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LETTER

# The globalization of riverine environmental resources through the food trade

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## Abstract

We quantify the impact of food consumption on local and foreign water resources through an indicator of the environmental value of the riverine water. This indicator takes into account both the local environmental relevance of the fluvial area where water is withdrawn (biodiversity richness, riparian vegetation, sediment transport, etc) and the downstream effects of water withdrawals. In the 1986–2013 period, food consumption has more than doubled its impact on foreign riverine environments, but still the international trade reduces the pressure of food consumption on global river system by 11%, as compared to an ideal situation where all food is produced locally. We also show the geography of country (or individual) responsibility on the environmental changes of world rivers. Hotspots of food-related river-environment degradation are found in Australia, Pakistan, South Africa, and Spain.

## 1. Introduction

Food production is the largest form of societal water consumption [1, 2] and agriculture's 'thirst' is expected to further increase in the next years, due to the growth of the world population, the rising of living standards, and climate change [2–4]. This pressing water demand is causing remarkable environmental impacts [5–7] that are growing to an unprecedented extent [8–10], in particular on surface water ecosystems [11] that globally contribute about 60% of the total water used for irrigation [12].

In the last few decades, the global agri-food system has exhibited a progressive geographical decoupling between food production and consumption [13, 14], as testified by the fact that about one quarter of the food produced for human consumption is currently traded internationally [15]. This has led to the so-called globalization of water [16–18]. Namely, (i) water resources used to produce food circulate in the global economic system virtually embedded in the internationally-traded products, (ii) food demand of several countries heavily rely on foreign water resources [18], and (iii) exported goods contribute to the water exploitation and degradation of environments far from the consumers [19–23].

The concept of virtual water has played a key role in shedding light on the links between the food consumption geography and the water resources (over) exploitation. However, a critique that has been made to the virtual water trade assessments is that they typically aggregate and compare water volumes without considering the place where water has been withdrawn, i.e. without differentiating among water-abundant and water scarce regions [24–27]. Some recent exceptions are the works by Lenzen *et al* [24], Yano *et al* [25], and Poore and Nemecek [28]. In order to focus on scarce-water trade flows, Lenzen *et al* [24] weighted the virtual water network adopting a country-specific water scarcity indicator. A similar metric (i.e. freshwater withdrawals weighted by local water scarcity) was adopted by Poore and Nemecek [28] to compare different types of supply-chains in the agri-food industry. Yano *et al* [25] proposed to scale the virtual water volumes by a factor expressing local water unavailability, which is calculated in each cell as the land area required to collect a reference amount of water. In spite of these studies investigate the virtual water network considering the availability (or unavailability) of the exploited resource, however they neglect: (i) the relevance of the impacted environment (e.g. its biodiversity richness, bio-geomorphologic characteristics, and habitat health) and (ii) the downstream propagation effect of an hydrological stressor.

Our work aims to overcome these limitations, focusing on the fluvial ecosystems. These are among the most precious, important and sensitive environments on Earth [29]; on the other hand, rivers are also the major water sources for irrigation and many of them show high stress levels due to local food consumption and trade [30]. For these reasons, it is urgent to disclose the environmental significance of the water extracted from river ecosystems. To this aim, we quantify the 'environmental value of the riverine water' (EVRW) embedded in food products. By considering 270 food and agricultural goods, we estimate the irrigation water supplied from surface water sources over the period 1986–2013. Then, building on the concept of Environmental Cost (EC) introduced by Soligno *et al* [31] we consider (i) the effect that the river discharge reduction has on different fluvial characteristics (e.g. transport of sediments and chemicals, riparian belt width, biodiversity richness, etc), and (ii) the downstream propagation effect of the withdrawal.

By this approach, we obtain the global picture of the EVRW involved in the food production and the corresponding network of the traded riverine resources. This allows us to unveil the main implications of the globalization of water resources on the health of surface water systems and to investigate the EVRW network efficiency over 28 years.

## 2. Methods

We provide here a brief description of the approach followed to obtain the EVRW network analysed in this work; a more detailed description of the procedure is given in the supplementary material (available online at stacks.iop.org/ERL/14/024020/mmedia).

From the FAOSTAT database, we collected 28 years (1986–2013) of food production and bilateral trade data for 270 food and agricultural (including cotton lint, natural rubber and tobacco) items. The yearly volumes of water used for the production (and for the export) of each item were estimated by employing the country-specific virtual water content (VWC), namely the amount of water required to produce a unit amount of product. The yearly country-specific VWC for each primary crop was assessed accordingly to the approach validated by Tuninetti *et al* [32] as

$$\mathrm{VWC}_{c,t} = \frac{\overline{\mathrm{VWC}}_{c,1996-2005} \cdot \overline{Y}_{c,1996-2005}}{Y_{c,t}} \quad \left[\frac{\mathrm{m}^3}{\mathrm{ton}}\right], \quad (1)$$

where subscripts $c$ and $t$ refer to the country and year considered, respectively, $\overline{\mathrm{VWC}}_{c,1996-2005}$ is the average VWC in the decade 1996–2005 [33], $\overline{Y}_{c,1996-2005}$ is the corresponding average yield, and $Y_{c,t}$ is the crop yield in year $t$ (provided by FAOSTAT).

The VWC of crop-derived products was based on the VWC of the primary input crops adjusted with the product fraction (i.e. ton of derived product obtained per ton of input crop) and the value fraction (i.e. the market value of the crop-derived product divided by the aggregated market value of all the products resulting from one input crop) [33], taking into account the difference between local and imported input products. For animal product, due to the lack of reliable data about country-specific animal diets we could not consider a time-varying VWC and, thus, we adopted time-averaged values at the country scale [34].

Trade and production data, expressed in metric tonnes, were multiplied by the corresponding VWC in order to convert them into the virtual water volumes used to produce and trade each commodity. Afterward, the yearly volumes of surface irrigation water for food production and trade were assessed using (i) the country- and product-specific ratio ($Rb$) of the average blue VWC to the overall VWC [33] and (ii) the country-specific percentage of the area equipped for irrigation served by surface water resources ($Rs$) [35]. The latter percentage was assumed constant for all the commodities produced in a given country due to a lack of reliable product-specific $Rs$ data. Finally, each cubic meter of water withdrawn from surface water systems was converted into its corresponding environmental impact, by considering the country-specific EC introduced by Soligno *et al* [31], in order to take into account the expected impact that these withdrawals have on local river ecosystems.

According to Soligno *et al* [31], in a given river section the EC per unit length ($\mathrm{ec}_w$) of a unitary water withdrawal $W$ is defined to be proportional to the river discharge reduction caused by the withdrawal. Namely, $\mathrm{ec}_w = \mathrm{ec}_{\max} \cdot W/Q$, where $\mathrm{ec}_{\max}$ is the maximum EC per unit length, which occurs when the entire river discharge ($Q$) is depleted. The value of $\mathrm{ec}_{\max}$ is related to the relevance of the river environment considered and is evaluated by considering fluvial site-specific characteristics (e.g. width of the riparian belt, biodiversity richness, transport of sediments and chemicals) related through power laws to the river discharge. Since the subtraction of $W$ alters the discharge from the section where water is withdrawn ($S_W$) down to the river mouth ($S_M$), the overall EC, $\mathrm{EC}_w$, is evaluated as the sum of the EC per unit length generated downstream by the withdrawal. Accordingly, $\mathrm{EC}_w = \int_{S_W}^{S_M} \mathrm{ec}_w(s)\,\mathrm{d}s$, where $s$ is the curvilinear abscissa along the river.

In this study, we computed the $\mathrm{EC}_w$ value related to a unitary surface water withdrawal with a 0.5° spatial resolution, adopting the annual average river discharges obtained from the pristine scenario of the WaterGAP 2.2c model [36–39]. Then, in light of the fact that water is generally withdrawn where it is more

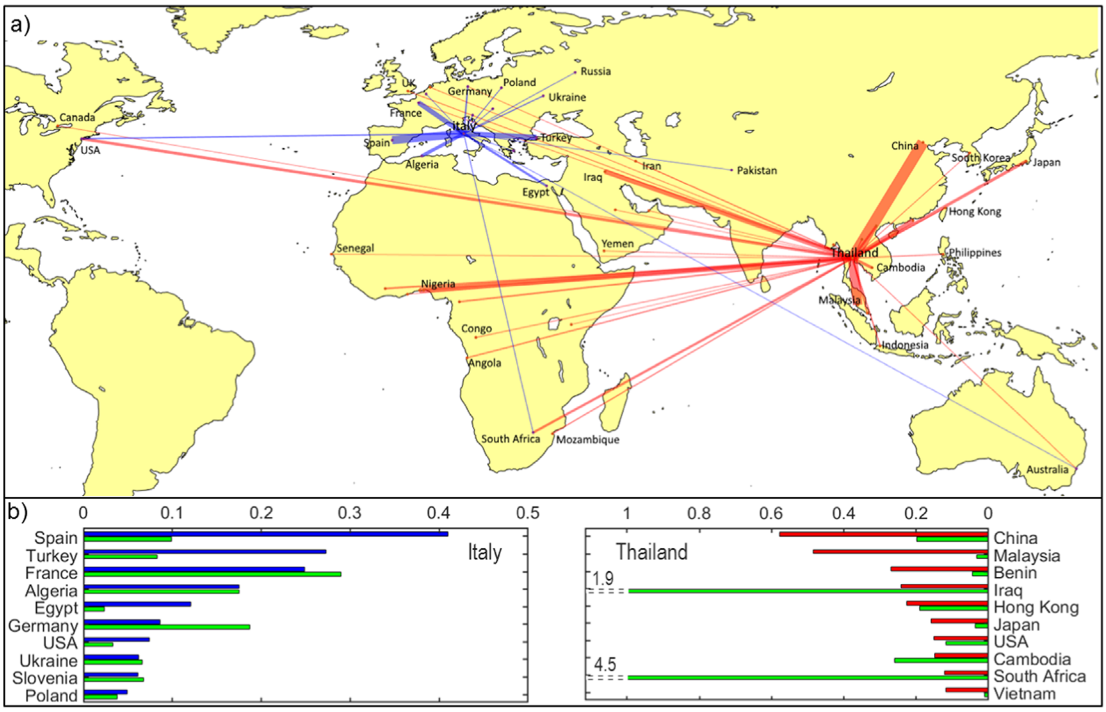


**Figure 1.** (a) Snapshot of the environmental value of the riverine water network in 2013, where the major import flows to Italy (in blue) and the major export flows from Thailand (in red) are shown. The line width is proportional to the EVRW flow. (b) The top 10 major countries from which Italy imports EVRW (blue bars) and the top 10 countries to which Thailand exports EVRW (red bars). The share of the global traded EVRW is reported. Green bars indicate the EVRW under a self-reliant scenario (i.e. if the same food had been produced locally, rather than imported). When the green bars exceed the others, the business as usual flow trade provides an advantage for the environmental status of global rivers.

abundant, we assessed the country-specific $EC_{w,c}$ as the weighted average of the $EC_w$ of the cells within the considered country, using the river discharge as the weight. Finally, by combining the EC values and the surface water volumes, food trade and food production data were converted into a common currency: the EVRW embedded in food products. In this way, we were able to study the impact of food consumption on the deterioration of world's river environments.

Conveyance and distribution losses in the irrigation system increase the water withdrawal with respect to the surface water volumes consumed by each agricultural item (i.e. surface water loss to the atmosphere by evapotranspiration); since we considered the latter volumes, we likely underestimated the real withdrawal from the river system.

## 3. Results and discussion

### 3.1. Food trade activates a global network of riverine value

In each nation, the riverine resources embedded in the food consumed has two components: one concerns food goods produced and consumed within the country (i.e. exploiting domestic rivers), the other concerns imported food commodities (i.e. exploiting foreign fluvial systems). We will call these the 'local' and the 'non-local', component respectively. Our results show that, globally, the EVRW involved in food consumption increased by 40% during the considered period, while the non-local EVRW embodied in trade has more than doubled since 1986. Therefore, threats to fluvial ecosystems are increasingly driven by consumer demand across the globe.

The global network of riverine resources (see figure S3 in the supplementary material) highlights the unexpected links by which final food consumers may impact rivers very far away from consumption places. A snapshot of the multiple routes in the network is shown in figure 1, where the non-local EVRW consumed by Italy and the environmental value of Thailand's surface resources 'eaten' by consumers abroad are depicted. Italy imports many food goods and is the eighth largest world EVRW importer; it follows that Italian consumers significantly affect several out-of-Italy river systems. Vice versa, a lot of the environmental value of the Thai riverine waters used for irrigation is exported through food trade, making Thailand the sixth largest world exporter of riverine resources.

Overall, the EVRW trade network can be interpreted as a network of responsibility of countries for the consumption of riverine resources beyond their national borders. In the case shown in figure 1(a) Italy is responsible for 2.2% of the overall riverine resources embedded in the world food trade and the largest share of its externalized pressure on surface waters concerns Spanish and Turkish rivers. Instead, China and Malaysia are among the main foreign countries that drain environmental value from Thai fluvial systems. Overall, China is the biggest importer of EVRW in the world with a share of 12% of the total EVRW traded internationally; in particular, the trade link from Australia to China is the highest recorded EVRW flow, involving a share around 4%.

The key point of looking at the environmental value of surface waters is that a same volume of surface water does not have the same environmental value in different countries. It follows that the amount of riverine resources imported by a country depends on the specific characteristics of the river environments of its trading partners. For example, the flow from Turkey to Italy conveys about $3 \times 10^5$ m$^3$ of surface virtual water (mostly embedded in shelled hazelnuts), which corresponds to an EVRW share of 0.27%; a smaller EVRW is imported from France (i.e. 0.24%) despite the surface virtual water volume ($8 \times 10^5$ m$^3$) being almost three times larger. This disparity is due to the greater environmental fragility of the Turkish river environments compared to the French ones.

Imports allow a region to satisfy its food needs without exploiting local resources and, thus, externalizing the impact of the national food consumption to foreign river systems. However, when food is imported by countries with a lower surface water productivity (i.e. m$^3$ of surface water used per unit of product) and a higher environmental value per unit of surface water than the trading partner, then there is a trade-induced global saving of EVRW. In order to highlight the occurrence of EVRW savings, in figure 1(b) we compare the business as usual trade situation with a scenario in which production and consumption spatially coincide, namely where all food is produced locally. The difference between these two scenarios provides a measure of the efficiency of the considered trade relationship.

Regarding Italy, figure 1(b) shows that significant EVRW inefficiencies occur in the import flows from Spain, Turkey, and Egypt; these inefficiencies depends on both the different sensitivity of rivers to withdrawals and the surface water productivity typical of the exporters. For example, in the case of Egypt, if the same imported-food had been produced in Italy, the impact on riverine systems would have been 5 times lower. This gap is due to a number of factors, where the main ones are (i) the greater environmental value per unit of Egyptian riverine water and (ii) the considerable share of blue water in Egypt. Also Thailand exports exhibit highly negative EVRW unbalances; e.g. flows towards China, Malaysia, Japan, and Vietnam. Conversely, although often the geography of food production and trade neglects the health of riverine ecosystem, there is a number of trade relationships that induce significant savings of EVRW. Two striking cases corresponds to the internationally-traded products that flow from Thailand to Iraq and to South Africa, which enable to cut down the EVRW by 8 and 37 times, respectively.

### 3.2. Importers, exporters, and consumers of riverine value

Our analyses reveal that there is a great spatial heterogeneity in EVRW dynamics (e.g. see figure 2(a)). For instance, food consumption in Australia, India and South Africa has little impact on foreign river systems; conversely, these countries are among the largest exporters of EVRW in the world and, therefore, there are foreign economies that have strong responsibilities on the river ecosystem health of these regions. 'Unpacking' each red bar of figure 2(a), the major trade links that drive the degradation of Australian rivers point towards China, Japan, Indonesia, and South Korea, while South Africa's major EVRW exportations go towards Zimbabwe, Botswana, Swaziland, Netherlands, and Namibia. Vice versa, China and Turkey are the most important importers of riverine resources, but also Germany emerges as a strong net importer (see figure S4 in the supplementary material to identify the largest Germany's imports flows over time).

The picture shown in figure 2(a) refers to year 2013 and it is a sample of the temporal dynamics exhibited by the EVRW flow network. Figure 2(b) discloses the temporal pattern of net exporters and importers during the period 1986–2013. Some countries maintain their characteristic of net exporter or net importer throughout the entire period (e.g. Australia, USA, and Germany), while other countries show substantial overturns, both pulsing (e.g. India and Turkey) or showing some persistence (China, at least in the last decades). These shifts are due to changes in both the basket of products imported/exported from each country and in the structure (topology and flows) of the trade partners. In the case of the USSR-Russia, the shift is due to the dissolution of the Soviet Union and the consequent EVRW import by Russia from neighbouring (ex-Soviet) countries characterized by fragile surface water systems (e.g. Kazakhstan and Uzbekistan).

The global EVRW network exhibits remarkable changes over time. In 28 years, the number of yearly active links in the network almost doubled, implying that year after year global food consumption relies on an increasingly interconnected network. A comprehensive description of the major temporal developments of the EVRW network is shown in figure 3, where trade flows are aggregated into nine world macro-regions and flows greater than 1% of the annual EVRW globally traded are displayed. Interestingly, from the year 2004 the number of (macro) links have decreased despite it has grown the total riverine value traded among the macro-regions. Therefore, the backbone of the network has strengthened by relying on a smaller number of (macro) links.

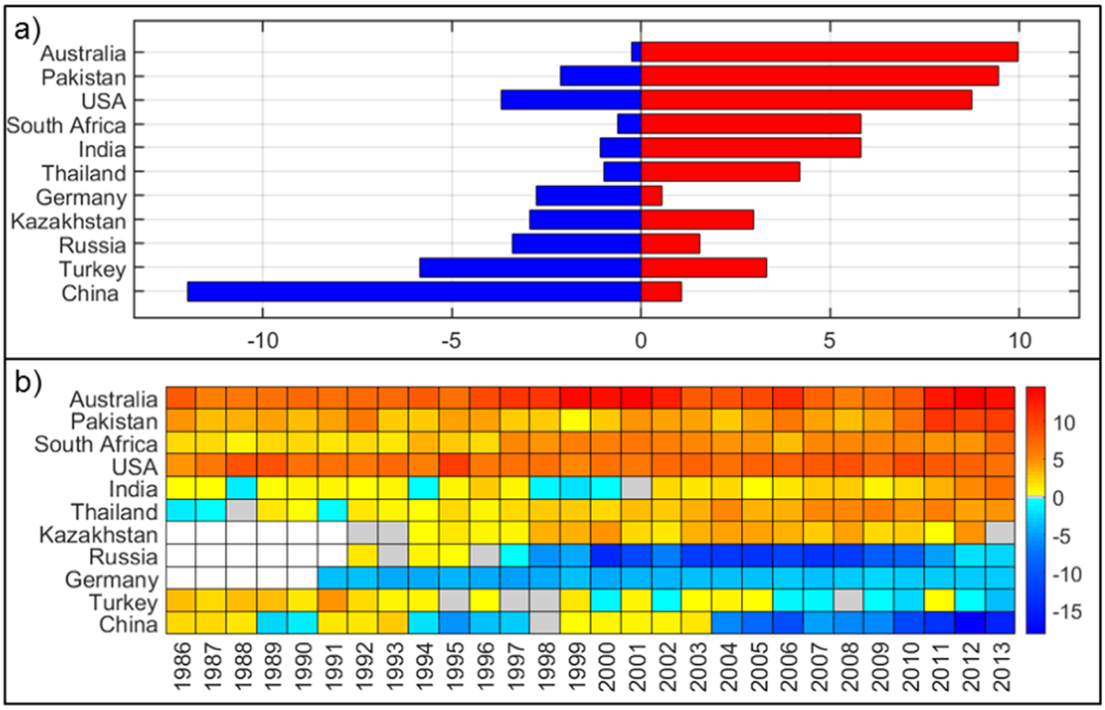


**Figure 2.** (a) Exports (in red) and imports (in blue) of EVRW for the six world major exporters and the six major importers in 2013 (the USA are both). The abscissa reports the shares of the overall EVRW exchanged during year 2013. (b) Global share of net export flows of EVRW embedded in food products for the same countries from 1986–2013. The shares are calculated based on the average EVRW yearly traded in the period 1986–2013. Figure S6 in the supplementary material provides the same information considering more countries.

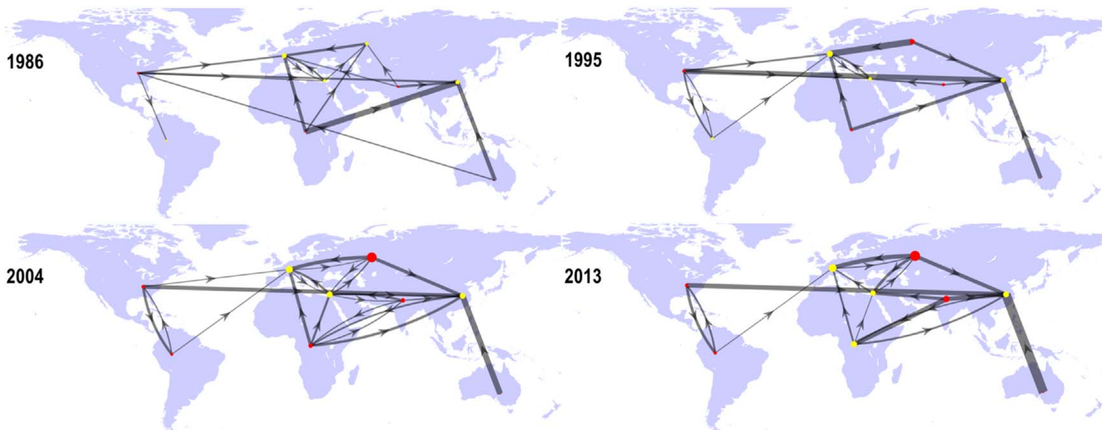


**Figure 3.** Four snapshots of the network backbone of the environmental value of the riverine water in 1986, 1995, 2004, and 2013. Nine world's macro-region are considered. The link width is proportional to the EVRW flow between macro-regions. The node are proportional to the EVRW flow within each macro-region. Yellow and red nodes indicate net importer and exporter of riverine value, respectively. Figure S7 in the supplementary material shows the geographical division into the nine macro-region; i.e. North America, Latin America and the Caribbean, Europe, Africa, North Africa and the Middle East, East Europe and Central Asia, South Asia, East Asia, and Oceania. The shares are calculated based on the average EVRW yearly traded in the period 1986–2013 and only the links higher than 1% are reported in the figure. About Africa, notice that the FAOSTAT database suffers of a lack of trade data related to Sudan and South Sudan in 2012–13.

During the 1986–2013 period, East Asia and Europe had have the highest influence on foreign rivers. Over the past three decades, European countries have maintained their strong dependence on the rivers of the Middle East and Africa and, meanwhile, they increased their impact in Central Asia (especially in Kazakhstan and Turkey). The latter region has significantly increased the trade of riverine resources both with neighbouring areas (in particular with Europe and East Asia) and within the region itself. Also the EVRW trading relationships among European countries have grown considerably since 1986 (threefold increased), despite the number of active trading links slightly decreased. Meanwhile, since 1986 South Asia has incessantly increased its exports of EVRW (e.g. India and Pakistan increased exports by

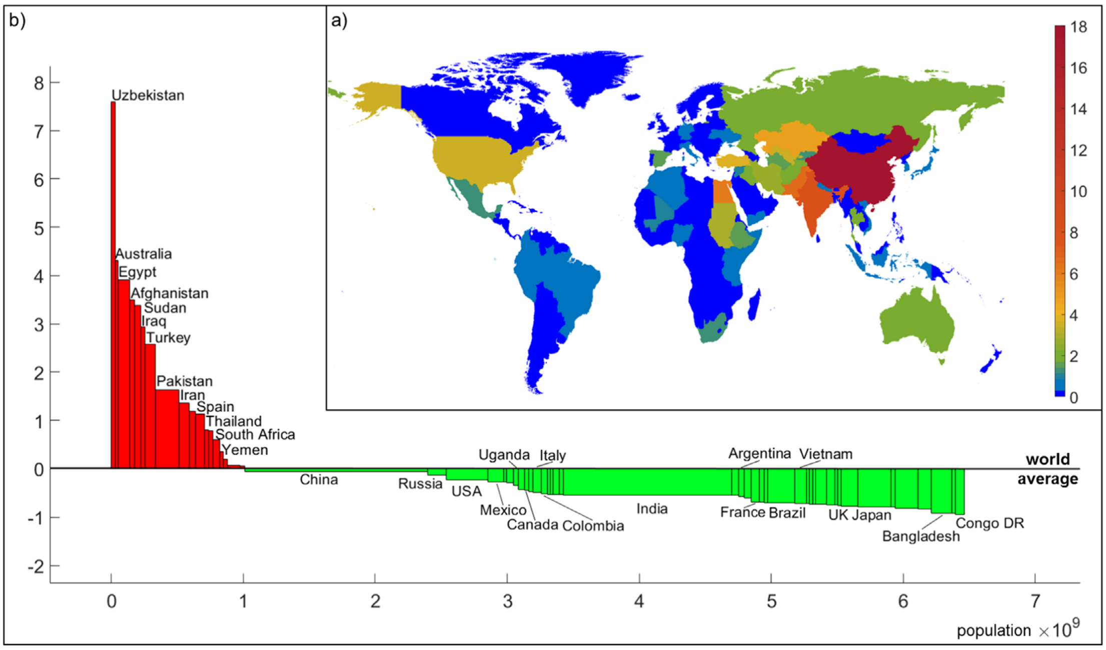


**Figure 4.** (a) The shares of responsibility of each country on the overall degradation of world river systems through the consumption of food goods in 2013. (b) Per capita impact related to the overall EVRW embedded in food consumption in 2013. The bar height indicates the normalized difference between the country's share and the world average. The normalization factor is again the global average. The width of each bar is equal to the total population of the considered country. Only countries with more than 20 million inhabitants are represented in the bar plot.

approximatelly eight and three times, respectively), becoming the largest world's net exporter. Although the EVRW network is highly dynamic and over time several new links appear (e.g. from the South Asian to the African countries) and other links disappear, overall the network has a global tendency to intensify trade relations between geographically close regions, e.g. see in figure 3 the links between East Asia and Oceania or between North and South America. This confirms that the physical distance between trading partners is one of the major drivers of virtual water flows [40].

The fluvial environmental value that we 'eat' typically includes both a local component—linked to the domestic food production—and a non-local component, due to the food import. Therefore, to assess the average EVRW related to the overall food consumption of a country (or of an average citizen), one has to consider the balance equation: consumption = (import + production − export), where stock variations are neglected for the sake of simplicity. Figure 4(a) shows the shares of responsibility of each country on the overall degradation of world river systems through the consumption of food goods in 2013 (while the time-series of the consumption values is reported in figure S5 in the supplementary material). The sum of all the EVRW shares equals 100%, which corresponds to the global impact of the world population on the health of surface waters worldwide. The picture is quite heterogeneous and testifies that country shares depend on multiple factors, including the typology of the exploited water sources (e.g. groundwater or surface water resources), the vulnerability of the involved river systems, the partition between local and imported food, and the import pattern. For example, countries like Saudi Arabia and Denmark use almost exclusively blue water from groundwater sources; this explains their low shares of consumption of riverine value. Many tropical countries have low EVRW share as they largely use rain-water for agriculture; a noteworthy exception is Indonesia, where food production mostly uses rainfall water (around 96%), but the share is quite high compared to other tropical countries since Indonesia imports large flows of EVRW from Australia, India, Pakistan, USA and South Africa.

Evidently, country population is one of the main drivers of EVRW consumption. In order to compare the citizens of different countries, figure 4(b) shows the per-capita impact. The world average value of the share is assumed as the reference value and citizens from countries in red (or in green) are responsible for a larger (or smaller) pressure on the global rivers. For example, Uzbekistan citizens show a high consumption per capita due (i) to the high vulnerability of the local surface water systems, which makes the local production of food and agricultural goods environmentally 'expensive', and (ii) to the significant import flows from Kazakhstan, which has analogous issues. Notice that also large river systems can become environmentally vulnerable when they are overexploited, as in the case of the Indus river. This is one of the largest water resources in the world, but its use to feed wide irrigation systems explains why Pakistan ranks among the largest consumers of riverine resources worldwide (see figure 4(b)). Finally, it is worth to notice that, despite

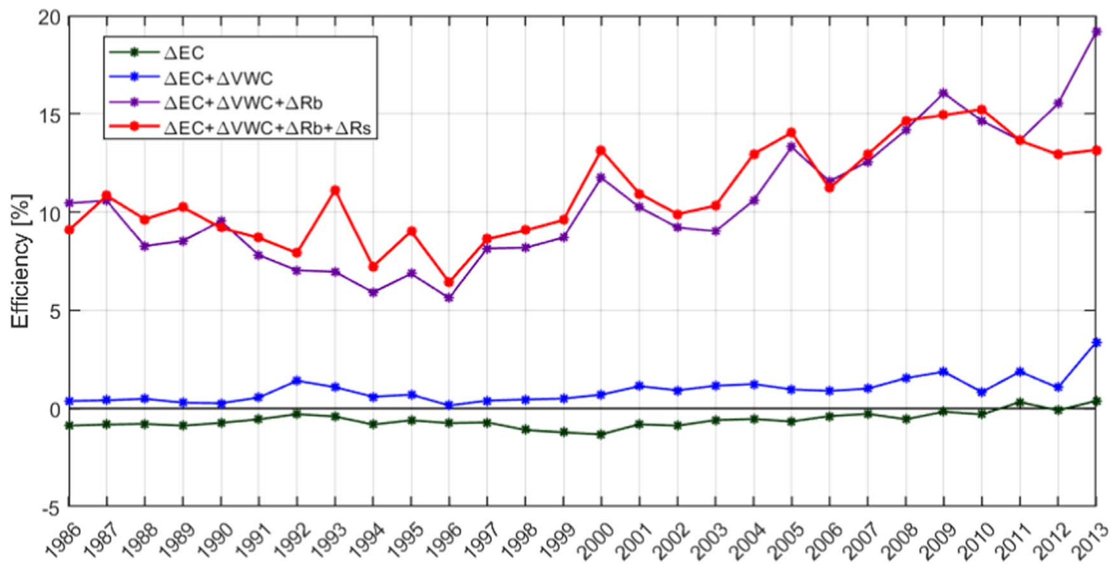


**Figure 5.** The global yearly efficiency of the EVRW trade network from 1986–2013. The ordinate axis reports the relative network efficiency (in percentage), which is the ratio between the global savings (or losses) of EVRW due to food trade divided by the total EVRW yearly embedded in the global food production under the business as usual scenario. The total efficiency (red line) is the result of a combination of disparities between importers and exporters related to: the environmental value per unit of surface water used (ΔEC), the virtual water content of each product (ΔVWC), the percentage of blue water in each product ($\Delta Rb$), and the surface water use ($\Delta Rs$).

China and India being in the first positions in the country ranking (see figure 4(a)), an average Indian or Chinese citizen has a lower EVRW consumption than the world average.

### 3.3. The sustainability of the food trade under a riverine perspective

From a riverine water resources point of view, when a trade flow saves EVRW it can be defined as an efficient flow; conversely, when the opposite happens the flow is defined as inefficient. For example, the flows from Turkey and Egypt to Italy in figure 1(b) are inefficient flows. Through these two flow categories, we can quantify the global efficiency of the EVRW network by evaluating, for each trade link, the difference in the EVRW under two different scenarios: a business as usual scenario and a self-reliant scenario, where all food is produced and consumed locally, at the country scale (the procedure to measure the EVRW efficiency is detailed in section S5 of the supplementary material).

Figure 5 reveals (see red line) that the international food trade has increasingly led to save environmental value over the considered period: annually, trade-induced EVRW saving ranged from 6% to 15% . However, despite the significant global EVRW efficiency of the food network, the number of trade efficient and inefficient connections is about the same, with the former being about 48% of the total number of links. The reason of the global EVRW efficiency is shown in figure 6: the higher are the riverine resources involved in the transaction the higher is the percentage of efficient links. Therefore, there is a global tendency to save riverine value when 'strong' trade relationships are considered. As a consequence when a high EVRW threshold is focused on, namely the upper tail of the cumulative distribution is considered, only few inefficient links occur (e.g. the products exported from Uzbekistan and Australia to China).

The global efficiency of the EVRW network is driven by a combination of differences between exporters and importers (see section S5 in supplementary material); it partly depends on the agricultural practices and climatic conditions of the main exporters and importers (i.e. the different amount of riverine water used per unit of product) and partly relies on the uneven distribution of riverine water resources worldwide. For example, the flow from Brazil to Egypt is saving EVRW (see the map in the inset of figure 6) both because Brazil has less vulnerable riverine ecosystems than Egypt and because a significant part of the Brazilian agriculture relies on rainwater rather than blue water.

In order to unveil the main disparities between exporters and importers that influence the overall EVRW efficiency value, figure 5 shows the global efficiency value assessed by progressively considering an increasing number of differences between trading partners. Firstly (see the green line in figure 5), only the different country-specific environmental values per unit of surface water use are considered while other differences among importers and exporters are neglected; in this case, the business as usual scenario is compared to a scenario where the same amount of surface water is consumed in the importer and exporter country for any food product. Under these conditions the global EVRW network is slightly inefficient. Thereafter, also the dissimilarity in the VWC are taken into

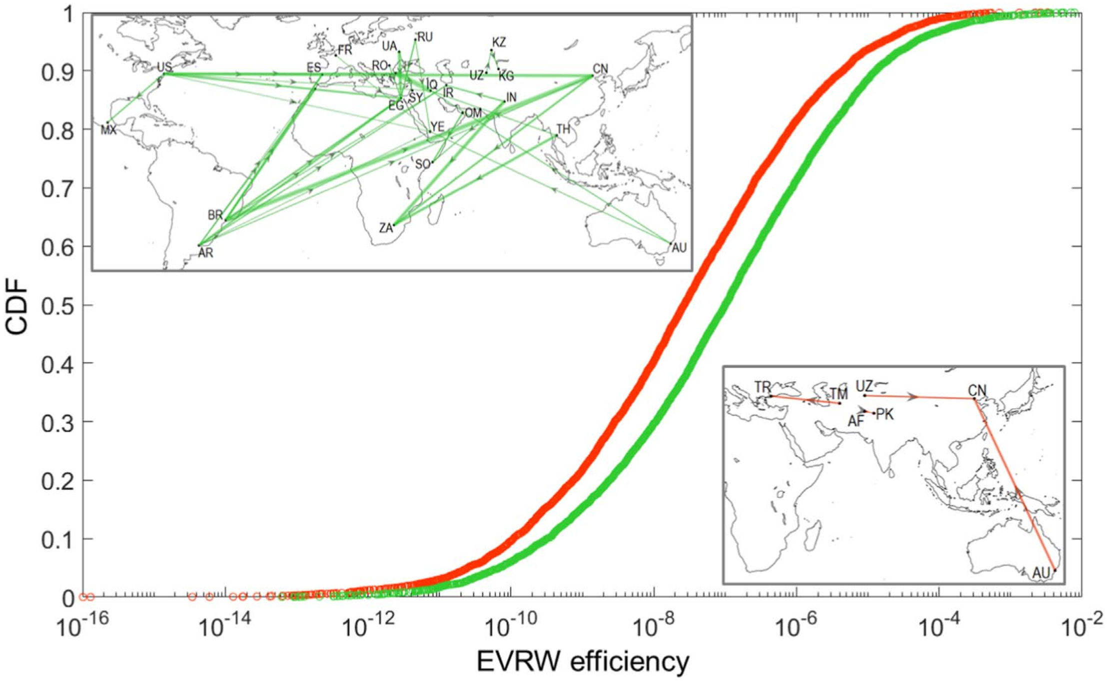


**Figure 6.** The cumulative distribution functions (CDF) of the EVRW efficiency of the international food trade links in 2013. The green curve refers to the efficient links (about 48% of the total number of links), while the orange curve is the CDF of the inefficient links (about 52%). The abscissa reports the absolute value of the EVRW efficiency (or inefficiency) of each trade link expressed in log scale and as a percentage of the total EVRW embedded in global food production, under the business as usual scenario in 2013. Maps in the insets show the major trade flows related to a global saving (in green) or loss (in orange); all values greater than $10^{-3}$ (in absolute value) are reported. The link widths are proportional to the efficiency/inefficiency value. Country acronyms are in ISO 3166-1.

account (e.g. in 2013 to produce one ton of wheat the amount of water used ranged from 400 to 9000 $m^3$, depending on the considered country). Finally, also the differences in the percentage of blue water and surface water use are considered (see purple and red lines in figure 5, respectively). Overall, the fact that exporters tend to exploit lower amounts of blue water than importers (see purple lines) appears to be the most significant factor that improves the global EVRW efficiency. The latter result is in line with the findings of Konar *et al* [41], who investigated the efficiency of the food trade network under a volumetric perspective and found a global saving of $119 \times 10^9$ $m^3$ of blue water in 2008 due to the international trade.

## 4. Conclusion

In the last decades, globalization of water resources by food trade has dramatically changed the human impact on riverine ecosystems. The scientific community is putting a lot of efforts to study this issue, in particular with a view to the traded (virtual) water volumes. However, the volumetric point of view tells only a part of the story, because the withdrawal of the same water volume can have very different environmental consequences depending on the particular ecosystem where water is withdrawn. Our work investigated this aspect and focused on the environmental value of the water withdrawn from riverine systems.

We have shown that water globalization drives a global trade of environmental value of surface water and local threats to fluvial ecosystems are induced by the food demand across the globe. Through food imports, consumers can affect river environments thousands of kilometres away from them. However, despite the growing geographical gap between food consumption and production, international trade has been increasingly reducing the impact of global food production on riverine systems over time, compared to an ideal situation where all food is produced locally. We find an average annual global saving of riverine environmental value of 11% due to the international food trade. Nevertheless, a large number of trade links is still largely inefficient and several countries rely on vulnerable riverine ecosystems locally or abroad.

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## Data availability

The data that support the findings of this study are available from the corresponding author upon request.

## Contributions

All authors designed the study, discussed the results and contributed to the manuscript. IS wrote the first draft of the manuscript and performed the analyses.

## Competing interests

The authors declare no competing financial interests.

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