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Published in:
Science of the Total Environment

Link to article, DOI:
10.1016/j.scitotenv.2023.161910

Publication date:
2023

Document Version
Publisher's PDF, also known as Version of record

Science of the Total Environment 870 (2023) 161910

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Advancing water footprint assessments: Combining the impacts of water pollution and scarcity

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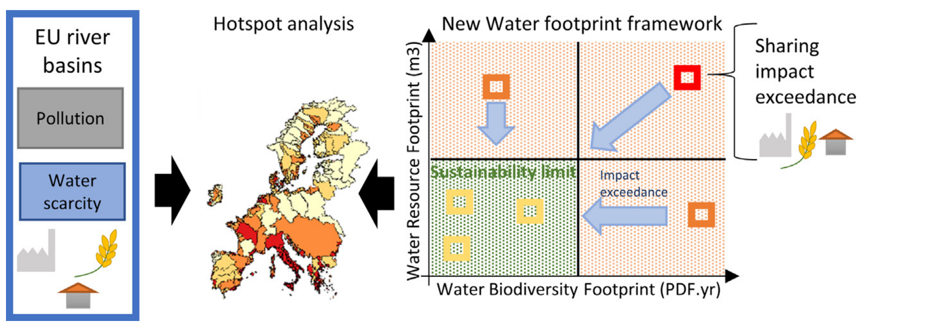
HIGHLIGHTS

- We integrate pollution and scarcity within a water footprint impact assessment.
- We model impacts on freshwater availability to humans and biodiversity.
- We test the framework assessing the sustainability of water use in the European Union.
- Pollution from industry and agricultural water consumption cause most impacts.
- Impacts exceed sustainability limits over 5 to 8 % of the area.

GRAPHICAL ABSTRACT



ARTICLE INFO

Editor: Jacopo Bacenetti

Keywords:
Life cycle assessment
Sustainability
Impact assessment
Freshwater
Water management

ABSTRACT

Several water footprint indicators have been developed to curb freshwater stress. *Volumetric* footprints support water allocation decisions and strive to increase water productivity in all sectors. In contrast, *impact-oriented* footprints are used to minimize the impacts of water use on human health, ecosystems, and freshwater resources. Efforts to combine both perspectives in a harmonized framework have been undertaken, but common challenges remain, such as pollution and ecosystems impacts modelling. To address these knowledge gaps, we build upon a water footprint assessment framework proposed at conceptual level to expand and operationalize relevant features. We propose two regionalized indicators, namely the water biodiversity footprint and the water resource footprint, that aggregate all impacts from toxic chemicals, nutrients, and water scarcity. The first impact indicator represents the impacts on freshwater ecosystems. The second one models the competition for freshwater resources and its consequences on freshwater availability. As part of the framework, we complement the two indicators with a sustainability assessment representing the levels above which ecological and human freshwater needs are no longer sustained. We test our approach assessing the sustainability of water use in the European Union in 2010. Water stress hampers 15 % of domestic, agricultural and industrial water demand, mainly due to irrigation and pesticide emissions in southern Europe. Moreover, damage to the freshwater ecosystems is widespread and mostly resulting from chemical emissions from industry. Approximately 5 % of the area is exceeding the regional sustainability limits for ecosystems and human water requirements altogether. Concerted efforts from all sectors are needed to reduce the impacts of emissions and water consumption under the sustainability limits. These advances are considered an important step toward the harmonization of *volumetric* and *impact-oriented* approaches to achieve consistent and holistic water footprinting as well as contributing to strengthen the policy relevance of water footprint assessments.

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http://dx.doi.org/10.1016/j.scitotenv.2023.161910
Received 1 September 2022; Received in revised form 19 January 2023; Accepted 26 January 2023
Available online 31 January 2023

1. Introduction

Water resources are under pressure due to the increase of human population, socio-economic development, changing consumption patterns and this pressure is expected to increase in the coming decades (UNESCO World Water Assessment Programme, 2019). The biodiversity crisis is profound in the freshwater realm, with indications that we are entering in the 6th mass extinction in the history of the planet (Ceballos et al., 2015; Cowie et al., 2022; WWF, 2020, 2021). Biodiversity loss is a major concern acknowledged by governments and economic stakeholders (World Economic Forum (2022), Sustainable Development Goals). The case of freshwater biodiversity is particularly sensitive, with a larger species abundance decline than for terrestrial species (WWF, 2020). The decline of freshwater biodiversity relates to the impacts of human activities, such as water pollution, river flow regulation, water consumption, and climate change (Reid et al., 2019). Besides the consequences on ecosystems, freshwater overuse also prejudices human development and welfare. Today, 10 % of the global population lives in areas where water scarcity is extreme, and half of the global population has limited access to safe drinking water (FAO and UN Water, 2021; UNICEF/WHO, 2021). Freshwater resources and ecosystems should be better preserved also because United Nations Sustainable Development Goals (SDG) for sanitation and clean water (SDG 6), zero hunger (SDG 2), and clean and affordable energy (SDG 7) all rely on water resources. A holistic water use assessment is needed to support decision-makers in achieving multiple, sometimes conflicting, goals involving water use (Berger et al., 2021; Boulay et al., 2021).

Several water footprint (WF) methodologies have been developed to address different issues in water management, such as scarcity, pollution, and impacts on ecosystems and human health (Boulay et al., 2018; Hoekstra, 2017; Mikosch et al., 2021; Motoshita et al., 2018; Wang et al., 2021). Two main approaches can be categorised. The first one, i.e. the *volumetric* WF assessment, focuses on water allocation and productivity (Hoekstra, 2017). It quantifies the blue (i.e., surface water and groundwater), green (precipitation stored in the soil available to crops) and grey water (pollutants dilution volume) appropriation of product systems (Hoekstra et al., 2011). The *volumetric* WF (blue and green) can then be compared to water availability after reserving environmental flow requirements that secure the functioning of terrestrial and freshwater ecosystems.

In contrast, a second approach, namely the *impact-oriented* WF (ISO, 2016), aims to reduce the consequences of water use on the environment. It builds on the *volumetric* WF and combines it with Life Cycle Impact Assessment (LCIA) models that translate water consumption volumes and pollution emissions into impacts on ecosystems, human health, and natural resources (Boulay et al., 2013). For example, the AWARE model has been used for quantifying water scarcity regional impacts throughout global supply chains (Boulay et al., 2018). Vivid discussions have occurred around the pros and cons of the *volumetric* and the *impact-oriented* approaches (Boulay et al., 2013; Pfister et al., 2017, 2022; Vanham and Mekonnen, 2021). Nonetheless, the overarching objective is the same: enabling sustainable water use. Researchers have undertaken efforts toward a harmonized approach (Boulay et al., 2021; Lathuillière et al., 2018), and highlighted several common challenges (Berger et al., 2021; Gerbens-Leenes et al., 2021; Gerbens-Leenes and Berger, 2021). For example, most WF studies focused on the quantitative aspects of water use (Mikosch et al., 2021). The ones considering pollution often considered the substance with the highest toxicity when determining the grey WF (Gerbens-Leenes et al., 2021). Common challenges to both water footprinting approaches therefore include modelling the impact of pollution on water availability, including all pollutants, quantifying comprehensively the impacts of water use on ecosystems, and integrating the social welfare dimension of water use (Berger et al., 2021; Gerbens-Leenes and Berger, 2021; Hoekstra, 2017; Mikosch et al., 2021; Van Vliet et al., 2017).

In this study, we aim to tackle the above-mentioned challenges and operationalize a framework that captures all impacts from consumptive and degradative water use. We take Lathuillière et al. (2018)'s harmonized WF assessment framework as a starting point, which combines the territorial and the product-system perspectives. In that framework, the impacts of water use, the sustainability of the consumed volumes, and water productivity are assessed to support decision-makers. We consider that such a harmonized water footprint framework that combines the benefits of a *volumetric* and *impact* based approaches is meaningful. Lathuillière et al. demonstrated in a case study the usefulness of such integrated framework to mitigate blue and green water scarcity impacts and enhance water productivity. Nonetheless, the operational impact assessment in the case study did not include the impacts of pollution on ecosystems and Lathuillière et al. pointed that integration between LCIA models for scarcity and pollution is needed. Moreover, the introduced volumetric WF sustainability assessment, which included absolute limits to freshwater availability, ecosystem water requirements, and human water needs, does not entirely mitigate the risk of damage to ecosystems and freshwater resources due to the crude representation of pollution impacts. Therefore, defining absolute limits for water use protecting ecosystems and resources remains a challenge and the water footprint sustainability assessment could benefit from life cycle impact assessment developments (Boulay et al., 2013). Our goal is therefore to revisit Lathuillière et al.'s harmonized water footprint assessment framework by enhancing the WF impact assessment and the WF sustainability assessment. We improve these parts of the framework by (i) developing environmental indicators quantifying the regional impacts of water scarcity and pollution on freshwater ecosystems and freshwater availability, (ii) providing absolute sustainability limits to environmental impacts protecting freshwater ecosystems and human welfare, and (iii) testing the operability of such improvements with a proof of concept assessing whether and where water use was sustainable in the European Union (EU) in 2010.

2. Methods

2.1. Overview of the revised harmonized water footprint assessment framework

In the following, we adopt the terminology from the water footprint ISO standard (ISO, 2016). We also designate surface water and groundwater as blue water, and precipitation stored in the soil as green water, following Hoekstra et al. (2011). Our starting point is the harmonized WF assessment framework proposed by Lathuillière et al. (2018). It includes the same stages as the ISO standard (ISO, 2016), i.e., a goal and scope definition, WF inventory analysis, a WF impact assessment. The first step determines the product system to be assessed by defining the functional unit and the reference flow. The inventory analysis delivers the volumes of blue, grey, and green water consumed, while the impact assessment translates the inventory into quantitative impacts on ecosystems, human health, and resources. Lathuillière et al. added a volumetric WF assessment, a volumetric WF sustainability assessment, and a policy decision stage. The volumetric WF assessment benchmarks water productivity by comparing the volume of water consumed per functional unit among similar products. Then, the volumetric WF sustainability assessment verifies whether the total water consumption volume of all unit processes in the hydrological unit (e.g., river basin) is smaller than the water available regionally and seasonally, following the guideline from the Water Footprint Network (Hoekstra et al., 2011). It defines environmental, social and economic sustainability criteria for the volumetric WF. The environmental sustainability assessment compared the blue and green WF with the freshwater availability subtracting environmental flow requirements and the grey WF is compared to the dilution capacity in the hydrological unit (Hoekstra et al., 2011). The social sustainability assessment verifies whether the water availability, removing the environmental flow requirements, can satisfy the water requirements for drinking, washing, cooking, and food (Hoekstra et al., 2011). Finally, the policy decision stage compiles the results from the previous stages to provide relevant insights to decision-making in water management and product eco-design decision contexts, similarly to the interpretation stage of the ISO standard. The revised harmonized WF assessment framework (Table 1) maintains the many elements

Table 1
Revision of the harmonized water footprint assessment step by step (modified from Lathuillière et al. (2018)).

WF assessment stage	Water Biodiversity Footprint (WBF)	Water Resource Footprint (WRF)
Goal and scope (ISO, 2016)	Defines the objectives of the study, the product system, the functional unit, the hydro-geographic extent, the intended audience, and how results will be used.	
WF inventory analysis (ISO, 2016)	Compiles all the inputs and outputs from the product system to obtain the elementary flows emitted into the environment, i.e., emissions to air, soil, and freshwater (regionalized, monthly values). Consumptive water use refers to water incorporated into the product, evaporated, transpired, released into a different watershed, or directly released into the sea. Surface water, groundwater (blue water), and green water (rain stored in the soil) consumption rates are calculated separately.	
WF productivity assessment (called volumetric WF assessment in Lathuillière et al. (2018)	Compares blue- and green- water consumption, and emissions per functional unit to similar product systems (equivalent to efficiency assessment). The objective is to identify possible water and chemicals savings to improve the efficiency of water use in the product system.	
WF impact assessment (ISO, 2016)	Translates the results of the WF inventory analysis into impacts on ecosystems. It represents the damage to global freshwater biodiversity and is expressed in potentially disappeared fractions of species integrated through time (global PDF yr).	Translates the results of the WF inventory analysis into impacts on freshwater resources. It represents the impacts to freshwater resources in terms of water deprivation potential and is expressed in m^3 unavailable water (m^3 yr).
WF sustainability assessment	Compares the total WBF with the absolute sustainability limit for freshwater ecosystems (SL_{WBF}). Impact exceedance indicates a potential risk of regional freshwater ecosystem collapse. Impact exceedance is shared among water users.	Compares the total WRF with the absolute sustainability limit for human welfare (SL_{WRF}). Impact exceedance indicates a potential risk to human health. Impact exceedance is shared among water users.
Decision-making (called policy decisions in Lathuillière et al. (2018))	Integrates the findings from WF productivity, impact assessment, and sustainability assessment, to provide recommendations for the product system (per functional unit from the WF productivity assessment and impact assessment) and the hydro-geographic extent (from the sustainability assessment). The uncertainty of the impact assessment, sustainability assessment, and the limitations of results should be assessed. Conflicting decisions should be highlighted with potential cost and benefit analyses.	

from Lathuillière et al. (2018) but we introduce some modifications which we present hereafter.

- Goal and scope definition

Following Lathuillière et al. (2018), the functional unit can be defined as all activities over a region or representing the function of a specific product system. In the first case, the WF assesses water use sustainability of territories for supporting water management and meso-scale decisions (Lathuillière et al., 2018; Loiseau et al., 2018). In the second case, the WF aggregates the results for all the unit processes in the product system, which eventually occur in different hydrological units, and support decision related to the product sustainability (Lathuillière et al., 2018).

- Water footprint inventory analysis stage

The collection of the blue and green WF inventory and emissions follows the scope determined by the functional unit. We do not calculate the grey water footprint because emission-based LCIA models are more accurate to represent the impacts of pollution in the WF impact assessment stage (Pfister et al., 2017).

- Volumetric WF assessment and decision-making stage

We propose to maintain the volumetric WF assessment and the policy decision stages in the revised framework while renaming them to WF productivity assessment stage, and decision making stage respectively, so that the terminology is aligned with the goal of each stage. Indeed, the results from the WF assessment can be used to support decision in policy making but also in product development decision contexts. Moreover, WF productivity should refer both to water consumption and chemicals (including fertilizers) use per functional unit. The decision making stage should also include an uncertainty assessment of the results of the previous stages, in particular the WF impact and WF sustainability assessments, and a statement of the limitations as required by the ISO 14046 (ISO, 2016).

- WF impact assessment stage

We expand the WF impact assessment stage so that it covers the impacts of water consumptive and degradative use in the river basin (Fig. 1). Fig. 1 includes the water footprint inventory and productivity assessment for completeness purpose but they are not modified in the revised framework. The ISO 14046 defines water availability as the "extent to which humans and ecosystems have sufficient water resources to satisfy their needs" (including quality degradation). Since the ecosystem and human water needs are different (timing-, quantity- and quality-wise), we propose two indicators to represent them: the Water Biodiversity Footprint (WBF) and the Water Resource Footprint (WRF). While, the WBF represents the combined damage to freshwater ecosystems from pollution and water scarcity, the WRF represents the competition between water users for fulfilling their water demand and includes the potential deprivation due to scarcity and pollution. Therefore, combined WRF and WBF cover the four dimensions of the ISO water availability footprint i.e. water quality degradation and scarcity impacts on water availability to humans and ecosystems.

- WF sustainability assessment stage

Unlike Lathuillière et al. (2018) and Hoekstra et al. (2011), who compared water consumption volumes with the freshwater availability, we propose comparing impact scores (WBF and WRF) with regional sustainability limits (SL_{WBF} and SL_{WRF}) in the WF sustainability assessment (Table 1). The WBF and WRF, better represent the possibility of fulfilling water needs of ecosystems and humans than volumetric WFs because WBF and WRF include pollution mechanisms comprehensively. Therefore, we argue that the volumetric WF sustainability assessment from Lathuilière et al.'s framework can be advantageously substituted by an impact sustainability assessment using WBF and WRF in the revised framework. The WBF and WRF of all facilities in the river basin are then compared to the absolute sustainability limits of the basin (Fig. 1), similarly to the carrying capacity approach developed by (Bjørn and Hauschild, 2015) and the geographical sustainability assessment in Hoekstra et al. (2011). The carrying capacities define the maximum impact that the environment can sustain without suffering irreversible impairment for each impact pathway. Setting absolute limits for environmental impacts (i.e., WBF and WRF) can better inform decision-makers and support preventing irreversible damages to ecosystems and human health. Defining absolute thresholds (for water use and the resulting impacts) also prevents rebound effects because the total impact is constrained, no matter the water efficiency. Because the impacts on freshwater ecosystems and resources are regional, SL_{WBF} and SL_{WRF} should be defined at the regional scale. The SL_{WBF} is the "ceiling"

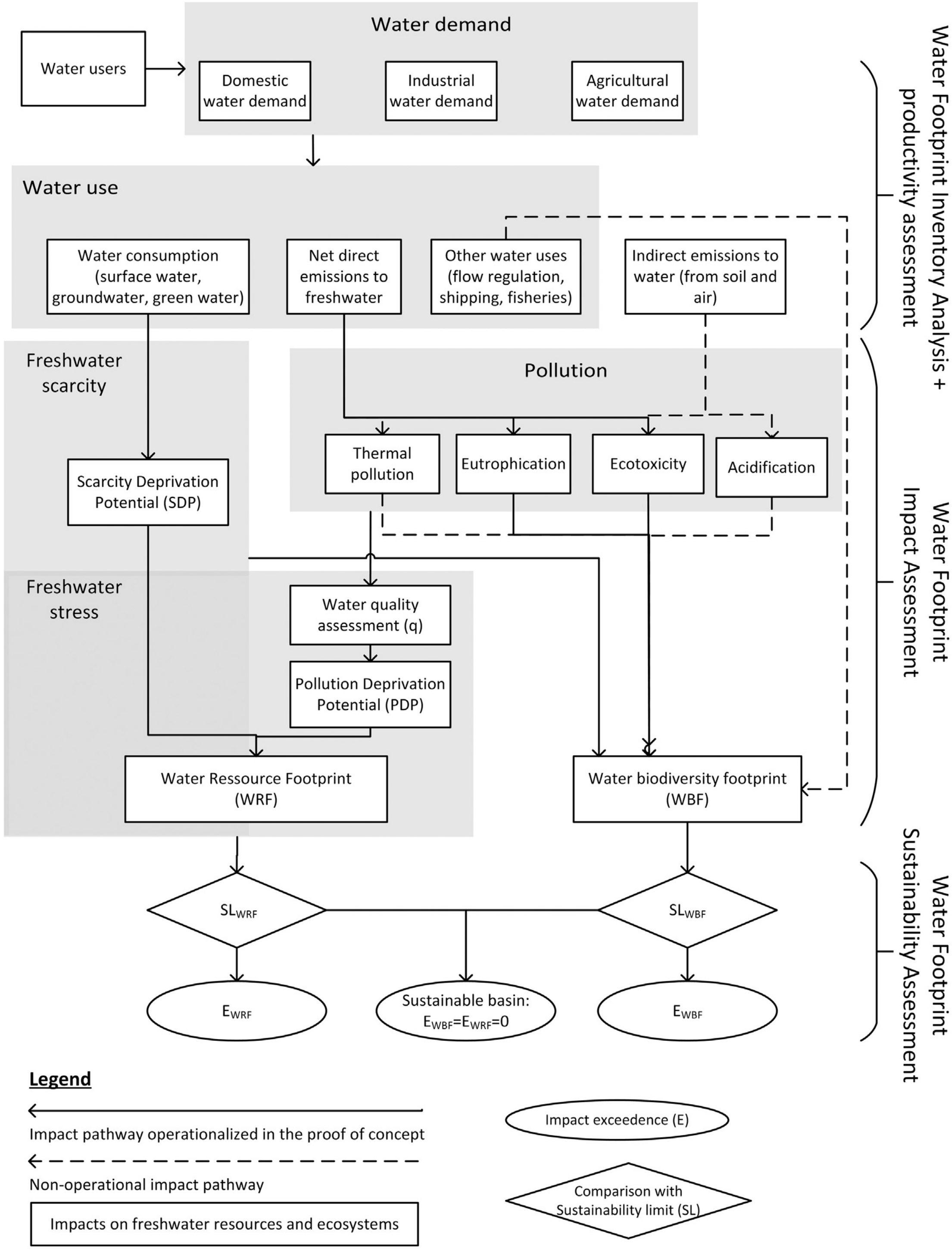


Fig. 1. Water footprint assessment overview and impact coverage.

of acceptable environmental impacts, while the SL_{WRF} is the "social foundation" guaranteeing human rights to freshwater (Raworth, 2012). Water overuse occurs when one or both limits are exceeded. As part of the WF sustainability assessment, we share the impact exceedance among water users to distribute the efforts to reduce impacts within sustainable limits. In doing so, we go beyond the previous sustainability assessment from Hoekstra et al. (2011) and Lathuillière et al. (2018), which did not include a practical approach to re-allocate water resources within sustainability limits.

2.2. Water footprint impact assessment

2.2.1. Water biodiversity footprint (WBF)

The WBF assesses the potential damage to freshwater ecosystems, defined as the species living in freshwater wetlands and water bodies. WBF compatible with LCIA is expressed in Potentially Disappeared Fraction of Species (PDF) integrated through time (i.e., PDF·yr) (Verones et al., 2017). It includes a transformation factor so that regional species losses can be converted into global species loss and can be aggregated throughout the life cycle (Verones et al., 2017), i.e., the Global Extinction Probability (Verones et al., 2022).

The operational WBF includes eutrophication and ecotoxicity impacts, for which LCIA models exist (Fig. 1 solid lines). It also covers the impacts of water consumption (Fig. 1 solid lines) from each water source, i.e., surface water, groundwater, and green water, are reported separately whenever possible. Water consumption associated with crops, i.e. soil moisture consumption (also called blue-green water consumption), can alter evapotranspiration over land and, consequently, precipitation regimes, which affects terrestrial and freshwater ecosystem (Boulay et al., 2021; Hoekstra, 2019; Quinteiro et al., 2015). Assuming that the water cycle and the ecosystems are in equilibrium with the current land use, only the change of green water use and land use affecting evapotranspiration are included in WBF (Boulay et al., 2021; Milà I Canals et al., 2009). The operational WBF supported by existing LCIA models (Fig. 1 solid lines) is expressed by Eq. (1). Impacts are calculated first per month then aggregated for the year.

$$WBF^{k} = 12\left(\sum_{m} CF_{wc} WC^{m,k} + \sum_{i,m} CF_{FET,i} r_{i}^{m,k} \frac{1}{V^{m}} + \sum_{m}\left(CF_{FE,P} \cdot r_{P}^{m,k}\right)\frac{1}{V^{m}}\right) GE \quad (1)$$

where: WBF^k is the water biodiversity footprint of water user k (PDF·yr), the characterization factors for freshwater ecosystem impacts are CF_{WC}, for water consumption (expressed in PDF·yr/m^3), $CF_{FET,i}$ for ecotoxicity (expressed in PDF·month·m^3/kg), $CF_{FE,P}$,is the CF of freshwater eutrophication impact on freshwater ecosystem from phosphorus (expressed in PDF·month·m^3/kg), V^m is the monthly river volume in month m (m^3), $WC^{m,k}$ is the water consumption of user k in month m (m^3), $r^{m,k}{}_i$ is the emission of toxic substance i from user k in month m (in kg/month), $r^{m,k}{}_P$ is the emission of phosphorus from user k in month m (in kg/month), GEP (dimensionless) is the conversion factor translating regional freshwater fish species extinctions into global extinctions. Background document behind Eq. (1) can be found in Section S1.1 of the supporting information.

Eq. (1) only represents the operational part of the potential damages of water use because globally-applicable LCIA models are missing to address a number of impacts covered in the framework, e.g., for thermal emissions, nitrogen emissions, green water consumption (Fig. 1, dashed lines). Eq. (1) should therefore be completed as soon as these become available. Likewise, atmospheric moisture travels across river basins' boundaries and surface water, groundwater, and soil are interconnected, thus water consumption impacts may extend beyond the river basin where consumption takes place (De Graaf and Stahl, 2022; Link et al., 2021; Pierrat et al., in review). However, in the current operationalisation, water consumption impact interactions between river basins could not be accounted for due to lack of data. Finally. multiple stressors can have synergistic or antagonist effects on freshwater organisms, which can change impacts on ecosystems significantly (Reid et al., 2019). For example, over-exploitation of groundwater can result in quality degradation (Gejl et al., 2018), and streamflow reduction can also concentrate pollutants in the environment (smaller volume of the exposure medium) and increase their residence time. More research is needed to enable modelling and integration of such multi-stressor mutual interactions in the footprint calculations. Until it becomes operational, Eq. (1) assumes additivity of the impacts, as is typically done in the LCIA framework (Hauschild, 2005) .

2.2.2. Water resource footprint (WRF)

We develop the indicator WRF to model the impacts of freshwater stress, i.e., pollution and scarcity, on water resource availability (Fig. 1, Table 1). The WRF is the water deprivation potential, expressed in m^3 of freshwater unavailable (m^3/yr), caused by a water user k (e.g., a farmer). The main innovation of the WRF, compared to other water stress indicators, is that it distinguishes the pollution deprivation potential (PDP) from the scarcity deprivation potential (SDP) (see section S1.2 for the operationalization) for the three main sectors using water (sector j), namely: the industry, domestic, and agricultural sectors (Fig. 2A, B). We report sectoral deprivation separately because the availability for each sector depends on its demand and quality requirements, and the consequence of depriving each sector from water are different. Therefore, the WRF of each user k (for instance, a farmer part of the agricultural sector; or a facility part of the industrial sector) on all sectors j (domestic, industrial, agricultural sectors) are reported.

First the SDP and PDP are defined at the river basin scale (Fig. 2). The PDP depends on the quality of the water present in the environment. The water quality is insufficient when the environmental concentration of at least one pollutant exceeds the limit of the sectoral water quality requirement. The water quality requirement considers a conjunction of chemical and biological parameters, typically devised to minimize risks to human health. Public authorities (e.g., EU Commission or national governments) or other authoritative organizations (e.g., FAO, UN-Water) often distinguish quality parameter thresholds according to the intended water use (Boulay et al., 2011). If the quality is insufficient, the PDP of a sector is equal to the water demand of that sector scaled by the water availability in the river basin (Fig. 2A, Eq. (2)). The PDP focuses on surface water resources because emission data and transport models are available while it is not the case for groundwater and soil moisture.

$$PDP_{j}^{m} = q_{j}^{m} \cdot s^{m} \cdot \Sigma_{k} WC_{j}^{m,k} \quad (2)$$

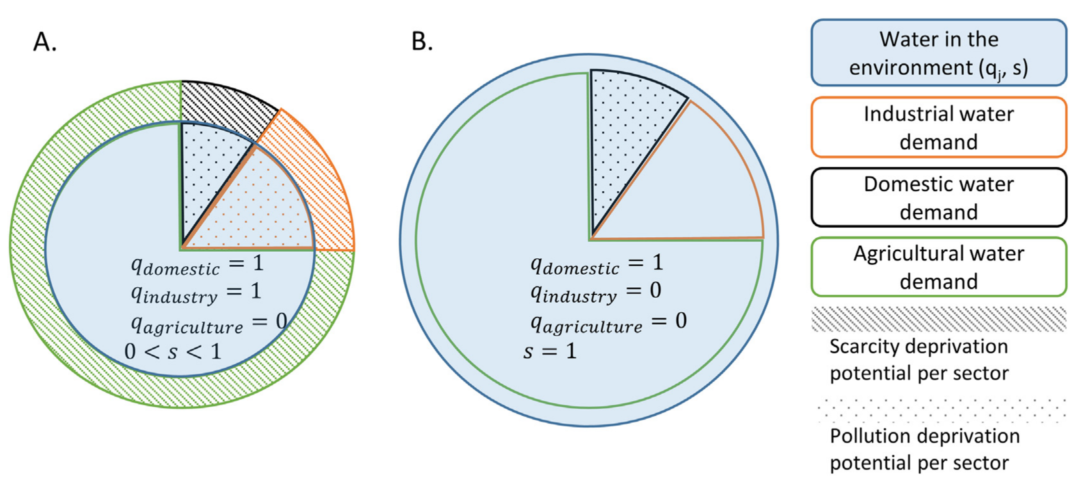


Fig. 2. Illustrative example of the water resource footprint at the river basin scale in the dry season with lower water quality ($q_j = 1$) and water shortage ($0 < s < 1$) (A) and wet season with higher water quality ($q_j = 0$) and no water shortage ($s = 1$) (B).

$$with\ s^m = \frac{A^m}{\Sigma_{k,j} WC_j^{m,k}}\ for\ 0 \leq \frac{A^m}{\Sigma_{k,j} WC_j^{m,k}} \leq 1$$
$$and\ s^m = 1\ otherwise$$

where PDP_j^m is the pollution deprivation potential of sector j in month m for the river basin, $q^m{}_j$ is a Boolean water quality assessment for sector j in month m (equals one if water quality is insufficient), s^m is the ratio of water availability to demand in month m (ratio between 0 and 1), A^m is the monthly water availability (m^3), $WC^{m,k}{}_j$ is the water consumption of user k in sector j in month m.

The SDP is defined only when the water available in the environment i.e., the surface water, groundwater, and green water availability, is lower than the total demand in the river basin (Fig. 2A). When there is a shortage, the SDP quantifies the fraction of the sectorial water demand that is not attended due to the water shortage (Eq. (3)) and we assume that the SDP is distributed across water users proportionally to their water demand.

$$SDP_j^m = (1 - s^m) \cdot \Sigma_k WC_j^{m,k} \tag{3}$$

where SDP_j^m is the scarcity deprivation potential of sector j in month m for the river basin, s^m is the ratio of water availability to demand in month m (ratio between 0 and 1), $WC^{m,k}{}_j$ is the water consumption of user k in sector j in month m.

The WRF associated with water user k is a fraction of the monthly PDP and SDP calculated at the river basin scale, and this fraction corresponds to the user's contributions to water scarcity and water quality degradation. We define a scarcity weighting factor (noted ws, see Eq. S13) to calculate the SDP of a specific water user, which is calculated as the ratio of the user's water consumption by the total sectoral demand. The pollution weighting factor (noted wp, see Eq. S10) is proportional to the user's emissions and the severity of the pollution compared to the quality requirements. It is calculated as the product of the ratios between the user's emissions by the total emissions, and the pollutant's concentration exceedance by the sum of all pollutants' concentration exceedances. Therefore, the operational form of WRF can be described as in Eq. (4).

$$WRF^k = \sum_{m,j} \left(wp_j^{m,k} PDP_j^m + ws_j^{m,k} SDP_j^m \right) \tag{4}$$

where WRF^k is the water resource footprint of water user k (m^3), $PDP^m{}_j$ is the pollution deprivation potential in sector j in month m (m^3), $SDP^m{}_j$ is the scarcity deprivation potential in sector j in month m (m^3), $wp^{m,k}{}_j$ is the pollution weighting factor of water user k in month m for sector j (expressed as a unitless ratio between 0 and 1), $ws^{m,k}{}_j$ is the scarcity weighting factor of water user k in month m for sector j (expressed as a ratio between 0 and 1). The detailed calculations with the intermediary steps leading to Eq. (2) are presented in Supplementary Methods Section S1.2. Thus, WRF is positive or null, and it cannot exceed the total water demand in the river basin. Positive WRF means that water consumption exceeds water availability, including pollution issues. Thus, SDP (hence WRF) already covers the volumetric WF sustainability assessment in Lathuillière et al. (2018), where the *volumetric* water footprint is compared to freshwater availability in the basin.

Eq. (4) can be derived for surface water, groundwater and green water because surface water stress, groundwater stress (PDP and SDP), and green water scarcity (SDP only) stress can differ in each water cycle compartment. Nonetheless, it requires monthly groundwater quality and crop data (e.g., irrigation efficiency, fertilizer and pesticide use) that are not always available. The blue water availability is the total blue water available, i.e., water in the river, lakes, reservoirs and groundwater storage. We consider that all freshwater can be used during a given month, even though this would jeopardize the ecosystem because ecosystem damage is quantified separately with WBF. It is noteworthy that it is possible to consider only the renewable part of water availability so that a positive SPD includes the risk of multi-annual groundwater depletion. However, it likely leads to overestimating the monthly water scarcity. In this case, the monthly groundwater recharge can be an estimate of renewable groundwater availability. Green water availability can be estimated by translating land use data and agronomy constraints into crop evapotranspiration budget (green water flow) (Schyns et al., 2019). Alternatively, blue-green water availability can be estimated using soil moisture over the cropland (Erlandsson et al., 2022). Here the assessment of the soil moisture scarcity (green or blue-green water) depends on the water accounting scheme for crop water consumption (e.g. Hoekstra (2019); Link et al. (2021)). However, the choice of a water accounting scheme is outside the scope of this study.

2.3. Water footprint sustainability assessment

The WF sustainability assessment stage compares the total WBF and WRF with the absolute sustainability limits of the river basin (Fig. 1 diamond and oval shapes). The proposed absolute sustainability limit for WRF is defined as the minimal volume of freshwater needed to fulfil human water requirements, including hygiene and irrigation demand for food (Hoekstra et al., 2011; Motoshita et al., 2020). This limit is aligned with the universal human rights to food, water, and sanitation recognized by the United Nations (United Nations, 1948, 2010). It represents the socio-economic limit under which freshwater in the environment cannot fulfil human physiological needs (see details in Section S1.3.2). Water deprivation exceeding this threshold can potentially affect human health if adaptation is insufficient (Debarre et al., 2022; Motoshita et al., 2018). The sustainability limit for WBF corresponds to the ecological carrying capacity beyond which the ecosystems are damaged irreversibly (Bjørn and Hauschild, 2015). Previous approaches were based on specific environmental mechanisms, e.g., carrying capacities and planetary boundaries for nutrients, chemicals, and water consumption (Bjørn et al., 2020a, 2020b; Kosnik et al., 2022; Steffen et al., 2015).

In contrast to these, in the current assessment, we propose defining SL_{WBF} in terms of maximum species losses because it should guarantee that the combined pressure does not threaten the ecosystem integrity. The planetary boundary framework proposed some regional biodiversity boundaries beyond which ecosystems start malfunctioning (Mace et al., 2014; Steffen et al., 2015). These proposals are valid, but incompatible with existing LCIA methods because most LCIA models currently quantify potential species richness loss (PDF) rather than functional biodiversity loss or extinction rates. Another solution compatible with the PDF metrics is to minimize the risk of ecosystem collapse, i.e., catastrophic ecosystem regime shift. Such shift can be triggered by snowballing species extinctions that propagate throughout the freshwater food web (Brook et al., 2013; Keyes et al., 2021). Based on Curtsdotter et al. (2011)'s study of food web response to species extinctions, a conservative estimate of biodiversity tipping point equals a PDF of 0.08. Beyond the threshold, the survival rate in a generic ecosystem is lower than 50 % of the species. A safe SL_{WBF} should include the uncertainty on the biodiversity threshold, a buffer to avoid reaching the threshold as in the Planetary boundary framework (Steffen et al., 2015), and the WBF (e.g., uncertainty on the inventory, LCIA models, impact coverage). Thus, we propose adopting a safety coefficient when determining SL_{WBF}.

When WBF or WRF exceed the sustainability limit, we calculate the impact exceedance as the difference between the water footprints and the sustainability limits (see Eqs. (7) and (8) and section S1.3.3 for the details). Therefore, reverting the river basin situation to a sustainable level of pressure requires to bring the impact exceedance down to zero. In the current framework, the impact exceedance is assumed to be shared following the widely used sharing principle of "polluters pay", which dictates that the responsibility of a polluter to an exceedance is assigned proportionally to the magnitude of its caused impacts (note that other sharing principles may be applied, e.g. see Section 4.3 and Ryberg et al. (2020)). The share of water user k to WBF (Eq. (4)) and WRF (Eq. (5)) exceedances are thus proportional to its environmental impacts.

$$S_{WBF}^k = \frac{WBF^k}{\Sigma_k WBF^k} \tag{5}$$

where S^k_{WBF} is the share of user k in the impact exceedance for WBF (expressed as a ratio between 0 and 1).

$$S^k_{WRF} = \frac{WRF^k}{\Sigma_k WRF^k} \tag{6}$$

where S^k_{WRF} is the share of user k in the impact exceedance for WRF (expressed as a ratio between 0 and 1). Therefore, the impact exceedances for WBF and WRF for user k are described by Eqs. (5) and (6). Additional details are available in in Section S1.3.3.

$$E^k_{WBF} = S^k_{WBF}\left(\sum_k \frac{WBF^k}{GEP} - SL_{WBF}\right) \geq 0 \tag{7}$$

where E^k_{WBF} is the impact exceedance for the freshwater ecosystem attributed to user k expressed in (expressed in local PDF·yr), S^k_{WRF} is the share of user k in the impact exceedance for WRF (expressed as a ratio between 0 and 1), WBF^k is the water biodiversity footprint of user k (expressed in global PDF·yr), GEP is the global extinction probability of freshwater species in the river basin, SL_{WBF} is the sustainability limit for WBF in the river basin (expressed in local PDF·yr).

$$E^k_{WRF} = S^k_{WRF}\left(\left(SL_{WRF,dom} - \sum_m\left(WC^m_{dom} - WRF^m_{dom}\right)\right) + \left(SL_{WRF,agri} - \sum_m\left(WC^m_{agri} - WRF^m_{agri}\right)\right)\right) \geq 0 \tag{8}$$

where E^k_{WRF} is the impact exceedance for the water resource attributed to user k expressed in (m^3); S^k_{WRF} is the share of user k in the impact exceedance for WRF (ratio between 0 and 1); $SL_{WRF,dom}$ is the human water requirement for domestic use (m^3), $SL_{WRF,agri}$ is the human water requirement for agricultural use (m^3), WRF^m_{dom} is the domestic water deprivation in month m (m^3), WRF^m_{agri} is the agricultural water deprivation in month m (m^3), WC^m_{dom} is the total water consumption for the domestic sector in month m (m^3), WC^m_{agri} is the total water consumption for the agricultural sector in month m (m^3).

The operationalisation of WF sustainability assessment is tightly related to the availability of LCIA models and their ability to calculate WBF and WRF. Ideally, the WF sustainability assessment should compare the sustainability limits with the total impacts on the ecosystems and freshwater resources, including the background stressors. The total impact on the ecosystems and freshwater resources should thus be the sum of WBF or WRF for all water use in the river basins with the WBF or WRF impacts of the background stressors. Background stressors (Fig. 1 dashed lines) include pressures on freshwater ecosystems unrelated to consumptive and degradative water use (e.g., fisheries, flow regulation for hydropower, etc.) and indirect pollution from soil and air-borne emissions deposition into freshwater (e.g., freshwater acidification). Although not yet possible due to lack of knowledge and data gaps, these could be included in E_{WBF} by introducing the impact of background stressors WBF background in Eq. (6) (Eqs. S16, S18), and accounting for indirect pollution in the calculation of the PDP and the pollution weighting factor (Eqs. (2) and S10). Data generation in this area (e.g., enabling modelling of all indirect pollution) and the development of new LCIA models to cover these pathways (e.g., flow regulation impacts on ecosystems) is therefore warranted to fill these gaps (see Section 3.1).

2.4. Proof of concept: water footprint assessment of the EU-27

To illustrate the workability of the proposed framework, we focus on the WF impact assessment and the sustainability assessment of 1031 river basins in the EU-27 in 2010. We adopted a territorial perspective, where the functional unit includes all the activities consuming water. We distinguish three major water users in the assessment: the domestic, the industrial, and the agricultural sectors (the users k: the domestic, the industrial, and the agricultural users aggregated at sector scale). We use Eqs. (1)–(8) to calculate the regionalized WRF, WBF, and impact exceedances for each sector (sectors j: domestic, industrial, and agricultural sectors).

The life cycle inventory (LCI) data and the LCIA models for WBF are presented in details in Section S2 of the Supplementary Material. We retrieved monthly sectorial water consumption and monthly river basin volume at grid cell scale (5 degree resolution) from a state of the art global hydrological model WaterGAP 2.2 (Müller Schmied et al., 2021). Moreover, we extracted yearly chemicals emissions at country scale from Leclerc et al. (2019). Annual crop harvest and nutrient emissions were collected from Eurostats (2022) and Scherer and Pfister (2015). Chemicals and nutrient emissions were disaggregated at basin and monthly scale assuming that they were uniformly distributed in space and time.

We used state of the art regionalized LCIA models to calculate WBF (Rosenbaum et al., 2008; Scherer and Pfister, 2015; Verones et al., 2022), except for the impacts of blue water consumption for which we used the most recent update (Pierrat et al., 2023). For WRF calculation we considered the water quality requirements from Boulay et al. (2011). We defined groundwater availability as the monthly groundwater recharge so that WRF captures groundwater depletion risk and surface water availability as the monthly river volume. We adopted the precautionary absolute sustainability limits equal to 0.04 PDF·yr for WBF (Sections 2.3 and S1.3.1) and to the domestic monthly water consumption for WRF.

We adopted the absolute sustainability limits equal to the domestic monthly water consumption for WRF. For WBF, we assumed that a safety coefficient of two is sufficient for buffering the uncertainty on WBF (Section 2.3). Several sharing principles based on ethical norms exist (Ryberg et al., 2018, 2020) and adopting one is always a value choice that should ideally involve all the stakeholders. We proposed to share the impact exceedance following the "polluters pay" principle where the impact mitigation efforts are shared among the sectors proportionally to their impacts.

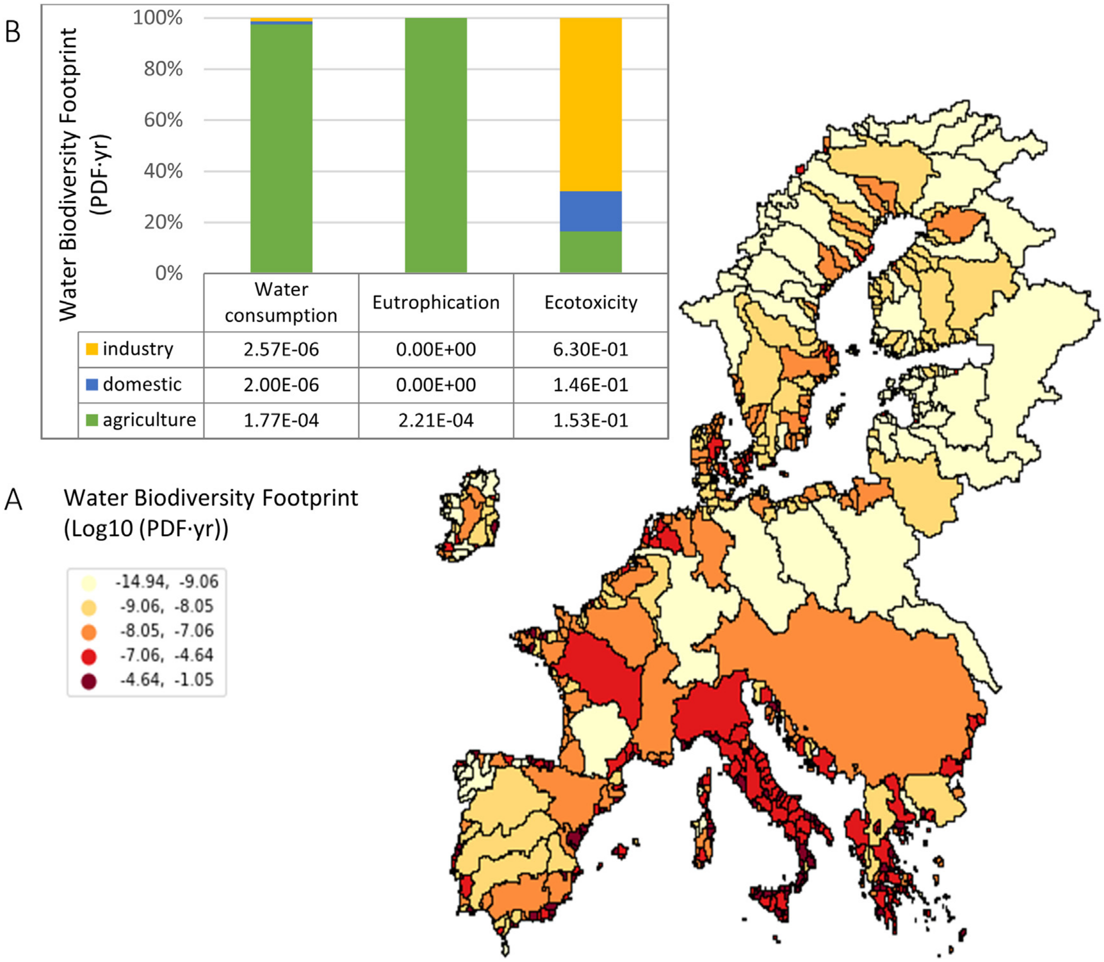
For this illustration, we focus on blue water consumption and surface water quality because we had sufficient data to show the combined effects of surface water pollution and consumption for this water resource. We disregard green water consumption and groundwater quality due to a lack of data and to simplify the case study.

The WF sustainability assessment stage includes several value choices that may influence the results: the sustainability limit definition and the sharing principle (Section 3.4). First, we investigate the effect of WBF selectivity on species using generic SL_{WBF} that correspond to average ecosystem robustness values from literature (Curtsdotter et al., 2011). One scenario corresponds to the best estimate of the sustainability limit where consumer species are randomly affected and we obtain $SL_{WRF} = 0.28/2 = 0.14$ PDF·yr. The other scenario corresponds to a conservative scenario where producer species are affected first, then $SL_{WRF} = 0.08/2 = 0.04$ PDF·yr. Second, we further analyze the spatial sensitivity of SL_{WBF} because each freshwater ecosystem has a different limit, aligned with the local species abundance, species richness, and the configuration of the food web (Brose et al., 2017; Curtsdotter et al., 2011). We arbitrarily divide by a factor ten SL_{WBF} to simulate the response of a fragile ecosystem. Third, we compare the impact exceedances for WBF obtained with the "polluters pay" and "gross value added" sharing principles. Adopting the "gross added value" sharing principle gives a higher share of the impact exceedance (i.e., more ambitious impact reduction targets) to wealthier users who can finance more efforts to reduce impacts, hence including equity questions partially.

3. Results

3.1. Water biodiversity footprint

The highest WBF occur around the Mediterranean, and Baltic seas, in western France and Italy (Fig. 3A and Fig. S1 for maps impacts for each pathway). Ecotoxicity (FET) impacts cause by far the greatest damage to freshwater biodiversity (99.99 % of total impacts), followed by eutrophication (FE) and water consumption (WC) (Fig. 3B). FET impacts dominate WBF due to the emissions of the industry sector, especially zinc and strontium compounds (substance contribution analysis in Table S1), while the agriculture sector dominates FE and WC (Fig. 3B). Nevertheless, the FE and WC impacts should not be neglected because their spatial distribution



	Water consumption	Eutrophication	Ecotoxicity
industry	2.57E-06	0.00E+00	6.30E-01
domestic	2.00E-06	0.00E+00	1.46E-01
agriculture	1.77E-04	2.21E-04	1.53E-01

Fig. 3. Map of the water biodiversity footprint where the colour scale corresponds to quantiles (number of river basins) (A) and contribution analysis of the industry, the agriculture and the domestic sector (B).

differs from FET's (Fig. S1). For instance, WC impacts are particularly strong in Spain, where irrigated agriculture is important (Fig. S1).

This impact map shows some resemblance to the EU ecological quality assessment (2009–2018) that identified Central Europe and Northern Europe, France, and Italy as regions with low to poor freshwater ecosystem quality (EU waters, 2018). Some identified areas strongly exposed to toxic emissions (e.g., Eastern Europe and North Germany), and areas exposed to FE (e.g., Northern Italy, Northern and Western France, Northern and Eastern Spain, Eastern Europe) (EU, 2018; Payen et al., 2021; Posthuma et al., 2019) do not stand out in our calculated pollution impact assessment. This discrepancy may come from the LCI data, which lacks spatial and temporal resolution, and is only available for the year 2010 and at the country scale (Sections 3.1, S2). Calculated WC impacts are consistently found in arid regions where irrigated agriculture is strong (e.g., Spain, Italy, Greece) and in small coastal river basins. The low contribution of WC to WBF is consistent with the correlation analysis of multiple anthropogenic pressures with water quality in the EU, but the high contribution of FET is contrasting (Lemm et al., 2021). Discrepancy in toxicity scores may come from inventory data, the assumption that all metallic emissions are biologically available (Section S2), the modelling of substances transport, which is not regionalized at the basin scale in USEtox.

3.2. Water resource footprint

A first estimate of water stress deprivation in the EU yielded 15 % of the total water demand (total water deprivation 10 km3 divided by total demand 65 km3) (Fig. 4A, B). SDP amounted to 6 km3 in 2010 of blue water shortage, representing 60 % of the total freshwater deprivation (Fig. 4B). Nonetheless, 40 % of the potential deprivation was caused by to pollution effects (PDP = 4 km3). Overall, scarcity caused 80 % of agriculture sector deprivation but <20 % of industry and domestic sectors deprivation (Fig. 4B). High PDP for domestic and industry sectors is consistent with more stringent water quality requirements here than for water for irrigation (Fig. 4B) (Boulay et al., 2011). The agriculture sector was the largest contributor to SDP and PDP (Fig. 4C). Where intensive agriculture takes place, the high demand for irrigation leads to SDP, and use of pesticides and fertilizers cause PDP that, in turn, limits the access to water for all sectors, including other farmers (western France, Spain, Italy, Croatia, Greece, Fig. 4A,D). Therefore, it is relevant to include pollution in sectoral water stress studies. Nonetheless, the agriculture sector is most affected by water stress (SDP) in absolute volume (Fig. 4B). In contrast, the industrial sector is the most deprived in relative terms (19 % of the demand) (Fig. 4B). Comparing our WRF map with the water stress assessment for the year 2018 by the UN (FAO and UN Water, 2021), our indicator captures well low and medium water stress regions in Southern France, Italy, and Spain and pinpoints regions in Greece that were assess as not stressed. Yet, the WRF fails to identify medium water stress regions in Germany and low water stress regions in Spain, central and eastern Europe. By design, our WRF pinpoints only regions where water demand exceeds availability in at least one month of the year (Fig. 2), which may lead to a different assessment from the water stress index used by the UN. Overall the WRF is biased toward medium water stress regions.

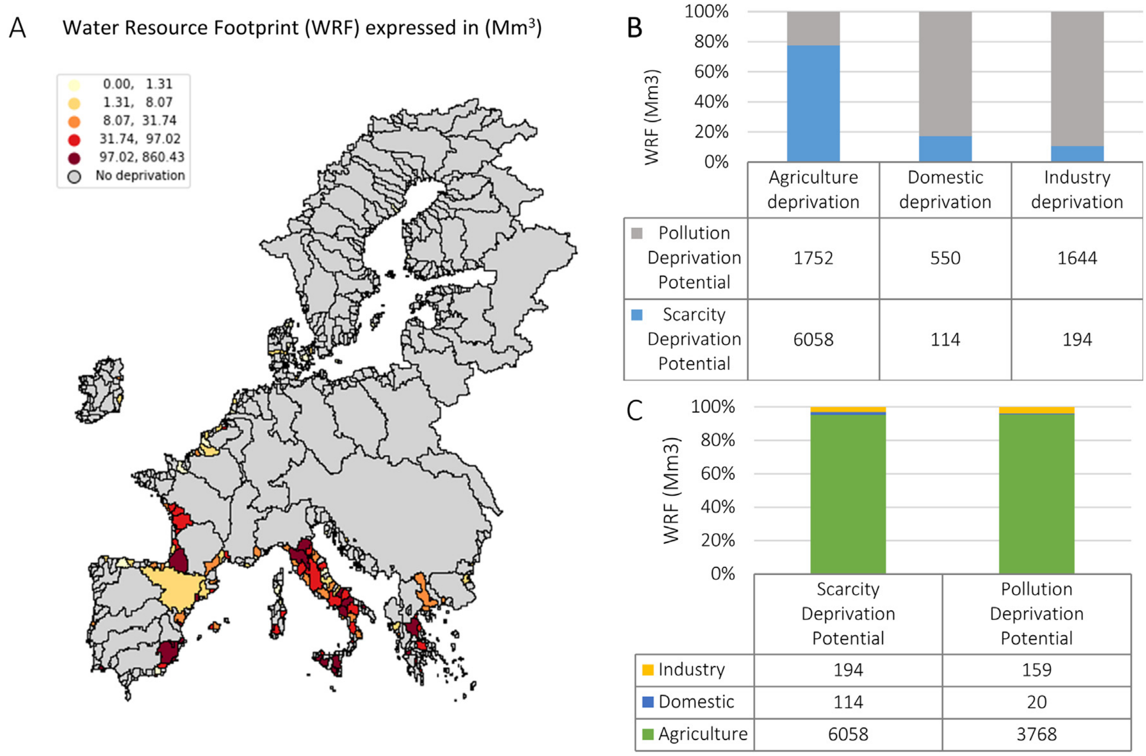


Fig. 4. Map of the water resource footprint where the colour scale corresponds to quantiles (number of river basins) (A); contribution of water pollution scarcity to sectorial deprivation (B); and contribution analysis of the industry, the agriculture and the domestic sector (C).

3.3. Sustainability assessment and impact mitigation strategy

Overall, 93 % of the EU landmass corresponded to ecologically and socially sustainable basins. WBF and WRF exceeded sustainability limits for 5 % (base scenario) and 8 % of the EU landmass. Combining the area of unsustainable WBF and WRF, the sustainability single score of the EU is 8 % (Eq. S22). The map of unsustainable basins for the WRF (Fig. 5B) resembles the map of impacts (Fig. 4A) because the sustainability limit is aligned with the domestic water demand (Section S2). Indeed, the domestic sector is more likely to be deprived than other sectors due to the high water-quality requirements and the assumption that water shortage affects all sectors. To bring impacts below sustainability limits, WBF and WRF impacts should be reduced by 80 % and 14 %, respectively. Adopting the principle that "polluters pay", the industry should bear 64 % of the ecosystem preservation effort (agriculture 20 %, domestic 17 %), while agriculture should assume 77 % of resource preservation efforts (industry 12 %, domestic 11 %) in unsustainable river basins.

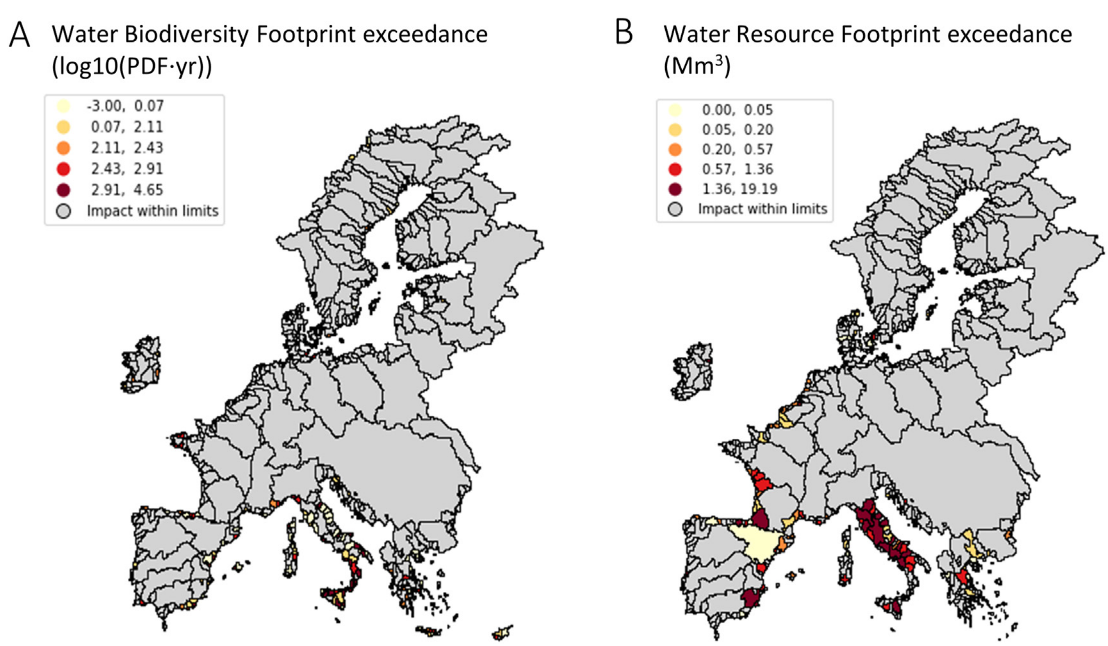


Fig. 5. Map of impact exceedance for ecosystems (A) and freshwater resources (B).

Table 2A
Sensitivity of the sustainability assessment results to the sustainability limit definition for the water biodiversity footprint (WBF) considering species extinction sequences.

Sensitivity on species selectivity on SL_{WBF}	Producer species extinctions	Consumer species random extinctions	Change (%)
Sustainability limit WBF (local PDF.yr)	0.04	0.14	+250 %
Percentage of the area exceeding SL_{WBF} (%)	4.67 %	5.67 %	+20 %
Total WBF exceedance (global PDF.yr)	0.85	0.85	0 %

3.4. Sensitivity analysis to value choices in the WF sustainability assessment

Analyzing the WBF impact exceedances in the different scenarios (Tables 2A-2C) we identify parameters of the WF sustainability assessment that can influence the results and therefore, the decision-making. We obtained similar results when considering that primary producers are extinct first (SL_{WBF} = 0.04 PDF.yr) and when random producers go extinct (SL_{WBF} = 0.14 PDF.yr), which indicate skewed results for WBF and a moderate sensitivity when considering global species extinction in the sustainability assessment (Table 2A). Nonetheless, the spatial uncertainty on SL_{WBF} is key to analyze because, if SL_{WBF} were 0.004 PDF·yr (instead of 0.04) then the area of unsustainable basins in the EU-27 area would double (Table 2B).

Changing the "polluters pay" sharing principle for the "gross added value" influences greatly the distribution of responsibilities to reduce impacts (Table 2B). In that case, the industry should reduce even more their impacts on freshwater ecosystems (+39 %) while the agriculture would do much less efforts (−90 %) than with the "polluters pay" principle (Table 2C).

4. Discussion

4.1. Relevance to decision making

The harmonized WF assessment framework (Table 1) can support decisions in product development and water management (Lathuillière et al., 2018) because results can be aggregated per water user (e.g., for a facility), sector, and river basin. For example, from a product-development perspective, the contribution analysis (Fig. 2B, Fig. 3C, D) shows the impacts of specific sectors (e.g., the industry) across Europe, while the maps (Fig. 2) highlight the regions where impacts are most intense. Moreover, the proposed WF impact and sustainability assessments bring additional information to support the policy decision stage (Table 1). We illustrate each contribution using examples from the proof of concept (Section 3).

First, including multiple pollutants improves the WF impact assessment accuracy. Impacts of nutrients and chemicals are modelled with specific LCIA methods for each environmental mechanism. Hence, WBF with multiple pollutants is more accurate despite the uncertainty of toxicity models (Rosenbaum et al., 2008). Moreover, the sectorial water quality requirements include many quality parameters, i.e. 169 in the guidelines we used for the proof of concept (Boulay et al., 2011). Therefore, including multiple pollutants allows distinguishing PDP for each sector, which is relevant because impacts on human health (and the economy) caused by domestic, agricultural, or industry water deprivation are different.

Second, the WBF and WRF indicators enable solving trade-offs between pollution and scarcity impacts. For example, impacts of water stress are minor compared to toxicity (and eutrophication) impacts on ecosystems in most river basins (Fig. 3A). Moreover, WRF models the competition for water resources, and sectorial PDP and SDP can be compared (Figs. 4, S3). Thus decision-makers can manage water resources knowing if scarcity or pollution dominates water stress. Eventually, the WF sustainability assessment stage can help prioritize WBF or WRF in specific river basins when one sustainability limit is exceeded. For example, actions to reduce SDP (i.e. water consumption) should be prioritized in the Ebro River (Northern Spain) because WRF exceeds the sustainability limit while WBF does not (Fig. 5). Theoretically, WRF and WBF could also pinpoint trade-offs between consuming water from surface water, groundwater and soil (green water) because the scarcity indicators include impacts on each water source separately (e.g., in Fig. S2).

Third, the WF sustainability assessment adds to the WF impacts and WF productivity assessment in the prioritization of economic resources between regional actions. Actions supporting sustainable water use should be directed toward the most unsustainable basins first. For example, Italy appears as one of the countries with the highest WBF and WRF in the EU-27 (Figs. 3A and 4A). Water productivity improvement should start with the unsustainable basins where the exceedance for WRF and WBF is the highest, e.g., on the Adriatic coast (Fig. 5). Because the WF sustainability assessment is holistic, the WF impact hotspot of the product system may point to other regions than the WF sustainability assessment. Nonetheless, river basins where impacts are assessed as sustainable should not be excluded from the action plan due to the uncertainty in the impact assessment and on the sustainability limit definition (Section 4.3). Refined hotspot analysis with impact exceedances maps (e.g. Fig. 5) are relevant to both water managers and industry decision-makers. For example, it allows for identifying unstainable elements of an existing supply chain or guide the development of one to ensure its sustainability.

Fourth, the shared impact exceedance for WRF and WBF (Fig. 5) could be used by decision makers to set sustainability targets in multi-annual water management plans, e.g., revising water allocations or redefining emission limits for specific users. For this, comprehensive LCI data is warranted. An unsustainable basin can revert to sustainable if the impact exceedance goes down to zero. Paths toward that goal can be delineated by identifying opportunities to increase water use efficiency. Nonetheless, the exceedances are null when the overall impacts are below the sustainability limits, which entails that the WF assessment does not prescribe quantitative impact reduction targets or new water allocations. Moreover, the WF sustainability assessment does not prescribe how much water each sector should use to maximize productivity. It only sets the "social foundation" and the "ecosystem ceiling" for sustainable water use, and

Table 2B
Sensitivity of the sustainability assessment results to the sustainability limit definition for the water biodiversity footprint (WBF) considering spatial variability.

Spatial sensitivity on SL_{WBF}	Higher	Lower	Change (%)
Sustainability limit WBF (local PDF.yr)	0.04	0.004	−90 %
Percentage of the area exceeding SL_{WBF} (%)	4 %	11 %	+175 %
Total WBF exceedance (global PDF.yr)	0.85	0.86	+1 %

Table 2C
Sensitivity of the sustainability assessment results to the sharing principle assuming a high sustainability limit (0.04 PDF.yr) for the water biodiversity footprint (WBF).

	Polluters pay (average)	Gross added value	Change (%)
Ratio of the total exceedance for the sharing principle S^k_{WBF} (%)			
agriculture	20 %	2 %	NA
industry and services	64 %	88 %	NA
domestic	16 %	10 %	NA
Shared WBF exceedance among sectors E^k_{WBF} (global PDF.yr)			
agriculture	0.17	0.02	−90 %
industry and services	0.54	0.76	+39 %
domestic	0.14	0.09	−37 %

Obs. The gross added value ratio of the domestic sector is calculated as the difference between the total GDP and the GDP of the agriculture, services, and industrial sectors. The GDP of the industry and services sectors were grouped. Source for the Gross Added Value sharing principle: (The world bank, 2021).

anything in between would be considered acceptable (Raworth, 2012). Therefore, the WF assessment does not substitute productivity, economic, and political decisions about water use allocation (Biswas, 2008).

4.2. Interpretation and potential application of the water resource footprint

To date, there is no consensus in the LCIA community on modelling damages to natural resources, unlike the damage modelling for ecosystems quality. Hence, we focus on modelling impacts to freshwater resources in the following. The impact to freshwater resources could represent the long-term (e.g., 100 years) loss of freshwater availability due to groundwater depletion and persistent pollution (Pradinaud et al., 2019). However, the multi-annual perspective may ignore intra-annual water stress issues and appear less appropriate to support water allocation decisions. There is therefore a multi-temporal perspective to consider when assessing damages to freshwater resources. Previously, domestic and agricultural water scarcity and human toxicity models have been used in LCIA models assessing the impacts of water use on human health (Debarre et al., 2022; Motoshita et al., 2018; Rosenbaum et al., 2008). The agricultural and domestic SDP and PDP may or may not result in human health damage depending on the adaptation capacity. Thus, the WRF overlaps partially with these models. The WRF components should be used with care in the context of LCA to avoid double counting issues when quantifying damages to human health and natural resources.

The WRF is a resource functionality impact indicator that represents the seasonal competition for freshwater. It shows how water use reduces freshwater availability for all users. The WRF bridges both the product and the territorial perspectives because WRF_k depends on the total pressure of all facilities on water resources in the river basin (Eq. (2)). Some authors have argued that a consistent water impact indicator should not reflect the impacts of other water users (Hoekstra, 2016, 2017). This view omits the feedback between water use and availability explained in Section 4.2.1. Moreover, managing water allowances efficiently can benefit from understanding the detail of water deprivation (what causes it and which sectors are deprived) because it becomes possible to model the benefit of allocating water to one user versus the loss of productivity due to the deprivation of another water user. Another concern regarding an *impact-oriented* WF has been that scarcity-weighted indicators are difficult to interpret (Hoekstra, 2016, 2017). In contrast, the proposed WRF corresponds to a fraction of the monthly water demand in the river basin, which is straightforward. Moreover, PDP and SDP have the same nature i.e. they represent water demand volumes. In contrast, the blue, green, and grey WF from (Hoekstra et al., 2011) are less comparable because the grey WF is a pollutant assimilation volume while the blue and green WF are freshwater consumption volumes.

The WRF intrinsically proposes an anthropocentric, instrumental perspective on freshwater availability, thus it is a relevant candidate for measuring impacts on natural resources, ecosystem services, and socio-economic assets in the LCIA framework (Verones et al., 2017). Humans adapt to water shortage with technology, infrastructures, and importation of water-intensive goods; or, if adaptation is insufficient, deprivation may lead to human health damage (Debarre et al., 2022; Motoshita et al., 2018). Freshwater consumption impacts on freshwater natural resources have been modelled in LCIA as the cost of adaptation, e.g., the cost of desalination to compensate for the water shortage (Pfister et al., 2009). The proposed PDP may be useful to specify better the cost of adaptation to water quality degradation. Moreover, the sectorial SDP (i.e., SDP_j^b), the river basin PDP, and the river basin WBF can be identified with some relevant freshwater ecosystem services in the river basin i.e., provision of water for agricultural, domestic, industrial water use, self-purification, and biodiversity support (Rinke et al., 2019). Depending on the reason for freshwater deprivation, the adaptation cost (in $\$/m^3$) can be combined with the water deprivation volumes (m^3) to obtain a damage score on freshwater resources in monetary terms ($) following a similar approach to Cao et al. (2015) for land use damage to ecosystem services. Combining the deprivation volumes with the cost of missed opportunity (e.g., loss of crop yield), the components of the WRF may be used to calculate monetary damages to socio-economic assets in LCA (Verones et al., 2017).

4.3. Defining sustainability limits and sharing principles

The choice of the sustainability limits in river basins can change the results, especially for WBF (Section 3.4). More research is needed to determine regional SL_{WBF} reflecting the local ecosystem configuration and to integrate food web models with LCIA, e.g., specifying the selectivity of each impact pathway toward specific functional traits. Such regional values should ideally be calculated for the pristine ecosystem. Nonetheless, there will probably be little regional data about the species composition in the pristine ecosystem. More pragmatically, the current ecosystem could also be taken for reference, assuming that it is in equilibrium with the current state of background pressures. Similarly, we proposed in Section 0 an ideal SL_{WRF} equal to the irrigation and domestic water for food and hygiene (Motoshita et al., 2020). However, the irrigation water requirement for food depends on water productivity and food import. Thus, the SL_{WRF} should also be regionalised to reflect location, time, and technology choices.

The sharing principle was also a key value choice in the WF sustainability assessment. Contrary to the "polluters pay" sharing principle, the gross added value is not necessarily correlated to water use or impacts and therefore this sharing principle might result in unfeasible impact reduction targets. Hence other sharing principles are relevant to explore. For instance, the WF productivity could be combined with the "polluters pay" so that water inefficient and impact-intensive unit processes would receive a larger share of the impact exceedance (Vanham and Mekonnen, 2021). The choice of the sharing principle is particularly important, because the perceived fairness of the cost distribution associated with reducing the impact exceedance may influence greatly collective action toward sustainable water use (Boyd et al., 2018).

All in all, the choice of the sustainability limit value should result from a consensus around an acceptable impact, e.g., how many species is it tolerable to lose to satisfy human demand? What is an essential need for humans? Prioritising human welfare or ecosystems is an ethical choice because the WBF and WRF impact exceedances are not comparable (different units). For example, should we first reduce scarcity that mostly affects humans or reduce pollution that primarily affects ecosystems (Section 3.1)?

4.4. Limitations and future research needs

The proof of concept put forward several caveats of the WF assessment, i.e., LCI regionalization and completeness requirements, LCIA of green water consumption, incomplete impact pathway coverage of WBF, and the resulting uncertainty of WF sustainability assessment.

The proposed water footprint assessment is data intensive. The proof of concept results showed that the poor spatial and temporal resolutions of LCI data hampers the hotspot mapping. For example, the case study failed to identify toxic impact hotspots in Northern Germany because emissions were assumed to be uniformly distributed over the country for lack of better information (Section 3.1). WBF is dominated by ecotoxicity impacts from industrial and urban emissions which are likely unevenly distributed in space but the regionalization of the LCI from country-scale to basin scale ignored the distribution of point source emissions relative to the basins. Moreover, the impacts in basins such as the Danube River, Rhine River, Rhone River basin are underestimated because the emissions from non-EU countries were not included (e.g., Bosnia, Macedonia, Swizerland).

The water quality requirement and the LCI chemical coverage influenced the WF impact assessment results in the proof of concept. For instance, contaminants dissolved in stormwater were not included despite a significant ecotoxicity impact related to metals such as copper (Brudler et al., 2019). In addition, only 59 out of 151 chemicals could be considered in the water quality assessment due to the lack of LCI data for the remaining chemicals for which concentration limits exist (Boulay et al., 2011). Other data gaps prevented a complete chemical coverage in WBF. Moreover, we did not account for other chemicals for which we had LCI data (321 substances) due to the lack of specific concentration limits in the water quality requirement (Boulay et al., 2011). Finally, due to the lack of characterization factors, we only assessed the ecotoxicity impacts (WBF) of 380 chemicals while LCI data for >1000 chemicals were available. Therefore, overall PDP and toxicity WBF are very likely to be underestimated in the proof of concept.

WF impact assessment accuracy may increase by using higher spatial resolution and temporally differentiated LCI data (at least at the river basin scale). Global geospatial models estimating environmental concentrations in freshwater dynamically (surface and groundwater) would also be helpful to refine the impact assessment of pollution and support achievement of the UN Sustainable Development Goal for water (SDG 6), especially in data sparse regions (Hofstra et al., 2019). We believe that these models would help, but obstacles are preventing their use. First, they are complex to operate; and second, they are specialized for one type of emissions (e.g., faecal coliforms, nutrients, toxic chemicals, and temperature). To support the WF assessment, we would ideally need software to integrate all substances transported in soil, air, surface water and groundwater, including water consumption.

Green water consumption impacts on WRF and WBF could not be quantified in the proof of concept. Indeed, LCIA models are currently lacking for quantifying terrestrial and freshwater species loss due to green water consumption (changing evapotranspiration). Understanding the consequences of blue and green water consumption on evapotranspiration rates, precipitation patterns, streamflow, and soil moisture would help develop such LCIA models (Link et al., 2021; Pierrat et al., in review). In future research works, the impact of green water consumption on freshwater ecosystems may be derived from existing LCIA models relating freshwater species loss with streamflow change (Pierrat et al., 2023) combined with models quantifying the consequences of evapotranspiration change on streamflow (Link et al., 2021).

Given the multiplicity of pressures affecting freshwater ecosystems, the total WBF in the river basin is likely underestimated, i.e. representing the combined impact of the background pressure and water use (as mentioned in Sections 2.2.2 and 2.3). This causes the absolute sustainability assessment of WBF to end up too optimistic even though the safety coefficient used in SL_{WBF} may buffer part of this uncertainty. For this reason, we recommend that the WF sustainability assessment should be used to prioritize sustainability and water use efficiency actions, but these actions should not be restricted to the unsustainable basins. The decision-making stage should also consider sustainable basins with low water efficiency and high impacts.

5. Conclusions

We further developed the impact assessment and sustainability assessment stages of the harmonized water footprint assessment proposed by (Lathuillière et al., 2018) and tested the revised framework on the EU-27 river basins. The main contributions to water use LCIA and water footprint assessments are that (i) the introduced Water Resource Footprint and the Water Biodiversity Footprint quantify the impacts of water use on freshwater availability to humans and ecosystems, (ii) we propose a practical approach to regionalize the impacts of multiple chemical emissions and combine them with the impacts of water consumption, and (iii) the WF sustainability assessment verifies whether water use impacts are small enough to ensure human welfare and ecosystem stability in river basins. The Water Resource Footprint represents the competition for water use, which is useful to allocate water efficiently between users. The proof of concept showed that the new framework helps pinpointing trade-offs between pollution and scarcity impacts. For example, the damage to ecosystems stemmed for >99 % from ecotoxcity impacts while scarcity only represented <0.01 %. In contrast, water scarcity caused 62 % of human water deprivation. The maps of impact exceedances pinpoint regions where water impacts are environmentally and socially unsustainable with a few river basins in southern Europe where impacts exceeded both thresholds. We find therefore that including pollution in water footprinting is relevant because it has often higher impacts on biodiversity than scarcity and reduce water availability to sectors with high water quality requirements.

The environmental relevance of the results can be expected to increase when new Life Cycle Impact Assessment models and comprehensive regionalized Life Cycle Inventory data become available. Further improvements require research, particularly in the field of LCIA, with for example the regionalization of pollutant transport models, development of characterization factors for green water consumption impacts on ecosystems and freshwater availability, and the spatially-differentiated determination of sustainability limits for freshwater ecosystems. Finally, future water footprint research should also investigate how to integrate the relationship between water productivity, water demand, and human water needs. Indeed, satisfying human needs for food and energy may require different amounts of water depending on productivity. Nonetheless, the proposed water footprint assessment is considered a step forward to supporting the implementation of water-related sustainable development goals.

Code availability statement

The code used to preprocess the data and generate the results of the proof of concept is available at: https://github.com/EleonorePS/water-footprint-assessment. The data that support the findings of this study were derived from resources available in the public domain and in scientific literature as described in the methods.

CRediT authorship contribution statement

EP conceptualized the study. All authors contributed the methodology development. EP has developed the dataset for the proof of concept, calculated the results, and prepared the visualizations. EP wrote the original draft. All authors have contributed to reviewing and editing the draft. EP did the project management. MZ, AL, MR, FV, MDo did the supervision.

Declaration of competing interest

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

Acknowledgements

EP acknowledges that this study was supported by the Fraunhofer-DTU project EDES (eco-design Stewardship), advancing eco-design excellence, supporting the aeronautics European - Clean Sky Programme.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2023.161910.

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