Environmental Research Letters

ACCEPTED MANUSCRIPT • OPEN ACCESS

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

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

Paolo D'Odorico[1], Joel Carr[2], Carole Dalin[3], Jampel Dell'Angelo[4,1], Megan Konar[5], Francesco Laio[6], Luca Ridolfi[6], Lorenzo Rosa[1], Samir Suweis[7], Stefania Tamea[6], Marta Tuninetti[6]

[1] Department of Environmental Science, Policy, and Management, University of California, Berkeley;

[2] Patuxent Wildlife Research Center, U.S. Geological Survey, Beltsville, MD,

[3] Institute for Sustainable Resources, University College, London, UK

[4] Department of Environmental Policy Analysis, Institute for Environmental Studies, Vrije Universiteit Amsterdam, NL

[5] Department of Civil and Environmental Engineering, University of Illinois at Urbana-Champaign

[6] Department of Environment, Land and Infrastructure Engineering, Politecnico di Torino, Turin, IT

[7] Dipartimento di Fisica "G. Galilei", Universita' di Padova, Padua, IT

Outline:

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 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 virtual water trade 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, Gleick and Palaniappan 2010, Gordon et al 2005, Oki and Kanae 2006, Rost et al 2008, Rockström et al 2009, 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 Rockstrom 2004, Allan and Castillo 2007, D'Odorico et al 2010a). 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 & Rockstrom 2004, Postel 2003, Rodell et al 2018, D'Odorico et al 2018). This fundamental and increasingly scarce resource (relative to increasing human demand) is crucial to agriculture, mining, energy production, manufacturing, and households (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 virtual water trade 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 virtual water trade; 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 or l/kg). For example, in the United States of America, the actual average water content of wheat is ~0.13 m^3/ton whereas the VWC is ~1961 m^3/ton.

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 or l/kg). 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 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 section5). Such an approach is usually named the "bottom-up" approach (e.g. Feng et al 2011).
2) 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).
3) 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., Arto et al 2016, Yang et al 2012). The product resolution, however, is often low as highlighted in Feng et al (2011) 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).
4) 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 FAOSTAT) 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) depends on both the virtual water footprint of crops from that location, $VWF_{c,i}$ (m^3/ton) 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 & 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 minute by 5 arc minute 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 MIRCA2000 (Portmann et al. 2010) to help ascertain rain fed 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 & 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 subnational trade. 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., subnational trade).

2. Virtual Water Trade: 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 2017). 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 $m^3 y^{-1}$ 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 transport 3 billion $m^3 y^{-1}$ (Cohen et al 2004) and the Great Man-Made River Project in Libya, which roughly transport 1.34 billion $m^3 y^{-1}$ (Sternberg 2016). There are about 155 inter-basin water transfer schemes in 26 countries around the world, with a total capacity of 490 billion $m^3 y^{-1}$ of which, 138 billion m^3y^{-1} are for water transfers in Canada alone, and 30 billion $m^3 y^{-1}$ in the rest of the Americas, 181 billion m^3y^{-1} in Asia, and 120 billion y^{-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 $m^3 y^{-1}$) − to supply water-stressed communities either on a regular basis or in periods of scarcity (Bevanger 1994, Lewis and Miller 1987). Drinking water can also be available as bottled water, which is increasingly transported over long distances around the world (Gleick and Coley 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 virtual water trade (D'Odorico et al 2014, Porkka et al 2017).

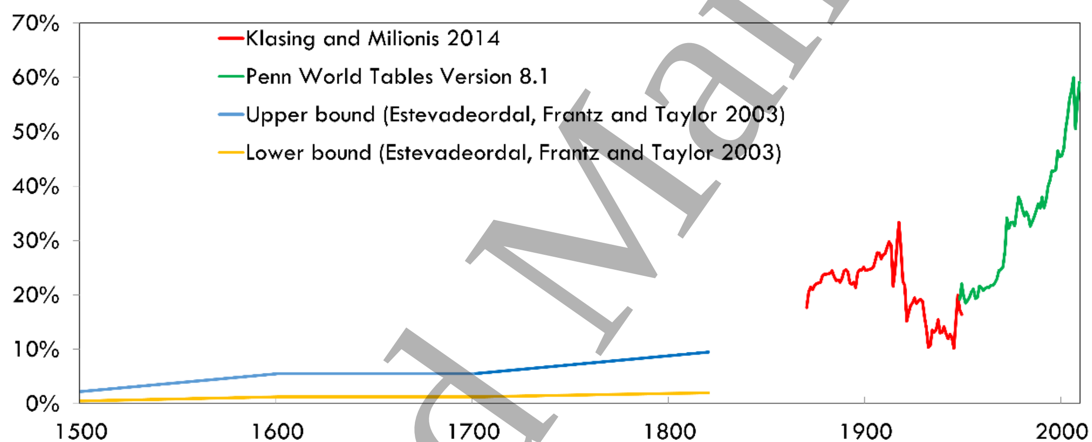

Figure 1. Globalization over 5 centuries (1500-2011 period). Shown as the world exports and imports expressed as a share of world GDP (%) (after Ortiz-Ospina and Roser 2018).

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 (Chapagain and Hoekstra 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 2010a, 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., Rosegrant et al 2009, Falkenmark and Rockström 2004, 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, D'Odorico et al 2017) and unconventional fossil fuels (Nicot and 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 2016, 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.

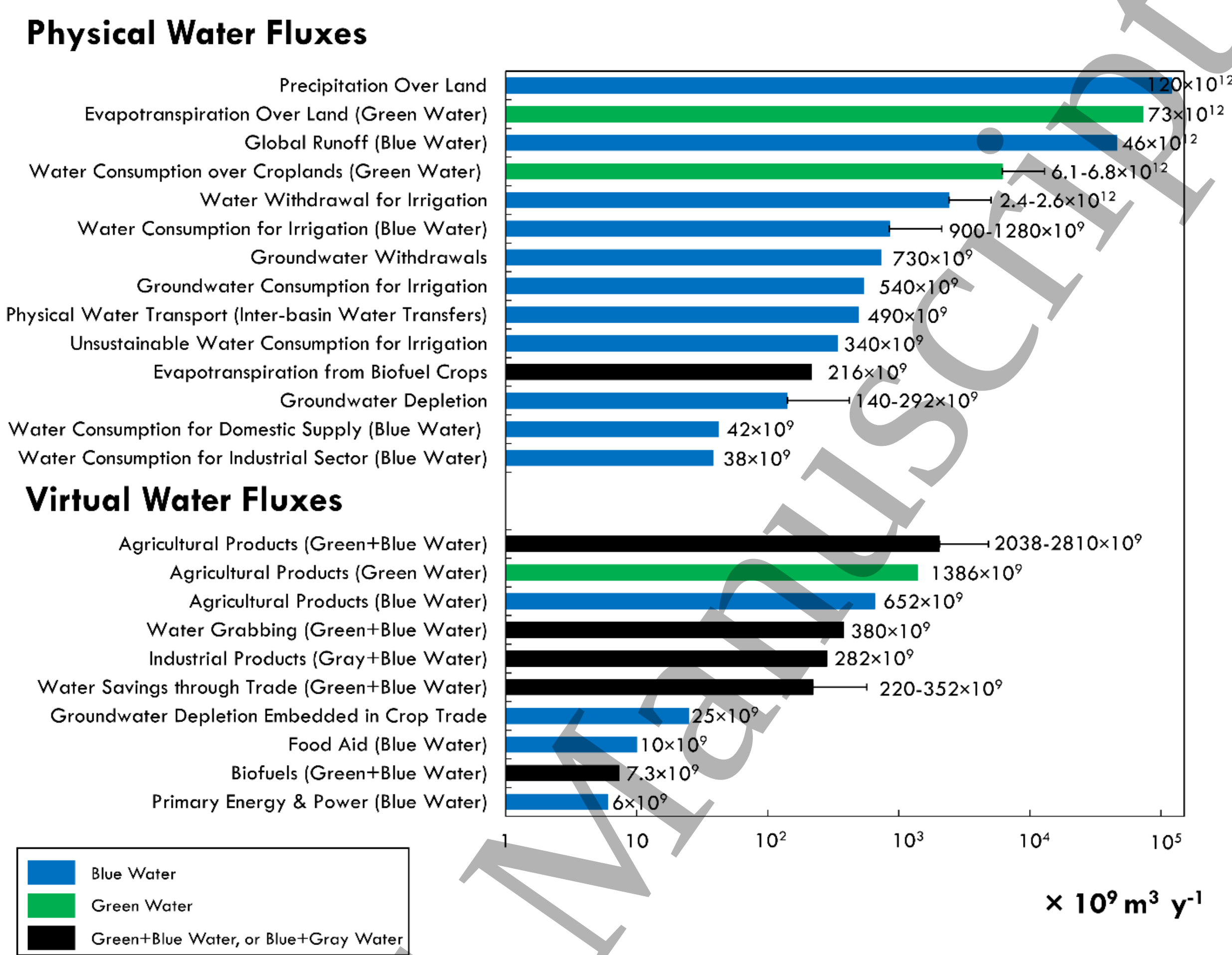

Figure 2. Physical and virtual water fluxes. Interval bars are computed by using different values from previous studies (see Tables 1 & 4). Note: The horizontal scale is logarithmic.

Table 1. Global virtual water flows.

Virtual Water Flows	Annual flow ($\times 10^9$ m^3 y^{-1})	Year	Source
Industrial Products (Blue+Gray Water)	282	1996-2005	(Hoekstra & Mekonnen 2012)
Agricultural Products (Green+Blue Water)	2038 (1386+652)	1996-2005	(Hoekstra & 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 2012)
	352	1997-2001	(Chapagain et al 2006)
Food aid	10	2005	(Jackson et al 2015)
Water Grabbing (appropriation through land investments)	380	2013	(Rulli & D'Odorico 2013)

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 virtual water trade 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 often misses a description of the phenomenon of virtual water trade 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 (GATT 1947), 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 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 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 in Japan for rice production and in 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 U.S. Therefore, the U.S. 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, 1993). 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 (Wang et al 2000, Nicholson 2000, Dai et al 2004) increased the cost of food 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 (FAOSTAT 2009). Overall, Brazil and other major agricultural countries such as Argentina and India, were then able to compete with the US for export markets.

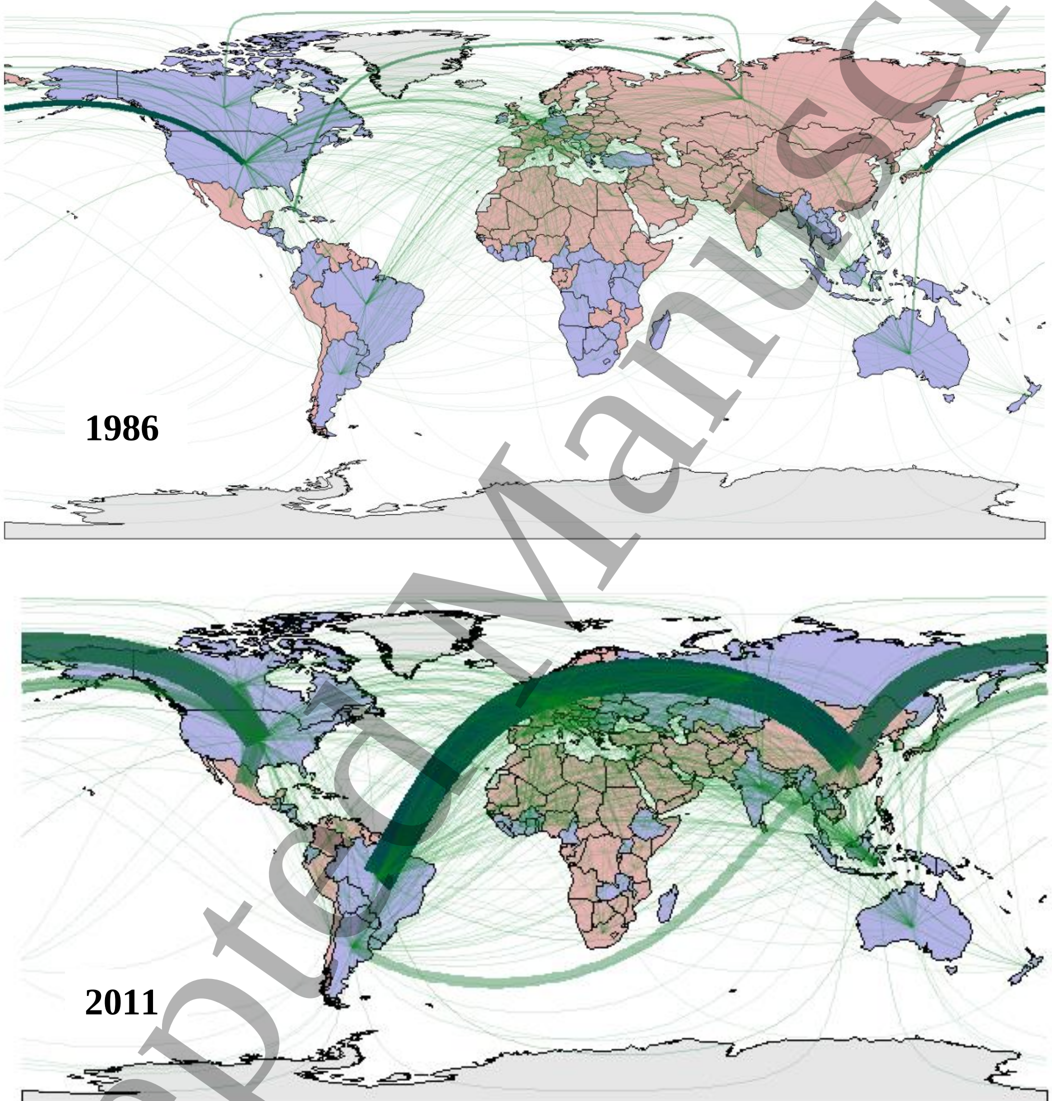

Figure 3. Global patterns of major virtual water trade in 1986 (top) and 2011 (bottom). In purple: net VW exporters. In pink: net VW importers.

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 2012). 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 2016), 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).

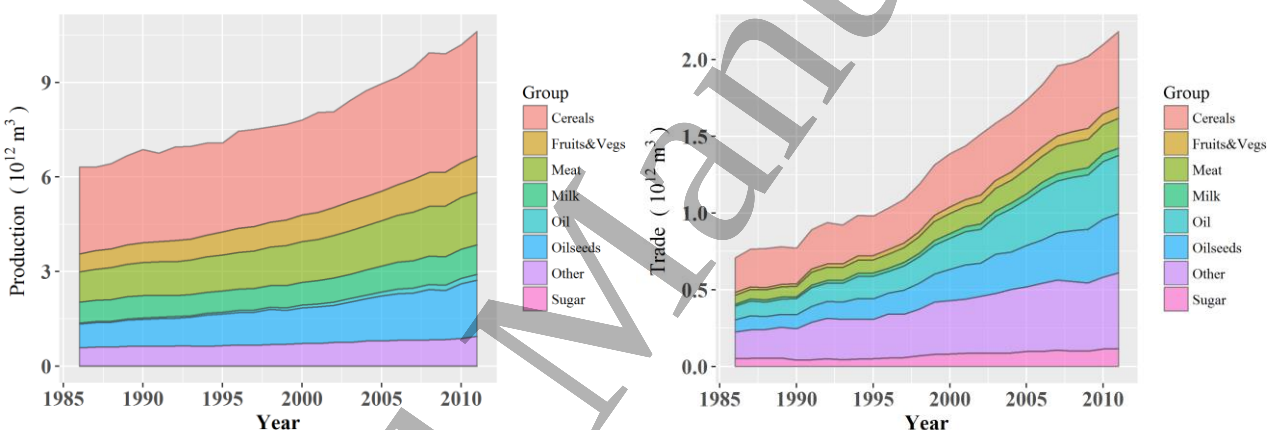

Figure 4. A) VW Production of 150 primary and B) subsequent VW Trade of 266 primary and secondary major food commodities removing feed and seed. (Analysis based on FAOSTAT data).

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, 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-Vasconez 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 by 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 to determine 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 14 potential bottlenecks (or chokepoints) worldwide (Table 2 & Figure 5). These chokepoints might be enhanced by intensifying meteorological events, underinvestment in infrastructure, increase in trade volumes, and conflicts.

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 has 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). United States and Ukraine are building new capacity and expanding grain terminals (Bailey and Wellesley 2017).

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%	-

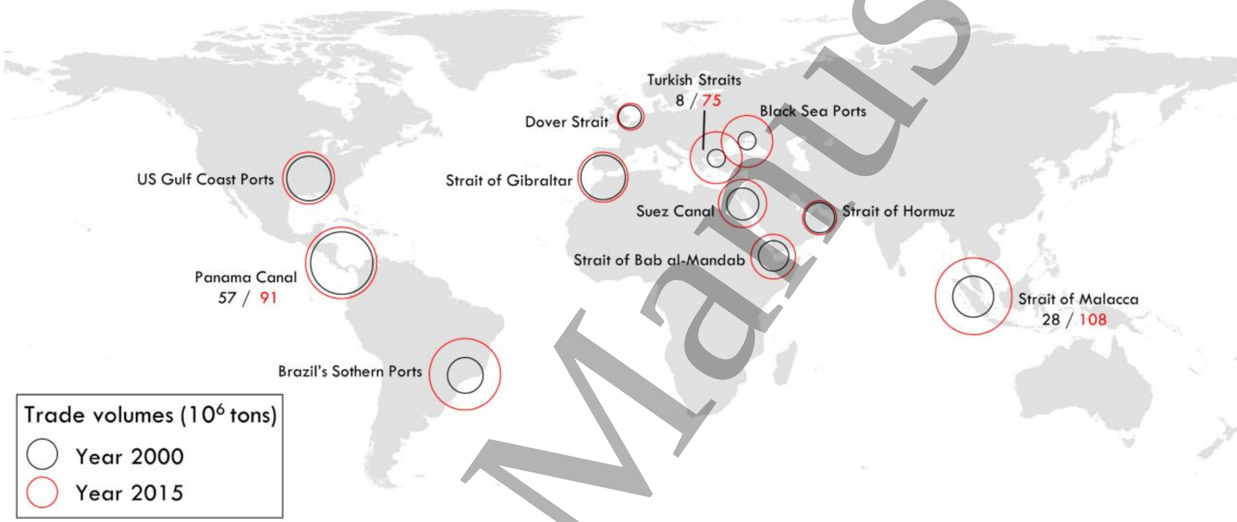

Figure 5. Major maritime chokepoints of agricultural commodity international trade. Redrawn from: Bailey and Wellesley (2017).

4. Patterns of Virtual Water Trade

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 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 virtual water trade 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 of the nation's participating within the network (Suweis et al 2011).

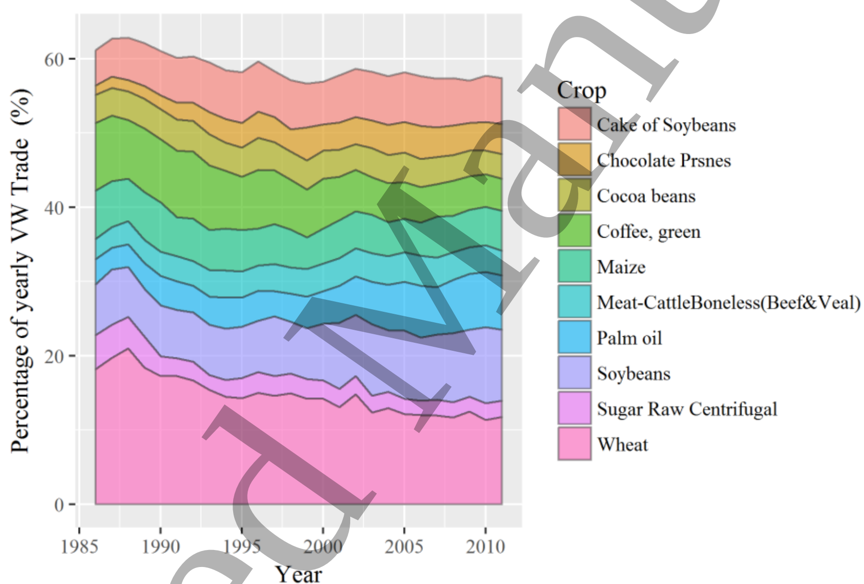

Figure 6. Relative contribution of different food commodities to virtual water trade. (Analysis based on FAOSTAT data).

The temporal reconstruction of the virtual water trade network (Carr et al 2012, Dalin et al 2012) has allowed for examination of changes in the geographic distribution of virtual water trade 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 virtual water trade 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 America, specifically Brazil and Argentina have decreased 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 exporting countries of animal and plant-based products.

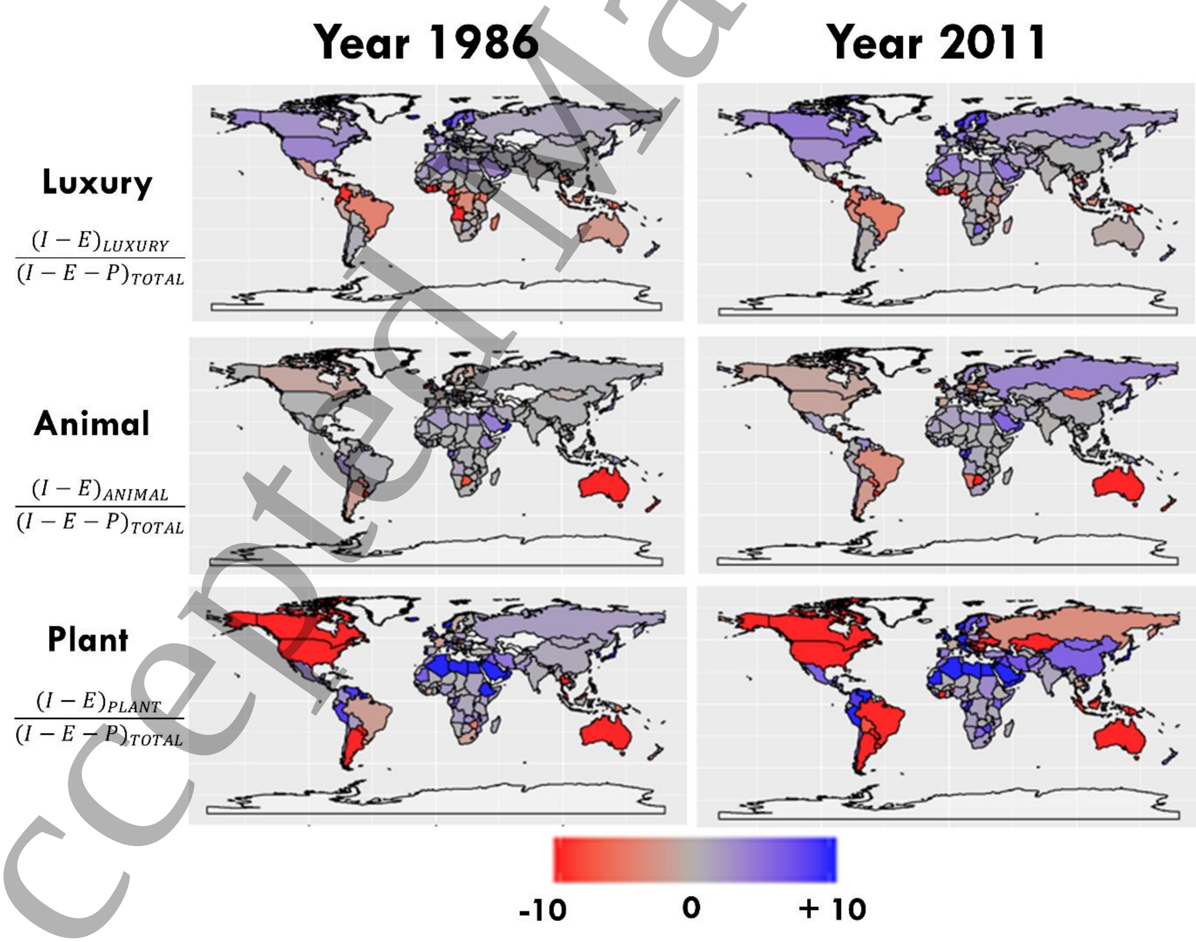

Figure 7. Major importers and exporters of luxury goods, animal products, and plant-based products in 1986 and 2011. Red countries are net exporters, blue countries are net importers. I, E, and P are imported, exported, and produced quantities of luxury, animal and plant-based commodities in each country,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

Carr et al (2012) investigated the global trade of 309 crop commodities using data from the Food and Agriculture Organization (FAO 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. (2012) 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 virtual water trade, 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 virtual water trade tends to reduce inequalities among countries in water use for food relative to wellbeing 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 virtual water trade. 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 virtual water trade (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 virtual water trade (see Box 3).

The literature on VWT began by tracking the water embodied in international trade. Early studies assumed that the virtual water content (VWC) of a product was constant within a country (Hoekstra and Hung 2002). This 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 was 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 combined with national trade data. Now, the current frontier is in further resolving trade flows in space.

To estimate crop water footprint, or virtual water content, VWC, most studies utilize a crop water model to calculate the consumptive water requirements (i.e., evapotranspiration, ET). CROPWAT (Allen, 1998) is a commonly used model (Mekonnen and Hoekstra, 2011, Tuninetti et al., 2015), though models such as the Global Crop Water Model (GCWM) (Siebert and Doll 2010, Hoff et al 2014), H08 (Hanasaki et al 2010, Dalin et al 2012), 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 km and ~50 km)(See Table 3; Tuninetti et al 2015).

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 Doll 2010	Global	5 arc min	1998-2002	5 arc min
Mekonnen & 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

Most studies on VW trade quantify international flows with commodity group resolution typically limited by the Harmonizing Commodity Description and Coding System (HS code) and the FAO food groups, since trade data is 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 2015, Hoekstra and Mekonnen 2016) or multi-regional input–

output (MRIO) models (Guan and Hubacek 2007, Dong et al 2014, Deng et al 2016, Serrano et al 2016, Zhang and Anadon 2014, 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 United States (Lin et al 2014, Dang et al 2015). Sub-national studies based upon modeled domestic transfers have also been developed (e.g. Brazil in Flach et al 2016; China in Dalin et al 2014).

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 United States, 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 modelling 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 United States (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.

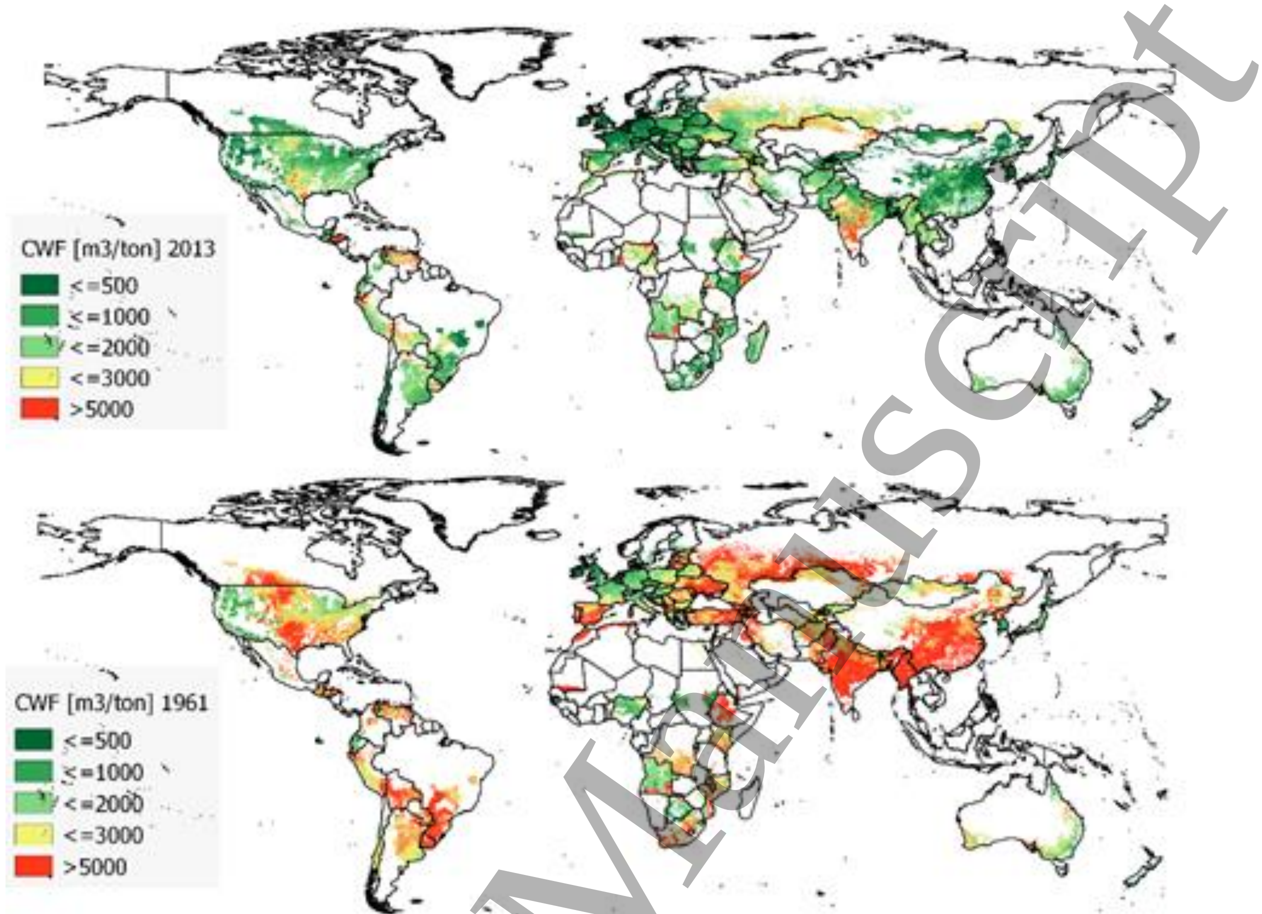

Figure 8. Comparison of the spatial distribution of the crop water footprint (CWF, also referred to as "virtual water content", VWC) of wheat in 1961 and 2013 based on the "fast-track" method from Tuninetti et al (2017b) using national yield data from years 1961 and 2013 (from FAOSTAT). The spatial distribution 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.

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 easily 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 et al. 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 2012). 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 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 virtual water trade estimates to the temporal variability in virtual water content of the main staple crops shows how, when the temporal variability of VWC is accounted for, the corresponding volumes of virtual water trade 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 2012, 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, Marston and Konar (2016) estimated annual VWT values. To do this, they calculated annual values of both trade and VWC. This study highlights the importance of time trends in both variables and provides a methodology for future time varying VWT studies to emulate.

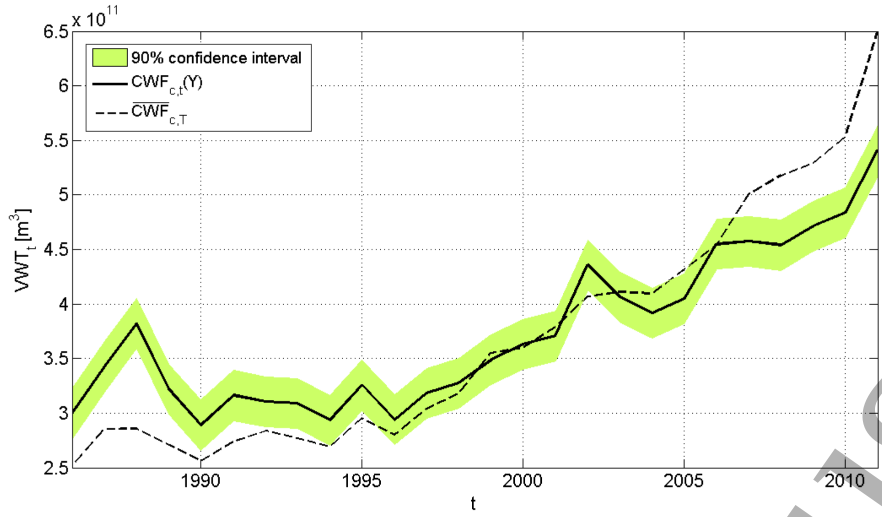

Figure 9. Virtual water trade of wheat, maize, rice, and soybean as in Tuninetti et al (2017b). Dashed line refers to VWT estimates based on a constant (i.e., time-averaged in 1996-2005) crop water footprint (or virtual water content). The solid line refers to estimates using time-varying crop water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

5.3 Commodity coverage

The VWT literature began by quantifying 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), natural rubber production (Chiarelli et al 2018) (Tab. 1). Virtual water flows have also been estimated for energy sources such as fossil fuels (Tab. 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 freshwater scarce 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 (Hoekstra and Mekonnen 2012, Mielke et al 2010, Gerbens-Leenes et al 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 investigate 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 Rockstrom 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 uses also 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, Zhuo et al 2016, Yano et al 2016, Rosa et al 2018a).

5.4.1 Green and blue virtual water trade

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,b, 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., ploughing, 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 by human activities globally, (Postel et al 1996, Falkenmark and Rockstrom 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 5 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 vs 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, Schyns et al 2015, Yano et al 2015, Marston 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 (Wada et al 2010, Siebert et al 2010, Gleeson et al 2012, Konikow and Kendy 2005, 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 United States (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 to 292 $km^3\ y^{-1}$), mainly in China (+102%) and the United States (+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 United States, and India alone. The trade of groundwater depletion by top crop exporters have greatly increased from 2000 to 2010 (100% increase in India, 70% in Pakistan and 57% in the United States), and the largest increase in the imports of groundwater depletion occurred in China (tripling), and were mainly associated with imports from the United States and India.

5.4.3 New vs 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 a 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 millions of years ago. Such a biomass contains energy from ancient photosynthesis and its growth 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 y^{-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 (D'Odorico et al 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 virtual water trade

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 blue and green water (O'Bannon et al 2014). This is because it 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) 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 greywater footprint associated with multiple pollutants (including other fertilizers and pesticides) would need to 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, by importing 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 and 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 baring 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 may have also negative environmental impacts in importing countries. For example, because if 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 2016). 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).

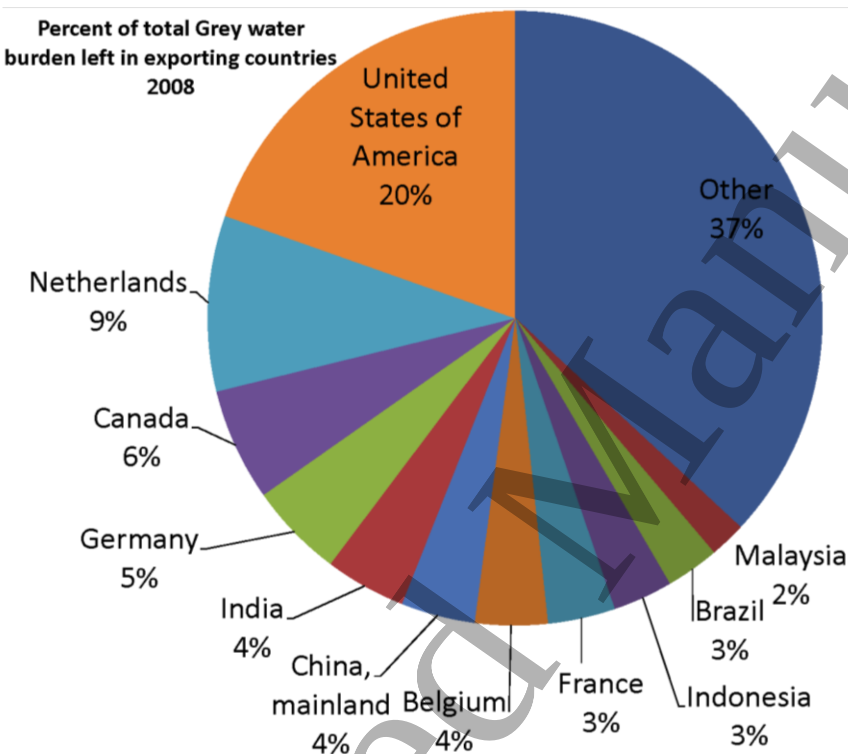

Figure 10. Share of total nitrogen pollution burden from agricultural trade on exporting countries in 2008 (based on analyses in O'Bannon et al 2014).

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

The previous sections have highlighted some important patterns and properties of virtual water trade. 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 (Table 4)? We define 'virtual water cycle' 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 inter-dependence 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, blue water consumption accounts for only part of water withdrawals from water bodies, with the remaining part being returned to water bodies by drainage and runoff processes. The blue water consumption (BWC) of humanity is dominated by water use in irrigation (0.85-1.28 $\times 10^{12}$ m^3 y^{-1}), which by far exceeds BWC by industrial production (0.038 $\times 10^{12}$ m^3 y^{-1}), and municipal uses (0.042 $\times 10^{12}$ m^3 y^{-1}). Collectively, these blue water uses account for 0.93-1.37 $\times 10^{12}$ m^3 y^{-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 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 y^{-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 y^{-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 y^{-1}) – i.e., sum of blue water consumption and crop uptake of root-zone soil moisture (or green water consumption, GWC $\approx$ 6.15 $\times 10^{12}$ m^3 y^{-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 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 × 10^{12} m^3 y^{-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 × 10^{12} m^3 y^{-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 × 10^{12} m^3 y^{-1} in the year 2000 and 2.04 × 10^{12} m^3 y^{-1} in 1996-2005 see Figure 5 & Table 1).

How have these figures been recently changing? Between 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 × 10^{12} m^3 y^{-1} in 2010, according to some estimates (see Figure 5 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 × 10^{12} m^3 y^{-1} in 2010 (see Figure 4 and D'Odorico et al 2018), which is again close to 20-24% 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, blue water consumption 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., 2018). 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.

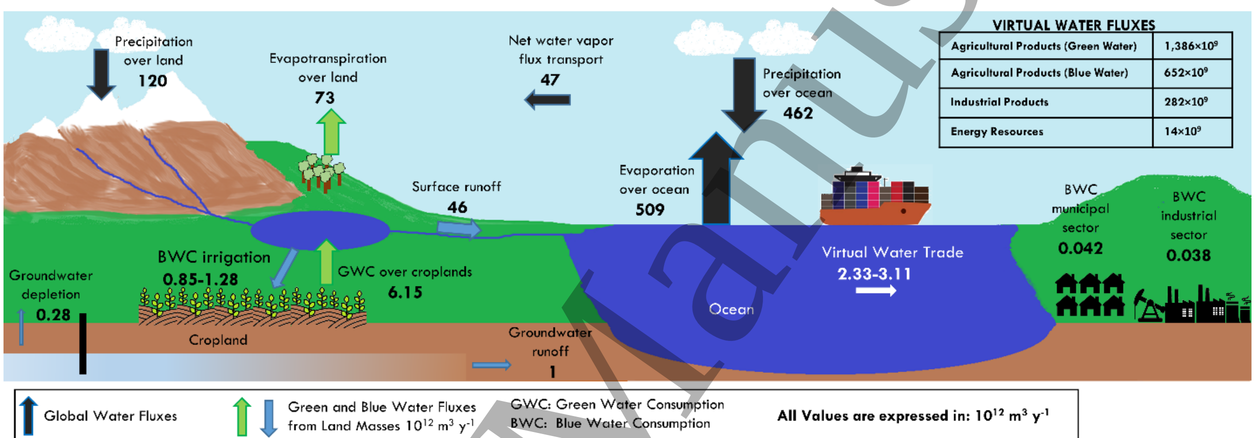

Figure 11. Comparison between physical and virtual water fluxes in integrative depiction of the global water cycle (based on data from Table 4).

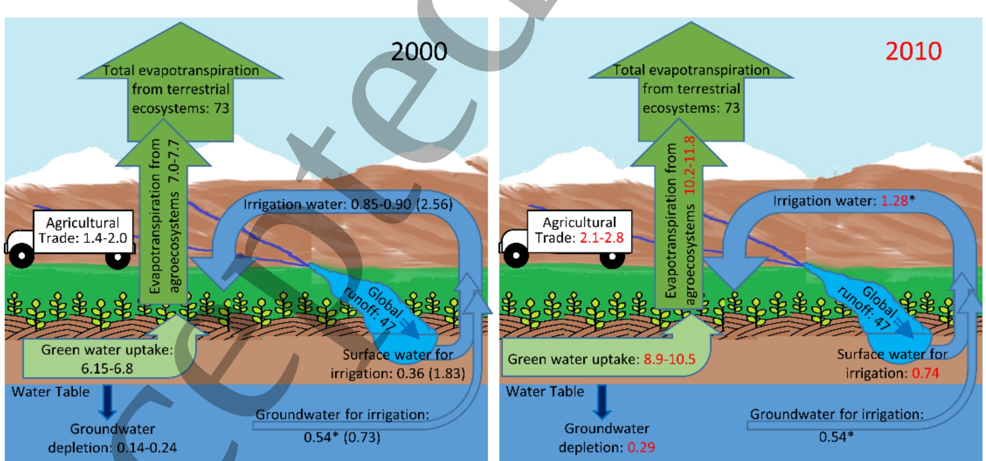

Figure 12. Global physical and virtual water fluxes in agriculture for the year 2000 (left) and 2010 (right) (source: Tables 1&4, Figure 4 and references cited in the text; all values are expressed in 10^{12} m^3 y^{-1}). Blue water flows are reported both as consumptive uses and blue water withdrawals (between

parentheses). The asterisk (*) denotes average values for 2000-2010. In red are values that have changed between 2000 and 2010.

Table. 4. Physical fluxes in the water cycle.

	Annual flow (m³ y⁻¹)	Year	Source
Precipitation over land **Evapotranspiration from land (Green water flows)** **Global runoff (Blue Water flows)**	120×10^{12} 73×10^{12} 47×10^{12}		(Chow, 1988)
Blue Water Withdrawal for irrigation	2.56×10^{12} 2.41×10^{12} 2.66×10^{12}	2000 1980-2009 2000	(Sacks et al., 2009) (Jägermeyr et al., 2017) (Oki and Kanae 2006)
Blue Water Consumption for irrigation	0.90×10^{12} 1.28×10^{12} 0.85×10^{12}	1996-2005 2000-2010 2000	(Hoekstra & Mekonnen, 2012) (Siebert et al., 2010) (Rosa et al 2018a)
Green Water consumption in croplands **For 16 major crops** **For 150 crops**	 6.15×10^{12} 6.79×10^{12}	 2000 2000	 (Rosa et al 2018a) (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 & Mekonnen, 2012)
Water consumption domestic supply	0.042×10^{12}	1996-2005	(Hoekstra & 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} 0.28×10^{12} 0.29×10^{12}	2001-2008 2000 2010	(Konikow et al., 2011) (Wada et al., 2010) (Dalin et al., 2017)

7. Models and drivers of VW trade

Modelling the virtual water trade 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 virtual water trade (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 2018). Suweis et al (2011) used country-specific values of gross domestic product (GDP) and average rainfall on agricultural areas to reproduce the undirected VW trade network (obtained by summing bilateral flows exchanged on the same link). Dalin et al (2012) 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. (2018) 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 & Singh (2005) identified the 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 & 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 (Fracasso 2014, Fracasso et al 2016, Lenzen et al 2013). 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, virtual water trade 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 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 2018, Abdelkader et al. 2018). Equilibrium models as well can be applied to assess different future scenarios.

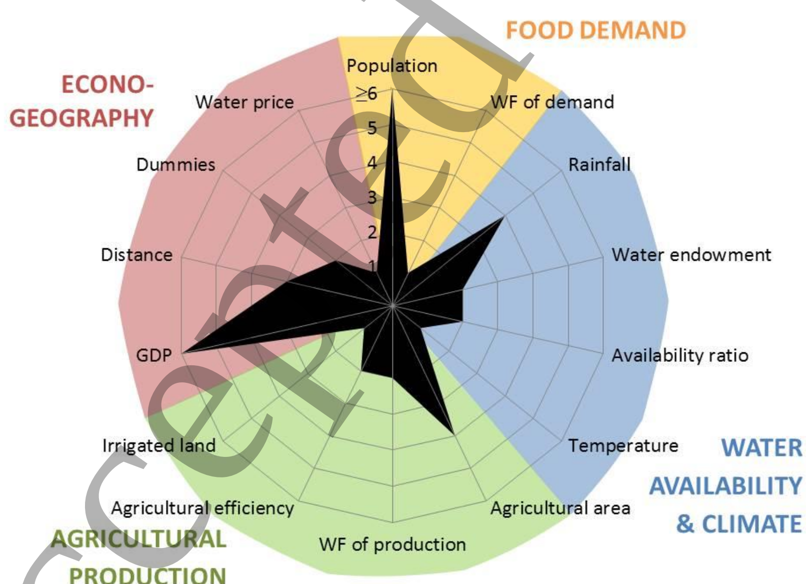

Figure 13. Drivers of VW trade identified in the publications reviewed in Section 7. The radial coordinate expresses the number of publications reporting a significant dependence on each variable. Details are provided in the Supplementary Information (Table S1).

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). Virtual water 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 Section 8.2 & 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 with in 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 might 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 determine the global patterns of production and trade (see Section 7).

On the other hand, even though virtual water trade 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 midcentury is expected to be an order of magnitude greater (e.g., Falkenmark and Rockstro, 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.

Virtual water trade 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 2010b, 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 virtual water trade.

8.1 Water savings

International trade can save national water resources by importing water-intensive commodities from other countries. National water saving 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-Mendelez and Bennett 2016, Brindha 2017). It has been estimated that virtual water trade saves 352 $km^3\ y^{-1}$ that would be otherwise consumed to produce agricultural products in the importing countries (Chapagain et al 2006) (Tab. 1). Other studies found smaller savings and reported the existence of a growing trend, from roughly 50 km^3/y in 1986 to 240 km^3/y in 2008 (Dalin et al 2012).
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, virtual water trade 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 commodities production if crops are grown where they are produced in less environmentally efficient and in more unsustainable ways (Martinez-Mendelez and Bennett 2016). Many countries produce commodities at the cost of additional pressures on their water resources. For example, agri-food products are sometime traded from an area with low water

productivity to and 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 et al 2013a).

Konar et al (2013b) 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 2013b). 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 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 virtual water trade

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 the ones 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., 2018b). 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 the ones 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 importat 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 at 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 virtual water trade 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 virtual water trade, found that bilateral and multilateral trade openness reduce the probability of interstate war. This is coherent 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 2010). A concern that has been raised though, is that the de-escalation of the risk of interstate water wars produced by virtual water trade could have other, neglected, yet important social implications. Dell'Angelo et al (2018b) discuss the notion of the 'neglected costs of water peace' pointing the attention to the issue that the hydropolitical understanding of virtual water trade 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 2018b). 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 and that they describe as "hidden socio-environmental costs of virtual water transfers". It is clear then that virtual water trade has strong societal influences, many that still need to be understood.

8.3 Virtual water trade 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 et al 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 Rockstrom, 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 be the locally available water resources (Suweis et al 2013). In other words, part of the global demographic growth has been sustained by virtual water trade and would have not been possible without an increasing reliance on food imports in 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, to be effective, water management strategies and policies need to 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 it 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 even lead in the long run to social unrest (D'Odorico et al 2010b, 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., Wathen 1993, Daly 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 they would need to comply to lower standards. Alternatively, they 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 (GATT, 1948 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 et al 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 et al 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 to 6.9×10^9 m^3 y^{-1} of groundwater depletion. As noted in Section 3, the negative foreign impacts of the US export policy have been more of 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 virtual water trade, 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, a 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 Virtual water trade 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 excess and deficit 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 of 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 2016). The expansion and intensification of international trade, thus, raises some concerns about the vulnerability of 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 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, Distefano et al 2018, Fair et al 2017), global food trade (Marchand et al 2016) or virtual water trade (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 are found to be determinant for 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 virtual water trade 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 largest 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 highlighted 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 of imports, and limit the effects on food availability to 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 (Marchand et al 2016, Headey 2011).

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, Rosengrant and IMPACT Development Team, 2012) 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 virtual water trade 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 non-linear 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 is "economically invisible and politically silent" (Allan, 2003), while 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 at the desired or undesired policy implications that emerge when opening the black box of water globalization (e.g., Kumand and Singh 2004, 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 (Kumand and Singh 2004) 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 virtual water trade 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. Virtual water trade 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 2018a).

The framework of "virtual water hegemony" (Sujamo et al 2012) developed on 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 happens is the opposite. It is the supposedly 'invisible hand' of neoliberal markets that impacts the direction, magnitude and dynamics of virtual water flows thorough transnational investments in land and agricultural commodities. Powerful agribusiness actors compete and cooperate in hydro-hegemony dynamics of persuasion, co-optation, and compromise that can include coercive leverages or incentives on multiple levels of politics (Sujamo et al 2012) with strong influence from powerful actors beyond the States (Selby 2007). The contemporary global land rush (Rulli et al 2013, Dell'Angelo et al 2017) is a good example of how the politics of virtual water trade can be concretely studied. In this context, the study of the global governance of land grabbing (*see* Special Issue in *Globalisations* (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 2018b). The main notion here is that in order to engage with the governance dynamics of virtual water trade, we need to move our focus on the politics of the different sectors that are associated and more visibly addressed by governance and 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 the 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 at 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, should also be considered in the context of neoliberal globalization where the dominant impositions of markets and profits puts in the shade many important needs 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. Virtual water trade 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 virtual water trade 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 virtual water trade 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 virtual water trade 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 virtual water trade 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 virtual water trade and demographic growth, water inequalities, environmental externalities, and the societal and political implications of virtual water trade, 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 Virtual Water Trade (VWT). More specifically, (1) more work needs to be done to investigate VWT at subnational 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 subnational 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 region 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 2018); (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 virtual water trade 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.

Acknowledgments

Francesco Laio, Luca Ridolfi, Stefania Tamea, Marta Tuninetti acknowledge funding support provided by the European Research Council (ERC) through the project "Coping with water scarcity in a globalized world" (ERC-2014-CoG, project 647473).

Lorenzo Rosa was supported by The Ermenegildo Zegna Founder's Scholarship.

References

Abdelkader, A., Elshorbagy, A., Tuninetti, M., Laio, F., Ridolfi, L., Fahmy, H., & Hoekstra, A. Y. (2018). National water, food, and trade modeling framework: The case of Egypt. *Science of the total environment*, *639*, 485-496.

Acreman, M., Arthington, A.H., Colloff, M.J., Couch, C., Crossman, N.D., Dyer, F., Overton, I., Pollino, C.A., Stewardson, M.J. and Young, W., 2014. Environmental flows for natural, hybrid, and novel riverine ecosystems in a changing world. *Frontiers in Ecology and the Environment*, *12*(8), pp.466-473.

Aldaya, M. M., Allan, J. A. and Hoekstra, A. Y. (2010) Strategic importance of green water in international crop trade, Ecological Economics, 69, 4, pp 887–894

Aldaya, M.M. and Llamas, M.R., 2008. *Water footprint analysis for the Guadiana river basin* (Vol. 3). Delft, The Netherlands: UNESCO-IHE.

Allan, J.A., (1993). Fortunately, there are substitutes for water otherwise our political futures would be impossible, in *ODA Priorities for Water Allocation and Management*, ODA, London.

Allan, J. A. (1996). Policy responses to the closure of water resources. *Water policy: allocation and management in practice. Chapman and Hill, London*

Allan, J. A. (1998). Virtual water: a strategic resource. *Ground water*, *36*(4), 545-547.

Allan, J. A. (2002). *The Middle East water question: Hydropolitics and the global economy* (Vol. 2). Ib Tauris.

Allan, J. A. (2003). Virtual water-the water, food, and trade nexus. Useful concept or misleading metaphor?. *Water international*, *28*(1), 106-113.

Allan, J.D., Castillo M.M. (2007), *Stream Ecology*, Springer, 2nd ed., Dordrecht, The Nederlands

Allen, R. et al., 1998. Crop evapotranspiration-Guidelines for computing crop water requirements-FAO Irrigation and drainage paper 56. *FAO*. Available at: http://www.academia.edu/download/40878584/Allen_FAO1998.pdf [Accessed December 6, 2017].

Ammerman AJ, Cavalli-Sforza LL. 1984. The Neolithic Transition, and the Genetics of Population in Europe. Princeton University Press.

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

Arto, I., Andreoni, V. & Rueda-Cantuche, J., 2016. Global use of water resources: A multiregional analysis of water use, water footprint and water trade balance. Water Resour. Econ., Volume 15, pp. 1-14.

Bailey, N.A., 1993. Foreign direct investment and protection in the third world. In: Zaelke, D., Orbuch P., and Housman, R.F. (Editors). *Trade and the Environment: Law, Economics, and Policy*, Island Press, Washington, D.C. pp. 133-143.

Bailey, R. and Wellesley, L. (2017), *Chokepoints and Vulnerabilities in Global Food Trade*, Chatham House Report, London: Royal Institute of International Affairs.

Barnes, J., 2013. Water, water everywhere but not a drop to drink: The false promise of virtual water. Critique of Anthropology, 33(4), pp.371-389.

Berrittella, M. et al., 2007. The economic impact of restricted water supply: a computable general equilibrium analysis. Water Res., Volume 42, pp. 1799-1813.

Bevanger, K. 1994. The North-South Carrier Water Project in Botswana. A review of environmental impact assessments. Trondheim: Norwegian Institute for Nature Research. ISBN 82-426-0531-9.

Bondeau, A. et al., 2007. Modelling the role of agriculture for the 20th century global terrestrial carbon balance. *Global Change Biology*, 13(3), pp.679–706. Available at: http://doi.wiley.com/10.1111/j.1365-2486.2006.01305.x [Accessed December 4, 2017].

Boserup, E. (1981). Population and technological change. Chicago, Ill: University of Chicago Press.

Brada, J. C. (1983). The Soviet-american grain agreement and the national interest. *American Journal of Agricultural Economics*, *65*(4), 651-656.

Brauman, K.A., Richter, B.D., Postel, S., Malsy, M. and Flörke, M., 2016. Water depletion: An improved metric for incorporating seasonal and dry-year water scarcity into water risk assessments. *Elem Sci Anth*, *4*.

Bren d'Amour, C. et al., 2016. Teleconnected food supply shocks. Environ. Res. Lett., Volume 11, p. 035007.

Brindha, K., 2017. International virtual water flows from agricultural and livestock products of India. *Journal of Cleaner Production*, *161*, pp.922-930.

Bruintjes, R.T., 1999. A review of cloud seeding experiments to enhance precipitation and some new prospects. *Bulletin of the American Meteorological Society*, *80*(5), pp.805-820.

Bruckner, M, G Fischer, S Tramberend, S Giljum (2015). Measuring telecouplings in the global land system: A review and comparative evaluation of land footprint accounting methods. *Ecological Economics*, 114, 11-21

Calzadilla, A., Rehdanz, K. & Tol, R., 2010. The economic impact of more sustainable water use in agriculture: A computable general equilibrium analysis. J. Hydrol., Volume 384, p. 292–305.

Carr, J. et al., 2012. On the temporal variability of the virtual water network. *Geophysical Research*

Letters, 39(6). Available at: http://doi.wiley.com/10.1029/2012GL051247.

Carr, J. et al., 2013. Recent History and Geography of Virtual water trade R. Huerta-Quintanilla, ed. *PLoS ONE*, 8(2), p.e55825. Available at: http://dx.plos.org/10.1371/journal.pone.0055825.

Carr, J.A., Seekell, D.A. & D'Odorico, P., 2015. Inequality or injustice in water use for food? *Environmental Research Letters*, 10(2), p.24013. Available at: http://stacks.iop.org/1748-9326/10/i=2/a=024013.

Carr, J. A., and D'Odorico, P. 2017. The water–food nexus and virtual water trade. In: S. Islam and K. Madani, Eds. Water Diplomacy in Action: Contingent Approaches to Managing Complex Water Problems, 95-109. Anthem Press, London.

Cereceda, P., R.S. Schemenauer and M. Suit. 1992. An alternative water supply for Chilean Coastal desert villages, *International Journal of Water Resources Development*, 8(1): 53-59.

Chapagain, A.K., Hoekstra, A.Y. and Savenije, H.H.G., 2006. Water saving through international trade of agricultural products. *Hydrology and Earth System Sciences Discussions*, *10*(3), pp.455-468.

Chapin, F. S., III, G. P. Kofinas, and C. Folke (2009), Principles of Ecosystem Stewardship: Resilience-Based Natural Resource Management in a Changing World, Springer, New York.

Charnovitz, S. 1993. Environmental harmonization and trade policy. In: Zaelke, D., Orbuch P., and Housman, R.F. (Editors). *Trade and the Environment: Law, Economics, and Policy*, Island Press, Washington, D.C. pp. 267-286.

Chiarelli, D. D., Rosa, L., Rulli, M. C., & D'Odorico, P. (2018). The water-land-food nexus of natural rubber production. *Journal of Cleaner Production*, 172, 1739-1747

Chini, C.M., Djehdian, L.A., Lubega, W.N. and Stillwell, A.S., 2018. Virtual water transfers of the US electric grid. *Nature Energy*

Chouchane, H., Krol, M.S. and Hoekstra, A.Y., 2018a. Virtual water trade patterns in relation to environmental and socioeconomic factors: A case study for Tunisia. *Science of the total environment*, *613*, pp.287-297.

Chouchane, H., Krol, M.S. and Hoekstra, A.Y., 2018b. Expected increase in staple crop imports in water-scarce countries in 2050. *Water Research X*, 1, 100001.

Chow, V. T., Maidment, D. R., & Mays, L. W. (1988). Applied hydrology, Series in Water Resources (572 pp.). McGraw-Hill Education, New York.

Clapp, J. (2016) "Trade and the Sustainability Challenge for Global Food Governance." *International Studies Association Annual Meetings in Atlanta, GA. March*. 2016.

Cohen, J. E. (1995), *How Many People Can the Earth Support?*, Norton, New York, 532 pp.

Cohen, R., Nelson, B.,Wolff, G. (2004) *Energy down the drain: The hidden costs of California's water supply*, Natural Resources Defense Council, New York, USA.

Cohen, A. and Ray, I., 2018. The global risks of increasing reliance on bottled water. *Nature Sustainability*, *1*(7), p.327.

D'Odorico, P. et al., 2012. Spatial organization and drivers of the virtual water trade: a community-structure analysis. *Environmental Research Letters*, 7(3), p.34007. Available at: http://stacks.iop.org/1748-9326/7/i=3/a=034007?key=crossref.b4884d1450a583ab113743cbad6e777a [Accessed December 4, 2017].

Dai, A., Lamb, P.J., Trenberth, K.E., Hulme, M., Jones, P.D. and Xie, P., 2004. The recent Sahel drought is real. *International Journal of Climatology*, *24*(11), pp.1323-1331.

Dalin, C. et al., 2012. Evolution of the global virtual water trade network. PNAS, 109(16), p. 5989–5994.

Dalin, C. et al., 2012. Modeling past and future structure of the global virtual water trade network. Geophys. Res. Lett., Volume 39, p. L24402.

Dalin C., N. Hanasaki, H. Qiu, D. Mauzerall and I. Rodriguez-Iturbe (2014) Water resources transfers through Chinese interprovincial and foreign food trade Proceedings of the National Academy of Sciences, 111 (27) 9774-9779, 2014.

Dalin C., H. Qiu, N. Hanasaki, D. Mauzerall and I.Rodriguez-Iturbe (2015) Balancing water resource conservation and food security in China, Proceedings of the National Academy of Sciences, 112 (15) 4588-4593, 2015

Dalin, C., and D.Conway (2016), Water resources transfers through southern African food trade: water efficiency and climate signals, Environ. Res. Lett., 11 (1), 15,005, 652 doi:10.1088/1748-9326/11/1/015005.

Dalin C., Y. Wada, T. Kastner and M. Puma (2017) Groundwater depletion embedded in international food trade Nature, 543, 700-704.

Daly, H.E. 1993. Problems with free trade: Neoclassical and steady-state perspectives. In: Zaelke, D., Orbuch P., and Housman, R.F. (Editors). *Trade and the Environment: Law, Economics, and Policy*, Island Press, Washington, D.C. pp. 147-157.

Dang, Q., X.Lin, and M.Konar (2015), Agricultural virtual water flows within the United States, Water Resour. Res., 51 (2), 973-986, doi:10.1002/2014WR015919.

Dang, Q. et al., 2016. A theoretical model of water and trade. Adv. Water Resour., Volume 89, p. 32–41.

Dang, Q. and M. Konar (2018), Trade openness and domestic water use, Water Resources Research, Vol 54, Issue 1, pp. 4-18, doi: 10.1002/2017WR021102.

Davis, K. F. and D'Odorico, P. (2015). Livestock intensification and the influence of dietary change: a calorie-based assessment of competition for crop production. Science of The Total Environment, 538, 817-823. https://doi.org/10.1016/j.scitotenv.2015.08.126

Davis, K.F., M.C. Rulli, A. Seveso, and P. D'Odorico, 2017. "Increase in food production and reduction in water use through optimized crop distribution", Nature Geosciences, doi.org/10.1038/s41561-017-0004-5.

da Silva, V.D.P.R., de Oliveira, S.D., Hoekstra, A.Y., Dantas Neto, J., Campos, J.H.B., Braga, C.C., de Araújo, L.E., Aleixo, D.D.O., de Brito, J.I.B., de Souza, M.D. and de Holanda, R.M., 2016. Water footprint and virtual water trade of Brazil. *Water*, *8*(11), p.517.

De Angelis, E., Metulini, R., Bove, V. and Riccaboni, M., 2017. Virtual water trade and bilateral conflicts. *Advances in Water Resources*, *110*, pp.549-561.

Debaere, P., 2014. The global economics of water: is water a source of comparative advantage?. Am. Econ. J.-Appl. Econ., 6(2), pp. 32-48.

de Fraiture, C. et al., 2004. Does international cereal trade save water? The impact of virtual water trade on global water use. Comprehensive Assessment Research Report n.4, Colombo, Sri Lanka: International Water Management Institute (IWMI).

de Koning, A, M Bruckner, S Lutter, R Wood, K Stadler, A Tukker (2015), Effect of aggregation and disaggregation on embodied material use of products in input–output analysis. *Ecological Economics*, 116, 289-299.

Delgado, C. L., Rosegrant, M. W., Steinfeld, H., Ehui, S., and Courbois, C. (1999). The coming livestock revolution. Background paper n. 6, Department of Economic and Social Affairs, Commission of Sustainable Development, Eighth Session.

Dell'Angelo, J., Rulli, M. C., & D'Odorico, P. 2018a. The global water grabbing syndrome. *Ecological Economics*, *143*, 276-285.

Dell'Angelo, J., D'Odorico, P. and Rulli, M.C., 2018b. The neglected costs of water peace. *Wiley Interdisciplinary Reviews: Water*, *5*(6), p.e1316.

Dell'Angelo, J., D'Odorico, P., Rulli, M.C. and Marchand, P., 2017. The tragedy of the grabbed commons: coercion and dispossession in the global land rush. *World Development*, *92*, pp.1-12.

Deng, G., Ding, Y. and Ren, S., 2016. The study on the air pollutants embodied in goods for consumption and trade in China–Accounting and structural decomposition analysis. *Journal of Cleaner Production*, *135*, pp.332-341.

Deng, G., Ma, Y., Li, X. (2016), Regional water footprint evaluation and trend analysis of China-based on interregional input- output model, *Journal of Cleaner Production*, 112 (5), 4674-4682.

Dermody, B. J., van Beek, R. P. H., Meeks, E., Klein Goldewijk, K., Scheidel, W., van der Velde, Y., Bierkens, M. F. P., Wassen, M. J., and Dekker, S. C., 2014. A virtual water network of the

Roman world. *Hydrol. Earth Syst. Sci.*, 18, 5025-5040, https://doi.org/10.5194/hess-18-5025-2014.

Distefano, T. and Kelly, S. (2017) Are we in deep water? Water scarcity and its limits to economic growth. *Ecological Economics*, *142*, pp.130-147.

Distefano, T., Laio, F., Ridolfi, L., Schiavo, S. (2018), Shock transmission in the international food trade network. A data-driven analysis. *PLoS ONE*, in press.

D'Odorico P., F. Laio, L. Ridolfi, 2010a. Does globalization of water reduce societal resilience to drought?, *Geophys. Res. Lett.* 37, L13403, doi:10.1029/2010GL043167.

D'Odorico P., F. Laio, A. Porporato, L. Ridolfi, A. Rinaldo, and I. Rodriguez-Iturbe, 2010b. Ecohydrology of terrestrial ecosystems. *Bioscience*, 60(11): 898–907.

D'Odorico, P., Carr, J., Laio, F. & Ridolfi, L., 2012. Spatial organization and drivers of the virtual water trade: a community-structure analysis. Environ. Res. Lett., Volume 7, p. 034007.

D'Odorico, P. et al., 2014. Feeding humanity through global food trade. Earth's Future, Volume 2, pp. 458-469.

D'Odorico P. and M.C. Rulli, 2014 The land and its people, *Nature Geoscience*, 4: 324-325.

D'Odorico, P., Natyzak, J.L., Castner, E.A., Davis, K.F., Emery, K.A., Gephart, J.A., Leach, A.M., Pace, M.L. and Galloway, J.N., 2017. Ancient water supports today's energy needs. *Earth's Future*, *5*(5), pp.515-519.

D'Odorico P, Davis KF, Rosa L, Carr J, Chiarelli DD, Dell'Angelo J, Gephart J, MacDonald G, Seekel D, Suweis S, Rulli MC. 2018 The Global Food-Energy-Water Nexus, *Reviews of Geophysics, 56(3)*, pp. 456-531.

Dogan, E. and Stupar, A., 2017. The limits of growth: A case study of three mega-projects in Istanbul. *Cities*, *60*, pp.281-288.

Dong, H., Geng, Y., Fujita, T., Fujii, M., Hao, D. and Yu, X., 2014. Uncovering regional disparity of China's water footprint and inter-provincial virtual water flows. *Science of the Total Environment*, *500*, pp.120-130.

Dorninger, C, A. Hornborg, (2015). Can EEMRIO analyses establish the occurrence of ecologically unequal exchange? *Ecol. Econ.* 119, 414–418.

Dorussen, H. (2006). Heterogeneous trade interests and conflict: What you trade matters." *Journal of Conflict Resolution* 50.1: 87-107.

El-Gafy, I., 2014. System dynamic model for crop production, water footprint and virtual water nexus. Water Resour. Manage., Volume 28, pp. 4467-4490.

Erisman, J. W., Sutton, M. A., Galloway, J., Klimont, Z., and Winiwarter, W. (2008). How a century of ammonia synthesis changed the world. Nature Geoscience, 1(10), 636. https://doi.org/10.1038/ngeo325

Estevadeordal, A., Frantz, B. and Taylor, A.M., 2003. The rise and fall of world trade, 1870–1939. *The Quarterly Journal of Economics*, *118*(2), pp.359-407.

Fader, M. et al., 2013. Spatial decoupling of agricultural production and consumption: quantifying dependences of countries on food imports due to domestic land and water constraints. Environ. Res. Lett., Volume 8, p. 014046.

Fader, M., Rulli, M., Carr, J. & al., e., 2016. Past and present biophysical redundancy of countries as a buffer to changes in food supply. Environ. Res. Lett., Volume 11, p. 055008.

Fair, K., Bauch, C. & Anand, M., 2017. Dynamics of the global wheta traed network and resilience to shocks. Sci. Rep., Volume 7, p. 7177.

Falkenmark, M. and J. Rockström. 2004. *Balancing water for humans and nature: the new approach in ecohydrology*. Earthscan.

Falkenmark, M., & Rockström, J. (2006) The new blue and green water paradigm: Breaking new ground for water resources planning and management. *Journal of Water Resources Planning and Management* 132(3): 129– 32.

Feng, K., Y.L. Siu, D. Guan, and K. Hubacek (2012) Assessing regional virtual water flows and water footprints in the Yellow River Basin, China: A consumption based approach, Applied Geography, 32, 2, pp. 691-701, doi: 10.1016/j.apgeog.2011.08.004

Flach, R., Ran, Y., Godar, J., Karlberg, L. and Suavet, C., 2016. Towards more spatially explicit assessments of virtual water flows: linking local water use and scarcity to global demand of Brazilian farming commodities. *Environmental Research Letters*, *11*(7), p.075003.

Foley, J.A., et al., 2011. Solutions for a cultivated planet. *Nature* 478, 337–342. http:// dx.doi.org/10.1038/nature10452 .

Food and Agriculture Organization (2017). FAOSTAT database. Rome, Italy: Food and Agriculture Organization of the United Nations.

Fracasso, A., 2014. A gravity model of virtual water trade. Ecol. Econ., Volume 108, pp. 215-228.

Fracasso, A., Sartori, M. & Schiavo, S., 2016. Determinants of virtual water flows in the Mediterranean. Sci. Tot. Env., Volume 543, pp. 1054-1062.

Friedmann, H. (1993). The political economy of food: a global crisis. *New left review*, (197), 29.

Galli, A., Weinzettel J., Cranston, G., Ercin, E., 2013, A Footprint Family extended MRIO model to support Europe's transition to a One Planet Economy, Science of the Total Environment, 461, 813-818

Galloway, J. N., Burke, M., Bradford, G. E., Naylor, R., Falcon, W., Chapagain, A. K., et al. (2007). International trade in meat: The tip of the pork chop. Ambio: A Journal of the Human Environment, 36(8), 622–629.

Gawel, E. and Bernsen, K., 2013. What is wrong with virtual water trading? On the limitations of the virtual water concept. *Environment and Planning C: Government and Policy*, *31*(1), pp.168-181.

Gephart, J. et al., 2016. Vulnerability to shocks in the global seafood trade network. Environ. Res. Lett., Volume 11, p. 035008.

Gephart, J. et al., 2017 The `seafood gap' in the food-water nexus literature—issues surrounding freshwater use in seafood production chains, Adv. in Water Res. Volume 110 https://doi.org/10.1016/j.advwatres.2017.03.025.

Gerlach C. (2015) Famine responses in the world food crisis 1972–5 and the World Food Conference of 1974, European Review of History: Revue européenne d'histoire, 22:6, 929-939, DOI: 10.1080/13507486.2015.1048191

Gerten, D., Hoff, H., Rockström, J., Jägermeyr, J., Kummu, M. and Pastor, A.V., 2013. Towards a revised planetary boundary for consumptive freshwater use: role of environmental flow requirements. *Current Opinion in Environmental Sustainability*, *5*(6), pp.551-558.

Giljum, S, M Bruckner, A Martinez. (2015). Material footprint assessment in a global input-output framework. *Journal of Industrial Ecology*, 19 (5), 792-804.

Gleeson T, Wada Y, Bierkens M F and van Beek L P 2012 Water balance of global aquifers revealed by groundwater footprint *Nature 488*(7410) 197-200

Gleick, P.H. and Cooley, H.S., 2009. Energy implications of bottled water. *Environmental Research Letters*, *4*(1), p.014009.

Gleick, P.H. and Palaniappan, M., 2010. Peak water limits to freshwater withdrawal and use. *Proceedings of the National Academy of Sciences*, *107*(25), pp.11155-11162.

Globalisations 2013. Special Issue on ‘‘land grabbing and global governance”. *Globalisations*, 10(1).

Gordon, L.J., Steffen, W., Jönsson, B.F., Folke, C., Falkenmark, M. and Johannessen, Å., 2005. Human modification of global water vapor flows from the land surface. *Proceedings of the National Academy of Sciences of the United States of America*, *102*(21), pp.7612-7617.

Guan, D. and Hubacek, K., 2007. Assessment of regional trade and virtual water flows in China. *Ecological economics*, *61*(1), pp.159-170.

Gustavsson, J., Cederberg, C. & Sonesson, U. *Global food losses and food waste: extent, causes and prevention*. (Food and Agriculture Organization of the United Nations, 2011).

Hale, A., & Opondo, M. (2005). Humanising the cut flower chain: Confronting the realities of flower production for workers in Kenya. Antipode, 37(2), 301-323.

Hamilton, H. A. et al. 2018. Trade and the role of non-food commodities for global eutrophication. *Nat. Sustain.* 1, 314–321.

Hanasaki, N., T.Inuzuka, S.Kanae, and T.Oki (2010), An estimation of global virtual water flow and sources of water withdrawal for major crops and livestock products using a global hydrological model, J. Hydrol., 384 (3), 232-244, doi:10.1016/j.jhydrol.2009.09. 880 028. 881

Hanasaki, N., Kanae, S., Oki, T., Masuda, K., Motoya, K., Shirakawa, N., & Tanaka, K. (2008). An integrated model for the assessment of global water resources–Part 1: Model description and input meteorological forcing. *Hydrology and Earth System Sciences*, *12*(4), 1007-1025.

Hassan, A., Saari, M.Y. and Ismail, T.T., 2017. Virtual water trade in industrial products: evidence from Malaysia. *Environment, Development and Sustainability*, *19*(3), pp.877-894.

Headey, D., 2011. Rethinking the global food crisis: The role of trade shocks. Food Policy, Volume 36, pp. 136-146.

Hegre, Ha., J. R. Oneal, and B. Russett. (2010). Trade does promote peace: New simultaneous estimates of the reciprocal effects of trade and conflict. Journal of Peace Research 47.6: 763-774

Heilmann, S., Rudolf, M., Huotari, M. and Buckow, J., 2014. China's shadow foreign policy: parallel structures challenge the established international order. *China Monitor*, *18*(Oct), pp.1-9.

Hoekstra A Y, 2006, *The global dimension of water governance: nine reasons for global arrangements in order to cope with local water problems*, Value of Water Research Report

Series 20, UNESCO–IHE Institute for Water Education, Delft, http://doc.utwente.nl/58371/1/Report_20.pdfHoekstra, A. Y., & Mekonnen, M. M. (2012). The water footprint of humanity. Proceedings of the National Academy of Sciences, 109(9), 3232–3237. doi:10.1073/pnas.1109936109

Hoekstra, A.Y. & Chapagain, A.K., 2008. *Globalization of water : sharing the planet's freshwater resources*, Blackwell Pub.

Hoekstra, A.Y. and A. Chapagain (2007) Water footprint of nations: Water use by people as a function of their consumption patterns, *Water Resour Manage*, 21:35-48, doi: 10.1007/s11269-006-9039-x

Hoekstra, A.Y. and Chapagain, A.K. (2008) Globalization of water: Sharing the planet's freshwater resources, Blackwell Publishing, Oxford, UK.

Hoekstra, A.Y. and Hung, P.Q., 2005. Globalisation of water resources: international virtual water flows in relation to crop trade. *Global environmental change*, *15*(1), pp.45-56.

Hoekstra, A.Y. and M.M. Mekonnen (2016) Imported water risk: the case of the UK, Environmental Research Letters, Vol 11, No 5.

Hoekstra, A.Y. and P.Q. Hung (2002) Virtual water trade: A quantification of virtual water flows between nation in relation to international crop trade, Value of water research report series No. 11.

Hoekstra, A.Y., Villholth, K.G., López-Gunn, E., Conti, K. and Garrido, A., 2018. Global food and trade dimensions of groundwater governance. In *Advances in Groundwater Governance* (pp. 353-366). CRC Press-Taylor & Francis group.

Hoekstra, A. Y., and T.O. Wiedmann. 2014. Humanity's unsustainable environmental footprint, Science 344(6188): 1114-1117.

Hoff, H., Döll, P., Fader, M., Gerten, D., Hauser, S., & Siebert, S. (2014). Water footprints of cities– indicators for sustainable consumption and production. Hydrology and Earth System Sciences, 18(1), 213-226.

Holland, R.A., Scott, K.A., Flörke, M., Brown, G., Ewers, R.M., Farmer, E., Kapos, V., Muggeridge, A., Scharlemann, J.P., Taylor, G. and Barrett, J., 2015. Global impacts of energy demand on the freshwater resources of nations. *Proceedings of the National Academy of Sciences*, *112*(48), pp.E6707-E6716.

Huang, H., von Lampe, M. & van Tongeren, F., 2011. Climate change and trade in agriculture. Food Policy, Volume 36, p. S9–S13.

IAASTD. 2009. Agriculture at a crossroads: the global report of the International Assessment of Agricultural Knowledge, Science, and Technology. Island Press: Washington, DC. ISBN 978-1-59726-539-3.

Ingram, J., 2011. A food systems approach to researching food security and its interactions with global environmental change. *Food Security*, 3(4), pp.417–431. Available at: http://link.springer.com/10.1007/s12571-011-0149-9 [Accessed December 11, 2017].

Jackson, N., Konar, M. & Hoekstra, A., 2015. The Water Footprint of Food Aid. *Sustainability*, 7(6), pp.6435–6456. Available at: http://www.mdpi.com/2071-1050/7/6/6435/

Jägermeyr, J., Pastor, A., Biemans, H., & Gerten, D. (2017). Reconciling irrigated food production with environmental flows for Sustainable Development Goals implementation. Nature Communications, 8, 15900. https://doi.org/10.1038/ncomms15900

Jia, S., Long, Q. and Liu, W., 2017. The fallacious strategy of virtual water trade. *International Journal of Water Resources Development*, *33*(2), pp.340-347.

Jones, A. & Hiller, B., 2017. Exploring the dynamics of responses to food production shocks. Sustainability, 9 (960)(doi:10.3390/su9060960).

Kaseke, K.F. and Wang, L., 2018. Fog and dew as potable water resources–maximizing harvesting potential and water quality concerns. *GeoHealth*, *2*(10), pp. 327-332.

Klasing, M.J. and Milionis, P., 2014. Quantifying the evolution of world trade, 1870–1949. *Journal of International Economics*, *92*(1), pp.185-197.

Konar, M. et al., (2011). Water for food: The global virtual water trade network. *Water Resources Research*, 47(5). Available at: http://doi.wiley.com/10.1029/2010WR010307 [Accessed December 13, 2017].

Konar M, Dalin C, Hanasaki N, Rinaldo A and Rodriguez-Iturbe I (2012) Temporal dynamics of blue and green virtual water trade networks Water Resour. Res. 48W07509

Konar, M. & Caylor, K., (2013a). Virtual water trade and development in Africa. Hydrol. Earth Syst. Sci., Volume 17, p. 3969–3982.

Konar, M., Z. Hussein, N. Hanasaki, D.L. Mauzerall, and I. Rodriguez-Iturbe (2013b), Virtual water trade flows and savings under climate change, Hydrology and Earth System Sciences, Vol 17, pp. 3219-3234, doi:10.5194/hess-17-3219-2013.

Konar, M., T.P. Evans, M. Levy, C.A. Scott, T.J. Troy, C.J. Vorosmarty, and M. Sivapalan (2016a), Water resources sustainability in a globalizing world: who uses the water? Hydrological Processes, Vol 30, Issue 18, pp. 3330-3336, doi: 10.1002/hyp.10843.

Konar, M., Reimer, J. J., Hussein, Z. & Hanasaki, N., (2016b). The water footprint of staple crop trade under climate change and policy scenarios. Environ. Res. Lett., Volume 11, p. 035006.

Konar, M., X. Lin, B. Ruddell, and M. Sivapalan (2018), Scaling properties of food flow networks, PLOS ONE, Vol 13, Issue 7, pp. e0199498, doi: 10.1371/journal.pone.0199498

Konikow L F and Kendy E 2005 Groundwater depletion: A global problem *Hydrogeology Journal 13*(1) 317-320

Konikow, L. F. (2011). Contribution of global groundwater depletion since 1900 to sea-level rise. Geophysical Research Letters, 38, L17401. https://doi.org/10.1029/2011GL048604

Kumar, M. & Singh, O., 2005. Virtual water in global food and water policy making: is there a need for rethinking?. Water Resour. Manage., Volume 19, pp. 759-782.

Lagi, M., Bar-Yam, Y., Bertrand, K. & Bar-Yam, Y., 2015. Accurate market price formation model with both supply-demand and trend-following for global food prices providing policy recommendations. 112(45), pp. E6119-E6128.

Lagi, M., Bertrand, K. & Bar-Yam, Y., 2011. The food crises and political instability in North Africa and the Middle East. [Online]

Laio, F., Ridolfi, L. & D'Odorico, P., 2016. The past and future of food stocks. Environ. Res. Lett., Volume 11, p. 035010.

Lamastra, L., Miglietta, P.P., Toma, P., De Leo, F. and Massari, S., 2017. Virtual water trade of agri-food products: Evidence from italian-chinese relations. *Science of the Total Environment*, *599*, pp.474-482.

Leach, A. M., Emery, K. A., Gephart, J., Davis, K. F., Erisman, J. W., Leip, A., et al. (2016). Environmental impact food labels combining carbon, nitrogen, and water footprints. Food Policy, 61, 213–223. https://doi.org/10.1016/j.foodpol.2016.03.006

Lenzen, M., ; Moran, D.; Bhaduri, A.; Kanemoto, K.; Bekchanov, M.; Geschke, A.; Foran, B., (2013), International trade of scarce water, Ecol. Econ., 94, pp. 78-85.

Lanari, N., Schuler, R., Kohler, T., & Liniger, H. (2018). The Impact of Commercial Horticulture on River Water Resources in the Upper Ewaso Ng'iro River Basin, Kenya. *Mountain Research and Development*, *38*(2), 114-124.

Lin, X., Q. Dang, and M. Konar (2014), A network analysis of food flows within the United States of America, Environ. Sci. Technol., 48(10), 5439–5447, doi:10.1021/es500471d.

Liu, J., Hull, V., Batistella, M., DeFries, R., Dietz, T., Fu, F., ... & Martinelli, L. (2013). Framing sustainability in a telecoupled world. *Ecology and Society*, *18*(2).

Liu, J., Mooney, H., Hull, V., Davis, S.J., Gaskell, J., Hertel, T., Lubchenco, J., Seto, K.C., Gleick, P., Kremen, C. and Li, S., 2015. Systems integration for global sustainability. *Science*, *347*(6225), p.1258832.

Liu, J., and H. Yang (2010), Spatially explicit assessment of global consumptive water uses in cropland: Green and blue water, J. Hydrol., 384(3), 187–197.

Liu, J., Yang, H., Gosling, S.N., Kummu, M., Flörke, M., Pfister, S., Hanasaki, N., Wada, Y., Zhang, X., Zheng, C. and Alcamo, J., 2017. Water scarcity assessments in the past, present, and future. *Earth's Future*, *5*(6), pp.545-559.

Lutter S, Pfister S, Giljum S, et al. 2016. Spatially explicit assessment of water embodied in european trade: A product-level multi-regional input-output analysis Global Environmental Change 38: 171-182.MacDonald, G.K. Brauman K. A., Sun S., Carlson K. M., Cassidy E. S., Gerber J. S. and West P. C., 2015. Rethinking agricultural trade relationship in an era of globalization. BioScience, 65(3), pp. 275-289.

MacDonald, G. K., Bennett, E. M., & Carpenter, S. R. 2012. Embodied phosphorus and the global connections of United States agriculture. Environmental Research Letters, 7(4), 044024. https://doi.org/10.1088/1748-9326/7/4/044024

Macknick, J., Newmark, R., Heath, G. and Hallett, K.C., 2012. Operational water consumption and withdrawal factors for electricity generating technologies: a review of existing literature. *Environmental Research Letters*, *7*(4), p.045802.

Malthus, T. R. 1789. *An Essay on the Principle of Population* (ed. Flew, A.)

Marchand, P., Carr, J., Dell'Angelo, J. & al., 2016. Reserves and trade jointly determine exposure to food supply shocks. Environ. Res. Lett., Volume 11, p. 095009.

Margulis, ME, N McKeon & S Borras Jr. 2013.Land grabbing and global governance: critical perspectives, Globalisations, 10 (1), 1-23.

Marston, L., Konar, M, Troy, TJ, and XM Cai 2015 Virtual groundwater transfers from overexploited aquifers of the United States, PNAS, 112 no. 28, pp. 8561–8566, doi: 10.1073/pnas.1500457112

Marston, L., & Konar, M. (2016). Drought impacts to water footprints and virtual water transfers of the Central Valley of California. Water Resources Research.

Marston, L., Y. Ao, M. Konar, M. Mekonnen, and A.Y. Hoekstra (2018), High-resolution water footprints of production of the United States, Water Resources Research, Vol 54, doi: 10.1002/2017WR021923

Martinez-Melendez, L.A. and Bennett, E.M., 2016. Trade in the US and Mexico helps reduce environmental costs of agriculture. *Environmental Research Letters*, *11*(5), p.055004.

Mayer, A., S. Mubako, and B. L. Ruddell (2016), Developing the greatest Blue Economy: Water productivity, fresh water depletion, and virtual water trade in the Great Lakes basin, Earth's Future, 4, 282–297, doi:10.1002/2016EF000371

Mekonnen, M. M., and A.Y. Hoekstra (2011) The green, blue and grey water footprint of crops and derived crop products, Hydrol. Earth Syst. Sci., 15, 1577-1600, doi: 10.5194/hess-15-1577-2011

Mekonnen, M.M. & Hoekstra, A., 2010. The green, blue and grey water footprint of crops and derived crop products. *Hydrology and Earth System Sciences*, 384(3), pp.230–236. Available at: http://doc.utwente.nl/76914 [Accessed December 11, 2017].

Mekonnen, M. M., & Hoekstra, A. Y. (2012). A global assessment of the water footprint of farm animal products. Ecosystems, 15(3), 401–415, https://doi.org/10.1007/s10021-011-9517-8

Mekonnen, M.M. and Hoekstra, A.Y., 2016. Four billion people facing severe water scarcity. *Science advances*, *2*(2), p.e1500323.

Meldrum, J., Nettles-Anderson, S., Heath, G. and Macknick, J., 2013. Life cycle water use for electricity generation: a review and harmonization of literature estimates. *Environmental Research Letters*, *8*(1), p.015031.

Mena-Vásconez, P., Boelens, R., & Vos, J. (2016). Food or flowers? Contested transformations of community food security and water use priorities under new legal and market regimes in Ecuador's highlands. Journal of Rural Studies, 44, 227-238.

Meng, J., Mi, Z., Guan, D., Li, J., Tao, S., Li, Y., Feng, K., Liu, J., Liu, Z., Wang, X. and Zhang, Q., 2018. The rise of South–South trade and its effect on global CO 2 emissions. *Nature communications*, *9*(1), p.1871.

Meyfroidt, P., Lambin, E. F., Erb, K. H., & Hertel, T. W. (2013). Globalization of land use: Distant drivers of land change and geographic displacement of land use. *Current Opinion in Environmental Sustainability*, 5(5), 438–444. https://doi.org/10.1016/j.cosust.2013.04.003

Mielke, E., Anadon, L.D. and Narayanamurti, V., 2010. Water consumption of energy resource extraction, processing, and conversion. *Belfer Center for Science and International Affairs*.

Montgomery D.D. (2007). Dirt: The Erosion of Civilization, University of California Press.

Moran, D.D., M. Lenzen, K. Kanemoto, A. Geschke, (2013). Does Ecologically Unequal Exchange Occur? *Ecol. Econ.* 89, 177–186.

Moran, D.D., M. Lenzen, K. Kanemoto, A. Geschke, (2015). Response to Hornborg et al. *Ecol. Econ.* 119, 419.

Mori, K., 2003, 'Virtual water trade in global governance, Virtual water trade,' in A. Y. Hoekstra (ed.), *Proceedings of the International Expert Meeting on Virtual Water Trade*, Value of Water Research Report series # 12.

Müller, J. and Colloredo-Mansfeld, R., 2018. Introduction: Popular Economies and the Remaking of China–Latin America Relations. *The Journal of Latin American and Caribbean Anthropology*, *23*(1), pp.9-17.

Murphy, S. (2008). Globalization and corporate concentration in the food and agriculture sector. *Development*, *51*(4), 527-533.

Nesme, T., Roques, S., Metson, G.S. and Bennett, E.M., 2016. The surprisingly small but increasing role of international agricultural trade on the European Union's dependence on mineral phosphorus fertiliser. *Environmental Research Letters*, *11*(2), p.025003.

Nicholson SE. Land surface processes and Sahel climate. *Rev Geophys*. 2000;38:117–39.

Nicot, J.P. and Scanlon, B.R., 2012. Water use for shale-gas production in Texas, US. *Environmental science & technology*, *46*(6), pp.3580-3586.

Northey, S.A., Mudd, G.M., Saarivuori, E., Wessman-Jääskeläinen, H. and Haque, N., 2016. Water footprinting and mining: Where are the limitations and opportunities?. *Journal of Cleaner Production*, *135*, pp.1098-1116.

Oberlack, C., Boillat, S., Brönnimann, S., Gerber, J. D., Giger, M., Heinimann, A., and Wiesmann, U. M. (2017). Polycentric governance in telecoupled resource systems: Is the tragedy of the grabbed commons unavoidable?.

O'Bannon, C. et al., 2014. Globalization of agricultural pollution due to international trade.

Hydrology and Earth System Sciences, 18(2), pp.503–510. Available at: http://search.proquest.com/openview/2164555db7aecc2f02a3fcde47f1537d/1?pq-origsite=gscholar&cbl=105724 [Accessed December 13, 2017].

OECD 2013, Global Food Security: Challenges For The Food and Agricultural System, Chapter 4: Improving Access to Food, pp.97-133.

Oita et al., 2013. Substantial nitrogen pollution embedded in international trade. Nat. Geosci. 9, 111–115.

Oki, T., and Kanae, S. (2006). Global hydrological cycles and world water resources. S*cience*, *313*(5790), 1068-1072.

Oki, T., Yano, S. and Hanasaki, N., 2017. Economic aspects of virtual water trade. *Environmental Research Letters*, *12*(4), p.044002.

Orlowsky, B., et al. 2014. Today's virtual water consumption and trade under future water scarcity." Environmental Research Letters, 9.7: 074007.

Ortiz-Ospina E. and Roser M. (2018) - "International Trade". *Published online at OurWorldInData.org*. Retrieved from: 'https://ourworldindata.org/international-trade' [Online Resource]

Patel J. K. and Fountain H., 2017. As Artic ice vanishes, new shipping routes open. The New York Times. May 2017. Available at: https://www.nytimes.com/interactive/2017/05/03/science/earth/arctic-shipping.html

Peen World Tables Version 8.1. Available at: https://www.rug.nl/ggdc/.

Pennock, D. M., G. W. Flake, S. Lawrence, E. J. Glover, and C. L. Giles (2002), Winners don't take all: Characterizing the competition for links on the web, Proc. Natl. Acad. Sci. U. S. A., 99(8), 5207–5211.

Pfister S. and Bayer P. (2014). Monthly water stress: Spatially and temporally explicit consumptive water footprint of global crop production *Journal of Cleaner Production* 73: 52-62.

Pingali, P.L., 2012. Green revolution: impacts, limits, and the path ahead. *Proceedings of the National Academy of Sciences*, *109*(31), pp.12302-12308.

Poff, N.L., Allan, J.D., Bain, M.B., Karr, J.R., Prestegaard, K.L., Richter, B.D., Sparks, R.E. and Stromberg, J.C., 1997. The natural flow regime. *BioScience*, *47*(11), pp.769-784.

Porkka, M., Guillaume, J.H., Siebert, S., Schaphoff, S. and Kummu, M., 2017. The use of food imports to overcome local limits to growth. *Earth's Future*, *5*(4), pp.393-407.

Porkka, M., Gerten, D., Schaphoff, S., Siebert, S. and Kummu, M., 2016. Causes and trends of water scarcity in food production. *Environmental Research Letters*, *11*(1), p.015001.

Porkka, M., M. Kummu, S. Siebert, and O. Varis (2013), From food insufficiency towards trade dependency: A historical analysis of global food availability, PLoS One, 8(12), e82714, doi:10.1371/journal.pone.0082714.

Portmann, F.T., Siebert, S. & Döll, P., 2010. MIRCA2000-Global monthly irrigated and rainfed crop areas around the year 2000: A new high-resolution data set for agricultural and hydrological modeling. *Global Biogeochemical Cycles*, 24(1), p.n/a-n/a. Available at: http://doi.wiley.com/10.1029/2008GB003435 [Accessed December 6, 2017].

Postel, S.L., Daily, G.C. and Ehrlich, P.R., 1996. Human appropriation of renewable fresh water. *Science*, *271*(5250), pp.785-788.

Postel, S., & Richter, B. 2003. *Rivers for life: Managing water for people and nature*. Washington, DC: Island Press

Puma, M., Bose, S., Chon, S. & Cook, B., 2015. Assessing the evolving fragility of the global food system. Environ. Res. Lett., Volume 10, p. 024007.

Raes, D. et al., 2009. AquaCrop the FAO crop model to simulate yield response to water: II. Main algorithms and software description. *Agronomy Journal*. Available at: https://dl.sciencesocieties.org/publications/aj/abstracts/101/3/438 [Accessed December 6, 2017].

Reimer, J., 2012. On the economics of virtual water trade. Ecol. Econ., Volume 75, p. 135–139.

Ren, D., Yang, Y., Yang, Y., Richards, K. and Zhou, X., 2018. Land-Water-Food Nexus and indications of crop adjustment for water shortage solution. *Science of The Total Environment*, *626*, pp.11-21.

Richter, B. (2014). Chasing water: A guide for moving from scarcity to sustainability. Washington, DC: Island Press.

Rice, J. 2007. Ecological Unequal Exchange: Consumption, Equity, and Unsustainable Structural Relationships within the Global Economy, *International Journal of Comparative Sociology*, 48(1): 43–72, DOI: 10.1177/0020715207072159.

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin III, F.S., Lambin, E., Lenton, T.M., Scheffer, M., Folke, C., Schellnhuber, H.J. and Nykvist, B., 2009. Planetary boundaries: exploring the safe operating space for humanity. *Ecology and society*, *14*(2).

Rodell, M., Famiglietti, J.S., Wiese, D.N., Reager, J.T., Beaudoing, H.K., Landerer, F.W. and Lo, M.H., 2018. Emerging trends in global freshwater availability. *Nature*, p.1.

Rosa, L., Rulli, M.C., Davis, K.F., Chiarelli, D.D., Passera, C. and D'Odorico, P., 2018a. Closing the yield gap while ensuring water sustainability. *Environmental Research Letters*, *13*(10), p.104002.Rosa, L., Davis, K.F., Rulli, M.C. and D'Odorico, P., 2017. Environmental consequences of oil production from oil sands. *Earth's Future*, *5*(2), pp.158-170.

Rosa, L., Rulli, M.C., Davis, K.F. and D'Odorico, P., 2018b. The water-energy nexus of hydraulic fracturing: a global hydrologic analysis for shale oil and gas extraction. *Earth's Future*, 6(5), pp. 645-656.

Rosa, L. and D'Odorico, P., 2019. The water-energy-food nexus of unconventional oil and gas extraction in the Vaca Muerta Play, Argentina. *Journal of Cleaner Production*, *207*, pp.743-750.

Rosegrant, M. & IMPACT Development Team, 2012. International Model for Policy Analysis of Agricultural Commodities and Trade (IMPACT): Model Description, Washington, D.C.: International Food Policy Research Institute (IFPRI).

Rosegrant MW , M. Paisner, S. Meijer, J. Whitcover (2001) *Global Food Projections to 2020: Emerging Trends and Alternative Futures*, IFPRI, Washington.

Rosegrant, M. W., Ringler, C., and Zhu, T. (2009). Water for agriculture: maintaining food security under growing scarcity. Annual review of Environment and resources, 34.

Rosegrant, M., Cai, X. & Cline, S., 2002. World water and food to 2025: dealing with scarcity, Washington, DC: International Food Policy Research Institute.

Rost, S., D. Gerten, A. Bondeau, W. Lucht, J. Rohwer, and S. Schaphoff, 2008. Agricultural Green and Blue Water Consumption and Its Influence on the Global Water System. Water Resources Research 44(9):W09405, doi: 10.1029/2007WR006331.

Rulli, M.C. & D'Odorico, P.D., 2013. The water footprint of land grabbing. *Geophysical Research Letters*, 40(October), pp.6130–6135. Available at: http://onlinelibrary.wiley.com/doi/10.1002/2013GL058281/full.

Rulli, M.C., D. Bellomi, A. Cazzoli, G. De Carolis, and P. D'Odorico, 2016 The water-energy-food nexus of first-generation biofuels, Sci. Rep., 6:22521 doi:10.1038/srep22521.

Rulli, M.C., Saviori, A. & D'Odorico, P., 2013. Global land and water grabbing. *Proceedings of the National Academy of Sciences of the United States of America*, 110(3), pp.892–897, doi: 10.1073/pnas.1213163110.

Rushforth RR, Ruddell BL. The vulnerability and resilience of a city's water footprint: The case of Flagstaff, Arizona, USA. Water Resour Res 2016; 52:2698–714. doi:10.1002/ 2015WR018006.

Sacks, W. J., Cook, B. I., Buenning, N., Levis, S., & Helkowski, J. H. (2009). Effects of global irrigation on the near-surface climate. Climate Dynamics, 33(2-3), 159–175. https://doi.org/10.1007/s00382-008-0445-z

Sartori, M. & Schiavo, S., 2015. Connected we stand: A network perspective on trade and global food security. Food Policy , Volume 57, p. 114–127.

Sartori, M., Schiavo, S., Fracasso, A. and Riccaboni, M., 2017. Modeling the future evolution of the virtual water trade network: A combination of network and gravity models. *Advances in Water Resources*, 110, 538-548.

Scanlon B R, Faunt C C, Longuevergne L, Reedy R C, Alley W M, McGuire V L and McMahon P B 2012 Groundwater depletion and sustainability of irrigation in the US High Plains and Central Valley *Proceedings of the national academy of sciences 109*(24) pp.9320-9325.

Schyns, J.F. and Hoekstra, A.Y., 2014. The added value of water footprint assessment for national water policy: a case study for Morocco. *PLoS One*, 9(6), p.e99705.

Schyns, J.F., Hoekstra, A.Y. and Booij, M.J., 2015. Review and classification of indicators of green water availability and scarcity. Hydrology and earth system sciences, 19(11), pp.4581-4608.

Seekell, D. A., D'Odorico, P., & Pace, M. L. (2011). Virtual water transfers unlikely to redress inequality in global water use. Environmental Research Letters, 6(2), 024017. https://doi.org/10.1088/1748-9326/6/2/024017

Seekell, D., Carr, J., Dell'Angelo, J. & al., e., 2017. Resilience in the global food system. Environ. Res. Lett., Volume 12, p. 025010.

Selby, Jan. "Beyond hydro-hegemony: Gramsci, the national, and the trans-national." *Occasional Paper* 94 (2007).

Serrano, A., Guan, D., Duarte, R., Paavola, J. (2016), Virtual Water Flows in the EU27 A Consumption-based Approach, Journal of industrial ecology, 20 (3), 547-558

Shumilova O., Tockner K, Thieme M, Koska A, and Zarfl C (2018) Global Water Transfer Megaprojects: A Potential Solution for the Water-Food-Energy Nexus? *Frontiers in Environmental Science* 6, 150, DOI=10.3389/fenvs.2018.00150

Siebert S and Döll P 2010 Quantifying blue and green virtual water contents in global crop production as well as potential production losses without irrigation. *Journal of Hydrology 384*(3) 198-217

Siebert, S., Burke, J., Faures, J.M., Frenken, K., Hoogeveen, J., Döll, P. and Portmann, F.T., 2010. Groundwater use for irrigation–a global inventory. *Hydrology and Earth System Sciences*, *14*(10), pp.1863-1880.

Smakhtin, V., Revenga, C. and Döll, P., 2004. A pilot global assessment of environmental water requirements and scarcity. *Water international*, *29*(3), pp.307-317.

Smil, V. (1994). How many people can the earth feed? *Population and Development Review*, 20(2), 255.

Sojamo, S., Keulertz, M., Warner, J. and Allan, J.A., 2012. Virtual water hegemony: the role of agribusiness in global water governance. *Water International*, *37*(2), pp.169-182.

Soligno I., Ridolfi L., Laio F. (2017), The environmental cost of a reference withdrawal from surface waters: Definition and geography. Adv. Water Res., 110, ,228-237

Steduto, P. et al., 2009. AquaCrop—The FAO crop model to simulate yield response to water: I. Concepts and underlying principles. *Agronomy Journal*. Available at: https://dl.sciencesocieties.org/publications/aj/abstracts/101/3/426 [Accessed December 6, 2017].

Stephenson, S.R., Wang, W., Zender, C.S., Wang, H., Davis, S.J. and Rasch, P.J., 2018. Climatic responses to future trans-Arctic shipping. *Geophysical Research Letters*, *45*(18), pp.9898-9908.

Sternberg, T., 2016. Water megaprojects in deserts and drylands. *International Journal of Water Resources Development*, *32*(2), pp.301-320.

Strogatz, S.H. 2014. Nonlinear dynamics and chaos: with applications to physics, biology, chemistry, and engineering. Hachette UK.

Sun, J., Mooney, H., Wu, W., Tang, H., Tong, Y., Xu, Z., Huang, B., Cheng, Y., Yang, X., Wei, D. and Zhang, F., 2018. Importing food damages domestic environment: Evidence from global soybean trade. *Proceedings of the National Academy of Sciences*, *115*(21), pp.5415-5419.

Suweis, S., Konar, M., Dalin, C., Hanasaki, N., Rinaldo, A. and Rodriguez-Iturbe, I., 2011. Structure and controls of the global virtual water trade network. Geophysical Research Letters, 38(10).

Suweis, S., A. Rinaldo, A. Maritan, and P. D'Odorico, 2013 Water-controlled wealth of nations, *Proc. Natnl Acad. Sci, USA*, PNAS, 110(11), 4230-4233, doi: 10.1073/pnas.1222452110, 2013.

Suweis, S., J.A. Carr, A. Rinaldo, A. Maritan, and P. D'Odorico, 2015. Resilience and reactivity of global food security, *Proc. Natnl. Acad. Sci., USA, PNAS*, doi: 10.1073/pnas.1507366112.

Tamea, S., Allamano, P., Carr, J.A., Claps, P., Laio, F., Ridolfi, L. (2013), Local and global perspectives on the virtual water trade, Hydrol. Earth Syst. Sci., 17, 1205–1215.

Tamea, S., Carr, J. A., Laio, F. & Ridolfi, L., 2014. Drivers of the virtual water trade. Water Resour. Res., Volume 50, p. 17–28.

Tamea, S., Laio, F. & Ridolfi, L., 2016. Global effects of local food-production crises: A virtual water perspective. Sci. Rep., Volume 6, p. 18803.

Taylor, R.G., Scanlon, B., Döll, P., Rodell, M., Van Beek, R., Wada, Y., Longuevergne, L., Leblanc, M., Famiglietti, J.S., Edmunds, M. and Konikow, L., 2013. Ground water and climate change. *Nature Climate Change*, *3*(4), p.322.

Tuninetti, M., Tamea, S., D'Odorico, P., Laio, F., Ridolfi, L. (2015), Global sensitivity of high-resolution estimates of crop water footprint. *Water Resources Research*, *51*(10), 8257-8272.

Tuninetti, M.., Tamea, S., Laio, F., Ridolfi L. (2017a), To trade or not to trade: Link prediction in the virtual water network, *Advances in Water Resources*, 110, pp.528–537.

Tuninetti, M., Tamea, S., Laio, F., Ridolfi, L. (2017b), A Fast Track approach to deal with the temporal dimension of crop water footprint, *Environmental Research Letters*, *12*(7), 074010.

Van Ittersum, M.K., Van Bussel, L.G., Wolf, J., Grassini, P., Van Wart, J., Guilpart, N., Claessens, L., de Groot, H., Wiebe, K., Mason-D'Croz, D. and Yang, H., 2016. Can sub-Saharan Africa feed itself?. *Proceedings of the National Academy of Sciences*, *113*(52), pp.14964-14969.

Vora, N., Shah, A., Bilec, M.M. and Khanna, V., 2017. Food–energy–water nexus: quantifying embodied energy and ghg emissions from irrigation through virtual water transfers in food trade. *ACS Sustainable Chemistry & Engineering*, *5*(3), pp.2119-2128.

Vörösmarty, C.J., Hoekstra, A.Y., Bunn, S.E., Conway, D. and Gupta, J., 2015. Fresh water goes global. *Science*, *349*(6247), pp.478-479.

Vörösmarty, C.J., McIntyre, P.B., Gessner, M.O., Dudgeon, D., Prusevich, A., Green, P., Glidden, S., Bunn, S.E., Sullivan, C.A., Liermann, C.R. and Davies, P.M., 2010. Global threats to human water security and river biodiversity. *Nature*, *467*(7315), p.555.

Yano, S., Hanasaki, N., Itsubo, N. and Oki, T., 2015. Water scarcity footprints by considering the differences in water sources. Sustainability, 7(8), pp.9753-9772.

Yano, S., Hanasaki, N., Itsubo, N. and Oki, T., 2016. Potential Impacts of Food Production on Freshwater Availability Considering Water Sources. Water, 8(4), p.163.

Wackernagel, M., Onisto, L., Bello, P., Linares, A. C., Falfán, I. S. L., Garcıa, J. M., et al. (1999). National natural capital accounting with the ecological footprint concept. Ecological Economics, 29(3), 375–390. https://doi.org/10.1016/S0921-8009(98)90063-5

Wada Y, van Beek L P, van Kempen C M, Reckman J W, Vasak S and Bierkens M F 2010 Global depletion of groundwater resources *Geophysical research letters 37*(20)

Wang, G. and Eltahir, E.A., 2000. Ecosystem dynamics and the Sahel drought. *Geophysical Research Letters*, *27*(6), pp.795-798.

Wang, R., & Zimmerman, J. (2016). Hybrid Analysis of Blue Water Consumption and Water Scarcity Implications at the Global, National, and Basin Levels in an Increasingly Globalized World. Environmental Science & Technology, 50(10), 5143-5153.

Wang, R., Hertwich, E. and Zimmerman, J.B., 2016. (Virtual) Water Flows Uphill toward Money. *Environmental science & technology*, *50*(22), pp.12320-12330.

Ward, J.R. 1993. Environmental strategies and agricultural trade. In: Zaelke, D., Orbuch P., and Housman, R.F. (Editors). *Trade and the Environment: Law, Economics, and Policy*, Island Press, Washington, D.C. pp. 247-256.

Warren, S. G. (2015). Can human populations be stabilized? Earth's Future, 3(2), 82–94. https://doi.org/10.1002/2014EF000275.

Warner, J.F. and Zeitoun, M., 2008. International relations theory and water do mix: A response to Furlong's troubled waters, hydro-hegemony and international water relations. *Political Geography*, *27*(7), pp.802-810.

Wathen, T., 1993., A guide to trade and the environment. In: Zaelke, D., Orbuch P., and Housman, R.F. (Editors). *Trade and the Environment: Law, Economics, and Policy*, Island Press, Washington, D.C. pp. 3-21.

Weidong, L., 2015. Scientific understanding of the Belt and Road Initiative of China and related research themes. *Progress in Geography*, *34*(5), pp.538-544.

Wichelns, D., 2001. The role of 'virtual water' in efforts to achieve food secrity and other national goals, with an example from Egypt. Agric. Water Manage., Volume 49, pp. 131-151.

Wichelns, D., 2004. The policy relevance of virtual water can be enhanced by considering comprative advantages. Agric. Water Manage., Volume 66, pp. 49-63.

Wiedmann, T. and Lenzen, M., 2018. Environmental and social footprints of international trade. *Nature Geoscience*, *11*(5), pp.314-321.

Winter, J.A., Allamano, P. and Claps, P., 2014. Virtuous and vicious virtual water trade with application to Italy. *PloS one*, *9*(4), p.e93084.

Wolf, A. T. (1998). Conflict and cooperation along international waterways. *Water policy*, *1*(2), 251-265.

Wolf, A. T. (2007). Shared waters: Conflict and cooperation. *Annu. Rev. Environ. Resour.*, *32*, 241-269.

Wood, R., Stadler, K., Simas, M., Bulavskaya, T., Giljum, S., Lutter, S. and Tukker, A., 2018. Growth in environmental footprints and environmental impacts embodied in trade: Resource efficiency indicators from EXIOBASE3. Journal of Industrial Ecology, 22(3), pp.553-564.

Yang, Z., Mao, X., Zhao, X. & Chen, B., 2012. Ecological network analysis on global virtual water trade. Environ. Sci. Technol., Volume 46, p. 1796−1803.

Yergin, D., 2011. *The prize: The epic quest for oil, money & power*. Simon and Schuster.

Yu, B. and A. Nin Pratt (2011), "Agricultural productivity and policies in Sub-Saharan Africa", IFPRI Discussion Paper 01150, International Food Policy Research Institute, Washington, DC.

Zaelke, D., Orbuch P., and Housman, R.F. 1993. *Trade and the Environment: Law, Economics, and Policy*, Island Press, Washington, D.C.

Zeitoun, M. and Warner, J., 2006. Hydro-hegemony–a framework for analysis of trans-boundary water conflicts. *Water policy*, *8*(5), pp.435-460.

Zhang, C. and Anadon, L.D., 2014. A multi-regional input–output analysis of domestic virtual water trade and provincial water footprint in China. *Ecological Economics*, *100*, pp.159-172.

Zhang, J.C., Zhong, R., Zhao, P., Zhang, H.W., Wang, Y. and Mao, G.Z., 2016a. International energy trade impacts on water resource crises: an embodied water flows perspective. *Environmental Research Letters*, *11*(7), p.074023.

Zhang, Y., Zhang, J., Tang, G., Chen, M. and Wang, L., 2016b. Virtual water flows in the international trade of agricultural products of China. *Science of the Total Environment*, *557*, pp.1-11.

Zhao, X., Li, Y.P., Yang, H., Liu, W.F., Tillotson, M.R., Guan, D., Yi, Y. and Wang, H., 2018. Measuring scarce water saving from interregional virtual water flows in China. *Environmental Research Letters*, *13*(5), p.054012.

Zhao, Y., Zhu, Y., Lin, Z., Wang, J., He, G., Li, H., Li, L., Wang, H., Jiang, S., He, F. and Zhai, J., 2017. Energy Reduction Effect of the South-to-North Water Diversion Project in China. *Scientific Reports*, *7*(1), p.15956.

Zhuo, L., Mekonnen, M. M., Hoekstra, A. Y., & Wada, Y. (2016). Inter-and intra-annual variation of water footprint of crops and blue water scarcity in the Yellow River basin (1961–2009). Advances in Water Resources, 87, 29-41.

Supplementary Materials

Table S1. Summary of explanatory variables used by different models of Virtual Water Trade.

	Suweis et al (2011)	*Dalin et al (2012)*	*Sartori et al (2018)*	*Tamea et al (2014)*	*Tuninetti et al (2014)*	*Chouchane (2018)*	*Kumar, Singh (2005)*	*Fracasso (2014)*	*Fracasso et al (2016)*
Population		x	x	x	x	x		x	x
WF of demand					x				
Rainfall	x	x	x					x	
Water endowment			x					x	
Availability ratio								x	x
Temperature									x
Agricultural area					x		x	x	x
WF of production				x	x				
Agricultural efficiency					x				x
Irrigated land						x			
GDP	x	x	x	x	x	x		x	x
Distance				x	x			x	

Dummies				x		x	
Water price							x
	(a)	(a)	(b,c)	(g)	(e)	(d,f,0)	(f,g,h)

NOTES: (a): rainfall on agricultural area; (b): rainfall per capita; (c): total renewable water resources; (d): water for agriculture, per capita; (e): gross and net cropped area; (f): arable land per capita; (g): efficiency as N-fertilizers; (h): efficiency as tractors; (0): all variables are per capita