage and pumped storage plants to calculate TWU.

Case 4b: We used the smallest TE value provided by Ecoinvent for storage and pumped storage plants to calculate TWU.

## 2.6 Case study at midpoint level

As currently, only consumptive water-use impacts are assessed at the midpoint level in LCA, we performed a case study to analyze if a high consumptive water-use would automatically lead to a high in-stream water-use or if consumptive water-use and in-stream water-use occur independently from each other.

The system boundary is limited to electricity production, and the functional unit is 1 kWh of storage hydropower produced in a certain basin. For the inventory, we included “Emissions to air, water,” representing evaporated water (consumptive water use), and “Water, turbine use,

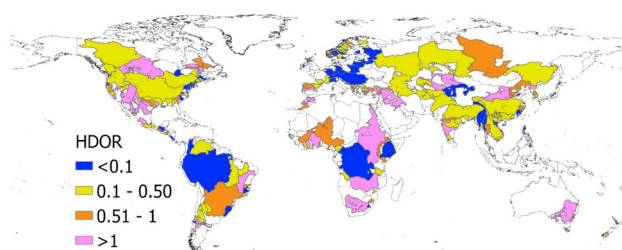
unspecified natural origin.” To quantify the impact of consumptive water-use, we used the midpoint CFs from Damiani et al. (2021). Their midpoint CFs model the monthly river habitat change potential (HCP) [ $\text{m}^2\cdot\text{s}/\text{m}^3$ ] induced by water consumption in Europe. For the TWU, we used the CF developed in this study. However, as Damiani et al. (2021) used a different basin delineation, we could only perform the case study for nine basins in Europe, as here the basin delineation between the two different CFs matched (Fig. S4). Our CFs are developed on a basin scale and with a yearly time dimension, while the TWU inventory is at the country level. The midpoint CFs of Damiani et al. (2021) are calculated on monthly scales. We, therefore, compared the impact for a minimum (Scenario 1) and maximum scenario (Scenario 2). To do so, we first identified with which country each basin overlapped.

For Scenario 1, we used the lowest available “Water, turbine use, unspecified natural origin” and “Emissions to air, water” country value and the lowest monthly CF value from Damiani et al. (2021). For Scenario 2, we used the highest available “Water, turbine use, unspecified natural origin” and “Emissions to air, water” country value and the highest monthly CF value from Damiani et al. (2021).

### 3 Results

#### 3.1 Hydropower regulation potential (HRP)

In total, we calculated a HRP value for 342 basins, where at least one hydropower reservoir and one power plant data point were present (SI2). The HDOR values (Fig. 2) range from 0.0015 to 16.66, with an average value of 0.77 and a median value of 0.27. 279 basins have HDOR values below 1, of which 89 basins have a HDOR smaller than 0.1. Ten basins have HDOR values larger than 5 (SI2). The underlying annual river discharge ( $Q$ ) ranges from 0.16 to 1758  $\text{km}^3/\text{y}$ , and the reservoir volume (RV) from 0.009 to 470.5  $\text{km}^3$  (SI2).



**Fig. 2** Hydropower degree of regulation (HDOR) per basin. Basin delineation from Lin et al. (2021) and country borders from the Word Bank (2021)

Overall, the HDOR is not catchment size-specific ( $R^2=0.005$ ), but in a basin with a relatively small  $Q$ , it is comparably easy to store a big fraction of  $Q$ , leading to a high HDOR value. Therefore, the 10 highest HDOR values can be found in basins with  $Q \leq 28.2 \text{ km}^3/\text{y}$ . In contrast, the 3 basins with the highest annual flow rate ( $Q > 946 \text{ km}^3/\text{y}$ ) have a HDOR value below 0.15. However, a low  $Q$  does not automatically lead to a high HDOR ( $R^2=0.007$ ). The lowest HDOR of 0.00151 is, for example, found in a basin with  $Q=305 \text{ m}^3/\text{s}$ . In parallel, the basin with the highest RV (470.5  $\text{km}^3$ ) has a HDOR of 0.85. This can be explained by the weak correlation between  $Q$  and RV ( $R^2=0.195$ ). For example, basins “5,001,649; Indonesia” and “7,002,898; Canada” with 34.12  $\text{km}^3/\text{y}$  and 34.59  $\text{km}^3/\text{y}$  have almost the same  $Q$ , but due to different RVs (6.6  $\text{km}^3$  and 0.07  $\text{km}^3$ ), the HDOR values of 0.19 and 0.0022, respectively, differ by two orders of magnitude. Furthermore, for example, basins “2,002,462; East Europe” and “7003613; USA/Mexico” with 18.07  $\text{km}^3$  and 18.12  $\text{km}^3$  have a similar RV, but due to the different  $Q$  values, the HDOR varies between 0.07 and 9.11.

#### 3.2 Turbine water use (TWU)

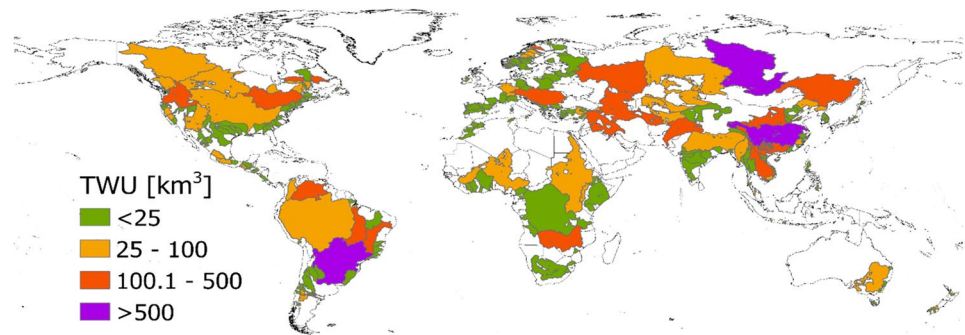
The TWU for the 342 basins varies between 0.0030  $\text{km}^3$  and 2824  $\text{km}^3$  per basin with an average value of 37.4  $\text{km}^3$  and a median value of 3  $\text{km}^3$  (Fig. 3).

The variation of the TWU can mainly be explained by the Electricity production (EP), ranging from 0.92 to 671,255 GWh. The general trend is that the higher EP, the higher TWU ( $R^2=0.97$ ). However, as the turbine efficiency (TE) varies between 0.56 and 11.58  $\text{m}^3/\text{kWh}$ , the same amount of EP can lead to different TWU. For example, basins “2,003,433; Spain” and “5,008,211; Australia” have an EP of 542 and 544 GWh, respectively. Since one is located in Spain (TE pumped storage and storage=8.1) and one in Australia (TE pumped storage=0.81 and storage=4.455), the basin in Australia has, with 0.44  $\text{km}^3$ , a much lower TWU than the basin in Spain with 4.39  $\text{km}^3$  despite the similar EP value.

#### 3.3 Characterization factors (CF)

The midpoint CFs, expressing the fraction of annual flow rate that reservoirs can store for future release in a certain basin per  $\text{m}^3$  TWU, range from 1.13E-13 to 3.28E10-7 HDOR·y/ $\text{m}^3$ . The CFs can be interpreted as an effectiveness measure, describing how much HRP is needed per unit of TWU. Since there is no correlation between HRP and TWU ( $R^2=0.0002$ ), the CF depends both on the HRP and the TWU (Figure S3), which themselves depend on local conditions (i.e., RV,  $Q$ , EP, TE). Consequently, a low HRP

**Fig. 3** Turbine water use (TWU) [ $\text{km}^3$ ] per basin. Basin delineation from Lin et al. (2021) and country borders from The Word Bank (2021)



does not automatically result in a low CF, and vice versa (Fig. 4).

For example, basin “7,003,654; USA” has, with 0.32, a comparably low HRP, but at the same time, it has, with  $0.003 \text{ km}^3$ , the lowest TWU of all basins, resulting in the 5th highest CF value ( $1.09\text{E-}07$ ). Basin “2,009,676; Irak/Tyrkia/Syria” has the 4th highest HDOR (8.77), but due to its high TWU of  $258 \text{ km}^3$  (rank 11), it gets a smaller CF of  $3.40\text{E-}11 \text{ HDOR}\cdot\text{y}/\text{m}^3$ . This shows that a comparably high TWU will lead to a comparably small CF value, independently of the HRP, which can be explained by the fact that the HRP varies by 4 orders of magnitude, while the TWU varies by 6 orders of magnitude. Consequently, for all basins with a TWU above  $10 \text{ km}^3$  ( $n=101$ ), the CFs lie below  $2.78\text{E-}10 \text{ HDOR}\cdot\text{y}/\text{m}^3$ . Furthermore, the lowest CF value of  $1.13\text{E-}07 \text{ HDOR}\cdot\text{yr}/\text{m}^3$  has, with  $2824 \text{ km}^3$ , the highest TWU. A comparably high HRP and a comparably low TWU will lead to a comparably high CF. This is the case for the four highest CF values ( $> 2.15\text{E-}07 \text{ HDOR}\cdot\text{yr}/\text{m}^3$ ), which have a HRP between 1.23 and  $16.66 \text{ HDOR}\cdot\text{y}$  and a TWU between  $0.004$  and  $0.05 \text{ km}^3$ .

### 3.4 Comparison of midpoint characterization factors

The results show for both scenarios (minimum and maximum) that a high consumptive water-use impact does not

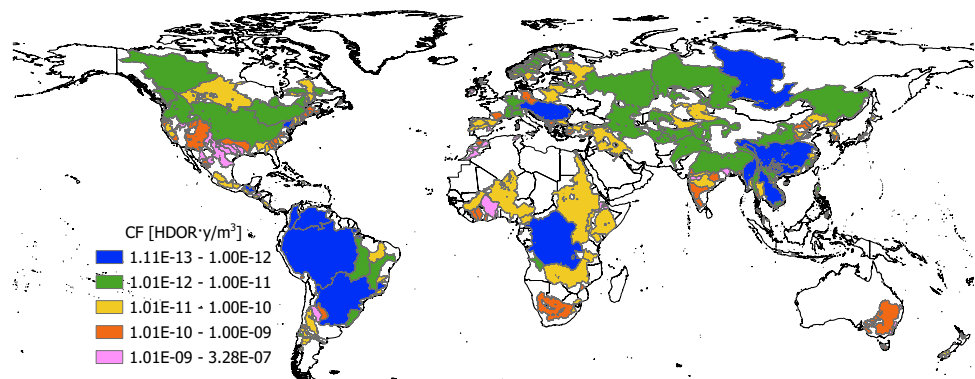
lead to a high TWU impact and vice versa ( $R^2$  Scenario 1 =  $0.0081$ ;  $R^2$  Scenario 2 =  $0.003$ ). In Scenario 1, this is specifically highlighted by basin “2,004,648; Poland,” where the impact of TWU has the second highest value, while the impact of consumptive water use has the second smallest value. In Scenario 2, basin “2,003,574; Germany/Czech Republic” has the highest impact due to TWU but the second smallest consumptive water-use impact. Only in Scenario 2 and basin “2,002,462; East Europe,” the ranks of impacts from turbine and consumptive water use are identical (Fig. 5).

### 3.5 Sensitivity

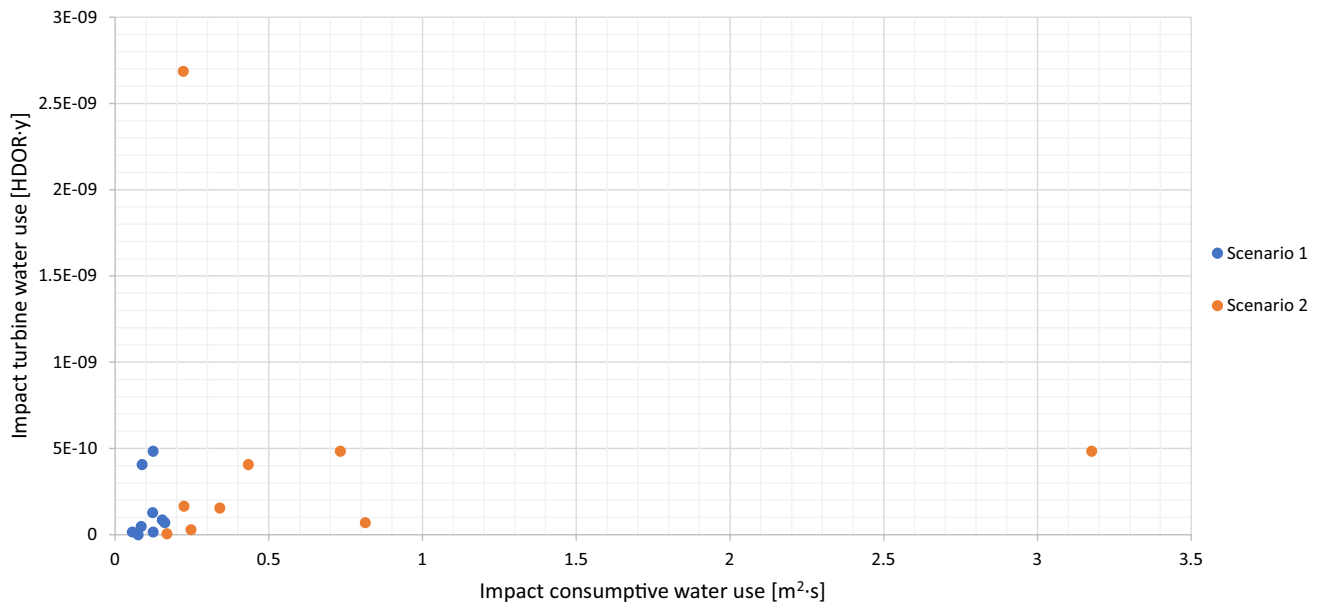
We tested the sensitivity of the calculated CFs for seven different cases (Fig. 6). Case 1a has an influence on all basin CFs and, on average, CFs are by a factor 2.14 smaller compared to original calculated CFs (Fig. 6). In Case 1b, no CFs could be calculated for seven basins due to  $Q=0$ . In all the remaining basins, Case 1b resulted in CFs that are, on average, larger by a factor of 2.97 (Fig. 6).

In Case 2a, in 134 basins, the CF value remained the same, and for the remaining 208 basins, the CFs are, on average, smaller by a factor of 2.07 compared to the originally calculated CFs (Fig. 6). This shows that, in most basins, reservoirs are used for multiple purposes. Overall, a reservoir with electricity production as the main use has, on average,

**Fig. 4** Midpoint characterization factor (CF) expressing the hydropower regulation potential (HRP) [ $\text{HDOR}\cdot\text{y}$ ] in a certain basin per turbine water use (TWU) [ $\text{m}^3$ ]. Basin delineation from Lin et al. (2021) and country borders from the Word Bank (2021)







**Fig. 5** Impact comparison of producing 1 kWh storage hydropower in nine European basins. X-axis: impact of consumptive water use [m²·s] based on CF values from Damiani et al. (2021) [m²·s/m³].

Y-axis: impact of turbine water use [HDOR·y] based on CFs from this study [HDOR·y/m³]. Scenario 1: lowest inventory values and lowest CF. Scenario 2: highest inventory value and highest CF

0.6 additional use purposes (maximum 6), and with electricity production as a secondary use, 2 additional use purposes (maximum 7).

In Case 3a, the CFs for 209 basins are identical, and for the remaining 132 basins, the CFs are, on average, smaller by a factor of 1.44 (lowest average factor) compared to the originally calculated CFs (Fig. 6). In Case 3b, for 44 basins, no CFs were calculated due to no remaining power plants (TWU = 0). For 223 basins, the CFs are identical, and for the remaining 73 basins, the calculated CFs are, on average, larger by a factor of 28.76 (highest average factor) compared to the originally calculated CFs (Fig. 6). This highlights that the developed CFs are very sensitive to hydropower plant information, as this is the basis for calculating TWU. The difference between Cases 3a and 3b can be explained by our assumption that all plants with a missing plant type and a capacity above 50 MW are storage hydropower plants. Since we already included 90% of the electricity of all power plants with “no type” information, adding the remaining power plants with “no type” as “storage plants” to calculate TWU in Case 3a has comparably little influence, while excluding all power plants with “no type” information in Case 3b has a comparably big influence on the CF value (SI1, SI2).

In Case 4a, the CFs for 36 basins are identical, and for the remaining 306 basins, the CFs are, on average, smaller by a factor of 4.25 compared to the originally calculated CFs (Fig. 6). In Case 4b, the CFs for 9 basins are identical, and for the remaining 333 basins, the CFs are, on average, larger

by a factor of 7 compared to the originally calculated CFs (Fig. 6). Case 4a gets a smaller factor because the median TE values of 3.43 m³/kWh for storage power plants and 4.3 m³/kWh for pumped storage power plants are closer to the minimum TE values of 0.56 m³/kWh and 0.81 m³/kWh, respectively, than to the maximum values of 8.1 m³/kWh and 11.58 m³/kWh.

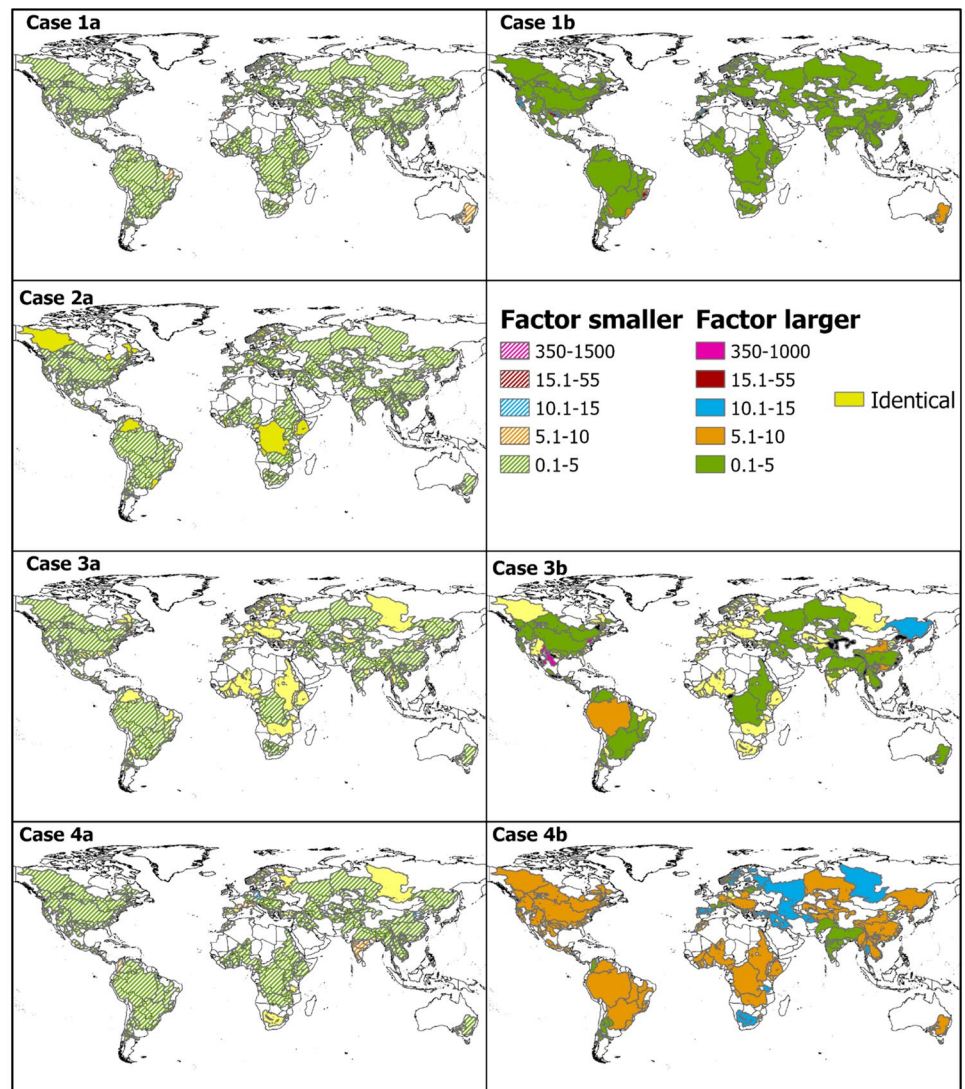
Overall, Case 1 (discharge) and Case 4 (turbine efficiency) are influencing almost all basins, while in Case 2a (use of reservoirs) and 3 (power plants “no type” information), many basins keep the same CF value, meaning that parameters Q and TE are most sensitive. But if the parameter in Case 3b has an influence, the factor is comparably higher than the sensitivity in Case 1 and 4.

## 4 Discussion

In this study, we present the first midpoint CFs for in-stream water use in LCIA, applicable to storage and pumped-storage hydropower. Lehner et al. (2011) classified rivers with a DOR ≥ 0.02 as “affected” rivers because this approximately represents a reservoir volume which is needed to store 1 week of annual river discharge.

When applying the same threshold to the HDOR, only 23 out of 342 basins have a HDOR below this threshold. That confirms that TWU of hydropower production can have important environmental impacts (Gracey & Verones 2016). The fact that the CF values vary by six orders of magnitude

**Fig. 6** Sensitivity analysis results showing by what factor smaller (Case Xa; left side, dashed color) or larger (Case Xb, right side, full color) the calculated CFs could be compared to the original input data. Country borders are obtained from the Word Bank (2021)



confirms the need for spatially differentiated CFs (Verones et al. 2017). Furthermore, the low correlation between TWU and HDOR ( $R^2 = 0.0002$ ) highlights that the pure amount of turbine water ( $\text{m}^3/\text{kWh}$ ) alone does not reflect the environmental impact (Flury & Frischknecht 2012).

In addition, our case study results indicate that a high consumptive water-use impact does not automatically lead to a high TWU impact and vice versa. This result is in line with (Scherer & Pfister 2016) who pointed out that there are trade-offs between water consumption (measured as water scarcity) and flow alteration impact (measured as environmental water stress). Consequently, our study highlights the necessity to simultaneously assess both consumptive (Boulay et al. 2018; Damiani et al. 2021) and in-stream water-use impacts. Hence, currently available storage and pumped storage hydropower (Dorber et al. 2020; Gibon et al. 2017; Mahmud et al. 2018), lacking to account for the environmental impacts of in-stream water-use, underestimate

the environmental impact, which ultimately could lead to wrong conclusions. This finding is in line with (Winemiller et al. 2016), who pointed out that many hydropower projects underestimate their related environmental impact, and Lyytimäki et al. (2020), who identified nonuse of indicators as one of three reasons for why we currently fail to reach sustainable development targets.

Since the unit of the CF is  $\text{HDOR} \cdot \text{y} / \text{m}^3$  turbine water use, our CFs are in line with the current inventory unit (Pfister et al. 2015) and are directly applicable for storage and pumped storage hydropower plants in LCA and should be interpreted as average CFs.

When interpreting the CF and the impact results, it is important to bear in mind that the CF consists of a proxy for the impact HRP [ $\text{HDOR} \cdot \text{y}$ ] divided by the TWU [ $\text{m}^3$ ]. When combined with the inventory, the final impact will be characterized as  $\text{HDOR} \cdot \text{y}$ .  $\text{HDOR} \cdot \text{y}$  can be interpreted as potential residence time, describing the timeframe during

which the release of discharge back into the natural river can be delayed. An HDOR above 1 indicates that discharge can be delayed for more than a year, assuming that the entire reservoir volume is used for flow regulation. However, there can be a high yearly variation in the actual use of storage volume (NVE 2023; Donchyts et al. 2022; Ashraf et al. 2018), influenced by the water inflow to the reservoirs (dry vs wet year). Hence, not all the available reservoir volume might be used each year. As a result, the HRP should be interpreted as a potential maximum value.

Additionally, the release of stored water can also lead to an increase in the natural river discharge. However, the HRP does not define when the regulatory effect leads to an increase or decrease in the flow in comparison to the natural flow. The actual flow regulation ultimately depends on local hydropower operation regimes, which can vary daily (Donchyts et al. 2022; Hirsch et al. 2014) and may also adhere to environmental flow requirements during reservoir operation to mitigate impacts (Poff 2017; Poff & Matthews 2013). If we assumed that each reservoir only uses 30% of its reservoir volume, each CF would be reduced by 70%, and the ranking of the basins would stay the same. However, when reservoirs use a different reservoir volume (Eloranta et al. 2018), this will cause a difference in CF value and ranking. Due to lacking global reservoir operation data, this information could not be included in the HRP calculation.

Therefore, the HRP should be interpreted as a quantitative proxy of how strongly the natural flow regime of a river may be affected by upstream reservoir operations but is not an absolute indicator. The higher the HRP, the higher the probability of a regulatory effect on the natural flow regime and the probability of an environmental impact.

Division of HRP with TWU also leads to the fact that basins, where plants have a comparably low turbine efficiency, will result in comparably high TWU, leading to comparably small CF values. However, when assessing the related impacts, areas with low turbine efficiency values will also get comparably high inventory values, leading to a comparably larger impact. In theory, the overall final quantified impact for one basin should equal the basin HDOR-y value shown in Sect. 3.1, unless the “Water, turbine use, unspecified natural origin” inventory value would be larger than TWU. However, when applying the CFs and interpreting the results, it is important to consider the limitations and uncertainty of our developed CFs, as discussed below.

We have chosen the DOR as the basis for the CF calculation, but we acknowledge that several other promising flow alteration indicators exist; for example, the eco deficit (ED) and ecosurplus (ES) from Vogel et al. (2007) (they implicitly also account for consumptive impacts) or the Dundee Hydrological Regime Alteration Method (DHRAM) from Black et al. (2005) (challenges with discharge data availability on a global scale). However, these indicators are unsuitable, as

the prerequisite for a CF is an indicator with global input data availability that exclusively focuses on in-stream water use to avoid an overlap with other impact categories. Still, if additional hydropower data availability increases, our proposed CF framework should be updated.

The sensitivity analysis for Case 2a revealed that, in 208 basins, reservoirs are not only used for electricity production. In these multipurpose reservoirs, an allocation of RV leads to lower CFs, meaning that we may overestimate the impact of hydropower reservoirs in these basins. At the same time, we assumed that electricity production ranks second if it is not the main user, as the GRanD v1.3 database (Lehner et al. 2011) does not distinguish a ranking between multiple secondary purposes. If the ranks of hydropower were lower, the overestimation of the impact would increase. As so far, no common allocation approach exists (Bakken et al. 2016), this highlights the need to agree on an allocation approach for the impacts of multipurpose reservoirs in LCA (Bakken et al. 2016).

Case 3b shows that the CF results in 117 basins are sensitive to the exclusion of hydropower plants with “no type” information, which highlights the need for global comprehensive hydropower plant data sets (Wan et al. 2021). The fact that the sensitivity of Case 3a is smaller than the one of Case 3b indicates that this data gap may lead to an underestimation of the impact because we may overestimate the number of power plants present in a basin.

Additional uncertainty is introduced by using the river mouth as representative of the entire basin. The ideal case would have been to calculate a HDOR for each river arm and then aggregate it to a basin value, following the River Regulation Index (RRI) developed by Grill et al. (2014). In this way, we could have accounted for the fact that a reservoir relatively far upstream can have a larger regulation influence on the discharge at the river mouth than a reservoir located right next to it (Grill et al. 2014). However, since the CF specifically needs to quantify the impact per m<sup>3</sup> turbine water use, this would, in parallel, have required data about how much electricity is produced with each reservoir and therewith information about the entire hydropower “cascade” (Bakken et al. 2013). This once more highlights the need for a better global hydropower database and further indicates that reservoir and hydropower plant databases should be linked.

Hence, our CFs are not meant to replace a local risk assessment of hydropower reservoir operation but should be used for relative comparison between basins, helping to identify environmental impact hotspots of hydropower production within LCA. This is currently very important, as several studies (e.g., Grill et al. 2019; Dorber et al. 2020) have pointed out that hydropower development with minimized environmental impacts, while meeting electricity production targets, can only be achieved with system-scale approaches,

including trade-off analyses of all relevant stressors. Since LCA is one tool to perform such an analysis (Hertwich et al. 2016; Liu et al. 2018), this study ultimately represents a step forward towards more sustainable hydropower development.

From a biodiversity point of view, it is known that larger rivers will generally contain more fish species. Hence, this raises the question of whether the same CF value could lead to a different biodiversity impact when  $Q$  differs. Ultimately, this question can only be answered when the impact is quantified at damage-level. To obtain an impact in PDF-y (Verones et al. 2017), a conversion factor for the here-developed midpoint with the unit PDF/HDOR would be needed. Since this factor is unitless, the conversion approach would be comparable to the PAF to PDF conversion. However, currently, no relationship between HDOR and PDF exists. Ultimately, the damage-level CF development would allow to directly compare the ecosystem quality impacts of TWU impacts with other impact pathways, such as consumptive water-use impacts. At the same time, flow regulation is only one of several cause-effect pathways leading to a potential freshwater habitat alteration related to hydropower reservoir operation (Gracey & Verones 2016). For example, CFs quantifying the impact on freshwater habitat fragmentation due to the related dam construction (Barbarossa et al. 2020) are currently also missing.

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**Data availability** The data supporting the findings of this study are available within the article and its supplementary information SI1 and SI2 can be found on Zenodo: <https://zenodo.org/records/11394252>.

## Declarations

**Competing interests** The authors declare no competing interests.

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