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Global virtual water trade and the hydrological cycle: patterns, drivers, and socio-environmental impacts

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Abstract

The increasing global demand for farmland products is placing unprecedented pressure on the global agricultural system and its water resources. Many regions of the world, that are affected by a chronic water scarcity relative to their population, strongly depend on the import of agricultural commodities and associated embodied (or *virtual*) water. The globalization of water through virtual water trade (VWT) is leading to a displacement of water use and a disconnection between human populations and the water resources they rely on. Despite the recognized importance of these phenomena in reshaping the patterns of water dependence through teleconnections between consumers and producers, their effect on global and regional water resources has just started to be quantified. This review investigates the global spatiotemporal dynamics, drivers, and impacts of VWT through an integrated analysis of surface water, groundwater, and root-zone soil moisture consumption for agricultural production; it evaluates how virtual water flows compare to the major 'physical water fluxes' in the Earth System; and provides a new reconceptualization of the hydrologic cycle to account also for the role of water redistribution by the hidden 'virtual water cycle'.

1. Introduction

The water cycle, the global-scale pattern of water circulation through, atmosphere, land masses, and oceans that strongly controls life on Earth, has been altered by human action since the onset of civilization as a result of water withdrawals from streams, lakes, and aquifers, river diversions, and damming. This disruption, however, has been exacerbated by the Industrial Revolution, the subsequent technological innovations of the Green Revolution, and the associated socio-economic dynamics. Not only climate change but also processes associated with shifts in land use and land cover—such as deforestation, large-scale irrigation, and dam construction—have strongly altered the water cycle (Postel *et al* 1996, Poff *et al* 1997, Gordon *et al* 2005, Oki and Kanae 2006, Rost *et al* 2008, Rockström *et al* 2009, Gleick and Palaniappan 2010, Runyan and D'Odorico 2016).

Freshwater resources are critical. Renewable, yet finite, they are central for ecosystem functions, human wellbeing, and societal development (Ammerman and Cavalli-Sforza 1984, Falkenmark and Rockström 2004, Allan and Castillo 2007, D'Odorico *et al* 2010b). As a consequence of increasing human pressure, in some regions, water use is exceeding sustainable levels (Rosa *et al* 2018a). Therefore, we are living in what has been described as an era of water scarcity in which water resources available to agriculture may limit the planet's ability to meet the growing crop demand by human societies (Falkenmark and Rockstrom 2004, D'Odorico *et al* 2018, Rodell *et al* 2018). This fundamental and increasingly scarce resource (relative to increasing human demand) is crucial to agriculture, mining, energy production, manufacturing, and residential use (Vörösmarty *et al* 2010, Brauman *et al* 2016, Mekonnen and Hoekstra 2016).

The complex interdependence between human societies and water, which tends to be thought of and treated as a socio-environmental dynamic between human needs and local hydrological resources, often occurs through distant interconnections that result through the globalization of water resources (Hoekstra and Chapagain 2008). Indeed, humanity affects and interacts with the global water cycle by modifying water stocks and moving substantial amounts of water, both spatially and temporally. Traditionally, though, scientists have evaluated the alterations to the water cycle focusing only on its physical water flows and stocks. Yet, this approach fails to account for an important aspect of the socio-hydrological interactions that shape the global water cycle, namely the existence of 'hidden' virtual water fluxes that should be accounted for in addition to the physical water flows. Understanding the drivers, processes and impacts of what we define as the 'virtual water cycle' becomes a constitutive aspect of understanding and redefining the notion of the global hydrologic cycle.

This paper aims at deepening the understanding of key elements of the main socio-hydrological dynamics that are associated with an increasingly interdependent globalized world. At the center of this endeavor, lies the study of the main drivers, processes and impacts of virtual water trade (VWT). Specifically, the goal of this article is to (1) review the impact of VWT on water resources (e.g. Pfister and Bayer 2014, Lutter *et al* 2016) by looking at global patterns of surface water, groundwater, and root-zone soil moisture consumption and trade; (2) analyze how virtual water flows fit into the 'natural' hydrological cycle by comparing their magnitude to those of major 'physical water fluxes' in the Earth System; (3) evaluate to what extent VWT establishes teleconnections (also known as 'telecoupling') in the global water system through dependencies on water resources available in other regions of the world; (4) review gaps in current knowledge, discuss about possible future research directions, and highlight emerging research trends related to VWT.

After an introduction of the general concept of VWT and its importance, we highlight the dynamics of global market integration and illustrate the main features of contemporary trade policies and their development. We then illustrate the key patterns of VWT; discuss the different resolutions at which the analysis of virtual water transfers occur; and reflect on the epistemological implications of the analysis of VWT and how these lead to a new analytical reconceptualization of the global water cycle that accounts also for a hidden 'virtual water cycle'. We then review the main drivers and models of VWT and discuss the major socio-environmental consequences of VWT. Finally, we conclude highlighting the key contribution of this review and point at new areas of research that we believe deserve more attention.

Box 1. Definitions.

Virtual water content (VWC) is the amount of water required to produce a good, considering all the steps involved in its production. The term 'virtual' emphasizes that the water is conceptually embedded though not physically present in the good. In this sense the term 'content', though commonly used, can be misleading and in contradiction with the adjective 'virtual' because virtual water is not really contained in the commodity. The VWC is generally expressed as the water volume per unit mass of product (in m^3 ton^{-1} or kg^{-1}). For example, in the United States of America, the actual average water content of wheat is ~0.13 m^3 ton^{-1} whereas the VWC is ~1961 m^3 ton^{-1}.

Crop water footprint (CWF) is the same as the VWC but specifically refers to crops. CWF is calculated as the ratio between total crop evapotranspiration in the growing season and crop yield and is expressed as a water volume per unit mass of product (in m^3 ton^{-1} or kg^{-1}). The footprint can be calculated either through a production-based accounting (PBA) or, most commonly, through a consumption-based accounting (CBA). PBA refers to impacts and resource inputs associated with production activity. CBA reallocates those produced goods to final consumers. In the case of crops, however, most of the water footprint is contributed by water losses by evapotranspiration in the course of the production process, while the water cost of processing and transportation is negligible. Therefore, there is no need to conceptually distinguish PBA from CBA and in this review the water costs of crop production will be simply expressed in terms of CWF (see box 2 for more details).

Water footprint (WF) identifies the volume of water associated with a certain (not unit) mass of a good and/or to a set of goods. Computation may follow different approaches, as detailed in box 2.

Green water footprint is the fraction of the WF that is contributed by green water (i.e. precipitation water directly contributing to the soil water balance in the crops' root zone in the absence of irrigation).

Blue water footprint is the fraction of the WF that is contributed by the consumptive use of blue water (i.e. irrigation water withdrawn from surface water bodies and aquifers).

Grey water footprint is an indicator of freshwater pollution defined as the water volume required to dilute pollutants to a concentration that meets the water quality standards.

Virtual water trade (VWT) is the (international or intra-national) trade of goods evaluated in terms of virtual water. Through the trade of goods, water resources that are physically used in the area of production are virtually transferred to the consumption region. This transfer generates a virtual water flux that links production to consumption. Sometimes, it is also referred to as the water footprint of trade.

Box 2. The Water Footprint calculation

The *water footprint* is the amount of water needed to produce a commodity or a set of (produced or consumed) commodities. When associated to the consumptive water use (i.e. water returned to the atmosphere as water vapor during the production process), it may include both rainwater (green water) and surface water or groundwater (blue water). For instance, crop production consumes both green water and, in the case of irrigated agriculture, blue water (see box 1). These consumptive uses of water by crops are due to evapotranspiration. Water footprint studies have used different approaches:

1) The biophysical approach, most widely used for crops and agricultural goods, estimates CWF as the ratio between evapotranspiration and the crop yield (see table 3). To estimate VWT (box 3), the CWF is then multiplied by the mass of

(Continued.)

product traded and the resulting volume of water is then summed across different goods. VWT is generally computed with the CWF of the country of origin of the trade flow, (see box 3 and section 5). Such an approach is usually named the 'bottom-up' approach (e.g. Feng *et al* 2012).

1) Life Cycle Analysis (LCA) approaches use LCA datasets that include a 'water footprint' library of products. The LCA approach includes multi-stage supply chains, so it would attribute cotton water use to the final purchaser of a t-shirt not to the textile producer country, and corrects for re-exports (e.g. Netherlands forwarding goods to Germany).

1) Top-down approaches refer to input–output analyses which have been used largely in economics to investigate the water use and allocation in countries or regions. Top-down approaches calculate the WF by tracing water use in regional, national or global supply chains using a Leontief demand-pull model. Multi-regional input–output (MRIO) analyses in particular use global supply chains and allow for an estimate of national totals (e.g. Yang *et al* 2012, Arto *et al* 2016). The product resolution, however, is often low as highlighted in Feng *et al* (2012) and for example all agricultural goods are usually considered as a bulk or classified in few categories. Recent efforts in the MRIO analyses of water footprint are oriented to improving the product resolution (Lutter *et al* 2016).

1) Finally, a good compromise between top-down and bottom-up approaches could be found in the development of combination methods that use both a monetary MRIO to track embodied goods and a physical model (e.g. based on FAO-STAT) to track physical flows (Bruckner *et al* 2015, de Koning *et al* 2015, Giljum *et al* 2015).

Box 3. Calculation of virtual water trade

The virtual water trade for a single crop *c*, from a given location (e.g. country) *i*, to another location *j*, $VWT_{c,ij}$ ($m^3\ yr^{-1}$) depends on both the virtual water footprint of crops from that location, $VWF_{c,i}$ ($m^3\ ton^{-1}$) and the trade amount of that crop $T_{c,ij}$ (ton yr^{-1}) or

$VWT_{c,ij} = VWC_{c,i}T_{c,ij}$.

Thus, in order to attempt to understand global patterns of virtual water trade, resolution of virtual water content of a crop from a given location and trade volume is required. Water use of a specific crop, both green (rain) and blue (water withdrawals from ground or surface water), is necessary to constrain the virtual content, or water footprint, of that crop for a specific location and growing season.

Biophysical approaches (Box 2) use a variety of grid based models, including, H08 (Hanasaki *et al* 2010), AquaCrop (Steduto *et al* 2009, Raes *et al* 2009), CROPWAT 8.0 (Allen *et al* 1998) and WaterStat (Mekonnen and Hoekstra 2010), LPJmL (Bondeau *et al* 2007, Rost *et al* 2008), that calculate potential evapotranspiration and the soil water balance at resolutions as fine at 5 arc min by 5 arc min scales using on global climate and soil datasets. There are differences in approaches and assumptions among these models, such as use of crop-specific evapotranspiration (ET) coefficients (Allen *et al* 1998, Mekonnen and Hoekstra 2011), versus calculating ET based on crop functional types (Bondeau *et al* 2007, Rost *et al* 2008), or the inclusion of calculations of a grey water footprint (Mekonnen and Hoekstra 2011). However, there are also similar underlying assumptions and databases, such as leveraging

(Continued.)

MIRCA2000 (Portmann *et al* 2010) to help ascertain rainfed versus irrigated agricultural areas and thus discriminate between blue and green water. In all of these models, water use and plant production over a growing season can then be summed over a given year, and crop yield estimates can be derived. Modeled yield can then be adjusted based on reported values, as in the case of Hoekstra and Mekonnen (2011). Yields and water use thus provide both the production volume, *P* (tons yr^{-1}), and the blue and green water use, WU ($m^3\ yr^{-1}$) necessary to calculate the VWC of a given crop as: $VWC_{c,i} = \frac{WU_{c,i}}{P_{c,i}}$. This provides a single year estimate, however, interannual variability can be high and temporally-averaged (1996–2005) values are typically used (Mekonnen and Hoekstra 2010). As agricultural production and trade data are, broadly speaking, estimated and reported, at the country scale (FAO), VWC of a given crop is typically calculated as a country-average value. Consequently, most studies to date have focused on international, rather than sub-national trade. Multi-regional input–output (MRIO)-based approaches go beyond the reconstruction of a trade matrix, tracing commodity flows across countries and across sectors, therefore allowing for a finer resolution in space (e.g. sub-national trade).

2. VWT: what is it and why does it matter?

Globalization increases the exchange and transfer of materials, energy and resources among distant countries. Through the integration of markets, systems of production and societal demands, globalization typically creates teleconnections (i.e. distant socio-environmental interactions) between coupled natural and human systems (Liu *et al* 2013, Oberlack *et al* 2018). Of all resources, water is virtually rather than physically mobilized (Allan 1996).

Water is too heavy and bulky and not valued enough to justify its transport costs. There are exceptions, like the South-to-North Water Diversion Project in China, where 9.5 billion $m^3\ yr^{-1}$ of freshwater are transferred to meet the burgeoning water demand in the North China Plains (Zhao *et al* 2017). Other examples of megaprojects with inter-basin water transfers are the California State Water Project, which roughly transports 3 billion $m^3\ yr^{-1}$ (Cohen *et al* 2004) and the Great Man-Made River Project in Libya, which roughly transports 1.34 billion $m^3\ yr^{-1}$ (Sternberg 2016). There are about 155 inter-basin water transfer schemes in 26 countries around the world, with a total capacity of 490 billion $m^3\ yr^{-1}$ of which 138 billion $m^3\ yr^{-1}$ are for water transfers in Canada alone, and 30 billion $m^3\ yr^{-1}$ in the rest of the Americas, 181 billion $m^3\ yr^{-1}$ in Asia, and 120 billion yr^{-1} in Europe (Verma *et al* 2009). Moreover, about 60 new projects are under study (e.g. Verma *et al* 2009, Shumilova *et al* 2018). Sometimes drinking water is carried by truck, boat, or pipelines—as in the case of the Botswana North–South Carrier project (16 million $m^3\ y^{r-1}$)—to supply water-stressed communities either on a regular basis or in periods of scarcity (Bevanger 1994). Drinking water can also be available as bottled water, which is increasingly

Table 1. Global virtual water flows.

Virtual water flows	Annual flow ($\times 10^9\ m^3\ yr^{-1}$)	Year	Source
Industrial products (blue + gray water)	282	1996–2005	(Hoekstra and Mekonnen 2012)
Agricultural products (Green + Blue Water)	2038 (1386 + 652)	1996–2005	(Hoekstra and Mekonnen 2012)
Agricultural products	2810	2010	(Carr *et al* 2013)
Biofuels (green +blue water)	7.31	2015	(Rulli *et al* 2016)
Virtual water trade of energy production (coal, oil, natural gas, and electricity) (blue water)	6	1992–2010	(Zhang *et al* 2016a)
Total	2333–3105		
Physical water transport (inter-basin water transfers)	490		(Verma *et al* 2009)
Groundwater depletion embedded in crop trade	25	2010	(Dalin *et al* 2017)
Water savings through trade	220	2007	(Dalin *et al* 2012a)
	352	1997–2001	(Chapagain *et al* 2006)
Food aid	10	2005	(Jackson *et al* 2015)
Water grabbing (appropriation through land investments)	380	2013	(Rulli and D'Odorico 2013)

transported over long distances around the world (Gleick and Cooley 2009, Cohen and Ray 2018). Moreover, humans have also tried to divert precipitation artificially through cloud seeding (Bruintjes 1999) and to harvest fog and dew (e.g. Cereceda *et al* 1992, Kaseke and Wang 2018).

However, the total volume of water consumed to produce traded commodities (table 1) is by far greater (and travels longer distances) than the volume of water that is physically transferred in the world (Oki *et al* 2017). Indeed, water remains a resource physically available mainly for local use (Konar *et al* 2016a, Hoekstra *et al* 2018) because transporting crops or other goods is considerably easier than transporting the water required for their production. For this reason, particularly important to the understanding of water resources in a globalized world is the contribution of Allan (1996, 1998, 2002), which elaborates on how water resources are appropriated through the transnational trade of agricultural commodities. The adjective 'virtual' is used to describe how such water is not physically present in the commodities that are traded but is embedded in their production (see box 1).

Trade of crops and other goods existed even in early civilizations. Estimates of the associated virtual water flows indicate that even in the Roman Empire water resources were shared through trade and the water costs of crop production were often externalized beyond regional boundaries (Dermody *et al* 2014). In the modern world, trade has greatly intensified, particularly in the last few decades (figure 1). Contemporary globalization dynamics have greatly enhanced the spatial and temporal dislocation of production and consumption through VWT (D'Odorico *et al* 2014, Porkka *et al* 2017).

The assessment of the net virtual water imports for a given nation, compared to the national consumption (figure 2), can be used to measure that country's reliance on trade for the food and water resources they consume (Tamea *et al* 2013, Winter *et al* 2014). In the course of the last century, the intensification of trade has led some regions of the world to become strongly dependent on food, energy, and materials produced or extracted with water resources existing elsewhere. This raises concerns about issues of national water security and control over the hydrological resources that are necessary for societal development (Carr *et al* 2012). Indeed, many countries are not self-sufficient and depend on imports from other regions to meet their needs (Hoekstra and Chapagain 2008, Carr *et al* 2013, Nesme *et al* 2016). Because of their reliance on trade, the populations of some of these countries often exceed their ability to feed themselves with their limited agricultural resources (D'Odorico *et al* 2010b, Van Ittersum *et al* 2016). For example, the scarce water resources existing in the Middle East are currently insufficient to meet the food demand of the local populations (Allan 1998).

Water is a vital resource controlling production, particularly in agriculture. Virtual water is embedded in agricultural, forestry, industrial, and mining products (Marston *et al* 2018, D'Odorico *et al* 2018). In particular, large volumes of water are required by agriculture, the largest water consumer globally (e.g. Falkenmark and Rockström 2004, Rosegrant *et al* 2009, Richter 2014). Virtual water flows (table 1) have also been investigated in the context of specific subsets of agricultural products used for biofuels (Rulli *et al* 2016), food aid (Jackson *et al* 2015), seafood (Gephart *et al* 2017), and natural rubber production (Chiarelli *et al* 2018). In 2005, virtual water transfers associated with food aid (Jackson *et al* 2015), accounted for only 0.5% of the water footprint of all food trade. Water is also required to produce electricity (Macknick *et al* 2012, Meldrum *et al* 2013) as well as to extract and process minerals (Northey *et al* 2016) and both conventional (Mielke *et al* 2010, Carr and D'Odorico 2017) and unconventional fossil fuels (Nicot and

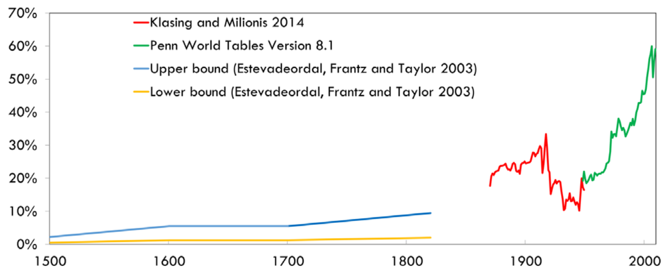

Figure 1. Globalization over five centuries (1500–2011 period). Shown as the world exports and imports expressed as a share of world GDP (%) (after Ortiz-Ospina and Roser 2018).

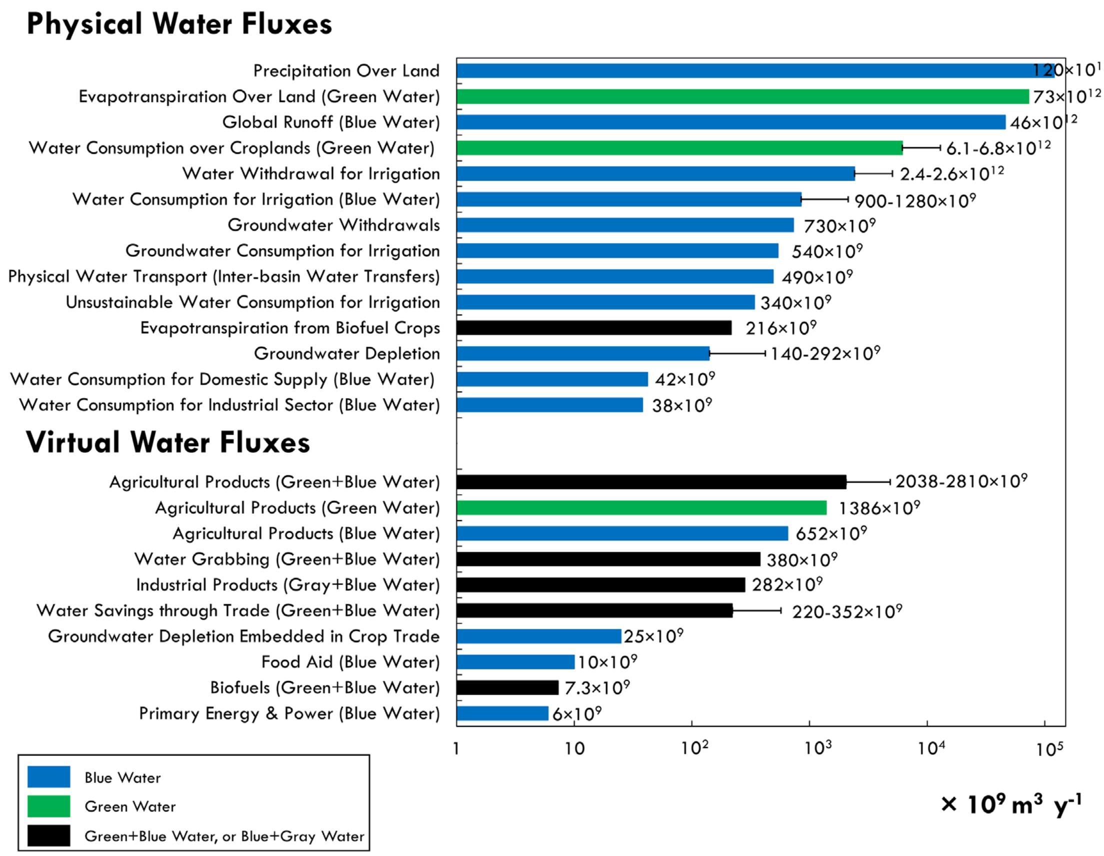

Figure 2. Physical and virtual water fluxes. Interval bars are computed by using different values from previous studies (see tables 1 and 4). Note: The horizontal scale is logarithmic.

Scanlon 2012, Rosa *et al* 2017, Rosa *et al* 2018b, Rosa and D'Odorico 2019).

Water is seldom explicitly accounted for in commodity trade analyses. Typically, labor, economic value, geographic location, and access to capital are the main inputs in trade models (see section 7). Recent trade analyses have considered environmental impacts such as those associated with CO_2 emissions (Deng *et al* 2016b, Vora *et al* 2017, Meng *et al* 2018), however, the study of environmental and social footprints of international trade (Wiedmann and Lenzen 2018) and associated spillovers (Liu *et al* 2015) has often failed to explicitly account for the impacts on water resources. Recent work has explicitly incorporated water as a factor of production into a theoretical trade model (Lenzen *et al* 2013, Lutter *et al* 2016, Dang *et al* 2016). This work incorporates key tradeoffs in agricultural production and decision making.

Most water scarcity indicators only account for local water consumption and local water availability (Liu *et al* 2017), while an important share of water consumption and pollution is due to global and regional trade (Vörösmarty *et al* 2015). With virtual water transfers affecting local water scarcity in importing and exporting regions, there is a need to integrate virtual water flows in water stress assessments (Lenzen *et al*, 2013; Pfister and Bayer 2014, Lutter *et al* 2016) and shed light on how water scarcity is embodied in international trade (Liu *et al* 2017). In this sense the virtual water concept has been criticized as a tool to advise policy-makers because it lacks relevant economic and environmental information about water resources (Gawel and Bernsen 2013). Indeed, the quantitative analysis of water footprint and VWT focuses on water consumption and therefore does not inform about the sustainability of water resource exploitation (Gawel and Bernsen 2013). Thus, water footprint and VWT analyses need to be integrated with a water balance approach to compare the consumption rates with locally available water resources (Lenzen *et al* 2013; Mekonnen and Hoekstra 2016; Soligno *et al* 2017; Rosa *et al* 2018a). Indeed, literature on the globalization of water resources may miss a description of the phenomenon of VWT in the context of its hydrological implications.

In order to understand the relevance and magnitude of the global VWT it is particularly important to appreciate recent developments of international trade. We will here synthesize some of the key moments of contemporary trade patterns and its policies.

3. Recent history of virtual trade and trade policies

In the last several decades, the global patterns of agricultural production often co-evolved with the international trade of agricultural goods and related policies. The distinctive aspect of food trade in the period after World War II with respect to the trade of other commodities was the absence of a general international agreement for liberalization and barrier removal. In fact, the General Agreement on Tariffs and Trade, which promoted liberalization of markets, elimination of trade barriers and expansion of international trade, did not include agricultural commodities. Trade of food products was included only in the 1994 Agreement on Agriculture (Clapp 2016). Between 1947 and 1972, the world's agriculture saw a big gap between national and international regulations, which led to the establishment of the so-called 'food regime', whereby the United States (US) protected its domestic economy (Friedmann 1993) following policies that were put in place after the Great Depression. To increase farmers' incomes, the New Deal (1933–1938) set minimum prices for commodities and maintained these prices constant through government purchases. This encouraged farmers to produce more, with a consequent problem of surpluses that needed to be disposed of, often by favoring exports through foreign aid and export subsidies (Friedmann 1993).

In addition to the effects of economic policies, major changes in global food production and trade resulted from the adoption of modern agricultural technology. After World War II production (and surpluses) further increased as a result of technological advances, and the industrialization of agriculture (e.g. Erisman *et al* 2008, D'Odorico *et al* 2018). Many agri-food corporations engaged themselves in intensive livestock operations as well as maize and soy farming sustained by the use of fertilizers, irrigation, new crop varieties, and other innovations of the Green Revolution (e.g. Delgado *et al* 1999, Pingali 2012). This transition in the agricultural production system significantly threatened the natural capital by inducing loss of biodiversity, soil erosion, freshwater pollution, and increased greenhouse gas emissions (e.g. Ward 1993, Montgomery 2007). It also provided an unprecedented excess in the supply of crops that were used as animal feed, thereby dramatically increasing livestock production often in concentrated operations, a phenomenon known as the *livestock revolution* (Delgado *et al* 1999, Davis and D'Odorico 2015). The intensification of crop production came at the cost of environmental damage (e.g. Ward 1993) and of a specialization in the production of a narrow range of products, which further increased the reliance on international trade of agricultural goods (Friedmann 1993).

The Marshall Aid to Europe (1947) was one of the cornerstones of the 'food regime'. In fact, the Marshall Plan was the first example of big foreign aid, which boosted the Atlantic agro-food relations; funds from the Marshall Aid to Europe were used to purchase US surplus commodities (maize and soybean, mostly) at rates 50% below domestic price and, at the same time, investments in the European livestock sector made it reliant on the US industrial feedstuffs. Thus, the Marshall Plan promoted an integration between the US and European agricultural economies. In the same period, Marshall Aid also replicated the US model for rice production in Japan, South Korea and Taiwan, but without achieving an integration with the US agro-food sector as in Europe.

The strict linkages between US and Europe did not hamper the European protection policy for some goods. In particular European wheat and dairy products were under import controls through the Common Agricultural Policy (CAP 1962), which made Europe less dependent on wheat imports from the US. Therefore, the US had to find other wheat importers outside Europe, targeting particularly developing countries in Asia and Africa (Friedmann 1993). By the 1970s, the developing world became therefore dependent on cheap wheat imports from the US, while tropical crops from developing countries (i.e. sugar and vegetable oils) were replaced by new industrial substitutes made in the US using subsidized maize and soybean surpluses (Friedmann 1993).

In the 1972–1973 period, the Soviet Union, taking advantage of the *Détente* period with the US (i.e. easing of the strained relations), bought 30 million tonnes of grain from the US (Brada 1983). The consequence was a sudden food scarcity worldwide (e.g. Gerlach 2015). This food shortage, combined with the concomitant oil crises (Yergin 2011), and the beginning of the multidecadal Sahel drought (Nicholson 2000, Wang and Eltahir 2000, Dai *et al* 2004) increased the cost of food

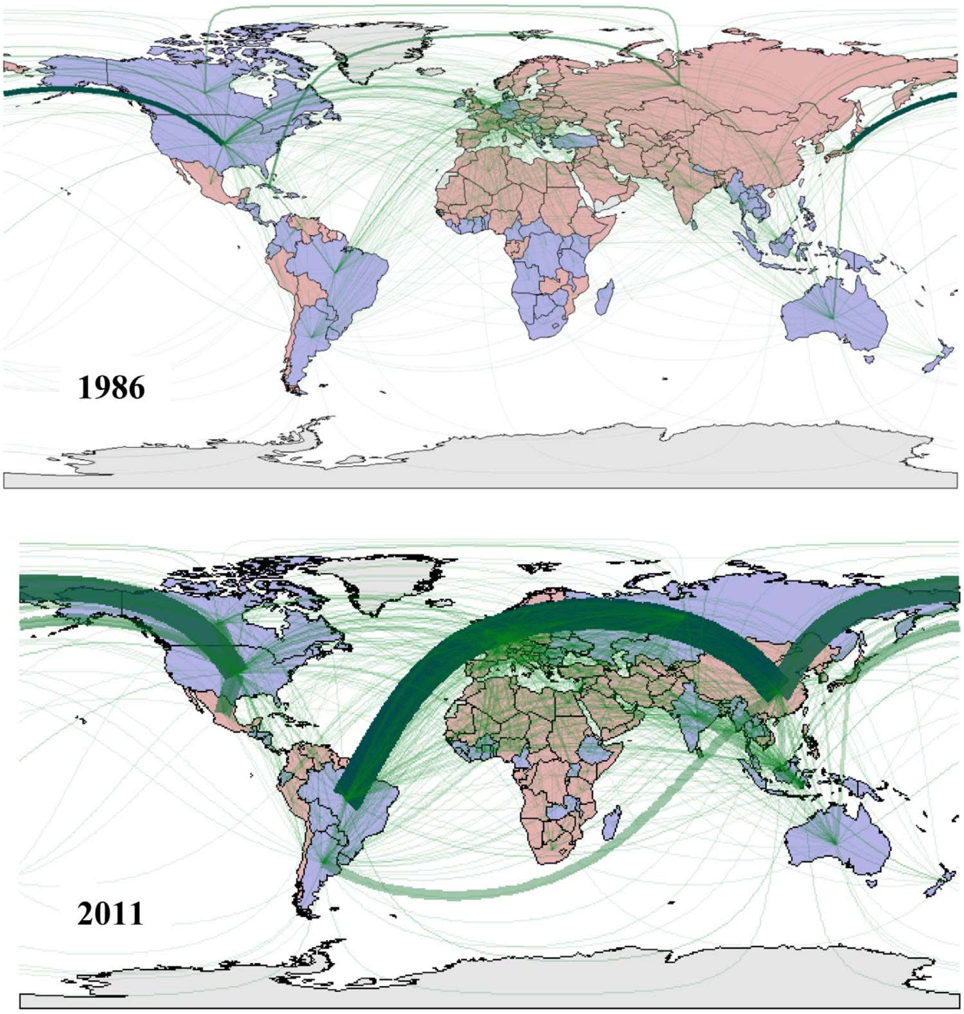

Figure 3. Global patterns of major virtual water trade in 1986 (top) and 2011 (bottom). In purple: net VW exporters. In pink: net VW importers. (From Carr and D'Odorico (2017). Used with permission from Anthem Press.)

in the world (Friedmann 1993). Therefore, the international market became unreliable for a number of import-dependent countries, which started to look for new suppliers (e.g. Japan found new suppliers in the developing world). The subsequent decades then saw a gradual decrease in the leading role of the US in the global agricultural trade and the end of the US-centered 'food regime' with the emergence of multiple new pivots. Countries of the developing world and of the former socialist block joined the multilateral trade negotiations at the GATT. A noteworthy example of the end of the 'food regime' can be found in the case of the soybean market. By the late 1980s, the US lost the control of this market (figure 3), while Brazil started to become a major exporter of soybeans and soy meals. Overall, Brazil and other major agricultural countries such as Argentina and India, were then able to compete with the US for export markets.

The emergence of new suppliers and new trade partnerships greatly enhanced international trade and the globalization of food, as reflected by the increase in the number of trade relationships and in the amount of food traded (D'Odorico *et al* 2014). Between 1986 and 2011 international food trade and the associated trade of virtual water almost tripled (figure 4) (Carr *et al* 2012, Dalin *et al* 2012a). In 2010 international trade accounted for 24% of global food production and associated virtual water (Carr *et al* 2013). Major changes in the recent history of agricultural trade include the ever increasing presence of the People's Republic of China as a major food importer (in year 2005), particularly from South America, the increase of soybean exports from Brazil and Argentina to southeast Asia (da Silva *et al* 2016, Zhang *et al* 2016b), and the escalating exports of palm oil from Indonesia and Malaysia to China, India, Pakistan, and Europe (Porkka *et al* 2013, MacDonald *et al* 2015) (figure 3).

The end of the US-centered agricultural trade era left some developing countries in conditions of strong food dependency, stagnating export revenues and debts (Friedmann 1993, Ward 1993). Decades of access to subsidized agricultural surpluses from the US impeded the development of domestic commercial farming (e.g. Ward 1993, IAASTD 2009, Yu and Nin Pratt 2011). In importer countries, local farmers, whose crops instead of being subsidized were taxed,

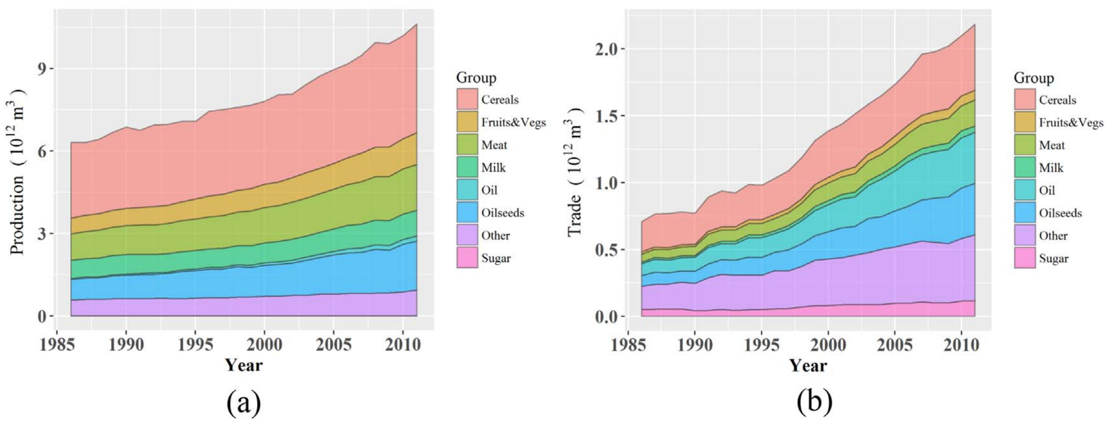

Figure 4. (A) VW Production of 150 primary and (B) subsequent VW Trade of 266 primary and secondary major food commodities removing feed and seed. (Analysis based on FAOSTAT data.)

were often run out of business by cheaper subsidized imports from the US (Ward 1993, IAASTD 2009). Thus, import dependency was often both a cause and an effect of limited agricultural development rather than of a shift to a more profitable non-agricultural economy (OECD 2013). In response to import dependency, debt, and import restrictions in developed countries, developing countries had to export non-traditional products such as exotic foods and flowers, which often contributed to land degradation, destruction of local food systems, and social inequality (Friedmann 1993, Hale and Opondo 2005, Mena-Vásconez *et al* 2016, Lanari *et al* 2018). At the same time, overproduction in developed countries often occurred at huge environmental cost in terms of pollution from use of fertilizers and pesticides, loss of habitat and biodiversity in intensive monocultures, and topsoil erosion (Ward 1993, Montgomery 2007). Thus, export subsidies benefited exporting companies but had environmentally harmful impacts in production regions and socio-economically detrimental effects on the receiving markets (Ward 1993).

Through their investments in multiple regions of the world, multinational agro-food corporations—whose interests were often neither aligned with those of producing nor importing countries—strongly contributed in determining the global patterns of transnational food trade (Murphy 2008).

3.1. Geography of trade routes and their vulnerability

Commodities are traded by road, rail, and sea, often through vulnerable routes (Bailey and Wellesley 2017). Disruption of these routes may cut off supplies while raising prices. Bailey and Wellesley (2017) identified 3 inland and 11 maritime potential bottlenecks (or chokepoints) worldwide (table 2 and figure 5). These chokepoints might be enhanced by intensifying meteorological events, underinvestment in infrastructure, increase in trade volumes, and conflicts.

Table 2. Food infrastructures chokepoints and percentage of global total key crops exported through each chokepoint (year 2015). Source: Bailey and Wellesley (2017).

	Wheat	Rice	Maize	Soybean
Brazil's ports	—	—	18%	33%
US Gulf ports	4%	5%	20%	17%
Black Sea ports	26%	—	18%	—

The increasing importance of international trade is also seen in the efforts to build new trade infrastructures. The Chinese government is developing the Belt and Road Initiative to connect Eurasian countries (Weidong 2015). For some time, there have been plans to build the channel of Nicaragua as an alternative of the Panama Canal (Heilmann *et al* 2014). The Turkish government with the Channel Istanbul is planning to build an artificial waterway channel to connect the Aegean and Mediterranean seas (Dogan and Stupar 2017). A new railroad is planned to connect Brazil to the Pacific Ocean (Müller and Colloredo-Mansfeld 2018). Climate warming is creating new routes in the Arctic that could reduce shipping time and pressure on the congested Turkish Straits (Patel and Fountain 2017). The US and Ukraine are building new capacity and expanding grain terminals (Bailey and Wellesley 2017).

4. Patterns of VWT

The globalization of water (Hoekstra and Chapagain 2008) associated with the transport of virtual water resources from one region to another is an interesting case of embedded complex systems. It results from the intertwined nature of production, distribution and consumption of products that leverage freshwater. Large-scale food systems, alongside production, consumption and population changes, also incorporate virtual water flows associated with trade (Hoekstra and Chapagain 2008), large-scale land

Figure 5. Major maritime chokepoints of agricultural commodity international trade. Redrawn from: Bailey and Wellesley (2017).

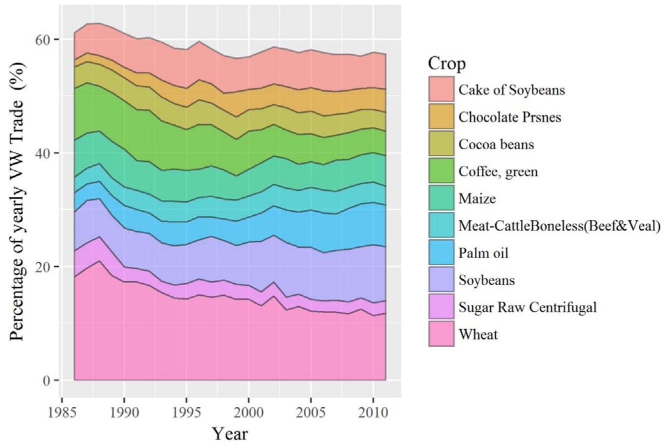

Figure 6. Relative contribution of different food commodities to virtual water trade. (Analysis based on FAOSTAT data.)

acquisitions (Rulli *et al* 2013), food waste (Gustavsson *et al* 2011), and more in general cultural behaviors (Ingram 2011).

The conversion of diverse elements (i.e. trade, land acquisitions, or waste) to a common currency (e.g. virtual water or virtual water per capita) allows us to explore the combined effect of the redistribution and disposal of agricultural commodities on the global food system and its impact on the global freshwater resources. Translation for a single year of detailed trade data of 58 major crop commodities from the Food and Agricultural Organization to the network of VWT allowed for the analysis of the topological properties of this network using methods from network theory (Konar *et al* 2011). The fluxes of virtual water associated with international trade follow a power law relationship with nodal degree (e.g. the number of export links a country has). Interestingly, VW flows remain concentrated to a small number of links and country nodes (or 'hubs') (Konar *et al* 2011). The structural properties of the VW network can be explained by geographic factors such as rainfall on arable land and economic indicators such as the gross domestic product (GDP) of the nations participating within the network (Suweis *et al* 2011).

The temporal reconstruction of the VWT network (Carr *et al* 2012, Dalin *et al* 2012a) has allowed for examination of changes in the geographic distribution of VWT and network properties in the last few decades. These analyses have highlighted that cereal grains tend to comprise the largest proportion of virtual water fluxes with soybeans, vegetable oils, and luxury goods such as coffee and chocolate also accounting for large portion of the traded virtual water (figure 6; Carr *et al* 2013). Simple VWT balances reveal that countries such as the United States, Brazil, Argentina, India, and Australia act consistently as net exporters, and Germany, Italy, Russia, and Japan act as net importers of virtual water (Carr *et al* 2013). Some regions, such as the Middle East have increased their importation of virtual water resources, while other regions such as Central Africa and China have switched from being net exporters to net importers of virtual water (Carr *et al* 2013). Interestingly, increased exports from South

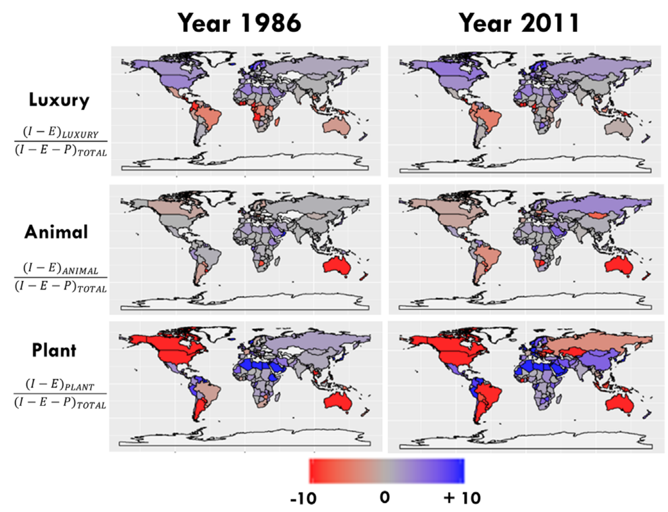

Figure 7. Major importers and exporters of luxury goods, animal products, and plant-based products in 1986 and 2011. Red countries are net exporters, blue countries are net importers. I, E, and P are imported, exported, and produced quantities of luxury, animal and plant-based commodities in each country, respectively. (Based on data analyses in Carr *et al* 2013.)

America, specifically Brazil and Argentina decreased the North American share of trade to both Asia and Europe from 1986 to 2007, which reflect historical changes in global trade and the loss of centrality of the US in agricultural exports (section 2).

The analysis of VWT by commodity classes (i.e. plant, animal, and luxury products, see Carr *et al* 2013) shows completely different VWT patterns across class (figure 7). Interestingly, many developing countries are net exporters of VW associated with luxury goods but importers of crops and animal products. In the 1986–2011 period, Brazil gained increasing importance as an exporting country of animal and plant-based products.

Carr *et al* (2012) investigated the global trade of 309 crop commodities using data from the Food and Agriculture Organization (2017) and showed that the associated total virtual water flux doubled from 1986 to 2008, and concurrently the number of links in the virtual water network increased 92%. Similarly, Dalin *et al* (2012a) leveraging the H08 model (box 3) also showed a doubling in both total virtual water flux and number of links among 58 major commodities. Both of these studies showed an almost doubling in average node strength (total flux) and degree (number of trade partners) over a similar 22–23 year period.

The VWT network is overall extremely dynamic and even links which carry large volumes of virtual water display intermittent behavior in the sense that their strength is not consistent from year to year (Carr *et al* 2012). Further, countries with few connections tended to remain less connected over time as exemplified by the broad lack of engagement of African nations in the food trade network. D'Odorico *et al* (2012) examined the temporal changes in community structure of the virtual water network demonstrating an increase in clustering of VWT, and that, while the network is highly variable, trade tends to be organized within communities.

The analysis of VWT has also led to ethical considerations regarding the inequalities in the distributions of water and population (Seekell *et al* 2011). It was found that VWT tends to reduce inequalities among countries in water use for food relative to well-being thresholds (Carr *et al* 2015). Moreover, international food trade provides access to nutrients and enables some poorer countries to be able to nourish hundreds of millions of people (Suweis *et al* 2013, Wood *et al* 2018). In these studies, it is unclear what the null model of trade equality might look like. Dang *et al* (2015) quantified the inequality within the US virtual water flow network. The US network is relatively homogeneous and social (e.g. normal node degree distribution, clustered Pennock *et al* 2002), making it a suitable null model for global VWT. The US virtual water flows have a Gini coefficient of 0.51 while the Gini Coefficient of global VWT is 0.63 (Dang *et al* 2015).

5. Refining resolution of virtual water transfers

When quantifying VWT, there are two main inputs: the water footprint (or virtual water content, VWC, see box 1) and trade (T). The resolution of each variable restricts the resolution of VWT estimates. This section analyzes the resolution in space, time, water source, water boundary, and commodity coverage that have been used in global reconstructions of VWT (see box 3).

The literature on VWT began by tracking the water embodied in international trade. Early studies assumed that the VWC of a product was constant within a country (Hoekstra and Hung 2002). This

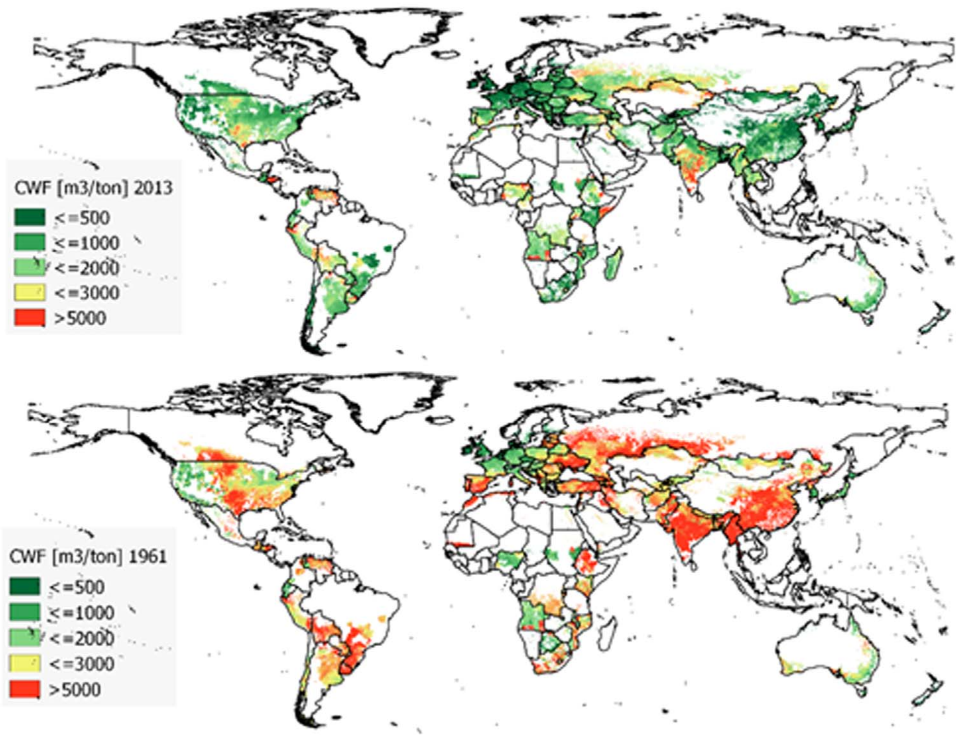

Figure 8. Comparison of the spatial distribution of the crop water footprint (CWF, also referred to as 'virtual water content', VWC) of wheat in 1961 and 2013 based on the 'fast-track' method from Tuninetti *et al* (2017b) using national yield data from years 1961 and 2013 (from FAOSTAT). The spatial distribution of wheat cultivations is kept constant in time and equal to the cultivated areas in the reference year (2000). Irrigation variability in time is indirectly accounted for as it contributes to the definition of national yields.

assumption is problematic for large countries, due to their high spatial heterogeneity in both climate and production patterns. In those countries internal redistribution of food and the attached virtual water can be significant and some studies have begun to examine this internal flow of virtual water (Guan and Hubacek 2007, Dang *et al* 2015). Data on trade initially included only agricultural and food commodities, ignoring non-food items that may also consume large volumes of water. Similarly, early research used temporal averages of VWC (Hoekstra and Chapagain 2007). This ignores the large interannual fluctuations in climatic conditions over time, as well as productivity trends, such as those induced by increasing demand or technological changes. Additionally, in initial VWT studies water use was lumped across source (i.e. rainfall, surface water, groundwater). In this way, differences in the source of water being used to produce commodities were ignored. Similarly, early work did not use naturally occurring hydrologic boundaries (i.e. watershed boundaries) to define the system, making it difficult to link to water resources management.

Water footprint and virtual water transfer estimates have seen recent advances to address all of the shortcomings outlined above. Here, we detail the state of the literature in terms of refining our estimates of VWT in space, time, commodity coverage, water source, and water body.

5.1. Spatial resolution

Initial VWT studies combined estimates of VWC at the national scale with trade data that were also national in spatial scale. Recently, great strides have been made in refining the spatial resolution of VWC (i.e. typically to a grid covering the globe, see figure 8) and then combining with national trade data. Now, the current frontier is in further resolving trade flows in space.

To estimate crop water footprint, or VWC, most studies utilize a crop water model to calculate the consumptive water requirements (i.e. evapotranspiration, ET). CROPWAT (Allen *et al* 1998) is a commonly used model (Mekonnen and Hoekstra 2011, Tuninetti *et al* 2015), though models such as the Global Crop Water Model (Siebert and Döll 2010, Hoff *et al* 2014), H08 (Hanasaki *et al* 2010, Dalin *et al* 2012a), and AQUACROP (Zhuo *et al* 2016) have been widely utilized as well. These models perform a calculation of potential and actual evapotranspiration relying on a simplified soil water balance. Their use of finer spatial scales is only limited by the availability of fine-grain information on crops, soil properties, and atmospheric variables, as well as by the computational time (e.g. figure 8). To date most of the global analyses of the water footprint of crops have been performed at resolutions ranging between 5 and 30 arc min (or between ~10 and ~50 km)(see table 3; Tuninetti *et al* 2015).

Most studies on VW trade quantify international flows with commodity group resolution typically

Table 3. Studies about the global VWC of crop production at high spatial resolution (after Tuninetti *et al* 2015).

Study	Scale	Resolution	Period	Crop yield
Rost *et al* 2008	Global	30 arc min	1971–2000	Country average
Hanasaki *et al* 2010	Global	30 arc min	1985–1999	Country average
Liu and Yang 2010	Global	30 arc min	2000	Country average
Siebert and Döll 2010	Global	5 arc min	1998–2002	5 arc min
Mekonnen and Hoekstra 2010	Global	5 arc min	1996–2005	5 arc min
Zhuo *et al* 2014	Local	5 arc min	1996–2005	5 arc min
Tuninetti *et al* 2015	Global	5 arc min	1996–2005	5 arc min
Rosa *et al* 2018a	Global	5 arc min	2000	5 arc min

limited by the Harmonizing Commodity Description and Coding System (HS code) and the FAO food groups, since trade data are predominantly available at this spatial scale (e.g. FAOSTAT, COMTRADE) and the paucity of detailed sub-national trade data is a major limiting factor. Sub-national VW trade studies typically pair VWC with modeled estimates of sub-national commodity transfers (e.g. Verma *et al* 2009, Zhang and Anadon 2014, Dalin *et al* 2014, 2015, Rushforth and Ruddell 2016, Hoekstra and Mekonnen 2016) or MRIO models (Guan and Hubacek 2007, Dong *et al* 2014, Zhang and Anadon 2014, Deng *et al* 2016a, Serrano *et al* 2016, Ren *et al* 2018).

Recently, sub-national studies of VWT based upon empirical sub-national commodity transfers (i.e. using Commodity Flow Survey (CFS) or Freight Analysis Framework (FAF) data) have been introduced for the US (Lin *et al* 2014, Dang *et al* 2015). Sub-national studies based upon modeled domestic transfers have also been developed (e.g. China in Dalin *et al* 2014, Brazil in Flach *et al* 2016).

VW trade estimates that are highly resolved in space provide the greatest opportunity to evaluate links between water scarcity, water resources sustainability, and complex supply chains (Flach *et al* 2016). For example, VWT resolved to the urban spatial scale enables the quantification of exposure and resilience of cities to direct and indirect water stress (Rushforth and Ruddell 2016). There is significant potential to evaluate high spatial resolution VWT within the US, due to the availability of sub-national empirical transfers (e.g. CFS and FAF databases), however these databases are limited in commodities and temporal resolution when compared to the international trade databases. Improvements in the spatial refinement of VW trade in other countries, however, will continue to be limited by a lack of data, making commodity flow modeling essential.

In hydrology, the watershed is the landscape unit typically used in the analysis and management of water resources. Increasingly, VWT studies are attempting to relate to this hydrologic unit of analysis. In this way, VWT studies will be more able to link with watershed scale hydrologic flows and management issues. Hoekstra and Mekonnen (2012) used the river basin as the unit of analysis to assess water scarcity globally. They found that roughly half of all basins evaluated are subject to severe water scarcity at least one month per year. Wang and Zimmerman (2016) quantified the impacts of VWT for water use and stress at both the national and watershed scale. To do this, they analyzed over 12,000 watersheds. Their study concluded that VWT mitigates water stress in some of the world's most stressed watersheds. VWT for the Great Lakes (Mayer *et al* 2016), Yellow River basin (Feng *et al* 2012), and major aquifers of the US (Marston *et al* 2015) have been evaluated. These are examples of VWT studies at the watershed scale that provide information at a scale that is meaningful for water resources managers.

5.2. Temporal resolution

The temporal variability of VWT depends on the annual trade patterns and on the VWC variability. While annual trade patterns are readily available from open dataset (e.g. FAOSTAT, COMTRADE), VWC values need to be estimated typically through an analysis of the consumptive use of water by crops and crop yields (Mekonnen and Hoekstra 2010). A number of studies (e.g. Carr *et al* 2012, Konar *et al* 2012, Tamea *et al* 2014, Tuninetti *et al* 2017a) adopted constant values of the VWCs to reconstruct the VWT network and its changes through time. This means that in these analyses temporal changes in VWT result from changes in trade patterns but not in VWC, which corresponds to considering constant crop yields and climate conditions. These studies showed a trend of increasing VWT, as globalization led to increased trade connections and exchanged volumes (Carr *et al* 2012, Dalin *et al* 2012a). These results point to the important role of temporal variability but only encapsulate time trends in T (i.e. trade) and not VWC (see also box 3).

Improvements to the temporal resolution of VWT, accounting for the interannual variability of the VWC would permit exploration of changes in time and in response to specific events (e.g. drought, political disruption, agricultural advances). For instance, Dalin and Conway (2016) show how socio-economic change and climatic variability in southern Africa propagated through the global VWT network. Importantly, implementing the temporal variation in the VWC is essential to evaluate sustainability issues that

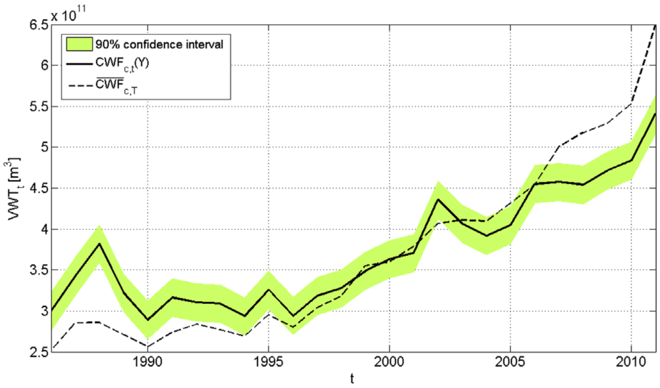

Figure 9. Virtual water trade of wheat, maize, rice, and soybean as in Tuninetti *et al* (2017b). Dashed line refers to VWT estimates based on a constant (i.e. time-averaged in 1996–2005) crop water footprint (or virtual water content). The solid line refers to estimates using time-varying crop water footprint.

may not be evident when average values are considered.

Recently, a 'fast-track' approach to deal with the temporal dimension of the VWC has been introduced and validated (Tuninetti *et al* 2017b). Accordingly, the VWC temporal variability is solely ascribed to the yield change, while the effect of evapotranspiration is assumed to be negligible compared to the yield effect. A comparison between the VWC of wheat in 2013 and 1961 (figure 8) shows a decrease in crop water requirement in the last 50 years, which reflects a concurrent improvement in crop yields. The sensitivity of VWT estimates to the temporal variability in VWC of the main staple crops shows how, when the temporal variability of VWC is accounted for, the corresponding volumes of VWT in the last few years are smaller than in the case with average VWC for the 1996–2005 period (figure 9).

Other studies on the temporal variability of VWT estimated annual values of VWC, allowing for both yields and the evapotranspiration to change (Hanasaki *et al* 2008) in global (Dalin *et al* 2012b, Konar *et al* 2012) and local (Dalin *et al* 2014, Dalin and Conway 2016, Marston and Konar 2016) scale VWT assessments. For instance, to evaluate the impact of the California drought for the years 2012–2014, Marston and Konar (2016) estimated annual VWT values. To do this, they calculated annual values of both trade and VWC, highlighting the importance of time trends in both variables and providing a methodology for future time-varying VWT studies to emulate.

5.3. Commodity coverage

Early VWT literature quantified the water embodied in agricultural and food trade (Hoekstra and Hung 2002) with commodity group resolution typically limited by the Harmonizing Commodity Description and Coding System (Harmonized System, HS) and the FAO food groups. Now, studies are increasingly including non-food commodities (both from agriculture and mining) as well, due to the realization that these commodities also use significant volumes of water. Virtual water flows have been assessed for industrial products (Hoekstra and Mekonnen 2012, Hassan *et al* 2017), biofuels (Rulli *et al* 2016), and natural rubber production (Chiarelli *et al* 2018) (table 1). Virtual water flows have also been estimated for energy sources such as fossil fuels (table 1). Zhang *et al* (2016a) estimated that 10% of the water needed to extract oil, natural gas, coal, and produce electricity is embodied in energy that is internationally traded. Chini *et al* (2018) quantified VWT related to electricity production in the US. Holland *et al* (2015) quantified the telecoupling between global energy demand and pressure on scarce freshwater resources in regions distant from the areas of energy consumption. Quantification of non-food VWT is possible due to recent advances in the calculation of non-food VWC (Mielke *et al* 2010, Hoekstra and Mekonnen 2012, Meldrum *et al* 2013). Commodity specific trade information is available for international trade. However, empirical information on sub-national commodity transfers typically lump commodities into groups (e.g. the CFS and FAF databases) (Lin *et al* 2014, Dang *et al* 2015). In this way, there is currently a tradeoff in the spatial resolution and commodity resolution/coverage available to VWT studies.

5.4. Water source

Identifying the water source is crucial to investigating its availability, opportunity cost, and potential variability under a changing climate. For this reason, it is increasingly important to distinguish 'blue' and 'green' components of VWT. Blue water (box 1) is comprised of water flowing through and stored in surface water bodies (streams, rivers and lakes) and aquifers, or, more simply, surface water and groundwater (Falkenmark and Rockström 2006). This water can be withdrawn (e.g. pumped through wells, or diverted from rivers and lakes), transported through channels and pipelines, and then used for municipal, industrial, and agricultural (i.e. irrigation) needs. Green water (box 1) refers to water in the root zone from precipitation supplies. In other words, green water refers to water stored in the soil and used by plants in both rainfed and irrigated agriculture (of course irrigated agriculture also uses blue water).

Recent studies have investigated the impact of water use in agriculture on the water source. Environmental flows describe the quantity, quality, and patterns of water flows required to sustain freshwater ecosystems and the ecosystem services they provide (Acreman *et al* 2014). Thus, blue water use can be analyzed based on its environmental and sustainability impacts (Mekonnen and Hoekstra 2016, Yano *et al* 2016, Zhuo *et al* 2016, Rosa *et al* 2018a).

5.4.1. Green and blue VWT

Green and blue water uses have different socio-environmental effects in terms of competition with other water needs and cost, though these two different water reservoirs are inter-connected (e.g. when the soil is filled to field capacity—with potential for green water use—excess water undergoes gravity drainage to the underlying aquifer and may eventually reach streams or other surface water bodies—potential for blue water use).

First, there is generally more competition for blue water use than for green water use. Competition may be particularly high for water resources stored in reservoirs, rivers, and lakes, as this water can be used for irrigation but also for hydropower generation, drinking water, energy extraction and production, mining, and other industrial purposes (Rosa *et al* 2018a, 2018b, D'Odorico *et al* 2018). Likewise, groundwater reserves are also often used for agriculture as well as industrial and drinking needs. The main competition that may arise for green water use is actually attached to the land. If no crops were planted, the soil moisture would have different fates depending on the land use type (forest, grassland, or built-up land), but once crops are planted, there is no other potential use of green water. Second, beside the cost of land, using green water in agriculture is a natural process and does not come with any additional direct operational cost. Indeed, green water becomes available at no cost through precipitation, though its productive use by crops requires indirect costs to prepare the soil (e.g. plowing, mulching, seeding, and weed removal) for rainfed agriculture. Conversely, the use of blue water comes with a direct cost, which is that of building, maintaining and powering irrigation infrastructure, such as canals, pumps, wells, and drip or sprinkler irrigation systems.

Much of the VWT literature has focused on trade of agricultural products, which not only are the main water consumers (about 90% of blue water consumption (BWC) by human activities globally, Postel *et al* 1996, Falkenmark and Rockström 2006) but also are the only products that may have both a blue and green VWC. Indeed, green water is only used in the agricultural and forestry sectors, while all other goods and services that are not related to agroforestry may only have a blue VWC. It should be noted that aquaculture and livestock use of agroforestry products (e.g. feed) accounts for the green water footprint associated with fish and animals. Interestingly, most cropland is rainfed (about 80%) and globally, blue water represents 12% of total (blue+green water) annual water consumption over cropland (Rosa *et al* 2018a), but irrigated land is twice as productive, accounting for 30%–40% of the global food calorie production (Rosa *et al* 2018a).

Some global studies have tried to separate blue and green water used in agriculture (Rost *et al* 2008, Aldaya *et al* 2010, Siebert and Döll 2010, Rosa *et al* 2018a). Blue and green water are virtually traded via crops with a similar ratio: Konar *et al* (2012) estimate that 12% of the global VWT is contributed by blue water, and this ratio has been stable over time, based on a study on five crops and three livestock products between 1986 and 2006 (Konar *et al* 2012). However, the shares of blue and green water in crop production significantly vary across products and locations. For instance, there is relatively more irrigation in some regions like South Asia than in other regions of the world. Likewise, the production of some commodities such as poultry uses much more blue water than others (Konar *et al* 2011). The share of blue and green water sources contributing to the total VWC of the same commodities may also greatly vary within countries. For instance, in China irrigation on average contributes to roughly 25% of the VWC of crops but in Xinjiang, Ningxia, and Inner Mongolia, crop production more strongly depends on irrigation (85%, 69%, and 49% of their VWC, respectively) (Dalin *et al* 2014). Similarly, the country-average blue water footprint of livestock accounts for 16% of its VWC, while in Ningxia the blue water share of the VWC of livestock is about 54%. This greater reliance on blue water reflects an arid climate with scarce growing-season precipitation. Conversely, other provinces, such as Chongqing and Guizhou, rely almost exclusively on rainfall with only 2% and 3% of water inputs from irrigation, respectively (Dalin *et al* 2014). These differences are then reflected in VW exports from these regions, and explain, for example, why Asia exports relatively more blue water than South America (Konar *et al* 2011).

5.4.2. Surface water versus groundwater

Key distinctions also exist within different sources of blue water, such as groundwater and surface water, which can both be used by all economic sectors (e.g. irrigation, industrial, and municipal uses).

A handful of studies have partitioned blue water into surface and groundwater, since the implications of using each is different (Aldaya and Llamas 2008, Aldaya *et al* 2010, Schyns and Hoekstra 2014 Marston *et al* 2015, Schyns *et al* 2015, Yano *et al* 2015). Now, several studies further distinguish between various types of surface and groundwater resources (e.g. renewable groundwater, groundwater depletion, small reservoirs, large reservoirs) (Hanasaki *et al* 2010, Dalin *et al* 2017).

Greater attention has recently been drawn to groundwater use, as it has been increasingly used for irrigation in many regions (Konikow and Kendy 2005, Wada *et al* 2010, Siebert *et al* 2010, Gleeson *et al* 2012, Scanlon *et al* 2012). The overuse of groundwater can lead to multiple environmental damages, including land subsidence, salt intrusion in coastal aquifers, or die-off of phreatophytes (Konikow and Kendy 2005, Taylor *et al* 2013). A few studies have recently focused on groundwater resources embedded in food trade. In the US, 46 km^3 of groundwater per year is withdrawn from three major aquifers and 13% of blue VWT from US (and 35% of blue water use for US production) come from three aquifers (Marston *et al* 2015).

As groundwater depletion becomes a more alarming issue in several regions across the world (Gleeson *et al* 2012), studies have analyzed the unsustainable use of groundwater due to withdrawal rates exceeding the rates of natural recharge. In some extreme cases, the recharge rates are very small (e.g. the Nubian aquifer in North Africa—see Konikow 2011) and non-renewable water resources accumulated during wetter epochs are 'mined'.

Wada *et al* (2012) found that unsustainable groundwater abstraction contributes to approximately 20% of the global gross irrigation water demand for the year 2000. The greatest rates of groundwater depletion are occurring in India (68 $km^3 yr^{-1}$) followed by Pakistan (35 $km^3 yr^{-1}$), the US (30 $km^3 yr^{-1}$), Iran (20 $km^3 yr^{-1}$), China (20 $km^3 yr^{-1}$), Mexico (10 $km^3 yr^{-1}$), and Saudi Arabia (10 $km^3 yr^{-1}$). In addition, globally, this contribution more than tripled from 75 to 234 $km^3 yr^{-1}$ over the 1960–2000 period.

In many countries, some aquifers are unsustainably mined as a result of crop production for the export market (Dalin *et al* 2017). Unsustainable groundwater use is not a local problem only, because increasingly global markets, companies and consumers worldwide depend on the products derived from unsustainable water supplies (Hoekstra *et al* 2018). Dalin *et al* (2017) estimated crop-specific groundwater depletion associated with irrigation globally, and determined the amounts of groundwater depletion embedded in international food trade in years 2000 and 2010. They found that global groundwater depletion for irrigation increased by 22% from 2000 to 2010 (240–292 $km^3 yr^{-1}$), mainly in China (+102%) and the US (+31%). About 11% of non-renewable groundwater use for irrigation is embedded in international food trade, of which two-thirds are exported by Pakistan, the US, and India alone. The trade of crops resulting from groundwater depletion by top crop exporters has greatly increased from 2000 to 2010 (100% increase in India, 70% in Pakistan and 57% in the US), and the largest increase in the imports of groundwater depletion occurred in China (tripling), and were mainly associated with imports from the US and India.

5.4.3. New versus ancient water

Water can be either physically or virtually transferred not only in space (through pipelines, trade, or foreign direct investments) but also in time. For instance, in some regions groundwater depletion (see previous section) may be contributed by the mining of ancient (or fossil) water that accumulated in aquifers during wetter epochs.

Groundwater mining (Konikow 2011, Taylor *et al* 2013) is an example of a physical use of ancient water. Water from the geological past can also be used in a virtual sense, by using commodities that were produced using ancient water. A notable example is the case of fossil fuels, which formed from the decay of biomass from organism that existed several million years ago. Such biomass contains energy from ancient photosynthesis which relied on the consumptive use of water. An indirect estimate of the ancient water virtually embodied in fossil fuels used worldwide (D'Odorico *et al* 2017) has shown how one year of fossil fuel use by human societies corresponds to a virtual consumption of an amount of ancient water of roughly 7.4×10^{13} $m^3 yr^{-1}$, which is close to the total annual evapotranspiration from terrestrial ecosystems. These results highlight how, to meet its present energy needs, humanity is borrowing water from a geological past. Constraints imposed by the global water cycle (in addition to land availability and food production) do not allow humanity to meet its energy demand by replacing fossil fuels with bioenergy (Carr and D'Odorico 2017). The reliance on ancient water is an example of highly unsustainable use of virtual water resources. Like in the case of groundwater depletion, such resources will not be available to future generations and will not be replenished.

5.5. Grey VWT

The notion of grey water was recently introduced by Hoekstra and Chapagain (2008). Grey water quantifies the theoretical volume of water polluted by agricultural production (see also section 8.4). It represents the volume of water needed to dilute pollutants (namely, nitrogen and phosphorous) to a given water quality standard. Estimates of grey VWT have not been as widespread as estimates of blue and green water (O'Bannon *et al* 2014). This is because estimation of grey water is a theoretical rather than an actual consumptive measure, making it difficult to combine directly with blue and green values. Moreover, the calculation of the grey water footprint depends on the number and type of pollutants that are accounted for, and the quality standards, which are both pollutant- and country-specific. To date, most studies on the grey water footprint (e.g. Hoekstra and Mekonnen 2012)

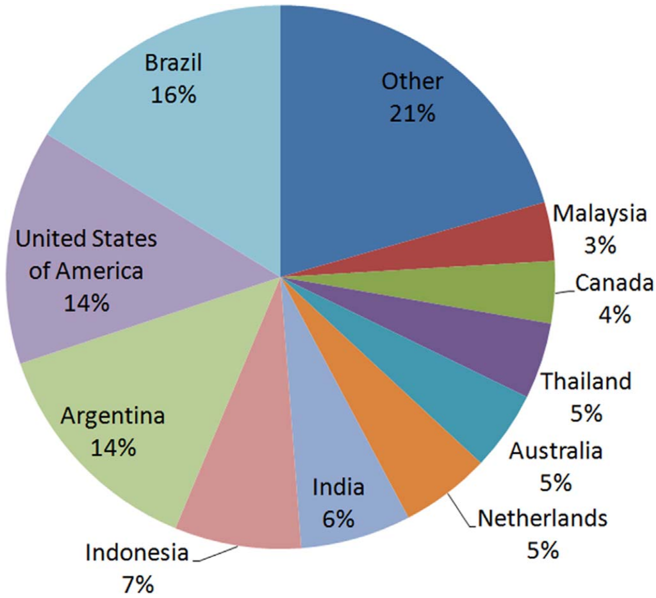

Figure 10. Share of total nitrogen pollution burden from agricultural trade on exporting countries in 2008 (based on analyses in O'Bannon *et al* 2014).

have concentrated on nitrate from fertilizer applications with nitrate concentrations in drainage and runoff water from agricultural field calculated as a fixed fraction of nitrate applications without modeling the underlying soil biogeochemical processes and their variability. It is still unclear how the grey water footprint associated with multiple pollutants (including other fertilizers and pesticides) would be calculated (i.e. as the sum of the grey water footprints of each pollutant or accounting also for their interactions?).

Grey water flows have been used as a proxy for the pollution left in the production region. Thus, when country B imports a certain agricultural commodity from country A, country B is virtually exporting pollutants to country A. Environmental degradation is avoided by diluting those pollutants with an amount of water defined as the grey water footprint. Thus, country A needs to allocate a fraction of its freshwater resources for the dilution of pollutants. A global analysis (O'Bannon *et al* 2014) of grey water flows associated with pollution from nitrogen fertilizers has highlighted the countries bearing the bigger shares of the planetary grey water footprint (figure 10). Interestingly, most of the burden is supported by more developed agricultural countries that make a relatively heavier use of fertilizers. However, the grey water concept has some limitations because it does not entirely describe pollution as a result of production processes. For instance, soybeans imported by Europe from Brazil are used as feed for pigs that are subsequently exported. This causes a manure and NO_3 excess that pollutes groundwater and surface water in Europe, not in Brazil. Thus, part of the environmental costs caused by fertilizers used in Brazil are exported to Europe, but the analysis of grey water footprints does not show this effect. Moreover, international food trade also may have negative environmental impacts in importing countries. For example, because of its reliance on imports, China is converting soybean croplands into corn fields and rice paddies with consequent increase in nitrogen pollution (Sun *et al* 2018). A related concept was developed by Galloway *et al* (2007), with the notion of 'virtual nitrogen' (or 'embodied nitrogen'). When applied to a geographic analysis of that scale (e.g. at the country scale) the nitrogen footprint of that region represents the nitrogen pollution (both of water bodies and of the atmosphere) caused by the consumption habits of the people living in that region. Like its water and ecological counterparts (Wackernagel *et al* 1999, Hoekstra and Chapagain 2008), part of the nitrogen footprint of a country falls outside the boundaries of that country, meaning that pollution is partly exported to other regions of the world (Oita *et al* 2013). Thus, the external component of the nitrogen footprint represents a virtual nitrogen export or, equivalently, a virtual import of grey water. Likewise, other authors have investigated the global phosphorus (and embodied phosphorus) flows associated with agricultural trade (MacDonald *et al* 2012, Nesme *et al* 2016, Hamilton *et al* 2018).

6. Reconceptualizing the global water cycle: accounting for the 'virtual water cycle'

The previous sections have highlighted some important patterns and properties of VWT. But to what extent is VWT redistributing (virtual) water resources around the globe? How do virtual water flows (table 1) compare to the major physical water fluxes in the water cycle

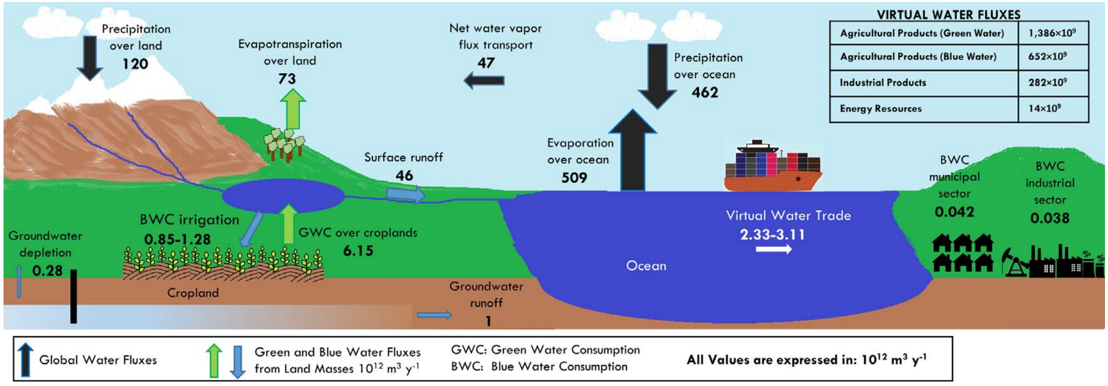

Figure 11. Comparison between physical and virtual water fluxes in integrative depiction of the global water cycle (based on data from table 4).

Table 4. Physical fluxes in the water cycle.

	Annual flow (m^3 yr^{-1})	Year	Source
Precipitation over land	120×10^{12}		
Evapotranspiration from land (green water flows)	73×10^{12}		(Chow et al 1988)
Global runoff (blue water flows)	47×10^{12}		
Blue water withdrawal for irrigation	2.56×10^{12}	2000	(Sacks et al 2009)
	2.41×10^{12}	1980–2009	(Jägermeyr et al 2017)
2.66×10^{12}	2000	(Oki and Kanae 2006)	
Blue water consumption for irrigation	0.90×10^{12}	1996–2005	(Hoekstra and Mekonnen 2012)
	1.28×10^{12}	2000–2010	(Siebert et al 2010)
	0.85×10^{12}	2000	(Rosa et al 2018a)
Green water consumption in croplands			
For 16 major crops	6.15×10^{12}	2000	(Rosa et al 2018a)
For 150 crops	6.79×10^{12}	2000	(Carr et al 2013)
Unsustainable blue water consumption for irrigation	0.34×10^{12}	2000	(Rosa et al 2018a)
Water consumption industrial production	0.038×10^{12}	1996–2005	(Hoekstra and Mekonnen 2012)
Water consumption domestic supply	0.042×10^{12}	1996–2005	(Hoekstra and Mekonnen 2012)
Groundwater consumption for irrigation	0.54×10^{12}	2000–2010	(Siebert et al 2010)
Groundwater withdrawals	0.73×10^{12}	2000	(Wada et al 2010)
Groundwater depletion	0.14×10^{12}	2001–2008	(Konikow 2011)
	0.28×10^{12}	2000	(Wada et al 2010)
	0.29×10^{12}	2010	(Dalin et al 2017)

(table 4)? We define 'virtual water cycle' as a representation of the hydrologic cycle that highlights the virtual water fluxes. Like the physical water cycle, its virtual counterpart includes water stocks (e.g. ocean, land, terrestrial water bodies, and glaciers), and (virtual) water fluxes. The virtual water cycle is powered not by solar energy and gravity forces as the physical water cycle, but by trade and the energy sources (for most part fossil fuels) used for transport by trucks, trains, and ships (figure 11). An integrative representation of the global water cycle, should take into account both the physical water fluxes (traditionally the 'natural water cycle') and the virtual ones (the 'virtual water cycle'). Early insights about the mutual presence and interdependence of the physical and virtual water can be found in a local case study about Egypt (Abdelkader et al 2018).

In this framing it is important to consider the distinction between the different types of water that are consumed. All consumptive water uses entail a loss of water to the atmosphere as water vapor fluxes due to evaporation and transpiration. Thus, BWC accounts for only part of the water withdrawals from water bodies, with the remaining part being returned to water bodies by drainage and runoff processes. The BWC of humanity is dominated by water use in irrigation ((0.85–1.28) $\times$ 10^{12} m^3 yr^{-1}), which by far exceeds BWC by industrial production (0.038 $\times$ 10^{12} m^3 yr^{-1}), and municipal uses (0.042 $\times$ 10^{12} m^3 yr^{-1}). Collectively, these blue water uses account for (0.93–1.37) $\times$ 10^{12} m^3 yr^{-1} (figure 11). Even though these estimates of BWC are only a small fraction (2.4%) of global surface and groundwater runoff, water withdrawals for

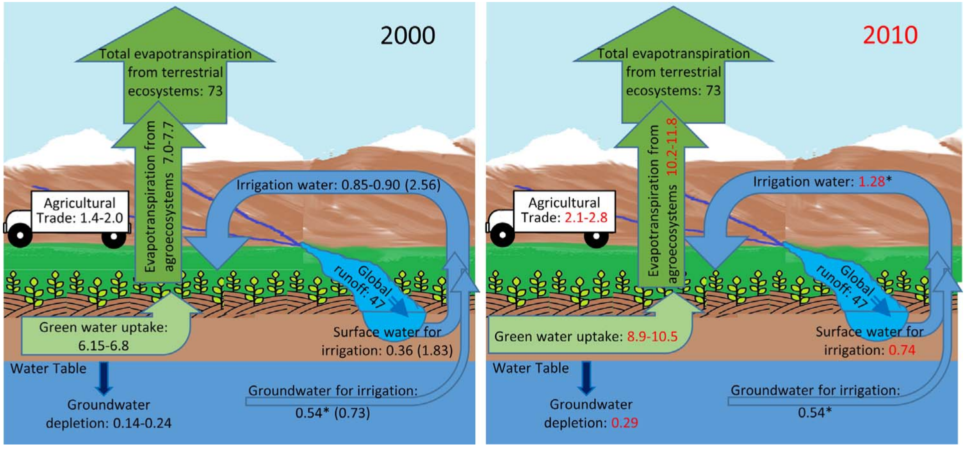

Figure 12. Global physical and virtual water fluxes in agriculture for the year 2000 (left) and 2010 (right) (source: tables 1 and 4, figure 4 and references cited in the text; all values are expressed in $10^{12}\ m^3\ yr^{-1}$). Blue water flows are reported both as consumptive uses and blue water withdrawals (between parentheses). The asterisk (*) denotes average values for 2000–2010. In red are values that have changed between 2000 and 2010.

agriculture and other uses are known for having strongly depleted several rivers, aquifers and other water bodies around the world—such as the Rio Grande or Colorado River in North America—with consequent destruction of aquatic habitat (e.g. Jägermeyr *et al* 2017) and depletion of groundwater resources ($(0.14–0.28) \times 10^{12}\ m^3\ yr^{-1}$). Blue water withdrawals, however, are a much bigger fraction of global runoff. In the case of agriculture (figure 12) blue water withdrawals account for $2.56 \times 10^{12}\ m^3\ yr^{-1}$, or roughly 5% of surface and groundwater runoff. Interestingly, according to these estimates, about 65% of these withdrawals are not consumed and are subsequently returned to aquifers and surface water bodies. Based on estimates for the year 2000, evapotranspiration from agroecosystems ($7.0 \times 10^{12}\ m^3\ yr^{-1}$)—i.e. sum of BWC and crop uptake of root-zone soil moisture (or green water consumption, GWC $\approx 6.15 \times 10^{12}\ m^3\ yr^{-1}$)—is roughly 10% of global evapotranspiration from continental land masses (figure 12).

Thus, agriculture contributes to the consumption of 2.4% of the blue water flows and 10% of the green water flows from the global land masses (figure 12). In other words, in the year 2000 human appropriation of water resources (blue and green) for agriculture accounted for 10% of terrestrial evapotranspiration, which is not a trivial amount of water if we consider that large land areas are not suitable for agriculture (e.g. D'Odorico *et al* 2018, Rosa *et al* 2018a). These estimates, however, are very conservative because they are based on a limited set of major crops (16 crops in Rosa *et al* 2018a, accounting for 73% of the planet's cultivated areas and 70% of global crop production) and do not account for many non-food crops, such as fibers, which would increase the total water consumption (i.e. evapotransiration) by agroecosystems to $(7.4–7.7) \times 10^{12}\ m^3\ yr^{-1}$ in the year 2000 (Oki and Kanae, 2006, Mekonnen and Hoekstra 2011, Carr *et al* 2013). If we include also water consumption for grazing (i.e. pastures) and direct water consumption by livestock, the total water consumption by agroecosystems in the 1995–2005 decade becomes $8.4 \times 10^{12}\ m^3\ yr^{-1}$ (Mekonnen and Hoekstra 2012), or 11.5% of terrestrial evapotranspiration. About 20%–24% of the water consumed by agriculture is virtually traded internationally ($1.4 \times 10^{12}\ m^3\ yr^{-1}$ in the year 2000 and $2.04 \times 10^{12}\ m^3\ yr^{-1}$ in 1996–2005 see figure 4 and table 1).

How have these figures changed recently? - Between the year 2000 and more recent years agricultural production has increased along with the blue and green water consumption by agroecosystems [up to $(10.2–11.8) \times 10^{12}\ m^3\ yr^{-1}$ in 2010, according to some estimates (see figure 4 and Carr *et al* 2013)], while the changes in total terrestrial evapotranspiration associated with climate warming and land use change impacts on the water cycle have likely been much smaller. Thus, the share of terrestrial evapotranspiration contributed by agroecosystems has increased since 2000 (up to 14%–16% by 2010; see figure 12). Likewise, trade volumes have dramatically increased in the last few years (see section 4), reaching about $(2.1–2.8) \times 10^{12}\ m^3\ yr^{-1}$ in 2010 (see figure 4 and D'Odorico *et al* 2018), which is again close to 20%–24% of the more recent estimates of water consumption in agriculture, while the share of terrestrial evapotranspiration that is virtually traded internationally has increased from 1.9%–2.4% to 2.8%–3.8% between 2000 and 2010 and is expected to escalate in the near future as a result of the increasing water demand by agriculture.

Future increase in human appropriation of freshwater resources will likely continue to be dominated by agriculture. Should the increasing crop demand be met through agricultural intensification (i.e. by enhancing crop yields on currently cultivated land) the green water consumption by agroecosystems would likely remain substantially unchanged. However, BWC would increase as a result of the expansion of irrigation on farmlands that are currently rainfed. Recent estimates have shown that irrigation water (i.e. blue water) consumption can sustainably increase at most by 48%, thereby increasing total water consumption (blue + green) for agriculture by 5% (Rosa *et al* 2018b). This 48% increase would bring irrigation water consumption close to 5% of global runoff. Therefore, unless agriculture is expanded to non-agricultural areas (an approach that has often led to soil degradation, 'dust bowls', and habitat loss) agriculture will not be able to appropriate a much greater share of the water cycle than what we see today.

7. Models and drivers of VW trade

Modeling the VWT enables the understanding of governing mechanisms, the identification of driving factors determining network topology and trade flows, and the prediction of future VW trade. The first models of VWT (Suweis *et al* 2011, Dalin *et al* 2012) were fitness models, which generated synthetic networks with similar properties to the observed patterns of VW trade (e.g. Sartori *et al* 2017). Suweis *et al* (2011) used country-specific values of GDP and average rainfall on agricultural areas to reproduce the undirected VW trade network (obtained by summing bilateral flows exchanged between any two nodes). Dalin *et al* (2012b) considered the directed VW trade network and included the population of each country as an additional explanatory variable, with rainfall being a determinant of agricultural production and exports, and population a determinant of food (and water) consumption and imports. Sartori *et al* (2017) identified country GDP, water endowment (or total renewable water resource), and precipitation per capita as drivers of the VW trade network structure. In all cases, the comparison of the real and reconstructed VW trade network is based on network's statistical properties, such as the degree distribution or trade flux distribution, while no attempt is made to evaluate the agreement on individual fluxes between model and data.

A different set of studies focused on the estimation of real fluxes using multi-regression, or gravity models. Tamea *et al* (2014) developed a gravity-like model establishing multi-regressive linear relations for the imports and exports of each country. Despite some differences among countries, a widespread significant dependency is found between VW flows, and drivers such as population, GDP, geographical trade distances, and the agricultural production of exporting countries (e.g. Wang *et al* 2016). A similar model was recently developed to describe the presence or absence of trade links between pairs of countries (Tuninetti *et al* 2017a) who highlighted that population, geographical distances and agricultural efficiency (e.g. due to fertilizers use) are the main factors driving the activation and deactivation of trade links over time. Multi-regression models have also been used to investigate the global relationship between VW trade, cultivated land and water resources. Kumar and Singh (2005) identified cropped land as a relevant factor, although agricultural land appears to have a minor role in other studies (e.g. Tamea *et al* 2014, Tuninetti *et al* 2017b). Irrigated land is found to be relevant for net VW flow associated with specific traded crops, even when they are produced in rainfed conditions (Chouchane *et al* 2018a). Many authors highlight that (blue) water scarcity is not a driver of VW trade (Kumar and Singh 2005, Fracasso 2014, Chouchane *et al* 2018a).

Gravity models have also been used to investigate to what extent VW trade is affected by the water endowment and water scarcity of countries (Lenzen *et al* 2013, Fracasso 2014, Fracasso *et al* 2016). In addition to determinants of bilateral trade flow such as country-specific values of population, GDP, distance and dummy variables about country-pair relationships, Fracasso (2014) found other possible drivers such as per-capita water endowment (measured by water volumes available for agriculture, freshwater availability of exporting countries, and the ratio of dietary requirement over total available water) and water demand (expressed as the ratio of water withdrawals and renewable water). Relevant drivers vary if one considers specific regions instead of the global trade network (Fracasso 2014). For example, in Mediterranean countries large water endowments do not lead to large VW exports, while exports may be hindered by high irrigation water prices (Fracasso *et al* 2016).

At the global scale, water-intensive goods across many different sectors tend to be exported by countries with relatively abundant water resources, in terms of per-capita freshwater resources, as shown by an econometric analysis of country exports by sector (Debaere 2014). Water is found to induce an international specialization of production and is a source of comparative advantage among countries (Wichelns 2004, Debaere 2014). However, VWT reflects more (and is possibly driven by) the opportunity cost of water, i.e. the cost of the best alternative, rather than its comparative advantage, i.e. the lower opportunity costs relative to other countries (Wichelns 2001). For a holistic view of international agricultural trade it is important to consider different metrics, i.e. monetary, nutritional and environmental resource metrics. Such metrics may offer complementary information on causes and implications of trade as well as on how countries' allocation of water and cropland resources determine the globalization patterns of agriculture and trade (Galli *et al* 2013, MacDonald *et al* 2015, Wiedmann and Lenzen 2018).

Other frameworks used to describe VW trade include general equilibrium models of trade economics that have been used, for example, to analyze the changes

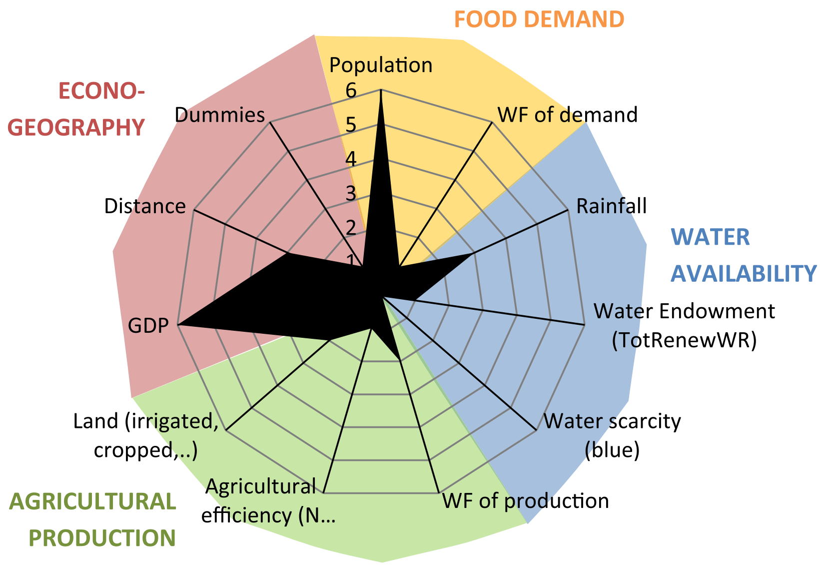

Figure 13. Drivers of VW trade identified in the publications reviewed in section 7. The radial coordinate expresses the number of publications reporting a significant dependence on each variable. Details are provided in the supplementary information (table S1 is available online at stacks.iop.org/ERL/14/053001/mmedia).

of VW trade induced by modifications (reductions) of local water availabilities (Berrittella *et al* 2007). Water reductions are expected to shift trade patterns of VW and induce large welfare losses, although possibly inducing an improvement of water use/allocation efficiency. Yet another model setup is based on complex system dynamics, exemplified by El-Gafy (2014) who proposed a multi-sector model including population, crop production, land use, water footprint of crop production and consumption, and VW balance. The model enables the accounting of sector dynamics and inter-sector feedbacks at the county level and the development of scenarios to support decision making. Likewise, partial equilibrium framework has also been proposed by (Dang *et al* 2016) to describe the effects of policies and decision making on water use in agriculture. This literature on the modeling of the impact of shocks on food prices and trade will be reviewed in the context of resilience analyses of VW trade (section 8.5.).

Figure 13 summarizes the drivers identified in the publications on VWT reviewed in this section. Many of the models presented above enable the development of future projections of the structure of the VWT network and/or of VW flows. Both fitness models and gravity models can be run with projected inputs to assess the possible evolution of trade network and flows (e.g. Suweis *et al* 2011, Sartori *et al* 2017, Abdelkader *et al* 2018). Equilibrium models as well can be applied to assess different future scenarios.

8. Socio-environmental consequences of VWT

Research on VWT has highlighted the existence of regional and global benefits for societies and the environment (Allan 1998). VW transfers are often used for famine relief and to mitigate the effects of regional food crises. VWT prevents massive migrations from arid regions of the world where water resources would be insufficient to meet the needs (food security) of local populations and for this reason it has been argued that they prevent conflict and wars (see sections 8.2 and 8.3) (Allan 1993, 1998). VWT is also associated with important water savings because the overall patterns of agricultural production and trade show that crops are planted in areas within which they have higher water use efficiency and the export to areas in which their production would require more water. This suggests that VWT entails a more efficient use of water resources, which results in a water saving (section 8.1). Other studies have also highlighted how trade either decrease the inequality existing among countries in their access to water for food production (Seekell *et al* 2011, Carr *et al* 2015, 2016). However, trade decisions are seldom directly driven by water needs, as many other factors (including the availability of capital, raw materials, labor, technical knowledge, and policies) contribute to the determination of the global patterns of production and trade (see section 7).

On the other hand, even though VWT can reduce local water deficits by virtually redistributing water resources (Suweis *et al* 2013), it is not a real long-term solution to water scarcity (Suweis *et al* 2013, Jia *et al* 2017) because water remains a globally limited resource that is subjected to increasing pressure from agricultural, industrial, and municipal uses (figure 12). As noted in section 6, human appropriation of freshwater resources for agriculture can sustainably increase in presently cultivated land only by up to 48% (Rosa *et al* 2018a), which would enhance total water consumption in agriculture by 5%. The projected increase in water demand for food production by mid-century is expected to be an order of magnitude greater (e.g. Falkenmark and Rockström 2006). Thus, curbing water demand by using water more efficiently—through soil water conservation, 'more crop per drop' methods, and adoption of water-efficient diets—, while reducing food demand and food waste, appears to be a much needed approach for long-term water sustainability.

VWT may have some negative impacts on societies and the environment (e.g. Carr *et al* 2013). Recent research has highlighted the impact of the globalization of water (and agricultural products) through trade on the resilience of the global food system (D'Odorico *et al* 2010a, Tamea *et al* 2016, Marchand *et al* 2016). The establishment of teleconnections between people and the resources they rely on, may distance consumers from the environmental impact of their decisions with the effect of undermining the ethic of environmental stewardship (Chapin *et al* 2009, Carr *et al* 2013, D'Odorico and Rulli 2014). Some of the environmental externalities of trade have only recently started to be investigated (see section 8.4).

In the following sections we review some of the benefits and impacts of VWT.

8.1. Water savings

International trade can save national water resources through the importation of water-intensive commodities from other countries. National water savings through trade can imply saving water at a global level if the flow is from sites with high to sites with low water productivity (Chapagain *et al* 2006, Martinez-Melendez and Bennett 2016, Brindha 2017). It has been estimated that VWT saves 352 $km^3\ yr^{-1}$ that would be otherwise used to produce agricultural products in the importing countries (Chapagain *et al* 2006) (table 1). Other studies found smaller savings and reported the existence of a growing trend, from savings of roughly 50 $km^3\ yr^{-1}$ in 1986–240 $km^3\ yr^{-1}$ in 2008 (Dalin *et al* 2012a).

Water-scarce nations and regions on average save water resources by importing food commodities. For example, it has been estimated that VWT alleviates water stress and promotes water sustainability in China (Zhao *et al* 2018). Moreover, VWT alleviates water scarcity in importing wealthy countries, while it has limited effects on water scarcity alleviation in poorer countries (Distefano and Kelly 2017). In some cases, international trade can also increase water consumption of agricultural commodity production if crops are grown where they are produced in less environmentally efficient and in more unsustainable ways (Martinez-Melendez and Bennett 2016). Many countries produce commodities at the cost of additional pressures on their water resources. For example, agri-food products are sometimes traded from an area with low water productivity to an area with higher water savings in production (Lamastra *et al* 2017). However, regional trade in Africa is much more efficient in terms of embodied water resources than any other region in the world. Thus, internal African trade patterns may be compensating for poor water productivities in their domestic production systems (Konar and Caylor 2013).

Konar *et al* (2013) have shown that the volume of water savings is likely to increase under a changing climate (see also Dermody *et al* 2014 for the case of the Roman Empire). This is despite the fact that the total volume of VWT is projected to increase under climate change, due to increased crop prices. Water savings occur under climate change because crop trade re-organizes into a more water-efficient structure (Konar *et al* 2013). When free trade policies are enabled, the volume of global water savings increases even more under a changing climate (Konar *et al* 2016b). This indicates that trade liberalization leads to water resources being more efficiently used in the global trade system, making it a potentially important adaptation measure to climate change (Konar *et al* 2016b). These findings are supported by recent causal inference work that shows that trade openness leads nations to use less of their domestic water resources on average (Dang and Konar 2018).

8.2. Hydropolitics of VWT

VWT is a concept that has radically influenced the development of hydropolitical theories. Allan introduced this concept as the result of several years of research on the role of embedded water in agricultural commodities to understand key questions on food security and social stability in water-scarce countries, such as in the Middle East and North Africa (Allan 1996, 1998, 2002). One of the key theoretical implications, is in its power to dispel the myth of future water wars.

The analytical definition of water wars, which should not be confused with the general notion of water conflicts, has several operational categorizations (Dell'Angelo *et al* 2018a). A key characteristic of the formal definition is that a water war occurs when violence is at the State level, specifically when there is interstate military confrontation (Wolf 1998, 2007). Referring to this precise analytical definition, many scholars have coherently worked to debunk the 'water leads to war thesis'. One of the strongest arguments in the literature that contributes to this theory of water peace is the one that derives from the VWT assessment. Allan explained how several water-stressed countries such as in the Middle East do not have enough water to grow locally the food that would be necessary to satisfy the needs of their populations (Allan 1996, 1998, 2002). This condition of hydrological scarcity should lead to expected social tensions, unrest and competition with neighboring countries which could lead to violent escalations and potentially to formal military engagement among different countries.

What the work of Allan demonstrates, is that international trade allows countries to circumvent their local physical water scarcity restrictions. His studies showed that countries' dependence on agricultural production, which on average is the most water-intensive sector in society, can be almost entirely satisfied by the importation of agricultural commodities (Allan 1996, 1998). Historical evidence about North Africa and the Middle East for example shows that the virtual water flows associated with grain imports from North America are larger than the actual water flows of the Nile river (Allan 1998). It is on the bases of these kinds of hydrological assessments and his observation of trade and agricultural policies, that Allan developed a coherent theory about the irrelevant likelihood that countries could go to war because of water. The underlying logic that Allan highlights is that for water-scarce countries it is much more convenient to benefit from virtual water through agricultural commodities importation than to compete and fight with other countries for direct control of physical water resources.

Despite some criticism (see Ansink 2010), the role of VWT represents one of the fundamental arguments that are invoked to refute the 'water leads to war' thesis. This perspective has been recently confirmed by quantitative tests. De Angelis *et al* (2017) analyzing data on VWT, found that bilateral and multilateral trade openness reduce the probability of interstate war. This is in agreement with the theories that show how trade openness, in general and not only of virtual water, reduces the likelihood of interstate conflicts (Dorussen 2006, Hegre *et al* 2010). A concern that has been raised though, is that the de-escalation of the risk of interstate water wars produced by VWT could have other, neglected, yet important social implications. Dell'Angelo *et al* (2018b) discuss the notion of the 'neglected costs of water peace' pointing out that the hydropolitical understanding of VWT might be ignoring some hidden but important social consequences. They raise the hypothesis that 'as water is a limited resource—both in local and global terms—, when competition over water is resolved by fetching it from abroad, the social tensions that can consequently emerge or escalate, are shifted elsewhere rather than being dissolved' (Dell'Angelo *et al* 2018a). Their central message is that the social tensions, that are believed to be dissolved by virtual water imports are in reality transferred to the countries where water is appropriated. This is described by the authors as a specific typology of environmental cost-shifting that takes place in an increasingly telecoupled world, described as 'hidden socio-environmental costs of virtual water transfers'. It is clear then that VWT has strong societal influences, many that still need to be understood.

8.3. VWT and population growth

The study of human demographic growth in relation to the resources available on Earth has been at the center of important debates since Malthus developed his theory that human population grows faster than increase in resource availability, a condition that should eventually limit population growth (Malthus 1789). This theory has been subsequently criticized on the grounds that technological innovations have historically allowed humanity to tremendously increase food production (Boserup 1981) and there is no evidence that food availability has constrained population growth at the global scale (Sen 1981). Therefore, most demographic models do not even account for resource limitation as a determinant of fertility and mortality rates (Lee 2011). In recent years, however, the question of whether the planet has enough natural resources to feed its increasing population (Cohen 1995) has resurfaced (Godfray *et al* 2010, Foley *et al*, 2011, Warren 2015). Because crop production requires water, a finite resource, and contributes to roughly 85% of freshwater use by humanity, the same question about resource limitation has been explicitly reformulated in terms of water (Falkenmark and Rockström 2006, Suweis *et al* 2013). Specifically, there have been concerns as to whether the planet had enough water resources to meet the increasing needs of the growing and increasingly demanding human population (e.g. Smil 1994). This challenge, however, is not only about a near future. Many countries today are already in conditions of water deficit and need to import food because they consume more virtual water than their water balance is able to provide (Allan 1998, Hoekstra and Chapagain 2008). This means that trade has allowed their population to grow way beyond the limits imposed by the locally available water resources (Suweis *et al* 2013). In other words, part of the global demographic growth has been sustained by VWT and would not have been possible without an increasing reliance on food imports by water-scarce regions such as North Africa and the Middle East (*sensu*, Allan 1998). It is unclear, however, to what extent trade patterns have historically been shaped by demographic dynamics or, vice versa, population growth affected by trade.

In recent years a number of studies have combined projections of population growth with predictions of water availability and agricultural productivity under a variety of climate change and land use scenarios (Rosegrant *et al* 2001, Foley *et al* 2011). These predictions have been used to assess whether mankind will run out of water in the next few decades, and to investigate possible strategies to deal with the global food-water-energy nexus (Hoekstra and Wiedmann 2014). These studies have highlighted how effective management strategies and policies can account for both global and local water resources. Suweis *et al* (2013) expressed the country-scale carrying capacity as a function of both local and virtual water resources on the basis of water footprint and trade calculations. Using these carrying capacities in country-specific logistic growth models fitted to population records, they highlighted the existence of a global water unbalance. In fact, they found that the long-term demographic growth of net exporter countries relies on local water resources, while in trade-dependent countries it relies also on virtual water imports. Thus, both water-rich and water-scarce populations are counting in the long run on the same pool of water resources (Suweis *et al* 2013). Therefore, there are some concerns that exporter countries might at some point reduce their exports as happened during recent food crises (e.g. Fader *et al* 2013). Moreover, exporters might have to reduce their exports if new policies impose a more sustainable use of water resources that prevents the depletion of groundwater stocks or environmental flows. Thus, while trade and globalization are crucial to increase the carrying capacity of water-scarce countries and improve their present food security, they also induce a dangerous loss in long-term resilience (see section 8.5) of the coupled water-food system which may, in the long run, lead to social unrest (D'Odorico *et al* 2010a, Orlowsky *et al* 2014, Puma *et al* 2015).

8.4. Water pollution and other environmental externalities of VWT

The environmental impacts of trade have been at the center of decades of research on trade policies (e.g. Zaelke *et al* 1993). One of the corollaries of the theory of comparative advantage—i.e. that in a free trade scenario every country specializes in the goods it can produce most efficiently—is that production is expected to shift to regions of the world in which socio-environmental regulations are loose, absent or poorly enforced (e.g. Daly 1993, Wathen 1993). Even though free trade does not necessarily require environmental deregulation, its combination with low socio-environmental standards (e.g. poor regulations on pollution or labor rights) may have detrimental effects on local environmental conditions because firms can relocate to countries where there are lower standards. Alternatively, companies could be outcompeted by those who are already operating under weaker environmental policies with consequently lower production costs. Therefore, there have been calls for the inclusion of environmental and worker protection standards in international trade agreements (e.g. Bailey 1993, Charnovitz 1993). The General Agreement of Trade and Tariffs (see section 3), did not adopt environmental regulations but recognized the right of countries to ban imports of goods made with prison labor (e.g. Charnovitz 1993). The same notion could be extended to environmental standards through a process of 'environmental harmonization' of trade policy with the adoption of similar product and production standards by different countries (Charnovitz 1993).

In the case of mining, manufacturing, or other industrial productions the avoidance of strict environmental laws often coincides with a shift of production to regions of the developing world, where it can occur at a lower cost because of unaccounted environmental externalities. The associated costs are often borne by the entire society or future generations, while profits remain with the corporations that invest in these systems of production and export (Ward 1993). This outcome is in agreement with the theory of 'ecological unequal exchange', whereby core industrialized countries disproportionately use natural resources of less developed countries and force them to sustain negative environmental costs (e.g. Rice 2007, Moran *et al* 2013, 2015, Dorninger and Hornborg 2015).

The case of agricultural commodities, however, is different because they are not necessarily produced in the developing world for export to more developed countries (figure 10). Rather, these commodities—which have a bigger water footprint than their industrial counterparts and therefore are major contributors to VWT—are often exported by developed countries, such as the US that have historically dominated the global production and trade of agricultural products. In the US, agricultural exports contribute up to 6.9×10^9 m^3 yr^{-1} of groundwater depletion. As noted in section 3, the negative foreign impacts of the US export policy have been more of an economic nature (through their impact on agricultural development) than environmental.

The main environmental costs of agricultural production are associated with soil and water pollution from pesticide applications and fertilizer overuse, as well as groundwater depletion, land use change, habitat destruction, and soil erosion (Montgomery 2007, Meyfroidt *et al* 2013). These environmental effects are often difficult to relate to VWT, except for the case of pollution because its impact can be expressed in terms of the grey water footprint, the amount of water that is needed to reduce the pollutant concentration within acceptable environmental standards. In section 5.5 we have already highlighted some of the limitations of the grey water framework (particularly in the case of multiple pollutants). Its applicability requires the homogenization of the environmental standards among countries operating under the same trade agreement.

A global assessment of grey water trade has highlighted patterns of externalization of agricultural pollution from net importing countries (O'Bannon *et al* 2014). Interestingly, this research found that agricultural exports from the US are virtually associated with a substantial importation of pollution and other environmental costs that remain in the production country (section 5.5). The more developed countries have promoted agricultural policies that have favored intensified models of production to enhance crop yields at the expenses of habitat and soil loss and environmental pollution (Ward 1993). In the US export subsidies have promoted surplus production and exports of agricultural commodities and other land-based resources (e.g. water and topsoil) to the benefit of agribusiness corporations while the remediation costs are or will be borne by the entire society (see section 3).

There are also other environmental externalities associated with water overuse. For instance. importing goods irrigated from overexploited water sources (e.g. lakes and aquifers), may have detrimental impacts on the aquatic habitats and water sustainability. Recent research has investigated the extent to which the closure of the yield gap of major crops collides with environmental health because it threatens environmental flows (Soligno *et al* 2017, Rosa *et al* 2018a). In these conditions the overexploitation of water resources may lead to increased pollution and irreversible losses of biodiversity (e.g. Postel and Richter 2003). Some studies have highlighted the existence of hotspots of water overuse, which are partly induced by trade exports (e.g. Dalin *et al* 2017).

Unlike fossil fuels as input to energy production, which can technically be replaced by solar or wind energy, there is no alternative to water as an essential input for agricultural production. Therefore, sustainable use of water resources could be attained by adapting both supply and demand of water consumption in agriculture to acceptable rates. For example, crop production could be optimally distributed across the planet to maximize efficiency of land and water use (e.g. Davis *et al* 2017), while national policies favoring food self-sufficiency in regions without adequate renewable water resources may need to be abandoned and replaced to enable food imports via multilateral trade agreements.

8.5. VWT and resilience in the global food system

As noted in the previous sections, the global food system strongly relies on international trade because there is a mismatch between the rates of food production and consumption in different regions of the world, which explains the existence of areas with surplus and deficits in food availability (Fader *et al* 2013, D'Odorico *et al* 2014). Because of the non-uniform distribution of resources (e.g. land, water, and energy) and population density only 15% of the world's countries are fully self-sufficient while the others rely on imports of agricultural goods (Puma *et al* 2015). Food imports allow countries to overcome resource limitations, compensate for temporary reductions of food supply, and partly adapt to changes in productivity induced by climate change (Huang *et al* 2011). International trade, however, exposes countries to possible shocks in food supply in response to production crises occurring in other regions of the world. In fact, countries tend to decrease their exports during crises, thereby decreasing the overall amount of food (and virtual water) available for trade (Puma *et al* 2015, Tamea *et al* 2016). The expansion and intensification of international trade, thus, raises some concerns about the vulnerability of the water-food system and its resilience to shocks.

While food production shocks are well studied, the response to them and the complex dynamics leading to larger-scale food crises are less understood (Jones and Hiller 2017). Network analysis tools have been applied to investigate the structure and dynamics of food trade. Scaling properties of food flow networks from the village to the global scale were found to have consistent statistical distributions, indicating that similar governing mechanisms may be driving the redistribution of food across spatial scales (Konar *et al* 2018). Other work concludes that the global food network is becoming more connected, but not necessarily less stable (Sartori and Schiavo 2015) and that shocks induce long-term structural changes leading to an evolution in the network's capability to absorb shocks (Fair *et al* 2017). At short time scales, the vulnerability, and resilience, of countries to external shocks has been analyzed using shock propagation models. These range from parsimonious ones, assessing only the direct effects of—partial or total—export reductions as in (Puma *et al* 2015), to more complex models mimicking the cascade effect of a crisis propagating in the trade network, for example applied to single commodity trade (Gephart *et al* 2016, Fair *et al* 2017, Distefano *et al* 2018), global food trade (Marchand *et al* 2016) or VWT (Tamea *et al* 2016). These models mimic the trade redistribution following a local crisis, i.e. a decrease in food supply, according to simple dynamics and a limited number of parameters. The number of network connections and the corresponding imported volumes can determine country vulnerability, which can be offset by the country adaptive capacity through governance, infrastructure, and socio-economic factors (Gephart *et al* 2016), as well as food reserves (Marchand *et al* 2016) or redundancies in the food production system (Fader *et al* 2016). The shock propagation model applied to VWT with data-based country-specific parameters reproduces well the propagation of observed shocks, such as the 2008–2009 crisis in Argentina (Tamea *et al* 2016). The analysis reveals that countries with the most abundant water resources have the strongest impact on international trade, while water-scarce, trade-dependent countries are among the most exposed to external crises. The analysis of shock propagation of single commodities highlights that least developed countries are likely to suffer more from import losses when they strongly depend on food imports (Distefano *et al* 2018).

Local food production and economic capacity (expressed as the ratio of low income levels and the cost of food) are the major factors determining the resilience of a country, defined as its ability to respond and adapt to food supply disruptions (Seekell *et al* 2017). Local agriculture also contributes to the accumulation of food reserves, which modify the short-term response to food supply shocks (Fader *et al* 2016, Marchand *et al* 2016). Food reserves, or stocks, have a key role in the dynamic balance between food demand

and food supply, with the former being quite rigid and the latter undergoing high variability (Laio *et al* 2016). During food crises, stocks buffer the temporary food shortage caused by a loss of local production or a decrease in imports, and limit the effects of food availability on the local population. Therefore, the spatial distribution of food stocks is as important as international trade in determining the impact of food supply shocks and must be taken into account when developing food crises propagation models (Headey 2011, Marchand *et al* 2016).

The dynamics of international food trade are tightly connected to the international economics of agricultural commodities and the dynamics of food prices. Trade shocks, together with other non-trade-related factors (e.g. crop failures due to droughts or pests), are likely to trigger food price spikes (Headey 2011), which in turn may cause food shortages (Bren d'Amour *et al* 2016) and socio-political instability, e.g. (Lagi *et al* 2011). Studies developed in the economics literature use global -or partial- equilibrium models to infer a system's response to alterations at longer time scales (to allow for the establishment of 'equilibrium conditions'), based on the behavior of rational individuals. An example is the Global Trade Analysis Project (GTAP) that provides a dataset and a modeling framework to simulate the global system of household behaviors, international trade and investments, in response to a change in policy, technology, population or endowments. Specifically for water resources, a GTAP-W global equilibrium model has been developed, considering (Calzadilla *et al* 2010) or not (Berrittella *et al* 2007) the separate role of irrigated and rainfed agriculture. This model allows for the assessment of the global effects of water-crisis or sustainable-water-use scenarios (Calzadilla *et al* 2010). Konar *et al* (2016b) applied the GTAP model in conjunction with a global hydrological model to investigate changes in trade under climate and policy scenarios. Konar *et al* (2016b) show that trade liberalization leads to more water savings under a changing climate.

Unlike global equilibrium models, partial equilibrium models focus on single sectors of the economy —which are described with greater accuracy—but do not consider the effects of perturbations outside the considered sector. For instance, the IMPACT model (Rosegrant *et al* 2002) mimics the link between food production and food demand. This model includes a hydrologic module with multiple water uses, and explicitly accounts for the availability of water and its role in food production. The IMPACT model was first applied to VWT to assess the water savings associated with agricultural trade (de Fraiture *et al* 2004). While these equilibrium models are suitable to predict trends in food prices, access to food and population dynamics over medium-to-long-term time scales, they may offer an incomplete picture about real crises and food shortages (Distefano *et al* 2018) when the dynamics of food supply, availability and related prices are extremely fast and not well reproduced by equilibrium conditions (Headey 2011, Lagi *et al* 2015). In these conditions, non-equilibrium approaches based on shock propagation and conservation of mass appear to provide a more realistic description of the food system's response to a crisis.

An alternative approach to investigate the long-term response of the system to perturbations uses a framework based on linear stability analysis of the coupled resource-population dynamics, with resources becoming available both through local production and global trade. The stability analysis uses mathematical tools developed by Lyapunov (e.g. Strogatz 2014) in nonlinear systems theory, to explain how their dynamics behave around an equilibrium state (or a local stationarity). Suweis *et al* (2015) applied this approach to the global food trade network coupled with a delayed logistic model for country-specific population dynamics. They found that globalization (increasing number of trade links) decreases the system's resilience and increase their fragility to perturbations (Suweis *et al* 2015, Porkka *et al* 2016, 2017).

8.6. Governing the invisible or invisible governance?

Virtual water has been described as 'economically invisible and politically silent' (Allan 2003). While virtual water it has been at the center of hydropolitical theorization, it received less attention in policy development. Studies on VWT often result in policy recommendations and point to the desired or undesired policy implications that emerge when opening the black box of water globalization (e.g. Hoekstra and Hung 2002). Virtual water strategies have also been presented as possible solutions for water-scarce countries in international water meetings such as the World Water Forum or as a way to influence consumption and production behavior through tools such as virtual water labeling (Mori 2003, Leach *et al* 2016). However, the awareness of the potential power of VWT to address issues of water scarcity and food security is difficult to translate in direct concrete policies and governance priorities. While the hydrological and environmental effects of VWT have a clear local biophysical manifestation, the governance of the phenomenon goes beyond the sphere of water management and enters other realms of political economy. Very relevant are the agricultural trade and transnational land investment dimensions. VWT is ultimately governed by the politics of agricultural trade and land investments which tends to playout with little or no consideration of hydrological conditions such as water stress or other important social dimensions such as those associated with food security and malnutrition (Dell'Angelo *et al* 2018b).

The framework of 'virtual water hegemony' (Sujamo *et al* 2012) developed using the approach of hydro-hegemony (Zeitoun and Warner 2006), is useful to understand that rather than a 'visible' governance of an 'invisible' socio-hydrological phenomenon, what may happen is the opposite. Some authors suggest the 'invisible hand' of neoliberal markets impacts the direction, magnitude and dynamics of virtual water flows thorough transnational investments in land and agricultural commodities. Powerful agribusiness actors may then compete and cooperate in hydro-hegemony dynamics of persuasion, co-optation, and compromise that can include coercive leverages or incentives at multiple political levels (Sujamo *et al* 2012). The contemporary global land rush (Rulli *et al* 2013, Dell'Angelo *et al* 2017) is a good example of how the politics of VWT can be concretely studied. In this context, the study of the global governance of land grabbing (Margulis *et al* 2013) provides deeper understanding of what ultimately impacts the governance of global water appropriation and water grabbing (Rulli *et al* 2013, Dell'Angelo *et al* 2018a). The main notion here is that in order to engage with the governance dynamics of VWT, we need to move our focus to transnational regulations such as industrial exports, trade, agribusiness and transnational land acquisitions.

Moreover, there can be some concerns with the 'virtual' aspect of VWT and the associated attempt to apply an abstract model to reality. By abstracting water from its material context, which includes traded commodities as well as the dynamics of human labor and environmental resources, virtual water ignores important functions of the human-water interaction (Barnes 2013). While virtual water studies point to problematic aspects of water globalization which might justify a call for a global water governance based on ethical and normative grounds (Hoekstra 2006) the concrete actionability of global governance approaches continues to raise several concerns (Gawel and Bernsen 2013). The validity of the VWT as a governance tool also could be considered in the context of neoliberal globalization where the dominant impositions of markets and profits may overshadow the need for stronger socio-environmental regulation.

9. Conclusions

Modern society has enabled the spatial and temporal dislocation of production and consumption. A community (e.g. a village, province or country) no longer has to consume only what it is able to produce but different locations specialize in commodities for which they have the comparative advantage, given the local resources and policies. VWT allows societies to feed people in areas where there are not enough water resources to produce sufficient food to feed everyone. The emergence of trade dependencies can be ascribed to a number of factors that are not necessarily related to water scarcity but include drivers of comparative advantage, trade policies, demographic dynamics, historical patterns of agricultural development and related legacies.

The notions of virtual water and VWT were developed more than 20 years ago (Allan 1998). In recent years these concepts have been investigated in the context of food and water security with an approach that has led not only to the quantification of traded virtual water volumes but also to the analysis of the topological properties of the virtual water network and how they have changed through time. Recent research has clarified to what extent the traded (virtual) water comes from rainfed or irrigated agriculture, or from surface water bodies and aquifers, thereby allowing for a better evaluation of the hydrologic implications of this phenomenon and its impacts on local and global water systems.

Through an analysis of the 'virtual water cycle' we have related physical water flows in the hydrologic cycle to virtual water flows, which allows for an evaluation of the magnitude of the VWT phenomenon. This integrative analysis completes a more comprehensive assessment of human impacts on and appropriation of the water cycle. Agriculture, consumes 2.4% of global runoff for irrigation and contributes to more than 10%–16% of the global evapotranspiration from terrestrial ecosystems; about *one fourth* of these water resources are virtually traded as water embodied in agricultural goods. Because the major physical water fluxes in the hydrologic cycle are changing at a much slower rate (e.g. as an effect of climate warming or land use change) than water consumption in agriculture or other uses, the share of water resources appropriated by human activities is expected to increase. Likewise, as international trade in commodities increases without changes in water productivities, the amount of water virtually (but not physically) transferred around the world will also increase. However, a changing climate and geo-politics will also impact this complex system.

What are the socio-environmental key aspects of the globalization of water resources? The literature has often highlighted the benefits of VWT as an approach to deal with local or regional water scarcity (either through trade or food aid) and feed populations living in water-stressed areas without engendering massive migrations or water wars. In this review we have critically discussed some of the socio-environmental impacts of an increasing reliance on VWT and associated dependency on water resources existing in other regions of the world.

A recent body of literature on the role played by trade on the resilience of the food system, has shown how the globalization of food and water through trade has increased the likelihood of global crises. Some authors have also re-examined the relationship existing between VWT and demographic growth, water inequalities, environmental externalities, and the societal and political implications of VWT, particularly with respect to conflict and food or energy security. Collectively, these results provide an integrated perspective on the phenomenon of the globalization of water.

This review has highlighted some major gaps in the analysis and understanding of global VWT. More specifically, (1) more work needs to be done to investigate VWT at sub-national scales, including both agricultural and industrial water uses. Therefore, there is the need to identify new data sources or proxies that can improve our understanding of the VWT and its hydrological consequences at sub-national scales; (2) likewise, previous studies have assessed the environmental consequences of VWT considering a 'well mixed' system of production within each country. This review has shown that to improve the analysis of the local environmental impacts of VWT there is a need to study intra-regional VWT and identify the exact location of production of exported commodities; (3) the analysis of the environmental impacts of VWT requires improved process-based tools for the estimate of grey water flows, based on mechanistic models of non-point source pollution from nitrates, phosphates, fungicides, pesticides and other chemicals; (4) while a relatively large body of literature has quantified the environmental impacts of exporting countries, the environmental effects of international trade on importing countries remain for most part unexplored (Sun *et al* 2018); (5) there are also more direct environmental impacts of VWT associated with the intensification of trade and the establishment of new and/or more frequently used shipping routes. These effects have only started to be evaluated (Stephenson *et al* 2018); (6) it is not clear to what extent VWT is contributing to the unsustainable use of water resources at the expense of environmental flows. Therefore, there is a need to evaluate the unsustainable fraction of crop production and the associated VWT; (7) with population growth and climate change exacerbating water scarcity in some regions of the world, it is not clear how VWT will evolve in the coming decades and whether it will be able to meet the growing demand for agricultural products (Chouchane *et al* 2018b); (8) the effect of VWT on water scarcity remains difficult to evaluate. In fact, adding net virtual water import to domestic production and subtracting water demand results in an overly simplistic approach that assumes that the demand (due to economic and population growth) does not depend on trade itself. Therefore, analyses based on integrated assessment modeling are likely needed to backcast past development with and without trade and evaluate the effect of VWT on water scarcity; (8) directly addressing a critical, yet 'invisible' phenomenon such as a VWT remains a governance challenge where research has to play a key role in informing policy decisions. There is a growing need for actionable research that translates knowledge on the VWT phenomenon into policies aiming at an environmentally more sustainable and socially more equitable water governance.

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