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The environmental cost of a reference withdrawal from surface waters: Definition and geography

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ABSTRACT

World freshwater ecosystems are significantly deteriorating at a faster rate than other ecosystems. Water withdrawals are recognized as one of the main drivers of growing water stress in river basins worldwide. Over the years, much effort has been devoted to quantify water withdrawals at a global scale; however, comparisons are not simple because the uneven spatiotemporal distribution of surface water resources entails that the same amount of consumed water does not have the same environmental cost in different times or places. In order to account for this spatiotemporal heterogeneity, this work proposes a novel index to assess the environmental cost of a withdrawal from a generic river section. The index depends on (i) the environmental relevance of the impacted fluvial ecosystem (e.g., bed-load transport capacity, width of the riparian belt, biodiversity richness) and (ii) the downstream river network affected by the water withdrawal. The environmental cost has been estimated in each and every river section worldwide considering a reference withdrawal. Being referred to a unitary reference withdrawal that can occur in any river section worldwide, our results can be suitably arranged for describing any scenario of surface water consumption (i.e., as the superposition of the actual pattern of withdrawals). The index aims to support the interpretation of the volumetric measure of surface water withdrawal with a perspective that takes into account the fluvial system where the withdrawal actually occurs. The application of the index highlights the river regions where withdrawals can cause higher environmental costs, with the challenge of weighting each water withdrawal considering the responsibilities that it has on downstream freshwater ecosystems.

1. Introduction

Freshwater resources are necessary for sustaining societal and economic activities and are essential to both aquatic and terrestrial ecosystems. The Millennium Ecosystem Assessment (2005) points out that freshwater ecosystems have been deteriorating consistently and at a faster rate than other ecosystems: the freshwater species (included in the Living Planet Index) declined on average by 50% between 1970 and 2000, compared to an average decline of 30% for both terrestrial and marine species over the same period. Moreover, aquatic habitats associated with 65% of global river discharge were classified as under moderate to high threat (Vörösmarty et al., 2010).

Human activities have considerably influenced most rivers of the world with respect to their habitat, water quality, river morphology and flow regimes (Dudgeon et al., 2006). Globally, freshwater withdrawals increased approximately sevenfold in the past century (Gleick, 2000) with irrigation accounting for around 70% of the total freshwater withdrawals (FAO, 2011); moreover, the expected economic and population growth in developing countries will steadily increase the exploitation of freshwater resources (Alcamo et al., 2007; Godfray et al., 2010; Vörösmarty et al., 2000). Water consumption has been highlighted as one of the main mechanisms that increases the intensity and frequency of hydrological droughts worldwide (Wada et al., 2013). Rising water withdrawals have been pointed out as the principal cause of increasing water stress on many river basins worldwide. In several regions, the growing water withdrawals are expected to have more consequences on fluvial ecosystems than climate change (Alcamo et al., 2007; Haddeland et al., 2014; Hanasaki et al., 2013). This picture highlights that meeting the competing water requirements of ecosystems and societies is a key global environmental challenge for both scientists and governments (Dudgeon et al., 2006).

In recent years, numerous studies have attempted to quantify the amount of water used to produce different commodities (e.g., Rost et al., 2008; Hanasaki et al., 2010; Liu and Yang, 2010; Siebert and Döll, 2010; Mekonnen and Hoekstra, 2011; Tuninetti et al., 2015). The concept of water footprint allows one to quantify the volume of water

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used to produce a good (Hoekstra and Mekonnen, 2012). Through the water footprint it is possible to examine the linkage between the consumption of any good and the water resources exploited. However, the accounting of the water volumes is not the only possible approach. It has been argued that to support decision-making, the use of the water footprint alone can be misleading without additional impact-oriented interpretations (Berger and Finkbeiner, 2013; Yano et al., 2015). In fact, the uneven distribution of freshwater resources implies that the same volume of water consumed does not have the same environmental consequences in different periods or places. In order to address this key issue, the purpose of our work is to evaluate the different environmental cost that a reference amount of surface water withdrawn has on different freshwater ecosystems, where the term ‘environmental cost’ refers to the degree of impact of water withdrawals on river systems, not related to economic estimates.

Our work aims to develop an effective and easy-to-apply tool for quantifying the environmental cost of any scenario of water consumption. The key point of our approach is to first consider a potential (fictitious) withdrawal corresponding to a unitary amount of surface water. We assume that this potential withdrawal occurs in a given river section, and we evaluate its environmental cost, which will depend on the specific river section considered. This procedure is repeated for every river section in the global hydrographic network and the global geography of the environmental cost associated with single unitary withdrawals is described. The environmental cost corresponding to a real pattern of withdrawals (or to possible future scenarios) is finally obtained as the linear spatial combination (i.e., as the superposition) of the environmental costs due to the unitary withdrawals. In our approach, a key issue is the evaluation of the environmental cost of the unitary water withdrawal. Since fluvial ecosystems are complex environments that exhibit a high number of interplaying processes, we designed an index able to manage such complexity in a parsimonious way, to allow for an easy evaluation of the impact of water consumption on fluvial ecosystems. The index is designed to allocate to the water withdrawal an environmental cost which accounts for the whole impacted ecosystem. To this aim, the proposed index considers: (i) the impacts that the reduction of the river flow causes on specific fluvial characteristics (e.g., the transport and dilution of chemicals, the width of riparian belt, the fish species richness), through suitable non-linear relations with discharge, and (ii) the whole portion of river network impacted by a withdrawal, that is from the section where water is withdrawn down to the river mouth. Thus, the environmental cost associated to a withdrawal does not depend exclusively on the status of local water resources, but it considers the downstream conditions as well. The downstream propagation of an hydrological stressor (e.g., a water withdrawal) was also accounted by Vörösmarty et al. (2010), who considered a number of drivers of stress (e.g., dam density, water consumption, livestock density) with the objective to study the current environmental status of the global river systems. More recently, these results were exploited to map the capacity of upstream freshwater provision areas to supply water for human populations downstream (Green et al., 2015). In the present work, instead of focusing on a specific scenario of water withdrawing, we propose a general approach that allows one to investigate and compare the environmental cost of any consumption pattern. The environmental cost of a reference withdrawal is computed globally with a 0.5° spatial resolution and using undisturbed river discharges (i.e., unaffected by human activities), which are obtained from the natural scenario of the WaterGAP model (Döll et al., 2014; Müller Schmied et al., 2014, 2016; Müller Schmied, 2017). The results obtained in this work are assessed considering an yearly average undisturbed river discharge, but overall the index proposed can be used with any spatial or temporal resolution depending on the available data and the application purpose.

Our work shares some ground with indices and metrics developed to quantitatively evaluate the average pressure on water resources by human activities. A frequently used measure of water stress is the Falkenmark indicator, which is equal to the annual water availability per person (Falkenmark, 1989). Raskin et al. (1997) estimated water stress at a country scale as the ratio of annual water withdrawals to the annual renewable water resources, defining as severely water scarce those countries having annual withdrawals greater than 40% of their annual available water resource. Smakhtin et al. (2004) developed a Water Stress Indicator (WSI) at the basin scale as the ratio of annual withdrawals to annual utilizable water, where the utilizable water is the difference between the total water available in the basin and the environmental water requirements. In recent years, also thanks to new global hydrological models (e.g., WaterGAP, PCR-GLOBWB, H08), different studies have adopted water stress indices with higher spatio - temporal resolution in order to identify water stress areas in current, past or future conditions (Alcamo et al., 2007; Hanasaki et al., 2013; Mekonnen and Hoekstra, 2016; Wada et al., 2011). Recently, also the historical development of water scarcity was investigated by Kummu et al. (2016). Overall, all these studies share the volumetric comparison of local resources and withdrawals (in Falkenmark (1989), withdrawals are proportional to population); withdrawals and water resources vary from region to region, and stress (or water scarcity) occurs when withdrawals significantly affect the available water resources. Therefore, the water stress measure is strictly related to local human water needs: the availability is affected by upstream withdrawals, but if locally there is no water consumption the region is not classified under water stress, even when the local river environment exhibits a strong impact due to upstream withdrawals. Conversely, our work focuses on the environmental consequences of a withdrawal and proposes an index that: (i) is designed to consider a withdrawal which can occur anywhere along a watercourse; (ii) depends on the impacted fluvial environment and not on socio-economic needs; and (iii) does not make a volumetric comparison between available and consumed water, but attempts to express the non-linear relation between water withdrawals and their impacts on river systems.

2. The index to assess the environmental cost of a surface withdrawal

2.1. Evaluation of the environmental cost per unit length

The health of a river ecosystem depends on multiple and complex biotic and abiotic processes. In this work, we focus on the impact related to water withdrawals in order to estimate their environmental cost. The environmental cost measure strictly depends on the impact that a water withdrawal has on river ecosystems. This work proposes an index which is function of (i) the water withdrawal and (ii) the undisturbed river discharge, that is the discharge existing before the river flow reduction due to water withdrawals.

We assume the environmental cost per unit length (ec_w) of a withdrawal in a generic section of the river network to be proportional to the river discharge reduction caused by the water withdrawal, W, in the same section. Thus, the proportion between the environmental cost and water withdrawal is

$$ec_w: W = ec_{max}: Q \qquad \Rightarrow \qquad ec_w = ec_{max} \cdot \frac{W}{Q} \tag{1}$$

where ec_w coincides with the impact of a water withdrawal W and ec_{max} is the maximum possible environmental cost per unit length, which occurs when the entire river discharge, Q, in the river section, is withdrawn (clearly, $W \leq Q$); ec_w has the dimension of an environmental cost per unit length. We model a linear proportion between ec_w and W, neglecting non-linear terms in order to keep the number of parameters at a minimum.

Therefore, ec_w estimates a local impact, which does not take into account downstream effects (i.e., ec_w considers only the impact in the river section where W occurs); in Section 2.3 we will introduce the index that considers the effects of a withdrawal on the whole fluvial system.

2.2. Assessing the maximum environmental cost per unit length

From Eq. (1), the environmental cost per unit length of a water withdrawal, ec_w, can be assessed once the undisturbed river discharge and the maximum environmental cost per unit length are known; thus, a method has to be defined to determine ec_{max}, which refers to the water depletion of the river section and occurs when the withdrawal equals the undisturbed river discharge.

The maximum environmental cost is strictly related to the relevance of the fluvial environment considered. However, such relevance cannot be reduced to a single number: a fluvial environment is characterized by multiple and interplaying processes and, thus, its environmental relevance is a multidimensional concept as well. It follows that the value of ec_{max} in a river section depends on the specific perspective adopted. A possible choice is to assume that the importance of a fluvial system linearly depends on the discharge of the considered river; other possible choices are to consider the fluvial biodiversity or the sediment transport capacity of the river. Overall, many fluvial characteristics can be related to the river discharge through suitable power-law relations (see Appendix) that lead to estimate ec_{max} as

$$ec_{max} = k(\alpha)\cdot Q^{\alpha} \tag{2}$$

where α is a parameter that typically varies between 0 and 1 (see Appendix) and $k(\alpha)$ is a proportionality constant, which is the same in all the river sections once the value of the parameter α has been defined.

Specific values of the parameter α correspond to the perspective chosen (e.g., the width of the riparian belt, the habitat richness, the dilution capacity, etc) to evaluate the relevance of a fluvial environment. In the Appendix, some emblematic cases are described and the corresponding values of α fall in the range [0 1].

The two limiting cases $\alpha = 0$ and $\alpha = 1$ embody the range of perspectives wherein ec_{max} moves. When $\alpha = 0$ all fluvial systems have the same environmental relevance everywhere independently of the discharge, namely the depletion of a large river has the same environmental cost as the depletion of a small stream. Conversely, when $\alpha = 1$, ec_{max} turns out to be directly proportional to the discharge flowing in the considered river section; with this formulation, the depletion of a large river has more impact than depletion of a small steam. Overall, the formulation of ec_{max} permits a change in the perspective adopted to determine the relevance of a river system, in line with the study targets.

A reasonable constraint is that the environmental cost of withdrawing all the world's surface water resources is unaffected by α and equal to a constant value, EC_{world}. If all the global surface water resources were consumed, the overall environmental cost EC_{world} would be equal to the summation of the maximum environmental cost per unit length along all river sections worldwide. Using Eq. (2), EC_{world} can be evaluated as

$$EC_{world} = L_w\cdot\int_{all\,Q} ec_{max}(Q)\cdot p(Q)\cdot \mathrm{d}Q, \tag{3}$$

where $p(Q)$ is the probability density function of the undisturbed river discharge, determined along all river streams in the world, and L_w is the total length of the world river network. Both are evaluated at a detail related to the scale of interest. Thus, EC_{world} is equal to the expected value of the maximum environmental cost per unit length multiplied by L_w. Substituting Eq. (2) into Eq. (3) and exploiting the fact that EC_{world} does not depend on α, $k(\alpha)$ can be assessed as

$$k(\alpha) = \frac{EC_{world}}{L_w\cdot\int_{all\,Q} Q^{\alpha}\cdot p(Q)\cdot \mathrm{d}Q}. \tag{4}$$

Therefore, Eqs. (2) and(4) allow EC_{world} to be subdivided among the different river sections according to the adoption of a specific environmental perspective (i.e., through the designation of the parameter α). Although EC_{world} is constant, the environmental significance of a specific river section (ec_{max}) can significantly change with α (see Fig. S1).

By introducing Eq. (2) into Eq. (1), the environmental cost per unit length of a water withdrawal, W, in a generic river section becomes

$$ec_w = k(\alpha)\cdot\frac{W}{Q^{1-\alpha}} \tag{5}$$

We note from Eq. (5) that there is a power-law relationship among the ec_w values calculated using different values of α. This relation is expressed as

$$ec_w(\alpha_1) = b_1\cdot ec_w(\alpha_2)^{b_0} \tag{6}$$

where $ec_w(\alpha_1)$ and $ec_w(\alpha_2)$ are the environmental cost per unit length values, which are assessed by employing $\alpha = \alpha_1$ and $\alpha = \alpha_2$, respectively; b_0 and b_1 are the power-law coefficients, where $b_0 = (\alpha_1 - 1)/(\alpha_2 - 1)$ and $b_1 = k(\alpha_1)\cdot k(\alpha_2)^{-b_0}\cdot W^{(1-b_0)}$. The relation in Eq. (6) implies that if a unitary reference withdrawal is considered in all the river sections, the ranking of the ec_w values among the different river sections is α-independent. However, the choice of α influences the range of variation of the ec_w values, which is minimum when $\alpha = 1$ and maximum when $\alpha = 0$.

Finally, notice that the proposed approach to define the environmental cost is able to embed different river characteristics at the same time. E.g., all the possible values of α between 0 and 1 can be considered through a kernel function, $Ker(\alpha)$, to weight them. By integrating Eq. (5) between $\alpha = 0$ and $\alpha = 1$ the environmental cost per unit length in this case turns into

$$ec_w = \int_0^1 Ker(\alpha)\cdot k(\alpha)\cdot\frac{W}{Q^{1-\alpha}}\mathrm{d}\alpha. \tag{7}$$

2.3. Evaluation of the environmental cost including the downstream river network

In the previous sections we assessed the environmental cost per unit length, by focusing only on the specific river section where water is withdrawn. However, the actual environmental cost of a water withdrawal also has to consider the impact on the portion of river network downstream to the point where water is withdrawn. In fact, the subtraction of W will alter the discharge in all the river sections from the section where water is withdrawn down to the river mouth (see Fig. 1a).

Therefore, the overall environmental cost of a water withdrawal W has to be evaluated as the sum of the environmental cost per unit length from the section where water is withdrawn, S_W, down to the river mouth. The impact per unit length in a generic downstream section, s, is assessed by employing Eq. (5) using the value of the undisturbed river discharge in s and a water withdrawal equal to W, since in each downstream section the undisturbed river discharge is reduced by the same amount of water that is withdrawn in S_W, namely W.

It follows that we can define the environmental cost, EC_w, of a water withdrawal, W, in a river section S_W of the river network as

$$EC_w = \int_{S_W}^{S_M} ec_w(s)\mathrm{d}s = k(\alpha)\cdot\int_{S_W}^{S_M}\frac{W}{Q(s)^{1-\alpha}}\mathrm{d}s \tag{8}$$

where S_M is the river mouth section and the argument of the integral is the environmental cost per unit length (ec_w) of a water withdrawal (W) in the river section s assessed with Eq. (5). EC_w and EC_{world} have the same units of measure, namely ec_w multiplied by a length.

In this regard, consider the Mekong river case study illustrated in Fig. 1a and b. In Fig. 1a, water consumption (W) affects the river from the section where water is extracted down to the river mouth section. Actually, in the same river a fixed water withdrawal has a higher environmental cost if this occurs in the upper part of the river (see the red line in Fig. 1b). This occurs for two reasons: (i) usually river discharge gradually increases from the spring to the river mouth and the same amount of W, thus, represents a very different share of the available

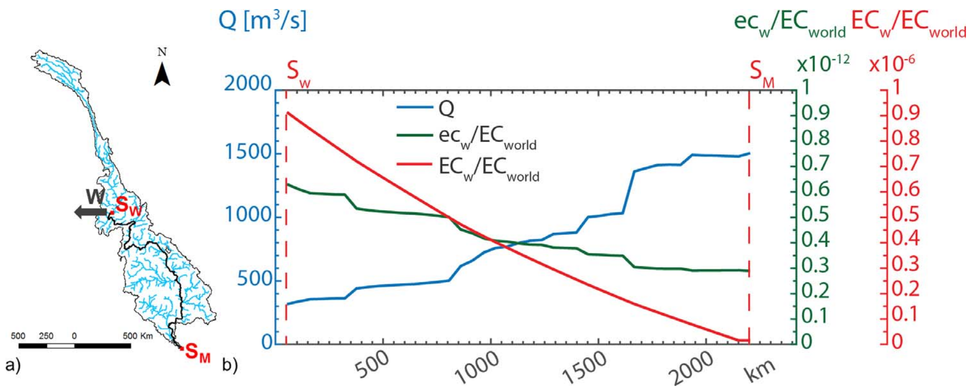

Fig. 1. (a) The example shows the main Mekong river network. A water withdrawal, W, located in a river section, S_W, of the river impacts all the downstream sections (marked with the black line) down to the river mouth following the flow direction along the curvilinear abscissa of the river. (b) Panel shows the average annual Mekong river discharge (Q), the environmental cost per unit length of the Mekong's course (ec_w) and the overall environmental cost in each cell of the Mekong's course (EC_w) from the section S_w down to the river mouth (S_M); both environmental costs are normalized by EC_{world} and are estimated with α = 0.5 and W = 1m^{3}s^{-1}.

water resources (see Eq. (5)); (ii) a water withdrawal impacts all the downstream sections (see Fig. 1a).

As mentioned in the previous section, instead of estimating the environmental cost per unit length by employing a specific value of α, ec_w can be assessed with Eq. (7) and hence in this case the environmental cost of a water withdrawal turns into

$$EC_w = \int_{S_W}^{S_M} \int_0^1 Ker(\alpha)\cdot k(\alpha)\cdot \frac{W}{Q(s)^{1-\alpha}} \mathrm{d}\alpha \mathrm{d}s. \tag{9}$$

In this section, the index has been described with the aid of an example related to a specific river segment (see Fig. 1); more generally, in this work the environmental cost of a surface water withdrawal has been assessed at a global scale as discussed in the following section.

3. Results

The environmental cost of a water withdrawal can be assessed once the river network and the undisturbed river discharge worldwide are known. In this work, EC_w is computed globally with a 0.5° spatial resolution using the global drainage direction map DDM30 (Döll and Lehner, 2002) and the undisturbed river discharges obtained from the pristine scenario of the WaterGAP 2.2c model (Döll et al., 2014; Müller Schmied et al., 2014, 2016; Müller Schmied, 2017). The average annual undisturbed river discharge has been evaluated over the time series 1901–2013. In order to assess the environmental cost EC_w of a water withdrawal in a generic river section, a discretization of the river network is needed. The environmental cost of a water withdrawal can be evaluated by discretizing Eq. (8) accordingly to the rectangular rule as

$$EC_w = \sum_{j=1}^{N} ec_{w,j}\cdot \Delta s_j \tag{10}$$

where $ec_{w,\,j}$ is the environmental cost per unit length, which is evaluated in the sections where the discharge is known, and Δs_j is the distance between sections ($j-1$) and j, which are two consecutive sections where the undisturbed river discharge is known (i.e., where $ec_{w,\,j}$ can be defined). N is the number of sections between the river section where the water withdrawal occurs and the river mouth. In the present study the index has been assessed at a global scale using an average yearly value of the undisturbed river discharge; Δs_j has been estimated as the square root of the area of the cell. The environmental cost in each cell is $ec_{w,\,j}\cdot\Delta s_j$, where ec_w has been estimated employing the undisturbed river discharge (Q) at the outlet of the cell, since the within-cell variations of Q are unknown. The discretization adopted in this work is suitable for the data used here (e.g., integration by trapezoidal rule does not change our results), but more refined discretizations can be adopted if more detailed data are available.

In order to evaluate the environmental value of a reference amount of surface water equal to W, the impact index is assessed considering everywhere a fixed value of W. Since the impact per unit length is estimated for $W \le Q$, setting W implies establishing a threshold of Q below which the environmental cost is not estimated. The reference water withdrawal has been fixed at 1 m^{3}s^{-1}, which can be considered a very small discharge in a cell of 30 × 30 arc min (~ 55 × 55 km at the equator).

An example of the evaluation of the proposed index is given in Fig. 2, which refers to the Danube river's undisturbed discharge behaviour from the spring to the river mouth. The trend of the

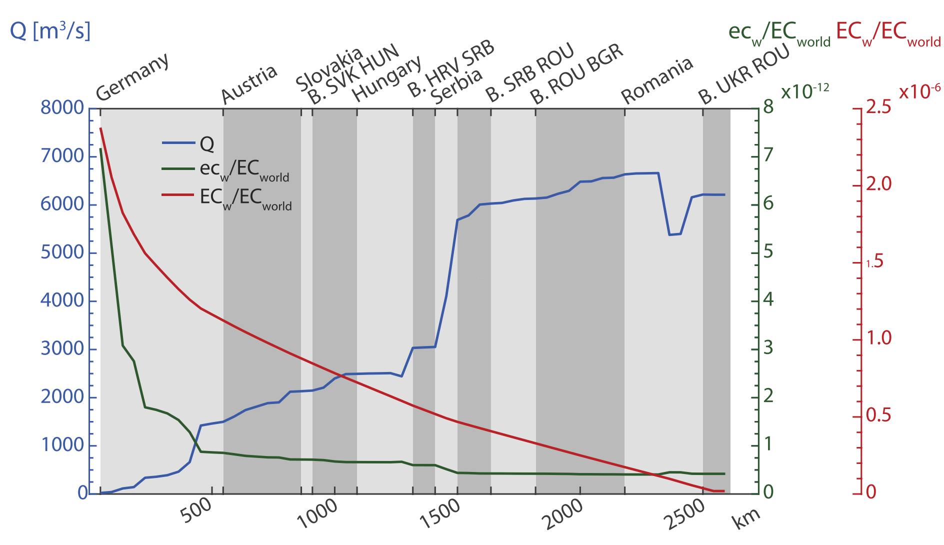

Fig. 2. The average annual discharge (Q) of the Danube river, the environmental cost per unit length (ec_w) at the closing section of each cell of the Danube's course, and the overall environmental cost (EC_w) in each cell of the Danube's course; both impacts are normalized by EC_{world} and estimated with α = 0.5 and W = 1m^{3}s^{-1}. The cells span 30 × 30 arc min (approximately 45 × 45 km at these latitudes). Abscissa reports the distance from the spring of the Danube river. In the top of the figure the countries crossed by the river are shown (the letter *B.* means that the river flows on the boundary between two countries). Close to the river mouth the discharge has a drop due to the Bala–Old Danube bifurcation.

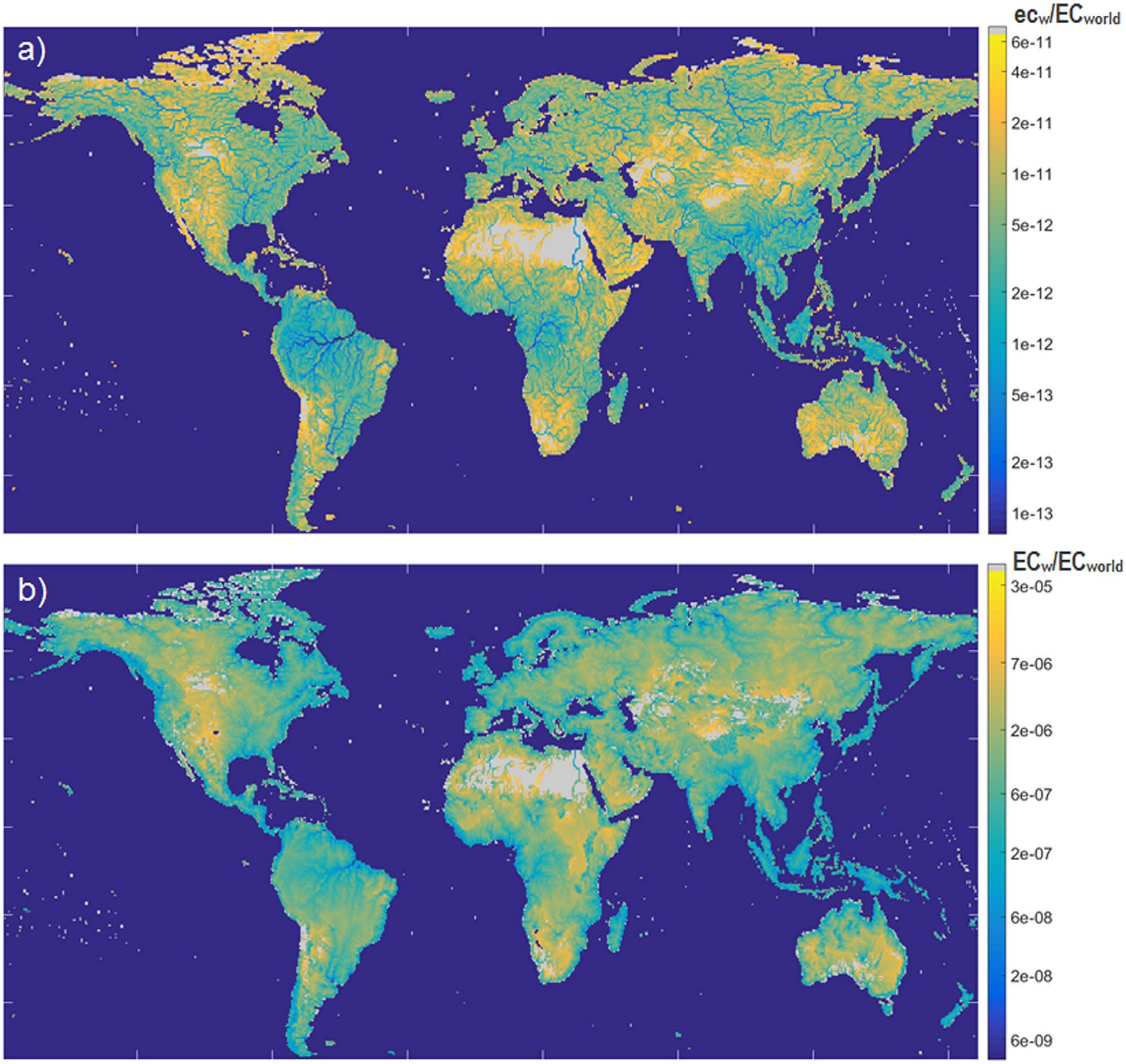

Fig. 3. (a) The environmental cost per unit length, ec_w; (b) the overall environmental cost, EC_w. Both maps are estimated with $\alpha = 0.5$ and $W = 1\text{m}^3\text{s}^{-1}$ in cells of 30 × 30 arc min and the environmental costs are normalized by EC_{world}. The colour bars are in log scale. Grey areas have $Q < 1\ \text{m}^3\text{s}^{-1}$, therefore in those areas the environmental cost is not computed.

environmental cost per unit length, ec_w, is strictly related to the undisturbed river discharge, although the relation between the two is non-linear (see Eq. (5)). The overall environmental cost, EC_w, is assessed applying Eq. (10) in each cell of the Danube's course. Since the Danube discharge increases along the flow direction and EC_w takes into account the discharge-depletion effects on the downstream cells, EC_w gradually decreases from the spring to the river mouth of the Danube river. The cumulative effect accounted in EC_w is highlighted as well observing the dissimilarities between the profiles of ec_w (green line) and EC_w (red line): (i) in the sections where Q sharply increases ec_w promptly decreases, conversely EC_w has a much more smoothed trend than ec_w, as expected by introducing an integral operator; (ii) the range between the maximum and the minimum values of the two impact measures is very different: $6.8{\cdot}10^{-12}$ for ec_w/EC_{world} and $2.2{\cdot}10^{-6}$ for EC_w/EC_{world}.

Clearly, the environmental cost can be analysed not only focusing on a specific river, but also examining ec_w and EC_w at a global scale, as illustrated in Fig. 3a and b, respectively. Both maps are obtained with $\alpha = 0.5$ and $W = 1\text{m}^3\text{s}^{-1}$ (see from Fig. S3 to Fig. S6 in the Supplementary Material for EC_w estimated considering further α values and Fig. S8 for ec_w and EC_w assessed by a $Ker(\alpha)$). The environmental cost per unit length (ec_w), which varies between 0 and $k(\alpha)$, expresses the local environmental cost of a water withdrawal without accounting for the impact on the downstream sections. It follows that the map in Fig. 3a is strictly related to the undisturbed river discharge geography. Since it is not admissible withdrawing more than the available water, in the river sections where $Q < W$, ec_w is not estimated (see grey areas in the maps). Worldwide, the cells having an average yearly undisturbed discharge equal or lower than 1 m^3s^{-1} are approximately 7.5% (grey cells in Fig. 3a), which together involve around 0.007% of the total global discharge; these cells belong to deserts or hyperarid areas. In contrast, most of the cells with ec_w under the 5th percentile (namely the areas that at a global scale have the lowest environmental costs per unit length) are located in the world's largest rivers.

Changing perspective from the local environmental cost (ec_w) to the overall environmental cost (EC_w) and, thus, accounting for the downstream propagation effect of a water withdrawal on fluvial systems, the environmental value of 1 m^3s^{-1} assumes worldwide the values shows in Fig. 3b. In this case, a water withdrawal in a cell having an high value of EC_w implies a significant impact on the overall downstream river course. Generally, the higher EC_w values are located far from the coastline in arid or semi-arid regions because of the combined effect of surface water scarcity and distance from the river mouth. Most of the cells with EC_w under the 5th percentile are located along the world's coastline and in the Amazon River; these are thus the areas that have the lowest environmental cost of withdrawing 1 m^3s^{-1} of surface water.

The downstream propagation effect considered in EC_w implies that in any cell $ec_w \leq EC_w$, where the equality holds only at the river mouth; as a consequence, the global mean value of EC_w is always considerably greater than the global mean value of ec_w for any α. For example, for $\alpha = 0.5, \overline{ec}_w/ec_{world} = 8.32 \cdot 10^{-12}$ and $\overline{EC}_w/EC_{world} = 1.31 \cdot 10^{-6}$, where the overbar indicates the global mean average. Overall, both $\overline{ec}_w$ and $\overline{EC}_w$ decrease with the value of α, as well as the coefficients of variation (CV) of $\overline{ec}_w$. In the case of EC_w, the variation coefficient reaches a minimum when $\alpha = 0.64$, then (for $\alpha \geq 0.64$) starts to gradually increase with α. Globally, EC_w has an higher variability than ec_w for any α value: for example, considering the case $\alpha = 0.5$, the CV of all the ec_w values is 0.91 and the CV of all the EC_w values is 1.26 (see Fig. S7 in the Supplementary Material for further details).

The average environmental cost of withdrawing 1 m^3s^{-1} of surface water in each country can be estimated (see Fig. 4a) by assessing the average environmental cost at a country level ($\overline{EC}_{w,c}$) as

$$\overline{EC}_{w,c} = \frac{1}{N_c} \cdot \sum_{j=1}^{N_c} EC_{w,j} \tag{11}$$

where $EC_{w,\,j}$ is the overall environmental cost in the cell $j-th$ within the considered country, and N_c is the number of cells in the country. The average environmental cost in Egypt might not be totally representative since this country has only the 21% of the cells with $Q \geq 1$ m^3s^{-1}.

Taking the average as in Eq. (11) implies that a potential water withdrawal can occur everywhere with the same probability within the country, without considering the fact that water is usually withdrawn where it is more abundant; as a consequence, a different approach to averaging can be considered: the environmental cost at the country scale can be assessed as a weighted average using the undisturbed river discharge as the weight

$$\overline{\overline{EC}}_{w,c} = \frac{\sum_{j=1}^{N_c} EC_{w,j} \cdot Q_j}{\sum_{j=1}^{N_c} Q_j}. \tag{12}$$

Adopting the weighted average according to Eq. (12) (see Fig. 4b) instead of the average index given by Eq. (11) entails that: (i) overall, the environmental cost at the country scale will be reduced, because in a river section the greater is Q the lower is ec_w (see Eq. (5)); (ii) countries having arid or semi-arid areas, but also crossed by significant rivers (e.g., Egypt and Pakistan) or, in alternative, large countries having both humid and arid regions (e.g., Argentina and China), will have significantly lower values of EC_w at the country scale than with the unweighted average (see from Fig. S9 to Fig. S13 in the Supplementary Material for further details and Fig. S14 for the weighted average environmental cost at the basin scale).

Assessing the surface water value of each country through the index proposed in this work implies taking into account the transboundary flows and, thus, the interdependency among countries: e.g., withdrawing 1 m^3s^{-1} of water in Sudan from the Nile will impact the Nile's course in Egypt too; as a consequence, the surface water value of that amount of water cannot be assessed considering exclusively the surface water resources of Sudan. Fig. 5 describes the Nile river's trends of the undisturbed discharge and of the two environmental cost measures down to the river mouth. The plot highlights the importance to take into account the downstream consequences of a water withdrawal. For example, even if the environmental costs per unit length after the 2000th kilometer are approximately constant, the values of EC_w are very different depending on the downstream river course impacted.

In Fig. 5 an example is also proposed of a hypothetical withdrawal that occurs in a Nile's river section S^* within South Sudan. The resulting overall environmental cost is divided in three portions, in correspondence of the country borders, which are the shares of $EC_w(S^*)/EC_{world}$ that affect South Sudan ($0.29 \cdot EC_w(S^*)/EC_{world}$), Sudan ($0.43 \cdot EC_w(S^*)/EC_{world}$), and Egypt ($0.28 \cdot EC_w(S^*)/EC_{world}$). Overall, this procedure can be extended by considering all the area under the curve EC_w/EC_{world}, which is divided according to the country borders (dark grey and bight grey areas). Therefore, comparing the areas under the curve of EC_w/EC_{world}, one can estimate the average percentage subdivision of the environmental cost of a generic potential withdrawal that can occur anywhere in the considered country, along the Nile river. The case study of the Nile's river network (see Table 1) underlines that the average environmental cost of a reference amount of surface water withdrawn can be considerably higher if the overall freshwater network is considered.

4. Discussion and conclusion

The index proposed in this paper, EC_w, aims to attribute an environmental cost of a reference withdrawal of water accounting for the impact that it would cause on river ecosystems. In order to develop an easy-to-apply tool to interpret any geography of the withdrawals, the index is referred to an unitary potential withdrawal. The effort in this work was devoted to design an index able to interpret, with a feasible level of complexity, the interaction between water consumption and its effects on fluvial ecosystems (i.e., taking into account the downstream effect of a water withdrawal and considering the non-linear relation between discharge and environmental significance of a river ecosystem). In an attempt to design a parsimonious index we introduced only one parameter, α, and kept at a minimum the number of variables employed, namely the river discharge and the river network, which are the main factors that influence the fluvial ecosystem equilibrium. Due to the small amount of data needed, the index can be consistently applied at a global scale. Depending on the available data and on the target of the application, other withdrawal scenarios and river discharge patterns can be used; in fact, the proposed method is general. As an example of this, the index was tested applying it to a different distribution of river discharges demonstrating consistent results compared to those shown in this work (see Fig. S16 and Fig. S17 in the Supplementary Material for further details).

The index can provide a further interpretation to the volumetric measure of surface water consumption, by evaluating the environmental cost of a potential water withdrawal on fluvial ecosystems. An example of application is shown in Fig. 6: the weighted average environmental cost at the country scale (shown in Fig. 4b) is compared with the yearly national surface water consumption per unit area. The latter is assessed employing the annual blue water footprint (BWF) of national production (Mekonnen and Hoekstra, 2011) divided by the country area (FAO, 2016) (thus mm/year). Since the BWF considers both surface and groundwater consumption, this value is multiplied by the average country percentage of surface water withdrawal in order to consider only the surface water component of the BWF. This percentage is evaluated at the country scale as the ratio of the surface water withdrawal to the total freshwater withdrawal (FAO, 2016); a percentage equal to 64% is employed in countries where these data were not provided, which is the weighted world average percentage of surface water withdrawal (where the BWF of national production is the weight). The value 64% is consistent with the results given by Döll et al. (2012) who estimated that 35% of the water withdrawn worldwide is groundwater.

Fig. 6 is divided into four quadrants by two black lines, which are: the global discharge-weighted average of the environmental cost, $\overline{\overline{EC}}_w$, and the global average of surface water consumption per unit area. Therefore, in the first quadrant one finds those countries that, despite

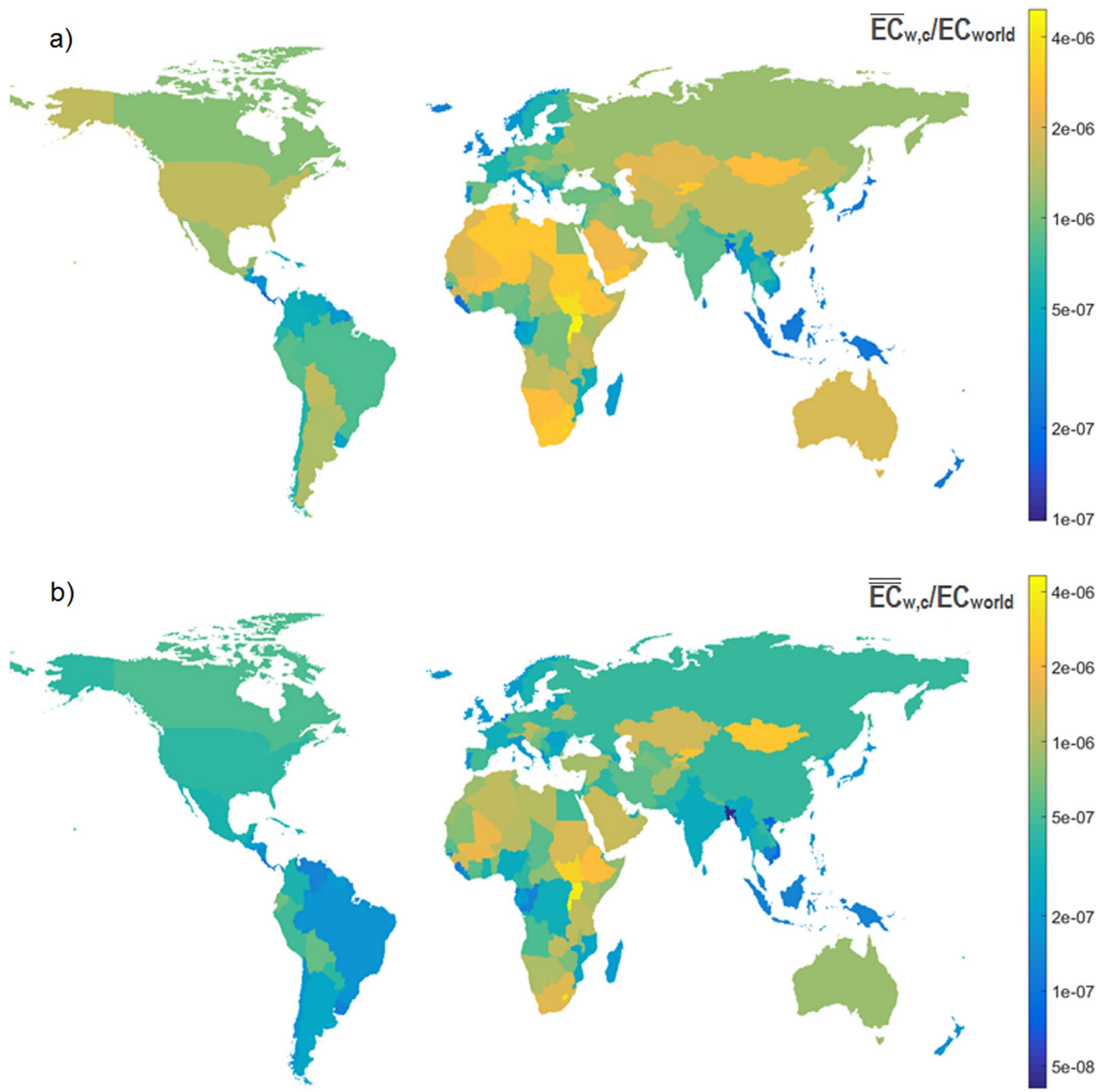

Fig. 4. (a) The average environmental cost at country scale ($\overline{EC}_{w,c}$) normalized by EC_{world}; (b) the weighted average environmental cost at country scale using the undisturbed river discharge as the weight ($\overline{\overline{EC}}_{w,c}$) normalized by EC_{world}. Both maps are obtained using $\alpha = 0.5$ and $W = 1\mathrm{m}^3\mathrm{s}^{-1}$ in cells of 30 × 30 arc min. The colour bars are in log scale.

the high environmental cost of withdrawing surface water (i.e., high $\overline{\overline{EC}}_{w,c}$ values), have an yearly surface water consumption per unit area higher than the global average (e.g., Kyrgyzstan, Egypt, Spain and Pakistan). Conversely, countries as Bangladesh and Vietnam, which fall in the fourth quadrant, have a high value of yearly surface water consumption per unit area, but their freshwater ecosystems are comparatively less impacted by water withdrawals than those in other countries. From a global point of view, a system that aims exclusively to minimize the impact on surface water resources, ideally, would require countries of the first quadrant to move toward the second one by reducing the volumes of surface water withdrawn. Those volumes could be compensated by moving the third quadrant countries toward the fourth quadrant, since these countries are characterized by low $\overline{\overline{EC}}_{w,c}$ and (currently) comparatively lower surface water consumption per unit area.

The application of the index highlights regions and countries more environmentally vulnerable to surface water exploitation. Since the index systematically assesses the environmental cost by accounting for the downstream propagation effect of a water withdrawal on the fluvial ecosystem, it aims to support decision-making in transboundary river basins as well, with the challenge to support water management strategies overcoming administrative borders. Moreover, the index is a possible novel tool to analyse the food trade network with an impact-oriented approach, evaluating the environmental cost of the surface water volumes consumed to produce a good by accounting for the freshwater ecosystems from which the volumes are removed. Similarly,

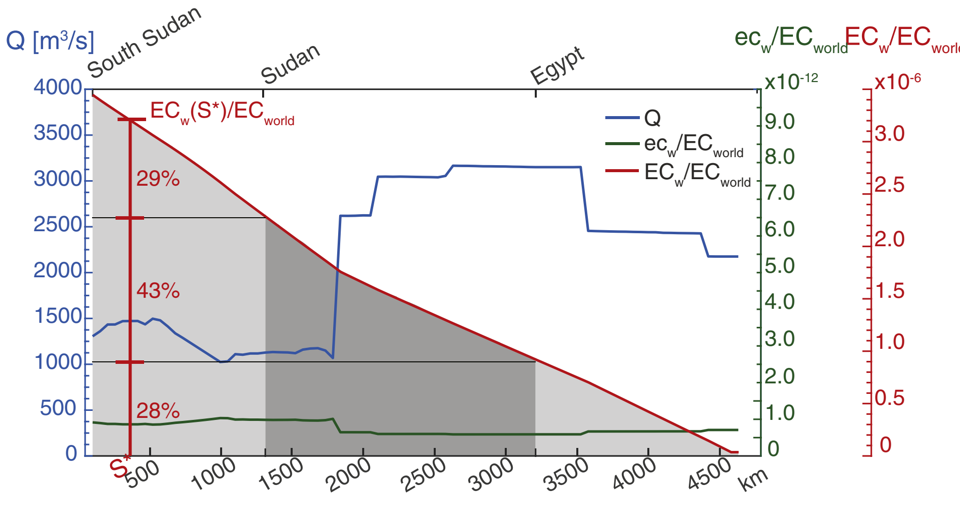

Fig. 5. The average annual Nile undisturbed river discharge, the environmental cost per unit length (ec_w) at the closing section of each cell of the Nile's course and the overall environmental cost (EC_w) in each cell of the Nile's course. Both environmental costs are normalized by EC_{world} and estimated with $\alpha = 0.5$ and $W = 1m^3s^{-1}$. The environmental cost of a water withdrawal that occurs in a generic river section S^* (i.e., $EC_w(S^*)/EC_{world}$) can be divided among the downstream countries affected. In the example, $EC_w(S^*)/EC_{world}$ is divided between: South Sudan (29%), Sudan (43%), and Egypt (28%). The cells span 30 × 30 arc min. Along the abscissa the distance from the Nile section at the border between Uganda and South Sudan is reported. In the top of the figure the countries crossed by the river are shown. In the range [800,1000] km, the WaterGap river discharge exhibits a quite irrealistic peak. Likely, it is due to the complex local hydrography (i.e., wide wetlands) that is very difficult to model. For this reason, the peak has been considered spurious and neglected.

Table 1
The average percentage subdivision of the environmental cost of a water withdrawal EC_w on the Nile river course shown in Fig. 5 among the impacted countries. The first column reports the location of the potential water withdrawal, while the corresponding row indicates the percentage subdivision of EC_w among the downstream countries. On the diagonal, the percentage of EC_w that has consequences within the country where the water withdrawal occurs is shown.

Location water withdrawal	Percentage of impacted countries		
	South Sudan	Sudan	Egypt
South Sudan	**23**	46	31
Sudan		**42**	58
Egypt			**100**

Yano et al. (2015) proposed a environmental-based factor to weigh the international food trade network, but, their approach does not consider the impact of the surface water withdrawal on the whole downstream river network.

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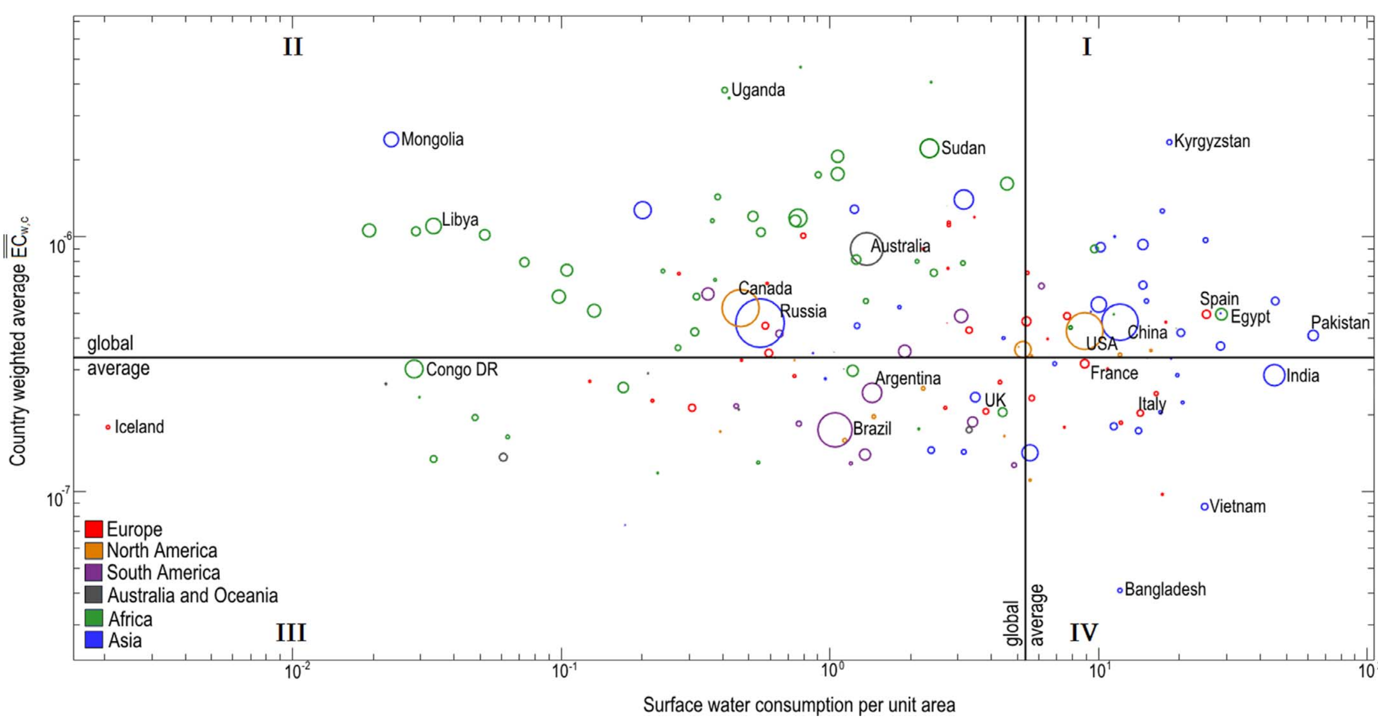

Fig. 6. Scatter plot of the yearly national surface water consumption per unit area (in mm yr^{-1}) and the average country impact index, $\overline{\overline{EC}}_{w,c}$, weighted in accordance to Q. The scatter plot is divided in 4 quadrants by two black lines. The horizontal line is the world weighted average of EC_w weighed in accordance to Q, and the vertical line is the world average surface water consumption per unit area. The circle size is proportional to the country area, while the color refers to the continent. See Fig. S17 in the Supplementary Material to visualize more country names in the plot.

Appendix A. Examples of possible choices of the parameter α

Once the discharge and the water withdrawal in the river section are knows, Eq. (5) allows one to evaluate the environmental cost per unit length through the choice of the parameter α, where the designation of α involves establishing the significance of a fluvial system accounting for a specific river characteristic. When $\alpha = 0$ it follows that $ec_{max} = k(0)$, thus ec_{max} does not depend on the considered river section; under this perspective in any river section the complete loss of the corresponding fluvial system is expressed by a constant value that is equal to EC_{world}/L_w. Differently, when $\alpha = 1$, one obtains $ec_{max} = k(1)\cdot Q$, and the significance of a fluvial system is directly proportional to the river discharge. In this case, from Eqs. (4) and (2), $ec_{max} = (EC_{world}\cdot Q)/(L_w\cdot\mu_Q)$, where μ_Q is the global average of the undisturbed river discharge. The case $\alpha = 1$ is reasonable, for example, when one focuses on the river water quality: it depends on the concentration of nutrients and chemical substances and, then, on the dilution capacity, which is in turn proportional to discharge.

By altering the river discharge, water withdrawals can lead to the impoverishment of riparian ecosystems: there is an intrinsic and significant sensitivity of riparian ecosystem to hydrological modifications (Camporeale et al., 2006; Doulatyari et al., 2014). Riparian areas are the transition zones between terrestrial and aquatic ecosystems and have a significant role in maintaining regional biodiversity; in fact, they have valuable plant communities, fisheries and wildlife (Naiman et al., 1993; Sabater and Tockner, 2009; Shafroth et al., 2002). Therefore, another criterion to estimate α may be to consider the impacted riparian area per unit river length. Such area can be approximately proportional to the channel width. This latter is in turn proportional to a power function of the discharge with an exponent α between 0.4 and 0.5 (Julien and Wargadalam, 1995).

River biodiversity can be another relevant issue: in fact, fish species richness increases as a power law of the river discharge (Oberdoff et al., 1995; Poff et al., 2001; Xenopoulos et al., 2005). Thus, the maximum environmental cost per unit length on river biodiversity can be evaluated considering the species-discharge relationship, where $fish - richness \propto Q^{\alpha}$. Xenopoulos et al. (2005) assumed $\alpha = 0.4$ for rivers between 42°N and 42°S, whereas Oberdoff et al. (1995) conducted a global scale analysis on variations in species richness finding $\alpha = 0.33$. However, the value of α changes depending on the specific site: e.g., Xenopoulos and Lodge (2006) obtained α in the range 0.1 - 0.2 for two different regions in the United States.

Another point of view about river characteristics is the bed-load transport, which controls the sediment erosion/deposition processes in fluvial systems, as well as the nutrient transport essential for the fluvial habitat health. For steady flows the bed-load transport is commonly represented as $\phi \propto (\theta - \theta_b{}^{\frac{3}{2}})$ (Meyer-Peter and Müller, 1948) where θ and θ_c are the effective and the critical Shields parameters, respectively, which are dimensionless bed-shear stress coefficients ($\theta \propto \tau_b$). Assuming a Chèzy friction law (that is, a steady and turbulent open channel flow) and the alluvial channel geometry relationships by Julien and Wargadalam (1995), the bed-shear stress can be evaluated as

$$\theta_b \propto R\cdot i_b \propto \frac{U^2}{R^{\frac{1}{3}}} \propto Q^{0.33} \tag{A.1}$$

where R is the hydraulic radius (equal to the river depth), i_b is the bed slope, and U is the mean stream velocity. Then, the sediment volumetric discharge can be estimated as

$$Q_s \propto q_s\cdot W_c \propto \theta_b{}^{\frac{3}{2}}\cdot W_c \propto Q^{0.5}\cdot Q^{0.44} = Q^{0.94} \tag{A.2}$$

where W_c is the channel width. Thus, in our framework the environmental cost considering the geomorphologic impact can be assessed employing $\alpha = 0.94$.

The examined cases show that typically values of α fall within the interval [0,1]. In any case, as no conceptual reason exists for limiting α within [0,1], values outside this range can be adopted in our framework.

Supplementary material

Supplementary material associated with this article can be found, in the online version, at 10.1016/j.advwatres.2017.10.016.

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