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Analysis

International trade of scarce water☆

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ABSTRACT

Recent analyses of the evolution and structure of trade in virtual water revealed that the number of trade connections and volume of virtual water trade have more than doubled over the past two decades, and that developed countries increasingly import water embodied in goods from the rest of the world to alleviate pressure on domestic water resources. At the same time, as demand continues to increase and climate change threatens to alter hydrological cycles, water scarcity is a growing problem. Does research into virtual water trade need to consider water scarcity and differentiate flows out of water-scarce regions from flows out of water-abundant regions? Previous studies sum and compare virtual water volumes originating in countries experiencing vastly different degrees of water scarcity. We therefore incorporate water scarcity into an assessment of global virtual water flows. We use input–output analysis to include indirect virtual water flows. We find that the structure of global virtual water networks changes significantly after adjusting for water scarcity.

1. Introduction

In the past, water policy schemes aimed at alleviating water shortage focused on the development of irrigation infrastructures for expansion of irrigated area. However, such expansionary policies have proven insufficient as demand for water continues to increase (Konar et al., 2012; Postel, 1999). Moreover, the development of further irrigation projects has been criticized given growing concerns over the adverse environmental effects of large dam projects (McCartney et al., 2000). As a result, water shortage affects 40% of the current global population (Hinrichsen et al., 1997). In the coming decades, population growth and economic development, coupled with increasing scarcity of water, may lead to further increase in costs of water supply development. This is threatening the economy of many river basins, and thus drawing countries that share these basins into possible water conflicts (Beach et al., 2000; Dinar and Dinar, 2000; Just and Netanyahu, 1998; Spulber and Sabbaghi, 1994). Global climate change may exacerbate scarcity problems as the variability of water supply is expected to change (Kenneth and Major, 2002). Coping with the effects of climate change on water will require stronger demand management measures to enhance the efficient usage of water.

International virtual water trade has been advocated by several researchers (Allan, 1997; Chapagain et al., 2006; Yang et al., 2006) to help to distribute uneven endowments of water in the world and achieve global water use efficiency. Ridoutt and Pfister (2010a) argue that 90% of water extraction is associated with the life cycle of products rather than with direct use by households, thus lending importance to the analysis of water-intensive supply chains. Dalin et al. (2012) showed that the number of trade connections and volume of virtual water trade have more than doubled over the past two decades, and that developed countries increasingly import water embodied in goods from rest of the world to alleviate pressure on domestic water resources. Some studies based on trade patterns for certain water-intensive crops support this view by showing a direct relationship between water scarcity and grain imports. However, many other authors have found no relationship between virtual water trade and water scarcity (Ansink, 2010; Kumar and Singh, 2005; Ramirez-Vallejo and Rogers, 2004; Verma et al., 2009). Examining global virtual water flows is hence useful for understanding the influence of international trade on water resources, and one question posed by this study is whether the consideration of water scarcity significantly affects the patterns of global trade in virtual water.

The impact of economic activity cannot be measured in terms of quantities of water used alone. The consumption of water entails a range of consequences such as for water quality, resources and availability, and more indirectly for example for biodiversity and human health. Probably the first researchers to propose a scarcity weighting for water use data were Frischknecht et al. (2006b) (see an update in Frischknecht et al., 2006a). Their Ecological Scarcity method was applied to water requirements of biofuels (Frischknecht et al., 2009) and even in an input–output analysis of Swiss

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consumption and production (Jungbluth et al., 2011). Since then, water use, water pollution or degradation, and water scarcity or (un)availability have recently been successfully included in Life-Cycle Assessment (LCA; Boulay et al., 2011; Pfister et al., 2009; Ridoutt and Pfister, 2013), notably by Pfister, Ridoutt and colleagues in Switzerland and Australia (Cooney, 2009). This is often done via scarcity weights or characterization factors (Hanafiah et al., 2011; Ridoutt and Pfister, 2013), and with additional research aims such as characterizing individual products (Gerbens-Leenes et al., 2012; Jefferies et al., 2012; Ridoutt and Pfister, 2010b), impacts on human health (Boulay et al., 2011), land resources stress (Pfister et al., 2011), or even fish species disappearance (Hanafiah et al., 2011). Kounina et al., 2012 provide a comprehensive review of methods applied in LCA for measuring freshwater use.

The approach most often used in LCA for quantifying the virtual water flows is a bottom-up technique called process analysis. A prominent detailed study with a global scope is that by Pfister et al. (2011). In process analysis virtual water flows are calculated by taking into account some but not all indirect virtual water requirements. While bottom-up techniques can offer high product resolution enabling for example to differentiate the water intensity of different crops, they may be affected by the truncation of the assessment's system boundary, resulting in indirect parts of virtual water flows remaining allocated to producers, not consumers (Feng et al., 2011). Approaches using Multi-Region Input–output (MRIO) tables linked to water accounts are able to make a clear distinction between direct and total indirect water consumption[1] (Arto et al., 2012; Feng et al., 2011), and can hence be used to derive the complete virtual water flows (see *Supporting Information* (*SI*) *S1*). The "classical" water footprint approach using process-based analysis addresses a slightly different research question, focusing often on water embodied in bilateral trade rather than on total virtual water in supply chains to final demand.

Steen-Olsen et al. (2012) published an interesting MRIO-based water footprint study using a hybrid approach pioneered by Ewing et al. (2012). The advantage of this hybrid approach is brought about by the detailed physical satellite accounts by country, type and sector that are appended to the conventional MRIO table. However this advantage applies only to the direct, and 1st-order indirect water footprint effects. This is because for indirect supply-chain effects, Steen-Olsen's hybrid analysis relies on the resolution of the MRIO just as any other MRIO water footprint study. In addition, the databases that are used to construct these physical satellites do not hold sufficient information on the identity of the using sectors, necessitating some prorating procedure, and leading to allocation errors.[2]

There exist a number of MRIO-based virtual water accounts and studies for the world (EXIOBASE, 2012; Feng et al., 2011; WIOD, 2012), however there are two specific shortcomings in these accounts which we seek to address here. First, many areas of critical water problems exist in developing countries that are not distinguished in existing MRIO databases. Second, existing MRIO databases group together countries characterized by widely varying degrees of water scarcity. Calculating global water footprints by adding the use of scarce water in one region to the use of abundant water in another region makes little sense because such footprints would not be able to indicate regions and/or commodities in need of policy measures to mitigate water-related problems (Feng et al., 2011; Pfister et al., 2009). In response, for the first time, we characterize national footprints and trade balances in terms of scarcity-weighted water for 187 individual countries. In addition, we apply Structural Path Analysis to identify major global routes conveying pressure on water resources from centers of consumption to regions of water scarcity.

So far, most virtual water concepts reflect water consumption without accounting for water scarcity. Indeed the Virtual Water concept was originally proposed to describe an alternative strategy to address water scarcity. Differences in resource endowment and demand conditions are some of the basic reasons for trade to take place between countries. It is clear that regions can gain from trade if they specialize in goods and services for which they have a comparative advantage. A region is therefore considered to have comparative advantage in producing a water-intensive good if the opportunity cost of producing it is lower in that country than in its trading partners (Verma et al., 2009). By reporting on total national water use, existing input–output satellite accounts ignore such comparative advantage in terms of water resource endowments and increasing water demand conditions. Further, it makes difficult to interpret a situation in which the opportunity cost of lower water consumption in terms lower water footprint could be higher than that of with higher water footprint, which depends upon where the water is sourced (Ridoutt and Pfister, 2010b). Our study addresses this concern by using a water scarcity index as a weight for converting total water use into scarce water use, incorporating water scarcity as a factor into global virtual water flow concept.

2. Methods

We employed the Eora global Multi-Region Input–output (MRIO) database (Lenzen et al., 2012a) containing an intermediate demand matrix **T**, final demand **y**, and value added **v**. The Eora MRIO provides a completely harmonized and balanced world MRIO table, drawn together from major sources such as the UN System of National Accounts (SNA), UN COMTRADE, Eurostat, IDE/JETRO, and many national input–output tables. It is publically available free of charge for research use at www.worldmrio.com. We extended this with a satellite account **Q** holding information on water use taken from the FAO's AQUASTAT database (FAO, 2012), which covers 204 countries (17 more than Eora). We choose the year 2000 for our analysis of global virtual water flows because the coverage of countries in the United Nations Official Country Database, on which the Eora MRIO relies, is best for years around 2000.

Crop water requirement is the total water required for evapotranspiration, from planting to harvest for a given crop under the condition that water resource availability does not have constraining effects on crop yield (Alexander and West, 2011). The crop water requirement of each crop is computed using CROPWAT developed by the FAO (2012). In the MRIO database, 187 countries are represented at a resolution of 25–500 sectors each, and 15,909 sectors in total. The per-crop water usage from the FAO was attributed to the corresponding sectors in each country by using correspondences matrices[3] for each country that allocate crops in detailed HS + classification to the less detailed sectoral classifications used in the MRIO. (This step introduces some loss of fidelity since the original data sources had to be aggregated to fit into the less

[1] The term 'virtual water' refers exclusively to indirect consumption, while the term Water Footprint includes both direct use (e.g. turning on the faucet at home or drinking imported Perrier) and indirect use.

[2] For example assume that for Australia and New Zealand only an aggregated 'Vegetable and fruit growing' sector existed in the MRIO, and New Zealand had manufacturing sectors called 'Vegetable products' and 'Fruit products'. Assume further that Australia exported grown vegetables and fruit to New Zealand, reflected in two data sources: a) in the UN ComTrade database, distinguished by traded product and country origin but not by using industry, and b) New Zealand's import matrix, distinguished by using industry but not by traded product and not by country origin. This means that in an unsupervised prorating procedure, the New Zealand vegetable products sector would end up using Australian fruit and the New Zealand fruit products sector would end up using Australian vegetables. Such issues can only be dealt with by manual correction of mis-allocated entries, thus rendering the hybrid approach perhaps not much less labor-intensive than pursuing a detailed expansion of a regular MRIO database (the strategy pursued in the Eora MRIO database).

[3] For more on correspondence matrices see Lenzen et al. (2012a).

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detailed agricultural classifications used in the MRIO.) In addition to crop water demand we also included water use for animal raising, grazing, industrial, and domestic uses. These data were taken from the Water Footprint accounts published by Mekonnen and Hoekstra (2011). The data we used distinguished four kinds of water use: crop water (a mix of green and blue water, about 75% green), blue water used for animals, industrial production, and domestic supply, green water use for grazing, and gray water used for industrial production and domestic supply. These definitions differ slightly from the categories used in other studies (e.g. Hoekstra and Mekonnen, 2012) in that crop water is not split into blue/green components, and industrial production and domestic supply are not always split into their blue and gray components. Our categories directly follow those used in the Water Footprint accounts. Using correspondence matrices these water uses were pro-rated, on the basis of annual monetary turnover, among the industrial, beef and sheep, and domestic final demand sectors.

To calculate virtual water flows Q we subjected the extended MRIO system to Leontief's demand-pull model $Q = \mathbf{qx} = \mathbf{q}\ (\mathbf{I} - \mathbf{A})^{-1}\ \mathbf{y1^y} = \mathbf{m}\ \mathbf{y1^y}$, where the vectors $\mathbf{q}$ and $\mathbf{x}$ hold the water use coefficients and gross output of the 15,909 industry sectors, $\mathbf{I}$ is an identity matrix, $\mathbf{A} = \mathbf{T}\ \hat{\mathbf{x}}^{-1}$ is a coefficient matrix describing inputs into the production of these sectors, the hat symbol denotes diagonalization of a vector, $(\mathbf{I} - \mathbf{A})^{-1}$ is the so-called Leontief inverse, $\mathbf{m} = \mathbf{q}\ (\mathbf{I} - \mathbf{A})^{-1}$ is the so-called water multipliers, and $\mathbf{1^y} = \{1,1,\ldots,1\}$ is a summation operator (Leontief, 1966). In this model, final demand $\mathbf{y}$ (for example of households) at a particular location in the world drives industrial production $\mathbf{A}$, along a complex system of international supply chains mapped by the Leontief inverse $(\mathbf{I} - \mathbf{A})^{-1}$, ultimately requiring gross output $\mathbf{x}$ at various global locations of production, in turn requiring water use $\mathbf{q}$, and leading to a virtual water flow Q.

In order to incorporate water scarcity into the virtual water flow calculus we construct a new satellite account where water use entries are weighted so that they reflect the scarcity of the water being used. As a weight we choose a measure of water withdrawals as a percentage of the existing local renewable freshwater resources. We use the Water Scarcity Index for converting total water use into scarce water use. Global data for this measure are provided by the FAO (2012). According to the FAO, "*this parameter is an indication of the pressure on the renewable water resources*". Note that we use resource and scarcity information only as an input into a weighting procedure, and that we do not determine water stress or water scarcity as a result of or calculations. A similar measure, the Water Exploitation Index, was developed by Alexander and West (2011); it compares the water stress for various countries, but data are only available for the Asia-Pacific region. From the AQUASTAT database we obtained the percentage of total actual renewable freshwater resources withdrawn, also called the Water Extraction Index (*WEI*), for 170 countries for 2000. Of these, 24 did not have *WEI* data for the year 2000 so we used the data for the closest adjacent year (between 1995 and 2005), depending on availability. To bring the *WEI* coverage up to 187 countries, for 17 additional smaller countries with no *WEI* data available we assumed water was essentially perfectly abundant, or $WEI = 0.01$.

Scarcity conditions may differ between green and blue water, but we would argue that overall the two tend to be correlated since blue and green water come from similar sources of terrestrial water. The degree of relationship between blue and green water scarcity depends on the source of the blue water (rainfall, river, aquifer, etc.), the degree of human intervention, and on basin characteristics such as discharge rate and supply by snowmelt. In the absence of reliable data on either the relationship between blue and green water scarcity, or any color-specific scarcity metrics, we assume that the scarcity of blue and green water are correlated and apply one scarcity weighting to both.

A Water Stress Index (WSI) was developed by Pfister et al. (2009) as a general screening indicator while assessing the environmental impacts of freshwater consumption using Life Cycle Assessment (LCA). The WSI was based on global hydrological and global water use models modified to account for variability of precipitation and corrected to account for watersheds with strongly regulated flows. The index follows a logistic function and is tuned to a particular threshold point between moderate and severe water stress. Ridoutt and Pfister (2010b) also assessed consumptive water use in another new water footprint calculation method calculation which considered the local water stress relative to the global average water stress. The methods applied in both papers are spatially differentiated on a watershed and a country level, and depend either on threshold or global water stress. This method could also lead to higher correlation between national water stress indices. Moreover, there is no broad consensus so far for the value of the threshold that defines water scarcity.

In this study we use the water exploitation index (WEI) which is based on national statistics and free of any particular thresholds, and thus is only sensitive to national water scarcity conditions. This water stress metric is used here to identify countries with relatively high abstraction in relation to water resources available, and subject to water stress. This indicator has been chosen in order to readily convey the notion that water use in a region of high water stress is more valued than water use in a region of low water stress. The water exploitation index only measures groundwater withdrawals, i.e. blue water stress. As a direct measure of water stress the WEI does not cover rainfall (green water) which accounts for $\approx$74% of total water use (Hoekstra and Mekonnen, 2012). While this is theoretically a limitation of WEI as an indicator of water stress, in practice farmers tend to only use extract groundwater after rainfall supply has been exhausted so measuring blue water stress is a suitable measure of actual water scarcity situation. In our paper the objective of using an indicator is not to capture the cause/effect relationship between water use and environmental impacts as reflected in midpoint and endpoint indicators of life cycle assessment (see see Kounina et al., 2012; Pfister et al., 2009), but rather to understand the structural flow of scarce water from one region to another. Hence we could rely on a simple water exploitation index to bring explicitly the scarce water endowments of the countries into the framework of international trade, and use the indicator as a weight in the virtual water flow accounting.

We use the water scarcity index directly as national scarcity weights $\mathbf{w}$ and simply element-wise multiplied (#) the water use account $\mathbf{Q}$ in order to obtain a scarcity-weighted water use account $\mathbf{Q}^* = \mathbf{Q}\ \#\ \mathbf{w}$. The scarcity-weighted account $\mathbf{Q}^*$ is then subjected to the same Leontief demand-pull calculus and Structural Path Analysis as the unweighted account $\mathbf{Q}$. Water use for crops and blue water used for animal water supply, industrial production, and domestic supply account for $>\approx$85% of water use. However given the paucity of regular data on subnational variations in scarcity or on water scarcity by use (how water scarcity affects agriculture, industry, grazing, etc.) we applied the same scarcity weighting to all four use categories.

Using Structural Path Analysis (*SI S2*, and *SI S4*), the Leontief inverse $(\mathbf{I} - \mathbf{A})^{-1}$ can be expanded into an infinite series $Q = \mathbf{q}(\mathbf{I} - \mathbf{A})^{-1}\ \mathbf{y1^y} = \mathbf{q}\ \mathbf{y1^y} + \mathbf{qA}\ \mathbf{y1^y} + \mathbf{qA}^2\ \mathbf{y1^y} + \ldots$, with each component of the series representing a particular supply chain, or structural path. Structural Path Analysis is able to provide a collectively exhaustive and mutually exclusive atomic representation of virtual water flows in a complex economic system. We also use Structural Path Analysis to generate the production layer decomposition shown in Fig. 5. For more details see *SI S2*.

3. Results

In 2000 the world consumed about 8000 TL (1 TL = 1 Teraliter = 10^{12} L = 1000 Gigaliters) of water, however of those only about

1500 TL can be classified as scarce. We find that 1900 TL (about 24% of global water use), was embodied in internationally traded goods, with about 480 TL being more or less scarce (32% of global scarce water).

Rankings of countries with respect to their total and scarce water use yield substantially different results (Fig. 1). As expected, large and/or populous nations such as India, China, the USA, Brazil, Russia and Indonesia occupy top ranks among countries in terms of total water use. However, in terms of scarce water, relatively water-scarce countries such as Pakistan, Iran, Egypt, Algeria and Uzbekistan become major users and consumers, while relatively water-abundant countries such as Brazil and Russia drop in their ranks (SI Table S3.1). In many countries in the Middle East and North Africa nearly all water is be classed 'scarce' (SI Table S3.2).

In contrast to the territorial water use perspective portrayed in the left panel in Fig. 1, the virtual water, or water footprint perspective shown in right panel sees developed countries such as the Japan, Germany, France, Italy and the UK gain in the ranking, both in terms of unweighted virtual water use and scarce water. Egypt, Iran, and Pakistan retain top positions, due to both their population size and their location in a water-scarce world region (SI Table S3.3). In the water footprint perspective, the relative positions of countries are determined not only by their domestic water use, but also by the virtual water embodied in their imports. The water footprints from our study, before the scarcity weighting step, agree with those determined in previous studies (Chapagain and Hoekstra, 2004; Feng et al., 2011), as documented in Lenzen et al. (2012a). As with water use figures, the most of the flows into water-scarce countries such as in the Middle East, Central Asia and North Africa are classified as 'scarce' while flows into water-abundant countries, often located in equatorial regions such as Central Africa and Central America, contain far less scarce water (SI Table S3.4).

Scarcity weighting alters the relative trade balances of net importers (top ten bars in Fig. 2). These are exclusively developed, relatively water-abundant countries (such as the USA, Japan, and Germany) that appear to import a significant part of their virtual water from water-scarce sources. However, scarcity weighting does elevate a number of countries towards a net importer status (SI Table S3.6). These countries appear more importing (or less exporting) after scarcity weighting. In other words, their imports are more water-scarce than their exports. Indonesia, New Zealand, and Papua New Guinea, for example, receive a major part of their imports (40%, 27%, and 9%, respectively) of their embodied scarcity-weighted water from water-scarce regions in Australia, for example in the form of wheat, cotton and live cattle to Indonesia, sugar, grapes and other prepared foods to New Zealand, and meat and other prepared food to Papua New Guinea. Mauritania, another water-scarce exporter, sends embodied water to Portugal (23% of exports), Algeria (18%), Tunisia (13%), Spain (6%), and Nigeria (5%). The USA, UK, and Germany are among the top recipients of embodied water from Kenya, Congo, Gabon, Senegal, Mali, and Chad. Our results often show that proximate and therefore important trade partners of water-stressed countries play an important role in exacerbating water scarcity. This effect is especially drastic along geographical divides of water scarcity and abundance, such as the Timor Strait, the Sahel, and the Kalahari.

Scarcity weighting also alters the relative trade balance of top-ranking net exporters (bottom ten bars in Fig. 2). Net exporters are almost exclusively (with the exception of Australia) developing, relatively water-scarce countries, however more Middle-Eastern and Central Asian countries rank high after scarcity weighting. Egypt exports its scarce water embodied in not only in cotton and cotton products, but also vegetables, fruit and their products to Saudi Arabia (16% of exports), Japan (12%), USA (9%), Germany (8%) and Italy (7%). Generally, water-scarce countries with water-abundant neighbors, such as the USA and Mexico, Mediterranean and Middle-Eastern countries, and South Africa, appear more exporting (or less importing) after scarcity weighting (*SI Tab, S3.8*). In other words, their exports are more water-scarce than their imports. This finding once again confirms the important role of geographical abundance/scarcity divides for regional water scarcity.

Fig. 3 reports the top ten importers and exporters in terms of scarce water. India, Pakistan, and China are the largest exporters of scarce water; the USA, Japan and Germany are the largest importers. Pakistan and Syria stand out as comparatively small exporters in traditional virtual water terms but are the 2nd and 5th largest exporters of scarce waters, respectively. Table 2 identifies the largest bilateral (country pair) flows both in terms of total virtual water (left side) and in terms of scarcity-weighted water (right side). In the top

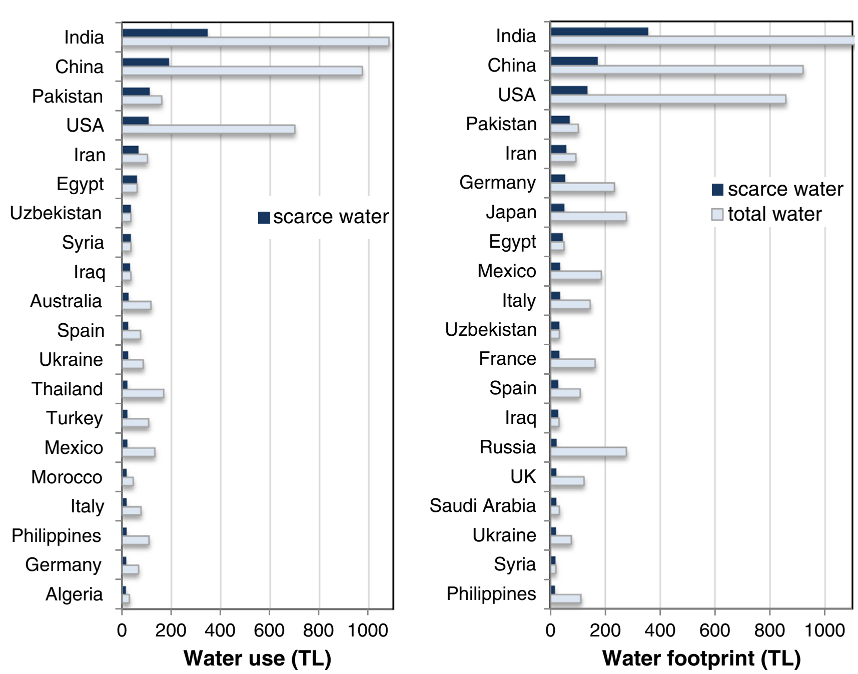

Fig. 1. Twenty countries top-ranked in terms of their water use **qx** and scarce-water **q*x** (left panel) and their water footprint **my** and scarce-water footprint **m*y** (right panel).

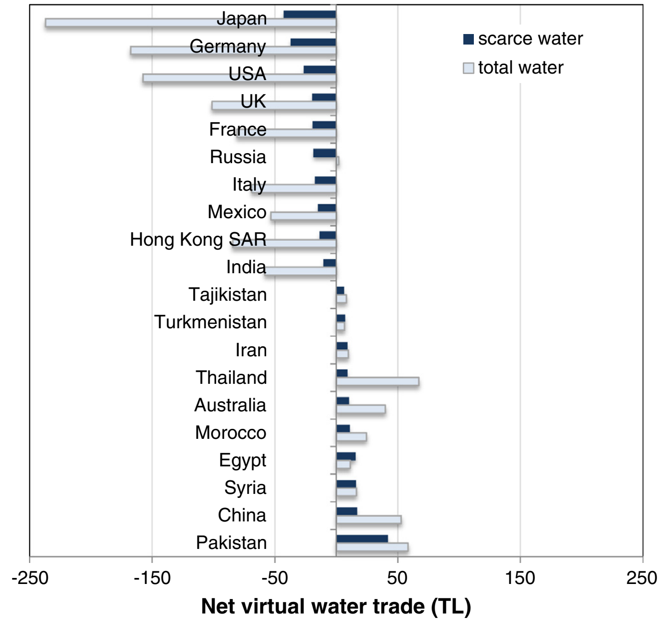

Fig. 2. Ten countries top- and bottom-ranked in terms of their net virtual water trade ($\mathbf{q}\ \mathbf{ex} - \mathbf{m}\ \mathbf{im}$) and net scarce-water trade ($\mathbf{q}^*\ \mathbf{ex} - \mathbf{m}^*\ \mathbf{im}$), where **ex** and **im** are vectors of exports and imports by product, respectively. Net importers (top ten bars) are characterized by negative trade balances, and vice versa.

list of scarce flows we see exports from India and Pakistan, the largest two exporters of scarce water, flowing into the USA and Japan, the two largest importers of scarce water.

Policy and business decisions are typically made at the industry or commodity level. In order to extract these more detailed results from the MRIO virtual water accounts we use Structural Path Analysis (SPA) to identify the largest global supply chains and find key scarce-water flows. We extracted and ranked those global structural paths that are most important in terms of the scarce virtual water embodied in them. The top path in Table 1 contains cotton from Pakistan (high-quality long fiber for business shirts) that is woven into cloth for high-quality Italian men's apparel designs, and made up into shirt and suit linings in Hong Kong. Just this 2-node global supply chain consumes 1 ML of scarce Pakistani water annually. The second path reflects 880,000 L of scarce Iraqi water pumped into medium-age oil fields in order to flood the deposit from below and float the crude oil to the top of the stratum. The oil thus extracted is refined in the USA, and then supplied by petrol wholesalers to consumers in Singapore. The path originating from Egypt represents citrus fruits, cane sugar and vegetable saps and extracts that are processed in the Netherlands and sent to soft drink factories in the USA. This chain is likely to include gum arabic, an important ingredient in soft drink syrups. Indian coconuts are processed in Germany and the Netherlands to give coconut oil that in turn provides the acidic taste to American soft drinks. This chain could also contain "coconut water", a new health drink with an isotonic concentration much like blood. One can also buy "instant coconut powder for soft drinks and desserts". Even though highly complex and specialized, this supply chain consumed 100,000 L of scarce Egyptian water. Sri Lanka most probably supplies coconut, abaca, ramie and other vegetable textile fibers to China for blending and weaving, and subsequent fabrication of clothes in Hong Kong. Most of Australian cotton is sent to Indonesia for spinning (into raw cotton yarn and staple yarn) and weaving, then to Taiwan for further processing and design, and finally to Hong Kong for apparel fabrication.

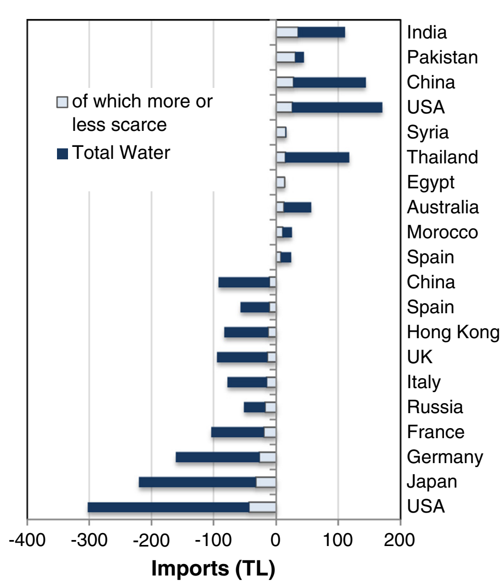

Fig. 3. Top ten exporters (upper bars), and top ten importers (lower bars) of scarce water, as ranked by their scarce-water exports ($\mathbf{q}^*\ \mathbf{ex}$) and imports ($\mathbf{m}^*\ \mathbf{im}$).

Examining the top sources of scarce water flowing to consumers in the USA (Fig. 4), the traditional major economic trading partners in Europe, Latin America and Asia are sharply outranked by water-scarce countries. India, Australia, China, Pakistan, Turkey, Thailand, and Argentina are major sources of scarce water embodied in imports, and suppliers in Africa (both North and Sub-Saharan) and Central Asia also stand out as exporters of scarce water. Among other water-intense agricultural products, these countries export cotton, beef, fruit, and rice.

Finally, we demonstrate the value added by MRIO analysis of process-analytical approaches. We exploit the series expansion of the Leontief inverse in order to enumerate the distribution of the water footprint across supply-chain stages (also called production layers, compare with Fig. 1 in Lenzen, 2000). We assume that process analyses capture water used directly in households (for example for drinking, washing etc.), by producers of products sold directly to households (supplier stage #1, for example water to make fruit juice), as well as by suppliers of producers of products sold directly to households (supplier stage #2, for example water to irrigate orchards to make fruit juice). We assume that further upstream supplier stages are being truncated by process analyses. Based on these assumptions, process analyses would cover about 50% of the total water footprint of nations, and hence regional and sectoral water footprints would be affected by truncation errors of about 50%. In MRIO terms this translates into an underestimation of water embodiments in international trade by 50%. Even if the assumption were to be relaxed by assuming that process analysis also captured water used by suppliers of suppliers of producers of products sold directly to households (supplier stage #3, for example water used in manufacturing machinery that is used to irrigate orchards for producing fruit juice), the coverage would increase to about 75%, leaving a truncation error of 25% (Fig. 5). For the entire world, this truncation means that process analyses would differently allocate (to countries and sectors) about 2000 TL of water used. Note that our MRIO framework distinguishes about 10,000 sectors worldwide, meaning that a Structural Path Analysis would be able to capture 10,000 stage-1 paths, $10{,}000^2 = 10^8$ stage-2 paths, and $10{,}000^2 = 10^{12}$ stage-3 paths. Given that stage-3 paths are responsible for about 500 TL of water used (difference between stage 2 and 3 in Fig. 5), an average stage-3 path would embody about only 500 L of water. Such paths will not show up dominantly in a Structural Path Analysis, and may hence appear as rather "exotic", however the relevance of such higher-order paths lies in their sheer number, and this circumstance is something that MRIO analysis is able to address.

Table 1
Selected results from a global Structural Path Analysis (SPA) of scarce virtual water. Supply-chains proceed from left to right, starting with the water-using industry, via intermediate trade and transformation steps, to the industry supplying final consumers. SPA traces supply chains to the final consumer, e.g. people buying cloths in Hong Kong, petrol in Singapore, and soft drinks in the US.

Water-using industry	Intermediate suppliers		Industry supplying final demand	Virtual water content of path (ML)
Pakistan agriculture	Italy textiles		Hong Kong wearing apparels	1.04
Iraq mining and drilling	USA petroleum refineries		Singapore petroleum products	0.88
Egypt agriculture	Netherlands food and beverages		USA soft drink and ice	0.12
India coconuts	Germany food	Netherlands food and beverages	USA soft drink and ice	0.10
Sri Lanka agriculture	China other textiles		Hong Kong wearing apparels	0.07
Australia cotton	Indonesia made-up textile	Taiwan other fabrics	Hong Kong wearing apparels	0.05

4. Discussion and conclusions

With water becoming scarcer globally, virtual water trade is taking in increasingly important place in water policy discussions, and is often advocated as one in a set of feasible policy options to mitigate the spatial variability in water availability. However, before concrete policy implications can be drawn it is pertinent to identify which flows are coming from water-scarce sources, and this is where the current literature lacks information. Studies published so far either indicate water scarcity without dealing with indirect effects that ripple through international supply chains, or quantify virtual water trade without considering scarcity. Our study is unique in that it has filled a research gap by using a Multi-Region Input–output framework to quantify both the direct and indirect consumption of scarce water. The approach adds value to the literature on virtual water by identifying major global routes conveying pressure on water resources from centers of consumption to regions of water scarcity, thus facilitating water policy dialog and formulation. Our findings confirm that global flows of virtual water look substantially different when scarcity is taken into account: a result that validates and underscores the importance of considering scarcity in water footprinting.

Overt emphasis on international trade in scarce water resources may distract from tractable responses within countries. Studies on China (Feng et al., 2012), Iran (Faramarzi et al., 2010), Egypt (Zeitoun et al., 2010), and Uzbekistan (Bekchanov et al., 2012) highlight national responses where water-rich regions could provide larger shares of water intensive food production allowing water in scarce regions to be re-allocated to products and services with higher value returns per unit of water. Hoekstra (2009) however emphasizes the high import-dependency in virtual water terms of many water-scarce countries on limited numbers of grain producers such as the USA, Brazil and Argentina. Thus the tension grows between calls for international 'virtual water' treaties and legal rights, and ensuring that each sovereign state makes its own required regional and industry adjustments to improve food security of its citizens. Structural Path Analysis is useful for translating national-level results from water footprint work into actual policy measures or business decisions, which are nearly always taken at the industry or commodity level. Structural Path Analysis can be employed to identify the most water-intensive production activities, trade flows, and final products, and target actions accordingly.

There are issues in our work that point towards future research needs. The first issue relates to a shortcoming in our methodology. Our approach – like some earlier global water footprint studies (for example van Oel et al., 2009) – follows political rather than river-basin or catchment boundaries. In essence, this is due to the non-availability of economic and especially input–output data referring to catchments. While water consumption figures could in principle be adjusted to sub-national areas (see for example studies based on regional statistics, aerial or satellite images, models (for example CROPWAT) or GIS databases, such as Hanafiah et al., 2011), the absence of economic data precludes the application of MRIO to catchments and river basins at the global level. In this respect, our analysis shares a particular shortcoming referred to as aggregation errors with virtually every generalized input–output analysis, and especially other generalized MRIO analyses of the environment such as using the GTAP, WIOD or EXIOBASE databases. This aggregation error problem relates to sectoral and regional aggregation alike. Take for example the aggregation of Australian copper and aluminium production into a sector called 'non-ferrous metals'. While aluminium smelting may consume large amounts of electricity entailing emissions from power plants, copper production usually does not, and due to the aggregation of the two, copper-containing motors are incorrectly allocated emissions from aluminium smelting. Or take emissions embodied in beef exports assessed in a national input–output model of Australia. Most beef is exported from the state of Queensland where intensive land clearing prevails, however in a national Australian analysis beef exports are multiplied with average Australian emission intensities which are lower that emission intensities in Queensland, hence emissions embodied in exports are being underestimated because of regional aggregation. Thus, regional and sectoral aggregation issues exist in all (MR)IO models, and are more severe in situations with high degrees of aggregation. More sectoral and regional detail (such as in Pfister's work) is desirable, for example for obtaining correct region-specific industry-averages of water scarcity. However at present such data are often unavailable because of constraints faced by statistical agencies (such as confidentiality of information).

Table 2
Top ten paths of total virtual water (left) and scarcity-weighted virtual water (right). Flows out of water-scarce Central Asian countries are prominent. The US has relatively little scarce water but is a major exporter of embodied water, hence its appearance as a top-ranked scarce water exporter.

Origin	Destination	Total water (TL)	Origin	Destination	Scarce water (TL)	Total (TL)
USA	Mexico	34.2	Pakistan	USA	7.9	11.3
USA	Canada	30.5	China	USA	5.8	29.9
China	USA	29.9	India	USA	5.5	17.3
China	Japan	27.5	China	Japan	5.4	27.5
Thailand	Hong Kong	27.1	USA	Mexico	5.3	34.2
China	Hong Kong	21.6	USA	Canada	4.7	30.5
India	USA	17.3	Tajikistan	Russia	4.4	5.9
Mexico	USA	17.0	China	Hong Kong	4.2	21.6
USA	Japan	16.3	Uzbekistan	Russia	4.0	4.0
Ethiopia	Japan	16.0	India	Japan	3.9	12.1

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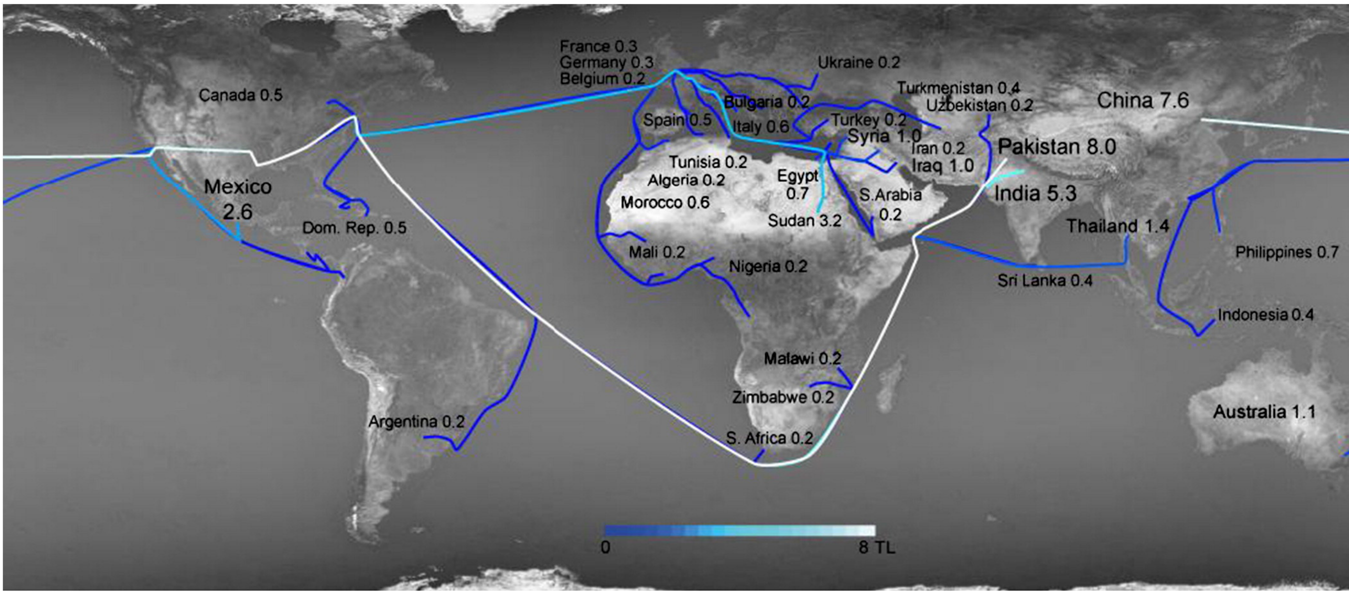

Fig. 4. Top sources of virtual water imports into the USA (values in TL shown on origin country). Flow line color encodes the magnitude of the flow: small flows are darker and large flows lighter. Lines directly connect the original source of scarce water to American consumers; intermediate processing stops are accounted for but not explicitly visualized.

The second issue is one of translating analysis into policy responses. While this study entrains the global complexity needed to adjust production chains and trade dynamics there are policy questions still beyond its reach. For example, the question of why and how to significantly adjust production chains that consume scarce water will be difficult. For countries such as Uzbekistan and Pakistan, nearly 25% of their total exports are raw cotton and yarns, derived from scarce water use and thus difficult to change while maintaining commerce and national stability. Elsewhere (Lenzen et al., 2012b), we have argued for a three-tier approach having producers utilize the best production methods, having intermediate agents trade only in certified goods, and empowering consumers through product labeling and education. Another question concerns how water, and scarce water, interacts with issues of rising nutrient use, biodiversity decline, and land clearance, among others. Rockström and Karlberg (2010) call for a green revolution focusing on rainfed systems and improved accounting of water at global and regional scales. Our study can only highlight many of the 'at risk' production chains and countries and so might become one of the starting points for further in-depth investigations feeding into decision-making. Improved water-extended MRIO accounts may eventually underpin a global certification framework that could lead to product labeling.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.ecolecon.2013.06.018.

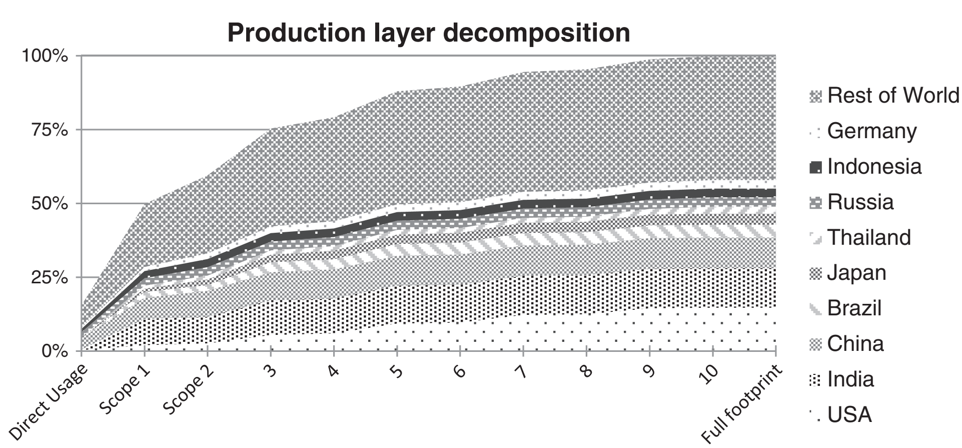

Fig. 5. As longer supply chains are evaluated the water footprint (the sum of indirect flows) grows. Leontief's IO matrix inversion method evaluates supply chains of infinite depth, however in most cases after evaluating supply chains to 8–10 stages the indirect flows associated with deeper stages approach negligible or zero size.

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References

Alexander, K., West, J., 2011. Water. In: Schandl, H., Turner, G.M., Poldy, F., Keen, S. (Eds.), Resource Efficiency: Economics and Outlok (REEO) for Asia and the Pacific. United Nations Environment Programme, Canberra, Australia, pp. 85–101.

Allan, J.A., 1997. Virtual Water: A Long Term Solution for Water Short Middle Eastern Economies? University of London, UK, Water Issues Study Group, School of Oriental and African Studies.

Ansink, E., 2010. Refuting two claims about virtual water trade. Ecological Economics 69 (10), 2027–2032.

Arto, I., Andreoni, V., et al., 2012. Water use, water footprint and virtual water trade: a time series analysis of worldwide water demand. 20th International Conference on Input–Output Techniques. C. Lager. Bratislava, Slovakia.

Beach, H.L., Hamner, J., et al., 2000. Transboundary Freshwater Dispute Resolution: Theory, Practice and Annotated Reference. United Nations University Press, New York, USA.

Bekchanov, M., Bhaduri, A., et al., 2012. Virtual water concept for sustainable economic restructuring: evidence from Uzbekistan, Central Asia. 20th International Conference on Input–Output Techniques. C. Lager. Bratislava, Slovakia.

Boulay, A.-M., Bulle, C., et al., 2011. Regional characterization of freshwater use in LCA: modeling direct impacts on human health. Environmental Science & Technology 45 (20), 8948–8957.

Chapagain, A.K., Hoekstra, A.Y., 2004. Water Footprints of Nations. UNESCO-IHE Institute for Water Education, Delft, Netherlands.

Chapagain, A.K., Hoekstra, A.Y., et al., 2006. Water saving through international trade of agricultural products. Hydrology and Earth System Sciences 10.

Cooney, C., 2009. LCA finally takes water into account. Environmental Science & Technology 43 (11), 3986-3986 (http://pubs.acs.org/doi/pdf/10.1021/es901078v).

Dalin, C., Konar, M., et al., 2012. Evolution of the global virtual water trade network. Proceedings of the National Academy of Science 109 (16), 5989–5994.

Dinar, S., Dinar, A., 2000. Negotiating in international watercourses: diplomacy, conflict and cooperation. International Negotiation 5, 193–200.

Ewing, B.R., Hawkins, T.R., et al., 2012. Integrating ecological and water footprint accounting in a multi-regional input–output framework. Ecological Indicators 23, 1–8.

EXIOBASE, 2012. EXIOBASE. Delft, Netherlands, Exiobase Consortium.

FAO, 2012. AQUASTAT — FAO's Information System on Water and Agriculture. Food and Agriculture Organization of the United Nations.

Faramarzi, M., Yang, H., et al., 2010. Analysis of intra-country virtual water trade strategy to alleviate water scarcity in Iran. Hydrology and Earth System Sciences 14 (8), 1417–1433.

Feng, K., Chapagain, A.K., et al., 2011. Comparison of bottom-up and top-down approaches to calculating the water footprints of nations. Economic Systems Research 23 (4), 371–385.

Feng, K., Siu, Y.L., et al., 2012. Assessing regional virtual water flows and water footprints in the Yellow River Basin, China: a consumption based approach. Applied Geography 32 (2), 691–701.

Frischknecht, R., Steiner, R., et al., 2006a. Swiss Ecological Scarcity Method: The New Version 2006. Swiss Federal Office for the Environment (FOEN), Bern, Switzerland.

Frischknecht, R., Steiner, R., et al., 2006b. The Ecological Scarcity Method — Eco-Factors 2006: A Method for Impact Assessment in LCA. Swiss Federal Office for the Environment (FOEN), Bern, Switzerland.

Frischknecht, R., Jungbluth, N., et al., 2009. UBP-Bewertung für den Wasserbedarf von Treibstoffen. ESU-services, Uster, im Auftrag des Bundesamtes für Umwelt (BAFU), Bern, Switzerland.

Gerbens-Leenes, P.W., Lienden, A.R.v., et al., 2012. Biofuel scenarios in a water perspective: the global blue and green water footprint of road transport in 2030. Global Environmental Change 22 (3), 764–775.

Hanafiah, M.M., Xenopoulos, M.A., et al., 2011. Characterization factors for water consumption and greenhouse gas emissions based on freshwater fish species extinction. Environmental Science & Technology 45 (12), 5272–5278.

Hinrichsen, D., Robey, B., et al., 1997. Solutions for a Water-short World. Johns Hopkins Bloomberg School of Public Health, Baltimore, USA.

Hoekstra, A.Y., 2009. Water Security of Nations: How International Trade Affects National Water Scarcity and Dependency Threats to Global Water Security. In: Jones, J.A.A., Vardanian, T.G., Hakopian, C. (Eds.), Springer, Netherlands, pp. 27–36.

Hoekstra, A.Y., Mekonnen, M.M., 2012. The water footprint of humanity. Proceedings of the National Academy of Science 109 (9), 3232–3237.

Jefferies, D., Muñoz, I., et al., 2012. Water footprint and life cycle assessment as approaches to assess potential impacts of products on water consumption. Key learning points from pilot studies on tea and margarine. Journal of Cleaner Production 33, 155–166.

Jungbluth, N., Nathani, C., et al., 2011. Environmental Impacts of Swiss Consumption and Production — A Combination of Input–output Analysis with Life Cycle Assessment. Federal Office for the Environment, Bern, Switzerland.

Just, R.E., Netanyahu, S., 1998. International water resource conflicts: experience and potential. In: Just, R.E., Netanyahu, S. (Eds.), Conflict and Cooperation on Transboundary Water Resources. Kluwer Academic Publishers, Boston, USA, pp. 1–26.

Kenneth, F.D., Major, D.C., 2002. Climate change and water resources. The Management of Water Resource, volume 2. Edward Elgar, New York, USA.

Konar, M., Dalin, C., et al., 2012. Temporal dynamics of blue and green virtual water trade networks. Water Resources Research 48 (W07509).

Kounina, A., Margni, M., et al., 2012. Review of methods addressing freshwater use in life cycle inventory and impact assessment. International Journal of Life Cycle Assessment. http://dx.doi.org/10.1007/s11367-11012-10519-11363.

Kumar, M.D., Singh, O.P., 2005. Virtual water in global food and water policy making: is there a need for rethinking? Water Resources Management 19, 759–789.

Lenzen, M., 2000. Errors in conventional and input–output-based life-cycle inventories. Journal of Industrial Ecology 4 (4), 127–148.

Lenzen, M., Kanemoto, K., et al., 2012a. Mapping the structure of the world economy. Environmental Science & Technology 46 (15), 8374–8381.

Lenzen, M., Moran, D., et al., 2012b. International trade drives biodiversity threats in developing nations. Nature 486, 109–112.

Leontief, W., 1966. Input–Output Economics. Oxford University Press, New York, NY, USA.

McCartney, M.P., Sullivan, C., et al., 2000. Ecosystem impacts of large dams. Thematic Review II.1: Dams, Ecosystem Functions and Environmental Restoration.World Commision on Dams, Cape Town, South Africa.

Mekonnen, M.M., Hoekstra, A.Y., 2011. National water footprint accounts: the green, blue and grey water footprint of production and consumption. Value of Water Research Report Series. UNESCO-IHE, Delft, the Netherlands (http://www.waterfootprint.org/Reports/Report50-NationalWaterFootprints-Vol1.pdf).

Pfister, S., Koehler, A., et al., 2009. Assessing the environmental impacts of freshwater consumption in LCA. Environmental Science & Technology 43 (11), 4098–4104.

Pfister, S., Bayer, P., et al., 2011. Environmental impacts of water use in global crop production: hotspots and trade-offs with land use. Environmental Science & Technology 45 (13), 5761–5768.

Postel, S., 1999. Pillars of Sand: Can the Irrigation Miracle Last? WW Norton and Company, New York, USA.

Ramirez-Vallejo, J., Rogers, P., 2004. Virtual water flows and trade liberalization. Water Science and Technology 49 (7), 25–32.

Ridoutt, B.G., Pfister, S., 2010a. Reducing humanity's water footprint. Environmental Science & Technology 44 (16), 6019–6021.

Ridoutt, B.G., Pfister, S., 2010b. A revised approach to water footprinting to make transparent the impacts of consumption and production on global freshwater scarcity. Global Environmental Change 20 (1), 113–120.

Ridoutt, B.G., Pfister, S., 2013. A new water footprint calculation method integrating consumptive and degradative water use into a single stand-alone weighted indicator. 18, 204–207.

Rockström, J., Karlberg, L., 2010. The quadruple squeeze: defining the safe operating space for freshwater use to achieve a triply green revolution in the Anthropocene. AMBIO: A Journal of the Human Environment 39 (3), 257–265.

Spulber, N., Sabbaghi, A., 1994. Economics of water resources: from regulation to privatization. In: Dinar, A., Zilberman, D. (Eds.), Natural Resource Management and Policy. Kluwer Academic Publishing, Boston, USA, pp. 235–269.

Steen-Olsen, K., Weinzettel, J., et al., 2012. Carbon, land, and water footprint accounts for the European Union: consumption, production, and displacements through international trade. Environmental Science & Technology 46 (20), 10883–10891.

van Oel, P.R., Mekonnen, M.M., et al., 2009. The external water footprint of the Netherlands: geographically-explicit quantification and impact assessment. Ecological Economics 69 (1), 82–92.

Verma, S., Kampman, D.A., et al., 2009. Going against the flow: a critical analysis of inter-state virtual water trade in the context of India's National River Linking Program. Physics and Chemistry of the Earth, Parts A/B/C 34, 261–269.

WIOD, 2012. World Input–Output Database. University of Groningen and 10 other institutions, Groningen, Netherlands.

Yang, H., Wang, L., et al., 2006. Virtual water trade: an assessment of water use efficiency in the international food trade. Hydrology and Earth System Sciences 10, 443–454.

Zeitoun, M., Allan, J.A., et al., 2010. Virtual water 'flows' of the Nile Basin, 1998–2004: a first approximation and implications for water security. Global Environmental Change 20 (2), 229–242.

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