Supplementary Material of “The globalization of riverine environmental resources through the food trade”

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S1. Assessment of the virtual (surface) water trade

From the FAOSTAT database, we collected twenty-eight years (1986 - 2013) of food production and bilateral trade data for 270 food and agricultural commodities, including crops, crop-derived goods, and animal products (the detailed list is reported in the Supplementary data as an excel spreadsheet). For each year, trade data were stored in (non-symmetrical) product-specific matrices, whose \((i, j)\) element corresponds to the export of the product from country \(i\) to country \(j\). The matrix size equals the number of countries and changes from year to year, according to political-administrative arrangements (e.g., the collapse of the USSR in 1990-91). In the FAOSTAT database, the trade from \(i\) to \(j\) is sometimes reported by country \(i\) and country \(j\) with discordant values; in these cases, we adopted a reconciliation method that takes into account the degree of reliability of each reporting country (see [1]). Further details about the matrix construction and country arrangements are given by Carr *et al.* [2] and Tamea *et al.* [3].

We adopted the approach described by Tuninetti *et al.* [4] in order to take into account the temporal variations of the water footprint per crop unit due to yield trends.
Accordingly, the yearly country-specific virtual water content (VWC) for each primary crop was assessed as

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

where subscripts \(c\) and \(t\) refer to the country and year considered, respectively, \(\overline{VWC}_{c,1996-2005}\) is the average virtual water content (over the period 1996 - 2005) given by Mekonnen and Hoekstra [5], \(\overline{Y}_{c,1996-2005}\) is the corresponding average yield, and \(Y_{c,t}\) is the country yield in the year \(t\). Yield time-series at the country scale for each primary crop are provided by the FAOSTAT database.

The evaluation of the virtual water content of crop-derived products required a number of steps. Firstly, we collected (i) the VWC of the primary input crops adjusted with the product fraction (i.e., ton of crop-derived product obtained per ton of primary input crop) and (ii) the value fraction (i.e., the market value of the crop-derived product divided by the aggregated market value of all crop products resulting from one primary input crop). Both fractions are provided by Mekonnen and Hoekstra [5]. Secondly, a crop-derived product is often obtained from both local and imported primary crops that have different VWCs, due to different yields and climatic conditions. To take these differences into account, we assessed the VWC of the primary input crop (at the national and yearly scale) as the weighted average of the VWC related to the domestic production and the VWCs of the imports, where the weights are the tons of the domestic production (minus exports) and the imported tons. Thirdly, some crop-derived products are obtained through successive process steps; in these cases, starting from the primary crops we estimated the VWC of the increasingly complex products at each step, considering the weighted average between domestic production and imports for each intermediate product. An example of the estimation of the VWC for a crop-derived product is reported in the next section.

Differently from the primary and crop-derived commodities - where the temporal variability of the virtual water content was considered - for animal products we used the time-averaged VWC given by Mekonnen and Hoekstra [6]. This choice is due to a lack of reliable data about the country-specific composition of feed for each animal type.

Once food trade and production data of each commodity were converted into virtual water values, the blue virtual water component was estimated using the country-specific ratio (herein called \(Rb\)) of the average blue VWC to the overall VWC, both given by Mekonnen and Hoekstra [5] for each food commodity. The blue water component includes both groundwater and surface water resources. To estimate the surface water
contribution, we used the “Global Map of Irrigation Areas” (version 5.0) dataset [7]; considering the percentage of the area equipped for irrigation served by surface water resources (herein called $Rs$).

Since the input products used to produce a crop-derived good usually have different geographical origins, the amount of surface blue water of each crop-derived product was assessed (at the national and yearly scale) as the weighted average of the surface blue water content related to domestic production and the surface blue water content of the imported input products, using the blue water volumes as the weights.

Once all the food commodities were converted into surface virtual water values, they were added together to obtain (i) the total yearly surface virtual water transferred between pairs of trading partners and (ii) the total yearly amount of surface water used within each country to produce food. The trade pattern was reconstructed considering all the food commodities, while the total amount of surface water used for food production was estimated without considering secondary products (e.g., bread) in order to avoid double-counting issues. For further details about this topic, see D’Odorico et al. [8].

**S2. The assessment of the virtual water content of a crop-derived product: a simplified example**

Section S1 presented the data and methods used to assess the global virtual surface water network. In order to provide more detailed information about this topic, this section illustrates an example (the dry pasta) of the estimation of the virtual water content of a crop-derived product.

The VWC of each crop-derived product is assessed based on the VWC of the primary input crop, adjusted with the product fraction and value fraction. The primary input product is the good required to produce the considered crop-derived product. The product fraction, $fp$, is defined as the weight of a derived product obtained per ton of root product. For example, if with one ton of wheat 0.8 tons of wheat flour are produced, the product fraction is 0.8. The value fraction, $fv$, is the market value of the crop-derived product divided by the aggregated market value of all crop products resulting from one primary input crop. The list of primary input products and the product and value fractions are given by Mekonnen and Hoekstra [5].

The alluvial diagram in Fig.S1 shows the steps to assess the virtual water content of the dry pasta, taking into account that usually a derived product is obtained from
Figure S1: Steps to assess the virtual water content (VWC) of the dry pasta considering a simplified trade network composed by 3 countries: A, B and C. Intra-panels numbers refer to the country-specific virtual water contents, while numbers inside panels refer to the tons of product. The graph was built using RAWGraphs (https://rawgraphs.io/).
both local and imported root products (in this case wheat and wheat flour). For the sake of simplicity, the example in Fig.S1 describes a very simple network composed by 3 countries: A, B and C. According to Mekonnen and Hoekstra [5] the main raw material of dry pasta is wheat flour and the latter derives from the milling process of wheat. Due to different yields and climatic conditions, wheat production in each country is associated to different VWCs; thus, to produce one ton of wheat different amounts of water are needed in country A, B, and C. In the example we assume 2500 m$^3$, 1000 m$^3$, and 1500 m$^3$ of water are used, respectively.

The VWC of the wheat flour produced within, say, country B (see Fig.S1) is assessed considering that the average amount of wheat used in B comes both from domestic production and imports from A and C. Therefore, the average VWC of the wheat employed in the production process of wheat flour is assessed as the weighted average between the VWC related to domestic production (1000 [m$^3$/ton]) and the VWC of the imports (2500 [m$^3$/ton] and 1500 [m$^3$/ton]), where the weights are the tons of the domestic production minus the exports (1.8 million tons - 1 million tons) and the imports tons of wheat (0.5 million tons from A and 0.5 million tons from C). Finally, the average VWC of the wheat employed in B to produce wheat flour is adjusted applying the value fraction (0.79) and the product fraction (0.80) as follows:

$$\frac{(1000 \cdot 0.8 \cdot 10^6) + (2500 \cdot 0.5 \cdot 10^6) + (1500 \cdot 0.5 \cdot 10^6)}{0.8 \cdot 10^6 + 0.5 \cdot 10^6 + 0.5 \cdot 10^6} \cdot \frac{0.79}{0.80} = 1540 \ [m^3/ton]. (S.2)$$

Similarly, the VWC of the dry pasta produced in B also depends on the wheat flour imported from foreign countries (see Fig.S1), and thus can be estimated as:

$$\frac{(1540 \cdot 0.6 \cdot 10^6) + (1950 \cdot 0.7 \cdot 10^6) + (1380 \cdot 0.3 \cdot 10^6)}{0.6 \cdot 10^6 + 0.7 \cdot 10^6 + 0.3 \cdot 10^6} \cdot \frac{1}{1} = 1690 \ [m^3/ton]. (S.3)$$

In our work, in order to assess the country-specific VWC of each crop-derived product, this conceptual scheme was applied to all products in the global trade network at the annual scale. Notice that the country-specific VWC of each crop-derived product needs to be reconstructed every year because: (i) in the food trade network, both the active links and the corresponding exchanged flows change; (ii) the VWCs of the primary crops are not constant in time but depend on the yield trends (see Eq.(S.1) in the previous section).
S3. Assessment of the environmental value of riverine waters embedded in food commodities

Surface water resources exhibit very heterogeneous characteristics and, thus, a same volume of withdrawn water does not have the same environmental value in different places worldwide. Therefore, in order to quantify the share of responsibility that each country (or individual) has on the overall degradation of world rivers through the consumption, production and trade of food commodities, we multiply the amount of surface water used to produce food goods by the Environmental Cost (EC) index proposed by Soligno et al. [9]. The index varies in space and quantifies the overall impact on fluvial ecosystems of a unitary amount of water withdrawn in a given river section. Here, we provide a brief description of the EC index and how we used it to obtain the EVRW network analysed in this work.

In a given river section, the local environmental cost per unit length ($ec_w$) of a unitary water withdrawal $W$ is defined to be proportional to the river discharge reduction caused by the withdrawal. Namely, $ec_w : W = ec_{max} : Q$, where $ec_{max}$ is the maximum environmental cost per unit length which occurs when the entire river discharge ($Q$) is depleted. The value of $ec_{max}$ is related to the relevance of the specific river environment considered. Since rivers are complex environments characterized by multiple and interplaying processes, such relevance depends on various fluvial characteristics (e.g., width of the riparian belt, biodiversity richness, transport of sediments and chemicals). Soligno et al. [9] demonstrated that $ec_{max}$ can be evaluated in each river section by considering site-specific characteristics, which can be related through physically-based power laws to the river discharge. Namely, $ec_{max} = k(\alpha) \cdot Q^\alpha$, where $\alpha$ is a parameter that typically falls within the interval [0,1] and depends on the focused river characteristic; $k(\alpha)$ is a proportionality constant. This latter is the same in all river sections, once $\alpha$ is chosen, and it is assessed according to the constraint that the EVRW of all the world’s surface water resources is unaffected by $\alpha$. The local environmental cost per unit length of a water withdrawal is therefore equal to

$$ec_w = k(\alpha) \cdot \frac{W}{Q^{1-\alpha}}. \quad (S.4)$$

The index $ec_w$ is a metric of the local impact of $W$ and does not consider downstream effects. However, the subtraction of $W$ alters the discharge from the section where water is withdrawn down to the river mouth ($S_M$). Therefore, the overall environmental cost of $W$ is evaluated as the sum of the environmental cost per unit length generated downstream by the withdrawal $W$. It follows that the overall environmental cost, $EC_w$,
of a water withdrawal in a river section $S_W$ of the river network is

$$EC_w = \int_{S_W}^{S_M} ec_w(s) ds = k(\alpha) \cdot \int_{S_W}^{S_M} \frac{W}{Q(s)^{1-\alpha}} ds,$$

(S.5)

where $s$ the is curvilinear abscissa along the river.

As the present study aims to describe the comprehensive impact on world rivers and not the impact related to a specific fluvial characteristic (i.e., a specific value of $\alpha$), we employed a formulation of $EC_w$ able to embed different features of the river environment at the same time. Therefore, different $\alpha$ values in the interval $[0, 1]$ are considered, and weighted through a triangular kernel. A sensitivity analysis (see Section S4) shows that the main results of this work do not change by using different peak positions in the triangular kernel. For this reason, reported results refer to a symmetric kernel with peak at $\alpha = 0.5$. The environmental cost of a water withdrawal is thus

$$EC_w = \int_{S_W}^{S_M} \int_{0}^{1} Ker(\alpha) \cdot k(\alpha) \cdot \frac{W}{Q(s)^{1-\alpha}} d\alpha ds,$$

(S.6)

where $Ker(\alpha) = 4\alpha$ when $\alpha \leq 0.5$, and $Ker(\alpha) = 4 - 4\alpha$ when $\alpha > 0.5$.

We computed the $EC_w$ value related to a unitary surface water withdrawal with a 0.5° spatial resolution, adopting the global drainage direction map DDM30 [10] and the annual average river discharges obtained from the pristine scenario of the WaterGAP 2.2c model [11, 12, 13, 14]. The pristine river discharges were averaged over the period 1901-2013. The $EC_w$ computation was obtained for each fluvial section of the world river network.

In order to evaluate the environmental value enclosed in the surface virtual water network (see Section S1) we adopted a country-specific $EC_{w,c}$ value. No global data are available about the specific river sections where water is withdrawn for irrigation; however, it is reasonable to assume that water is generally withdrawn in river sections where it is more abundant. Therefore, we assessed the country-specific $EC_{w,c}$ as the weighted average of the $EC_w$ of each cell within the considered country using the river discharge of each cell as the weight. Finally, surface water trade links were multiplied by the $EC_{w,c}$ value of the exporting countries, while the volumes of surface water related to national food production were multiplied by the $EC_{w,c}$ of the corresponding country. In this way, we obtained the dataset (at the country and yearly scale) of the productions, exports and imports of the environmental value of riverine waters.
**Figure S2:** Influence of the peak position of the triangular kernel on the evaluation of the country-specific weighted average environmental cost, $EC_{w,c}$. The case with the peak in $\alpha = 0.5$ (adopted in the main text) is compared with kernels with peak in $\alpha = 0$ (a), $\alpha = 0.25$ (b), $\alpha = 0.75$ (c), and $\alpha = 1$ (d). Panel (e) compares the weighted average environmental costs estimated using a triangular kernel (peak in $\alpha = 0.5$) and a uniform kernel within the interval [0 1]. Circle sizes are proportional to the country area; axis are in log-scale.

**S4. A sensitivity analysis of the average environmental cost of a water withdrawal at the country scale**

In Section S3, we described the approach followed in this work to quantify the environmental value of the riverine water involved in the production and trade of food commodities. To this aim, we adopted the Environmental Cost (EC) index [9] and used a formulation of $EC_w$ able to concurrently consider different river characteristics (i.e., we did not focus on a specific value of $\alpha$). Therefore, all the values of $\alpha$ within the range [0 1] were considered and weighted through a triangular kernel with the lower vertexes in $\alpha = 0$ and $\alpha = 1$, and the peak in $\alpha = 0.5$ (see Eq.(S.6)). Finally, the country-specific $EC_{w,c}$ was assessed as the weighted average of the $EC_w$ of each cell within the considered country, using the river discharge in the cell as the weight.

Fig.S2 compares (at the country scale) the $EC_{w,c}$ values adopted in this work
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(i.e., assessed with the peak of the triangular kernel in $\alpha = 0.5$) with those obtained considering different positions of the peak (i.e., in $\alpha = 0, 0.25, 0.75, \text{and } 1$. The case of a uniform kernel is also considered). The scatter plots in Fig.S2 show that $EC_{w,c}$ values are quite insensitive to the exact shape of the kernel.

Notice that when $\alpha$ tends to one the natural river discharge has a limited (or null) effect on $ec_{w}$ (see Eq.(S.4)). As a consequence, when high values of $\alpha$ are adopted, the length of the downstream river network turns out to be the main factor that influences the environmental cost of a surface water withdrawal. Hence, for countries such as Brazil and Canada – which are characterized by long river networks – the more the peak of the distribution approaches $\alpha = 1$, the more the country-specific $EC_{w,c}$ increases. The opposite occurs for countries such as South Africa and Spain.

S5. Assessment of the EVRW efficiency of the food trade network

We assessed the EVRW efficiency by comparing the business as usual trade network (see Fig.S3) with a scenario in which countries are self-reliant. This latter scenario assumes that all food is produced and consumed within the same country instead of being imported from foreign regions. The difference between these two scenarios provides a measure of the efficiency (or inefficiency) of each trade flow. The global EVRW efficiency of the food trade network is obtained by summing up the contributions of all the trade links.

Consider the export of a basket of food products $P_{i,j}$ from country $i$ to country $j$. The difference between the EVRW for the production of the products in the importer country $j$ (i.e., the self-reliant scenario) and in the exporter country $i$ (i.e., the business as usual scenario) defines the EVRW efficiency of the trade flow, $\Delta EVRW_{i,j}$. It can be assessed as

$$\Delta EVRW_{i,j} = \sum_{p \in P_{i,j}} (F_{i,j}^p \cdot VWC_{j}^p \cdot Rb_{j}^p \cdot Rs_{j} \cdot EC_{w,j} - F_{i,j}^p \cdot VWC_{i}^p \cdot Rb_{i}^p \cdot Rs_{i} \cdot EC_{w,i})$$

where the superscript $p$ refers to the considered product, which belongs to the product basket $P_{i,j}$; $F_{i,j}^p$ are the tons of the product $p$ that currently flow from $i$ to $j$; $VWC_{j}^p \cdot Rb_{j}^p$ denotes the blue virtual water content of product $p$; and $Rs$ is the ratio of surface water to the total blue water consumed within the considered country. Finally, $EC_{w}$ stands for the environmental value per unit of surface water used. When $\Delta EVRW_{i,j} > 0$, the trade relationship reduces the environmental value of the riverine water embedded in food products (i.e., the considered trade flow leads to save EVRW).
Eq.(S.7) enables to distinguish the different contributions to $\Delta EVRW_{i,j}$. For each product, $VWC_j^p$ and $VWC_i^p$ influence the EVRW efficiency through the difference in the virtual water productivity between $i$ and $j$. This contribution reduces the efficiency of the trade connection when the importer country requires less water per ton of $p$ than the exporter (i.e., when $VWC_j^p < VWC_i^p$). The blue water share depends on the water supply system adopted to produce $p$ in the considered country; it can range from fully irrigated ($Rb = 1$) to fully rain-fed ($Rb = 0$). Therefore, the efficiency value improves when the exporter (importer) country reduces (increases) the irrigation water use. $Rs_j$ and $Rs_i$ focus on the different country-specific exploitation rate of surface water in agriculture. Finally, the $EC_w$ typical of each country depends on the sensitivity of domestic rivers to water withdrawals and it is the only term in Eq.(S.7) unrelated to agricultural practices.

Since there are imported-goods that cannot be offset by domestic production (e.g., tropical fruits imported by Northern European countries), in the generic link from $i$ to $j$ the efficiency was calculated considering only the imported-goods that could be conceivably produced in $j$. To this end, for each trade flow we included only the products with known $VWC_{1996-2005}^p$ in both countries (for the crop-derived commodities the $VWC_{1996-2005}$ value of their input products was considered). Therefore, in Eq.(S.7) $P_{i,j}$ is a subset of the products actually exported from $i$ to $j$.

In order to identify the main differences between exporters and importers that influence the overall efficiency value, the EVRW efficiency was assessed in four steps by considering an increasing number of disparities between $i$ and $j$. Firstly, only the dissimilarities in terms of $EC_w$ were considered, while the other differences among importers and exporters were neglected (see green line in Fig.5 in the main text); in this case, the business as usual scenario was compared to a scenario where country $j$ consumes the same amount of surface water of country $i$ to produce any commodity, thus

$$\Delta EVRW_{i,j}^{step_1} = \sum_{p \in P_{i,j}} (F_{i,j}^p \cdot VWC_i^p \cdot Rb_i^p \cdot Rs_i) \cdot (EC_{w,j} - EC_{w,i}).$$ (S.8)

Thereafter, also the dissimilarities on the VWCs typical of each pair of countries were taken into account, while the impact on the EVRW efficiency of both $Rb$ and $Rs$ were still neglected, thus $Rb_j^p = Rb_i^p$ and $Rs_j = Rs_i$ (see blue line in Fig.5). The third step was performed including also the effect of the different product-specific blue water shares of importers and exporters (see purple line in Fig.5). Ultimately, the EVRW efficiency was estimated as in Eq.(S.7), where $EC_{w,j} \neq EC_{w,i}$, $VWC_j^p \neq VWC_i^p$, $Rb_j^p \neq Rb_i^p$, $Rs_j \neq Rs_i$ (see red line in Fig.5).
Figure S3: the EVRW world network in 2013. Blue and light blue links refer to EVRW flows higher than 0.25% and 0.001% of the total EVRW internationally traded in 2013, respectively. The line widths are proportional to the embedded EVRW. The network nodes correspond to the barycentre of the population distribution within each country (available at http://www.cepii.fr). Country names are in ISO 3166-1.

Finally, our efficiency measure focuses exclusively on the impact of food production on riverine systems and it does not take into account any social, economic or resource limitations (e.g., the availability of agricultural land).

S6. Supplementary Figures

In this section, we report the EVRW network (i.e., Fig.S3) and some figures that highlight the temporal dynamics of the inter-country network of the environmental value of riverine waters.

The global EVRW network in Fig.S3 points out the links by which final food consumers may impact rivers very far away from consumption places. In 2013 the number of countries involved in the food trade (as importers and/or exporters) was about 200, whereas the major EVRW flows involved a limited number of countries, such as Australia, Thailand, USA, Kazakhstan, and South Africa as exporters and China, Italy and Germany as importers.

Fig.S4 is analogous to Fig.1 in the main text but focuses on Germany and India; it shows three snapshots of the various routes between local fluvial ecosystems and foreign consumer demand corresponding to different years (i.e., 1991, 2001 and 2013). In the period 1986-2013, Germany has always been in the top 10 importers of EVRW (see
Figure S4: Snapshots of the major EVRW import flows of Germany (in blue) and the major EVRW export flows of India (in red) in 1986, 1999, and 2013. The line widths are proportional to the embedded EVRW; the network nodes correspond to the barycentre of the population distribution within each country.
Figure S5: The major consumers of EVRW embedded in food products from 1986 to 2013. The shares in the colorbar are related to the average annual EVRW globally consumed during the considered period; the colorbar is in log-scale.

also Fig.2b) and Fig.S4 shows the remarkable pattern of major foreign river systems impacted by an average German citizen though his/her consumption of imported food commodities. The main temporal changes concern African partners, even if the disappearance of the link with Sudan in 2013 is due to a lack of data in the FAOSTAT database regarding Sudan in 2012 and 2013 (the same issue can be noticed in Fig.S6 as well). India, on the contrary is a great exporter of EVRW; this despite the production of food in India being strongly reliant on groundwater resources. Indeed, only 35% of the total blue water consumed in India for food production comes from surface water resources. EVRW flows spread heterogeneously from India and their number and intensity have strongly increased over time, in particular in the last years.

The rising living standard and the the growth of the world population have led to increase the consumption of food goods over time; this fact has clearly influenced the amount of EVRW impacted by the food sector, as can be seen in Fig.S5 where the patterns of the EVRW embedded in the food consumed within each country over the period 1986 - 2013 are reported. Although the EVRW embedded in consumed food exhibits a growing or stable trend for most countries, a much more heterogeneous pattern has characterized the net export flows over time, as already shown in Fig.2b and as displayed in Fig.S6 (where more countries are considered).
**Figure S6:** Net export flows of EVRW embedded in food products of the main importers and exporters from 1986 to 2013. The shares in the colorbar are related to the average annual EVRW exchanged during the considered period.

**Figure S7:** The geographical division into the nine macro-region analysed in Fig.3; 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.
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