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Virtual Water and Water Footprints: Overreaching Into the Discourse on Sustainability, Efficiency, and Equity

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ABSTRACT: The notions of virtual water and water footprints were introduced originally to bring attention to the large amounts of water required to produce crops and livestock. Recently, several authors have begun applying those notions in efforts to describe efficiency, equity, and the sustainability of resources and production activities. In this paper, I describe why the notions of virtual water and water footprints are not appropriate for analysing issues pertaining to those topics. Both notions lack a supporting conceptual framework and they contain too little information to enhance understanding of important policy issues. Neither notion accounts for the opportunity cost or scarcity value of water in any setting, or the impacts of water availability and use on livelihoods. In addition, countries trade in goods and services – not in crop and livestock water requirements. Thus, the notions of virtual water and water footprints cannot provide helpful insight regarding the sustainability of water use, economic efficiency, or social equity. Gaining such insight requires the application of legitimate conceptual frameworks, representing a broad range of perspectives from the physical and social sciences, with due consideration of dynamics, uncertainty, and the impacts of policy choices on livelihoods and natural resources.

KEYWORDS: Agriculture, economics, food security, livelihoods, risk, trade, uncertainty

INTRODUCTION

The notion of virtual water was first presented as an interesting description of how arid countries satisfy their annual food demands by importing substantial amounts of grain and other products (Allan, 1996, 2002). Water footprints were introduced somewhat later, with the goal of describing the direct and indirect water use by consumers and producers (Chapagain and Hoekstra, 2004; Hoekstra and Chapagain, 2007a). Both virtual water and water footprints characterise the volume of fresh water used to produce the goods and services consumed by an individual or community (Ercin and Hoekstra, 2014). The depiction of the estimated water footprints of an array of goods and services, particularly in the popular press, likely has enhanced public awareness of water resource issues in some areas. However, much of the discussion of virtual water and water footprints in the scholarly literature involves normative prescriptions regarding land and water allocation, consumer choices, and international trade. It is not clear that the notions of virtual water and water footprints contain sufficient information to support such prescriptions.

In recent years, several authors have extended the discussion of virtual water and water footprints to include the topics of sustainability, efficiency, and equity. Such discussion moves the proposed applications of virtual water and water footprints even further beyond their initial use as simple descriptors of the volumes of water used in production. This is not a trivial extension, as there is substantial interest among policy makers and international donors in the topics of sustainability, efficiency, and equity. My goal in this paper is to describe why estimates of virtual water and water footprints do not provide insight regarding those topics. I first review briefly the earlier applications and critiques of virtual water and water footprints. I then consider several recent papers in which the authors apply the notions of virtual water and water footprints in the analysis of sustainability, efficiency, and equity.

Individual authors offer different perspectives regarding sustainability, efficiency, and equity. In this paper, I consider a somewhat narrow definition of sustainability, in the context of ensuring that water supplies and water quality are sufficient to meet the demands for water, over time, in agricultural, commercial, residential, and environmental sectors. Efficiency involves maximising the net values obtained in all sectors, subject to resource constraints and while addressing all pertinent externalities. Equity pertains to the notion of a fair distribution of outcomes that are influenced by access to land and water resources, such as income generating opportunities and household welfare.

EARLIER APPLICATIONS AND CRITIQUES

Many authors have presented estimates of water footprints for a wide range of goods and services, and many have described the 'flows of virtual water' between countries that engage in trade (Hoekstra and Chapagain, 2007a, b; Ercin and Hoekstra, 2014; Zoumides et al., 2014; Marston et al., 2015). Several have suggested that the 'flows of virtual water' should be considered when designing national strategies regarding water and agriculture, and when evaluating guidelines pertaining to international trade (Rudenko et al., 2013; Duarte et al., 2014; Shi et al., 2014; Zhang and Anadon, 2014; Zoumides et al., 2014). Some have suggested that water-short countries should import water-intensive products from countries with larger water endowments, and that water-abundant counties should focus on producing and exporting water-intensive goods (Yang and Zehnder, 2002; Hoekstra and Hung, 2005; Velázquez, 2007; Chapagain and Hoekstra, 2008; Winter et al., 2014). Some propose changes in the locations or methods of production, to produce the goods and services consumed worldwide with the smallest possible water footprints (Ercin et al., 2012).

At one level, the presentations of water footprints and descriptions of virtual water are merely descriptive, and they might be viewed as simply highlighting the role of water in production. The prescriptions regarding production and trade, which often accompany the presentations, might seem compelling at first glance, but the supporting analysis generally considers only the water input, with no consideration of other inputs, resource conditions, or the potential impacts on livelihoods. There are many inputs in most production processes, and the incremental value of any one input often is a function of the amounts and quality of several other inputs. Production strategies and resource policies based on just one input will be inappropriate in most settings. In addition, firms and countries do not trade in the volumes of water required to produce goods and services, just as they do not trade in the amounts of labour, energy, or capital used in production. Decisions regarding imports and exports are based appropriately on relative prices, reliability of delivery, and product quality. Traders respond also to tariffs, subsidies, and non-price barriers to trade, but they have no incentive to consider a single input used to produce the goods and services they trade.

Countries do not trade in virtual water

There is no conceptual or empirical support for deriving policy implications based on the 'flows of virtual water' between countries. The notion of virtual water is not consistent with the concept of comparative advantage, which is a fundamental tenet of trade theory (Wichelns, 2004, 2011a, b). The relative size of a country’s water endowment is not a sufficient criterion for determining optimal trading strategies. Thus, the prescriptions of virtual water analysis regarding international trade do not offer useful advice for firms or policy makers, and can result in misleading and inefficient recommendations (Gawel and Bernsen, 2011a).

Many of the policy prescriptions based on virtual water would impose production and trading regimes on firms or countries, to achieve some notion of global equity with respect to water consumption, or to minimise the 'global water footprint', without considering local and regional preferences, opportunities, and constraints (Gawel and Bernsen, 2013; Gawel, 2014). The prescriptions, if adopted, might require economically harmful adjustments in production and trade, while also potentially denying developing countries the opportunity to determine and implement optimal production and trading strategies (Gawel and Bernsen, 2013). Those countries, even in water-scarce regions, must consider much more than their water endowment when selecting viable paths of economic development.

Several authors have described the 'flow of virtual water' between sub-national units, such as states and provinces. Bulsink et al. (2010) suggest that some provinces of Indonesia export virtual water to other provinces, while some provinces are net importers of virtual water. Others have described the 'flow of virtual water' between provinces and counties in China (Dalin et al., 2014, 2015; Zhang and Anadon, 2014; Shi and Zhan, 2015; Jiang et al., 2015; Liu et al., 2015) and between states in the United States (Mubako et al., 2013). The notion of virtual water trade at the sub-national level suffers from the same conceptual inadequacies as the notion of virtual water trade among countries. States and provinces do not engage in trade on the basis of a single input in production, and there is no consideration of opportunity costs when calculating virtual water volumes.

It is also not clear that states, provinces, or counties engage in trade, in the traditional sense that they import and export goods and services from each other. In market economies, such as the United States, commercial firms produce, buy, and sell products, moving them across state lines in response to market signals. The states of Illinois and California do not engage in bi-state trade, as suggested by Mubako et al. (2013). Fruit and vegetables grown in California are purchased in large amounts by firms and brokers who distribute the products across markets in many states. Similarly, the corn and soybeans grown in Illinois are sold to large firms that process the grain and beans into products that are distributed across the country and internationally. It seems implausible to consider that Illinois and California engage in virtual water trade.

The notion of sub-national trade seems misplaced also in quasi-market economies, in which the government intervenes in the pricing, purchasing, and distribution of selected agricultural commodities. The Government of Indonesia has a long history of intervening in the rice market to motivate greater domestic production, while also stabilising the price of rice for consumers (Timmer, 1996; Arifin, 2008; Warr, 2011). Indonesia presently restricts rice imports and encourages domestic production by subsidising the farm-level price of fertiliser (Warr and Yusuf, 2014). In such an environment, rice production and marketing are notably influenced by the prevailing policy parameters. It is not clear that provinces in Indonesia engage in rice trade, first within island groups, and then with more distant provinces, as suggested by Bulsink et al. (2010). It is more likely that rice brokers purchase rice from farmers and they sell the rice where they can obtain the highest net price.

Liu et al. (2015) calculate the 'virtual water flows' between five counties in the very large Hetao Irrigation District in China. The authors use the same methodology as that of Bulsink et al. (2010), in which counties with consumption:production deficits are assumed to trade first with the nearest counties that have consumption:production surpluses. Neither the original promoters of this method (Ma et al., 2006), nor Liu et al. (2015) provide empirical evidence that trade between provinces or counties occurs in this manner. In a country as large as China, with a well-developed transport system, it seems unlikely that the full range of products considered in Liu et al. (2015) – wheat, coarse cereals, sugar crops, and vegetables – would move largely between nearby counties. Substantial commerce likely does take place across county lines, but it is not clear that such commerce would be considered as trade between counties.

Zhang and Anadon (2014) also examine sub-national 'virtual water trade' in China, using an input-output model, rather than the method proposed by Ma et al. (2006). The authors examine information for 30 provinces, which they divide into six geographic regions. Given the large population density in eastern China, most of the estimated 'flows of virtual water' move into the east region. The authors find no empirical support for the notion of virtual water trade between Chinese provinces, as their estimates of 'virtual water exports' are larger in drier regions of the country.

Regarding empirics at the national level, Kumar and Singh (2005) find that the observed trading patterns for 146 countries are not consistent with those predicted by the virtual water perspective. Some water-abundant countries import food, while some water-scarce countries export food. De Fraiture et al. (2004) also find limited empirical support for the virtual water perspective. They caution against inferring that international trade can mitigate water scarcity, in part, because political and economic considerations can have greater influence than water scarcity in determining national trading strategies. Lopez-Gunn and Llamas (2008) also observe that international trade in food is driven largely by factors other than water.

Upon examining the estimates of virtual water imports and exports prepared for 77 countries by Chapagain and Hoekstra (2004), Wichelns (2010a, b) finds that the amount of arable land per person in a country is a better descriptor of international trade patterns than is the amount of renewable water resources available, per person or per hectare. A country's arable land endowment is not a sufficient predictor of trade patterns, but it is a better descriptor of trade in crop and livestock products than a country's water endowment.

Ramirez-Vallejo and Rogers (2004), while considering the Heckscher-Ohlin model of international trade, which has fewer restrictions than the model of comparative advantage, find little empirical support for predicting trade patterns on the basis of national water endowments. Rather, they find that average income, population, irrigated area, and the amount of value added in agriculture explain the observed variation in traded agricultural commodities.

The impacts of water use are local and regional

Several authors have suggested that water footprints provide insight regarding the impacts of consumers in one country on the water resources of another (Chapagain et al., 2006; Chapagain and Hoekstra, 2007; van Oel et al., 2009; Hoff et al., 2014). Some have calculated the internal and external water footprints of selected countries (Hoekstra and Chapagain, 2007b; Fader et al., 2011), with the goal of quantifying the international impacts. The internal water footprint is defined as the volume of domestic water resources used to produce the goods and services consumed by a country's residents (Ercin and Hoekstra, 2014). It is calculated by subtracting the volume of virtual water exported to other countries from the volume of water required to produce all goods and services within a country. By analogy, the external water footprint is defined essentially as the volume of water used in other countries to produce the goods and services consumed by residents of the importing country (Ercin and Hoekstra, 2014).

Van Oel et al. (2009) report that 89% of the estimated per capita water footprint for the Netherlands is external, reflecting the large role of imports in the economy of the Netherlands. The authors suggest that the external water footprint imposes negative impacts on the countries from which the Dutch import goods and services. Chapagain et al. (2006) suggest that Japan's demand for cotton exerts pressure on water resources in Pakistan, China, and India. The authors suggest also that European consumers, through their consumption of cotton products, contribute indirectly to about 20% of the desiccation of the Aral Sea. In aggregate, about "half of the water problems in the world related to cotton growth and processing can be attributed to foreign demand for cotton products (p. 201)".

The characterisation of water scarcity and water quality as global phenomena is compelling, but inaccurate. While water can be viewed as an international resource, particularly in areas where countries share rivers, aquifers, and watersheds, water scarcity and water quality are largely local and regional issues (Gawel and Bernsen, 2011c; Gawel, 2014; Perry, 2014). Water scarcity arises when the demands on local and regional resources exceed the available supply. Water quality is degraded most often due to inappropriate practices within a given country. While acknowledging important issues regarding transboundary resources, generally there is little relationship between water consumption in one region and water scarcity or water quality in another.

It is not likely that Dutch consumers who switch from eating rice to potatoes, those who replace cane sugar with beet sugar, or those who choose clothing made from rainfed – rather than irrigated – cotton, will have any impact on water scarcity conditions or water quality in the countries of origin, as suggested by Hoekstra (2008). Similarly, a reduction in cotton imports into Japan or the European Union likely will have little impact on water scarcity or water quality in the cotton producing regions of India, Pakistan, or Uzbekistan, as suggested by Chapagain et al. (2006). The scarce water resources in those countries will be used to produce goods for sale in local, regional, or international markets, in response to market demands, provided that the returns exceed the costs of production and marketing. If consumers in distant countries discontinue their purchases of cotton from water scarce regions, consumers in other countries likely will take their place. If all consumers discontinue their purchases, the farmers in water scarce regions will consider alternative production activities. Perhaps they will produce other crops that generate smaller net returns. Yet, the water resources will be used, as long as the net returns are positive. Issues regarding water scarcity and water quality must be addressed by local officials who must implement the right policies to motivate wise use of natural resources. It is their responsibility to ensure that the externalities of water use are internalised in firm-level decisions.

The same reasoning regarding local v. global perspectives applies also to the comparison of carbon and water footprints. While some authors have suggested the two types of footprints are similar (Hoekstra, 2009; Ercin and Hoekstra, 2012), the characteristics of each are quite different. Carbon emissions essentially have the same impact on the atmosphere, regardless of where the emissions are generated. The sum of global carbon emissions is of interest when considering implications for the global climate. By contrast, the impacts of water scarcity and water quality degradation are realised in local and regional settings (Gawel and Bernsen, 2011a; Wichelns, 2011c; Perry, 2014). Thus, the two footprints are not analogous. Reducing the carbon footprint of an activity generates a globally relevant impact. The same is not true when reducing a water footprint.

Estimates of water footprints are inaccurate and potentially misleading

Perry (2014) has shown that the water footprint perspective fails to account for inherent complexities in describing and managing water resources, and that estimates of water footprints do not account correctly for evapotranspiration. Regarding complexities, the notions of green and blue water are presented in the literature as distinct forms of water made available to crops, either by rainfall or irrigation. Yet, the ultimate source of both forms of water is rainfall, which can infiltrate into soils or run off the surface into streams and lakes. Some soil moisture contributes to plant growth directly, while some of the water percolates into aquifers. In addition, shallow aquifers interact with streams, such that green water can become blue water, and vice versa. As Perry (2014) describes,

> [w]ater that is held in the unsaturated layer between the root zone and the (saturated) aquifer has no colour status, but will eventually be blue if, as a result of further rainfall or excessive irrigation, it reaches an underlying aquifer or appears as a spring, further down slope. Yet the water might become green again if the underlying aquifer is recharged and rises toward the root zone.

The notions of green and blue water, as presented in the literature, are not distinct and separable. They are interdependent components of the same hydrologic system (Perry, 2014).

Many of the authors who present estimates of water footprints rely on the estimation procedure described in the Water Footprint Manual (Hoekstra et al., 2009, 2011), which involves several shortcomings, particularly with regard to irrigated crops (Perry, 2014):

- The assumption that crop evapotranspiration (ET) from any source is equal to potential crop ET is incorrect when crop yields are less than potential. When plants are stressed, transpiration slows, such that actual ET will be less than potential ET. As a result, estimated water footprints generally overstate actual evapotranspiration.

- Because actual ET generally is less than potential ET, while the contribution of rainfall is fixed, any overestimation of ET is assigned to the blue water component.
- The sources and uses of moisture in the soil before and after the crop season affect estimates of the contribution of green and blue water to total consumption.

In sum, the discussion of green and blue water can be inaccurate and misleading, regarding crop water use and water movement in soils and waterways. Yet, there is no need to rely on such terms and calculations. Specialists in hydrology and agronomy have well-established terms, procedures, and standards for describing the impacts of rainfall, irrigation, and crop production on the hydrologic cycle and on soils and aquifers (Perry, 2014). Terms such as effective rainfall, infiltration, soil moisture, deep percolation, runoff, and conjunctive use provide better information and greater insight regarding water management than do the notions of green and blue water (Wichelns, 2011c).

The inherent lack of information in water footprints renders them inappropriate for comparing resource use and production outcomes at any scale. It is not meaningful to compare estimates of water footprints across countries or across river basins within countries. The opportunity costs of water, production technology, and the prices and availability of complementary inputs vary widely across such settings. Even within a river basin or irrigation district, where the empirical values of those parameters might be similar, water footprints will not convey information that allows one to evaluate the efficiency of water use or crop production. This inadequacy can be demonstrated using an example involving three farms, perhaps operating in the same river basin or the same irrigation district.

Suppose we are told that the water footprint of maize production for two farms in the most recent season is precisely the same, at 500 m^3 per ton. At first glance, it might seem that both farms are operating in similar fashion, with regard to water use and maize production. Yet, suppose we learn further that Farm A has achieved a yield of 8 tons per ha, with a consumptive water use of 4000 m^3 per ha, while Farm B has achieved a yield of 10 tons per ha, with a consumptive water use of 5000 m^3 per ha. These are very different input levels and maize yields. The estimated water footprint of 500 m^3 per ton does not reveal these differences. Nor does it enable one to assess the efficiency of water use and crop production on either farm. It is not possible to know if the additional 1000 m^3 consumed in maize production on Farm B represents efficient water use, without knowing the opportunity cost of water, the price of maize, and the incremental costs of other inputs.

Suppose, in addition, we are told that the seasonal water footprint for Farm C, which is in the same location, is 556 m^3 per ha. Considering only the water footprint, it might seem to some observers that Farm C uses water less efficiently in maize production, than do Farms A and B, given the larger water footprint on Farm C. Yet, suppose Farm C's water footprint represents a maize yield of 9 tons per ha, with a consumptive water use of 5000 m^3 per ha (Table 1). This is more water use than that of Farm A, yet the yield also is larger. It is possible that Farm C's production generates a higher net return per ha than Farm A's production, if the value of the incremental yield is larger than the incremental cost of the additional water, including its scarcity value.

Farm C's outcome also might be preferred to that of Farm B. Suppose the lower yield at Farm C is due to the farmer's decision to plant her crop early in the season, to capture higher prices before the bulk of the regional maize crop arrives in the market. If so, her 9 tons of maize might have a higher market value than the 10 tons of maize produced on Farm B. Alternatively, perhaps less nitrogen was used on Farm C than on Farm B, resulting in a lower yield, but also a lower production cost, and possibly a smaller impact on water quality. None of this information is conveyed in the estimated water footprints. It is not possible to rank Farms A, B, and C in order of preferred input use or production outcomes, based only on the estimated average water consumption per ton of output.

Table 1. Examples of maize yields, consumptive water use, and the corresponding water footprints for three farms.

	Farm A	Farm B	Farm C
Water use (m^3/ha)	4000	5000	5000
Maize yield (t/ha)	8	10	9
Water footprint (m^3/t)	500	500	556

An additional, substantial concern regarding the comparison of water footprints, even within a river basin or irrigation district, is the inherent randomness of both the numerator and denominator. Crop yields and consumptive water use are random variables, such that water footprints are ratios of random variables, which generally do not follow Gaussian distributions (Marsaglia, 2006; Park, 2010; Díaz-Francés and Rubio, 2013; Khoolenjani and Khorshidian, 2013). Thus, typical test statistics are not sufficient for evaluating statistical significance. In addition, when random shocks influence both the numerator and denominator, and when one is also a function of the other (i.e. yield depends on ET, and ET also varies with yield), it is rather unclear just how one should interpret observed differences in the ratio, over time or across locations.

Given the inherent randomness in crop yields and consumptive water use, it would be difficult to predict the maize water footprints for Farms A, B, and C in a subsequent season. A perfectly plausible set of production outcomes might generate the crop yields and water use depicted in Table 2. In that scenario, Farm A has the smallest water footprint, while Farms B and C have larger water footprints. Yet, for the reasons described above, it is not possible to assess the efficiency of water use or maize production on any of the farms, using only the calculated water footprints. It is not possible to know which outcome is preferable, or to gain insight regarding the production issues or random shocks that have contributed to generating the observed crop yields and water use.

Table 2. Additional, plausible examples of maize yields, consumptive water use, and the corresponding water footprints for three farms.

	Farm A	Farm B	Farm C
Water use (m^3/ha)	4800	4400	5400
Maize yield (t/ha)	9	8	10
Water footprint (m^3/t)	533	550	540

Some observers might suggest that crop yields per hectare carry the same inadequacies as water footprints. However, there is a notable difference between the two ratios. As noted above, crop yields per hectare are, indeed, random variables. Yet, only the numerator in the ratio is random, while the denominator is not. A two-hectare field is essentially a two-hectare field, year after year, and across an agricultural region. There can be measurement errors and small variations in the actual number of square metres planted each year or season, but the size of the field generally is not subject to random shocks from weather, pests, or rainfall. Thus, the ratio describing crop yields per hectare is fundamentally different from the ratio describing a water footprint. The significance of observed differences in crop yields can be evaluated using standard test statistics.

Recent extensions and a further critique

Several authors have begun applying the notions of virtual water and water footprints in discussions of sustainability, efficiency, and equity. Given the nature of earlier critiques regarding both conceptual and empirical shortcomings, it seems reasonable to ask whether the notions of virtual water and water footprints contain sufficient information to enhance understanding and support policy recommendations regarding sustainability, efficiency, and equity. Several articles published recently provide the opportunity to examine this question.

Considering sustainability

Mekonnen et al. (2015) present estimates of national water footprints across Latin America and the Caribbean, with the stated goals of enhancing understanding of current water allocations and pollution in the region, and assessing the environmental sustainability, economic efficiency, and social equity of water use. The authors calculate water footprints pertaining to agriculture, industry, and domestic water supply, and they estimate the water required to produce goods that are traded internationally. The authors use the proportion of population considered to be under-nourished as a proxy for the equity of water allocation within a country, and they compare national rates of under-nourishment with their estimates of water footprints.

Water scarcity is not necessarily unsustainable

Mekonnen et al. (2015) consider sustainability from two perspectives: water scarcity and water pollution. Based on their estimates of water footprints and water supplies, the authors suggest that 3 of 77 river basins in the region are characterised by year-round 'severe water scarcity', while an additional 26 river basins would be considered severely water scarce in one or more months of the year. The authors define the severity of water scarcity as the degree to which the demands for surface water and groundwater exceed the available supply, on a monthly basis. Using that metric, more than 60% of the river basins in the region do not exhibit severe water scarcity in any portion of the year.

The ratio the authors construct for assessing water scarcity (an estimated water footprint divided by the available water supply) does not describe how the implied imbalances between water supply and demand arise, how they are resolved, or what impacts the imbalances have on households, communities, or the environment. In addition, it is not clear that the region should be characterised as water scarce, if only 29 of 77 river basins exhibit severe water scarcity in one or more months of the year. In aggregate, Latin America and the Caribbean are well endowed with water resources. In areas where persistent or periodic water scarcity exists in the region, local communities likely have developed measures for sustaining livelihoods and maintaining commerce through the dry periods. Water scarcity is largely a local issue requiring local attention and policy intervention, as needed.

One might also inquire whether water scarcity is indicative of a non-sustainable situation. Water has been scarce in the Middle East and North Africa for millennia, and likely will remain scarce for additional centuries. Living with water scarcity requires wise management, efficient practices, and appropriate policy interventions. Commerce and livelihoods can be sustained in areas where water is perpetually scarce. Some activities in water scarce areas, such as the continuous expansion of irrigated lands, excessive groundwater pumping, or the development of new residential or industrial sites, might not be sustainable, but simply noting that water demand exceeds the available supply in one or more months of the year does not enhance understanding of sustainability.

Chouchane et al. (2015) estimate water footprints of crop production in Tunisia, using information pertaining to 1996 through 2005. They report that the annual sum of groundwater and surface water footprints of crop production range from 290 million cubic metres (Mm^3) in central Tunisia to 650 Mm^3 in northern Tunisia. The sum of the estimated annual water footprints for central, northern, and southern Tunisia is 1330 Mm^3. The authors compare those estimates with estimates of renewable groundwater and surface water. In each region, the estimated water footprint is smaller than the renewable water supply. The estimated ratios of water footprints to water supply are 23%, 32%, and 78% in northern, central, and southern Tunisia, respectively. The authors characterise southern Tunisia as severely water scarce, while northern and central Tunisia are moderately and significantly water scarce, respectively. Considering only groundwater, all regions are characterised as severely water scarce, even though the estimated ratios of groundwater footprints to renewable groundwater supplies are 47% in the northern and central regions, and 62% for the country. Groundwater overdraft is indicated only in southern Tunisia, where the estimated ratio of withdrawals to supply is 123%.

The authors conclude that Tunisia suffers from 'significant water scarcity', even though the estimated annual national use of groundwater and surface water is just 31% of the estimated renewable supply (Chouchane et al., 2015). While groundwater overdraft might require policy attention in southern Tunisia, it is not clear that agricultural water use in any region of Tunisia is unsustainable, based only on the authors' calculation of regional water footprints. Perpetual groundwater overdraft is not sustainable, yet conjunctive use of groundwater and surface water can be sustained, if well managed.

As in the case of water scarcity and water quality, discussed above, the sustainability of water resources also must be considered locally. One must consider the sum of competing demands, the outlook for growth in those demands, and the outlook for water supplies. For example, saline intrusion can impair the sustainability of groundwater pumping in localised areas, even if water use appears to be sustainable in the larger region or at the national level (Garcia-Ruiz et al., 2011; Uddameri et al., 2014; Ngo et al., 2015).

As described in the literature, sustainability involves resilience, adaptability, and learning (Anderies et al., 2013; Rist and Moen, 2013; Sidle et al., 2013). Thus, sustainability analysis requires consideration of dynamics and uncertainty, and some sense of the social and economic objectives for a given river basin and region (Bacon et al., 2012; Ni et al., 2012; Prosperi et al., 2014; Duić et al., 2015; Salvati and Carlucci, 2015). Water footprints are simply ratios created by dividing some measure of water input (water applied or transpired) by some measure of crop production (output or value) or some other activity. As such, water footprints are not fit for the task of evaluating the complexity of issues that must be considered when evaluating sustainability (Wichelns, 2010a, b, 2011c, 2015).

Sustainability analysis must allow also for the pulsing of dynamic systems in response to changes in resource conditions and environmental stresses. Rainfall and transpiration in any river basin are random variables, and the patterns describing these variables can change, over time. In many areas, in years when rainfall is less than average, groundwater serves as a replacement or supplemental source of water for irrigation and domestic use. Overdrafting aquifers in dry years, while allowing them to regain water volume in wet years, when managed appropriately, is a sustainable strategy that can generate substantial value. Conjunctive use programs, managed aquifer recharge, and water banking are common in many water-scarce areas (Maliva, 2014a, 2014b; Megdal et al., 2014; Singh, 2014). Estimates of water footprints, which at best provide only a snapshot of water use at a given point in time, cannot capture or portray the dynamics involved in the optimal, conjunctive use of groundwater and surface water over long periods of time. Yet, such strategies have much to do with truly achieving sustainability.

'Grey water footprints' do not properly characterise water pollution

Regarding water pollution, Mekonnen et al. (2015) calculate 'grey water footprints', a construct defined in an earlier publication (Hoekstra et al., 2011), as a proxy for describing the degree to which the ambient concentrations of nitrogen and phosphorus differ from water quality standards (Chapagain and Tickner, 2012). As described by the authors, grey water footprints involve the volumes of freshwater required for dilution to achieve ambient concentrations of pollutants that are within acceptable or regulated ranges. Yet, the authors provide no evidence that dilution is the current or preferred method for reducing water pollution in any of the countries they consider. Thus, the estimates of water volumes required to dilute ambient pollution, labelled by the authors as grey water footprints, do not represent actual water demands or water use. They also do not contain information describing the causes, extent, or possible remedies for water pollution in any country. Suggesting that the estimated grey water footprint for the Salado River Basin in Argentina is 1541 Mm^3 per year, while that for the Magdalena River Basin in Colombia is 4500 Mm^3 per year (Table 2 in Mekonnen et al., 2015), does not provide insight regarding the sources of pollution, the impacts on natural resources or communities, or the incremental costs and benefits of efforts to reduce pollution. In addition, the grey water footprints are calculated with no consideration of the seasonal variability in stream flows and in the loads of pollutants discharged from various sources. It is not likely that the loads and concentrations of pollutants, or the impacts of water quality degradation, are constant across or within seasons.

Chenoweth et al. (2014) and Launiainen et al. (2014) also find grey water footprints to be inappropriate descriptors of water pollution, as they do not account for any downstream impacts of water quality degradation. Perry (2014) notes that water quality requirements and impacts will vary with water uses. In-stream fisheries, irrigators, aquatic wildlife, and hydropower plants have different water quality requirements, which are not considered in the calculation of grey water footprints. Morrison et al. (2010) consider grey water footprints to be theoretical, as they do not involve actual water volumes. In their view, summing grey water footprints with estimates of green and blue water footprints is misleading and of little value. Gawel and Bernsen (2011b) share the view that grey water footprints are fictional measures of water pollution, noting also that most wastewater is treated, rather than diluted.

Considering efficiency

Mekonnen et al. (2015) suggest that improvements in the efficiency of rainfed agriculture are needed, in part, to reduce the extent of irrigated agriculture in water scarce portions of Latin America and the Caribbean. Yet, there is no inherent trade-off involving rainfed and irrigated agriculture. Both forms of crop production have increased in recent years in the region, in response to increasing demands for crop, livestock, and biofuel products in domestic and international markets. In Brazil, the area equipped for irrigation increased by 6.4% between 2000 and 2012, while the much larger, rainfed area increased by 4.7% (FAOSTAT, 2015). It is likely that both irrigated and rainfed areas will continue increasing in future, subject to continuing increases in market demands, and subject also to the availability of affordable land and water resources and other productive inputs.

Improving the efficiency of rainfed production will not necessarily lead to a reduction in either rainfed or irrigated agriculture. Rather, increases in efficiency might lead to an expansion in rainfed production, given its improved efficiency, while having little or no impact on the growth rate of irrigated production. Regional economics, international markets, policy initiatives, and resource constraints will determine the growth trajectories of rainfed and irrigated agriculture in Latin America and the Caribbean. Estimates of water footprints contain too little information to enhance understanding of the numerous issues and parameters that influence farm-level production and marketing strategies, regional economic development, and natural resource management.

Mekonnen et al. (2015) suggest that "if all countries in Latin America and the Caribbean would reduce the green-blue water footprint of crop production to the level of the best 25th percentile of current global production, the water saving in [regional] crop production would be about 37% compared to the reference water consumption". Furthermore, the authors state that

> if every Latin American and Caribbean country would reduce the nitrogen-related grey water footprints in crop production to the level of the best 25th percentile of current global production, water pollution related to crop production in [the region] would be reduced by 44% compared to the current situation.

These perspectives are somewhat difficult to interpret, given the great diversity in crop production systems, natural resource endowments, technology, and incomes worldwide. It is difficult also to find pertinence in the notion of global benchmarks pertaining to water use or water pollution in agriculture. In addition, this discussion seems misplaced under the heading of efficiency in Mekonnen et al. (2015), as the statements regarding water use and water pollution benchmarks do not reflect any analysis of pertinent economic considerations. Furthermore, the notion of reducing water use by 37% is offered without considering the potential impacts on crop and livestock production or on the livelihoods of those engaged in agriculture. In a region that is generally well endowed with water resources, the motivation for reducing water use by 37% is not evident.

The notion of using global water footprints as benchmarks for water use and water pollution is not supported by an evident conceptual framework or a compelling empirical rationale. The social and economic aspects of water allocation and use vary substantially across regions, with differences in water availability, production opportunities, incomes, and livelihood status. One cannot evaluate the appropriateness of a given water allocation programme without knowing the opportunity costs of water in a given setting. Those costs will be different for a farmer in Jordan than a farmer in Honduras, or an urban household in Nepal. Given large differences in the inherent availability of water, and in opportunity costs and scarcity values, it is not meaningful to consider a 'global average water footprint' that might serve as a benchmark for evaluating water allocation decisions. Such decisions must be made locally, within regions, and with due consideration of local and regional scarcity values.

Mekonnen et al. (2015) also calculate average values of crop and livestock products sold in domestic and international markets. Such estimates of dollar-denominated water footprints are not enlightening or useful in evaluating efficiency. One should expect to observe higher average values in the production of fruits and vegetables, than in the production of cereals, yet each production activity has its place in a given market setting. It is not useful or correct to consider shifting land and water resources from cereal production to vegetables, in search of higher valued water footprints. Such metrics represent average calculations, and thus they do not pertain in discussions of economic efficiency, which require consideration of incremental costs and benefits. In addition, as the Latin America and Caribbean region is relatively well endowed with water resources, optimal cropping patterns will not be those that maximise either dollar-denominated or output-denominated estimates of water footprints. Market prices for inputs and outputs, the prices and availability of non-water inputs, access to markets, technological advances, and risk management strategies will guide smallholders and larger scale producers toward optimal cropping patterns.

Chouchane et al. (2015) suggest that they examine the water footprint of Tunisia from an economic perspective. The authors multiply their estimates of green and blue water footprints in agriculture by producer prices, to obtain ratios they characterise as the 'economic water productivity' of green and blue water in rainfed and irrigated agriculture. There are two notable shortcomings in the authors' analysis: 1) Multiplying a ratio describing the average amount of output per unit of a single input, by output price, does not constitute economic analysis, and 2) Estimating blue and green water productivities in irrigated agriculture in additive fashion is inappropriate. Water footprints are ratios of output to the amount of a single input used in production. Thus, they represent the average productivity of a single input. When multiplied by output price, the new ratio represents average revenue. Economic analysis requires consideration of incremental revenues and costs, rather than averages. Water footprints represent averages, rather than increments, and thus they are unsuitable for economic analysis, even if multiplied by producer prices.

The procedure used by Chouchane et al. (2015) to estimate green and blue water productivities in irrigated agriculture is depicted in their Figure 1. The authors essentially consider the impacts of rainfall and irrigation on crop yields to be sequential and separable; i.e. rainfall generates a certain yield per hectare, which can be augmented by applying irrigation water. That framework is inconsistent with the way in which crop water use and plant growth occur during a season. Rainfall and irrigation generally do not occur in fully separable sequence. Rather, they occur most often in repeated sequences, with irrigation taking place between rainfall events. For some crops, farmers apply a pre-irrigation before planting, and they might irrigate-up the small plants. Additional irrigations are applied, as needed, depending on the size and timing of rainfall events during the season. In such a setting, the incremental productivity of irrigation is a function of soil moisture, plant status, weather, and rainfall dynamics. Similarly, the incremental productivity of rainfall is a function of soil moisture, plant status, weather, and irrigation dynamics. The two incremental productivities are intertwined and inseparable. The estimates of green and blue water productivities in Chouchane et al. (2015) are not appropriate for addressing issues of economics or water use efficiency. The more important question to address in irrigated agriculture is how to optimise irrigation deliveries, given the opportunity costs of water and the stochastic nature of rainfall. Water footprints are not suitable for conducting such analysis.

Schyns and Hoekstra (2014) also endeavour to examine economic efficiency, using estimates of green and blue water footprints pertaining to crop production in Morocco. Using the same dataset as used by Chouchane et al. (2015) for 1996 through 2005, the authors estimate 'economic water productivities' by multiplying estimates of water footprints by output price. Although such ratios are not appropriate for economic analysis, the authors conclude that "from a strictly water-economics point of view, it would be worthwhile to reconsider which crops to grow in Morocco (due to the low value in US$/m^3 and US$/ha for some crops compared to others)". Based largely on their estimates of water footprints, Schyns and Hoekstra (2014) suggest further that Morocco could save substantial volumes of water by relocating crop production activities across the country's river basins. In particular, the authors suggest that the country can save 939 Mm^3 of water annually by moving all maize production to the Moulouya river basin, where the estimated average water footprint for 1996 to 2005 is the smallest. Such a move would increase maize production in Moulouya from 838 ha in the base period to more than 220,000 ha in the water saving scenario. The authors do not discuss the potential implications on livelihoods of such a large change in cropping patterns or the substantial additional risk inherent in producing all of the country's maize in a single river basin. Nor do they consider the fact that farmers must evaluate many agronomic and economic factors when selecting the crops they produce.

Schwarz et al. (2015) propose that the trade of goods and services can be evaluated in terms of the economic efficiency of water used in production. The authors define the 'economic water efficiency' of imports and exports as the "average amount of money spent per unit of virtual water inflow into a region" and the average amount of money earned per unit of virtual water outflow, respectively. They suggest that the "the higher the economic water efficiency of trade, the more financial value is being generated per unit of virtual water associated with a traded product". This framework might suggest to some observers that importers engage in trade to obtain virtual water, and that exporters consider the revenue they earn per unit of water used in production. As noted in earlier critiques of the notion of virtual water trade, there is no compelling rationale and no empirical evidence that such strategies are pertinent (Wichelns, 2004, 2011a, b). In addition, simply dividing the dollar value of imports and exports by the amount of a single input used in production does not provide insight regarding economic efficiency. One needs to know much more about the comparative advantages of producing goods and services in the trading countries. One might also consider that financial values do not necessarily reflect the full economic values of trade in goods and services.

Considering equity

Mekonnen et al. (2015) divide their estimates of water footprints by national populations to obtain estimates of water footprints per capita, and they calculate the average values of commodities produced per unit of water consumed in production. They also compare estimates of water footprints and water scarcity indicators for 20 river basins that together span more than 10,000 km^2 across the region. However, estimates of water footprints per capita provide no insight regarding individual, household, or community welfare. They convey no information regarding a household's access to water, the income of a household, the health status of its residents, or a household's food security status, either annually or seasonally. It is not possible to know if individuals or households in Bolivia are better-off or worse-off than individuals or households in Nicaragua, by considering only the authors' estimates of the average consumer water footprint in each country. Lacking a supporting conceptual framework, it is not clear if higher or lower water footprints are better, or if achieving more or less variation in water footprints would be a desirable policy goal.

Mekonnen et al. (2015) suggest that water allocation in Latin America and the Caribbean is inequitable, in part, because the average water footprint of consumers in the region is 28% larger than the global average water footprint. To many observers, this observation might seem plausible, given that the region is home to 10% of the global population and receives 30% of global rainfall (FAO, 2015; Flachsbarth et al., 2015). In a region that receives more rainfall per capita than other regions of the world, one might reasonably expect higher estimates of consumer water footprints.

The notion of a global average water footprint, described by Mekonnen et al. (2015), is not a meaningful benchmark or target. Water endowments vary substantially within countries and across regions, as do production conditions, employment opportunities, input availability, and access to markets. One should expect that estimated water footprints will reflect the considerable variation in these explanatory factors (Gawel and Bernsen, 2011a). It is likely that the estimated water footprints in a water abundant region, such as Latin America and the Caribbean, will be higher than the estimated water footprints in a much drier region, such as the Middle East and North Africa. They likely will be higher also than the estimate of a global average water footprint.

Estimated water footprints also vary widely within the region, ranging from 912 m^3 per person per year in Nicaragua to 3,468 m^3 per person per year in Bolivia (Mekonnen et al., 2015). However, this is not a meaningful comparison. Consumer water footprints are simply ratios depicting the estimated average amount of water required to produce an array of goods and services. Such ratios do not describe the incremental costs or benefits of production or consumption, or provide information regarding household or community welfare. Water certainly plays a role in urban and rural livelihoods, yet water footprints do not address critical issues that actually influence equity, such as reliable access, price, water quality, gender, land tenure, and the availability of complementary inputs that enable households to use water productively (van Houweling et al., 2012; Veldwisch et al., 2013; Roa-García, 2014).

Mekonnen et al. (2015) briefly discuss the notion of a 'fair share' water allocation, as if to suggest that consumer water footprints should be similar within and across countries. It is not clear that such a notion is appropriate or feasible, or that achieving similar water footprints would enhance sustainability, efficiency, or equity. Water footprints are simply calculations of the estimated average amounts of water required to produce goods and services. Such estimates do not pertain to equity. In addition, there is no inherent, policy relevant inequity in the naturally uneven distribution of water resources across regions.

Hoekstra (2014) also proposes the notion of a global 'fair water footprint' share per community that should be derived in accordance with the "limited maximum sustainable water footprint per global citizen". The author suggests that consumers in the United States and southern Europe "use nearly two times more water than the global average". That statement likely reflects differences in annual incomes per capita and the purchase of goods and services, rather than direct water consumption (Gawel and Bernsen, 2013). Differences in income and welfare have more to do with economic systems, institutions, governance, and opportunities, than the notion of a water footprint. It is not clear that one could or should attempt to equate water footprints across individuals, countries, or regions, in the interest of improving equity.

Hoekstra (2014) suggests also that to achieve an "equal water footprint share for all global citizens", China and India would need to reduce their current water footprints per capita by about 22% during the

21st century. The United States would need to reduce its average water footprint per capita by 70%. The rationale for such suggestions is not evident. Many of the impoverished, rural residents of China and India likely need to achieve a larger water footprint, rather than a smaller one. They need to produce more goods and services, and they need to consume more food and nutrition. Areas of water scarcity and excessive withdrawal of surface water and groundwater in China and India are well known, and pertinent policies and investments are needed to achieve sustainable use. Yet, a national effort to reduce the aggregate water footprint, in the interest of achieving some notion of a global average standard would be misguided and potentially quite harmful to millions of poor households. Similarly, an effort to reduce the aggregate water footprint in the United States by 70% could have substantial negative economic impacts across the country and among trading partners.

In the context of their discussion of equity, Mekonnen et al. (2015) consider the extent of under-nourishment in Latin America and the Caribbean. Yet, one might expect to observe very little correlation between estimates of national water footprints and the extent of under-nourishment in each country. Indeed, the authors find no statistically significant relationship. Notable areas of under-nourishment exist in the region, even where water is abundant and aggregate food production is substantial. Under-nourishment and other forms of food insecurity are due largely to persistent poverty, which prevents households from purchasing the food and nutrition needed to support their health and welfare (Misselhorn, 2005; Dávila, 2010; Naylor and Falcon, 2010; FAO, 2013). Food security and livelihood status vary with differences in access to education, health care, finance, and income earning opportunities. Estimates of consumer water footprints do not convey information describing those attributes. Thus, estimates of water footprints generally will not enhance understanding of food and nutritional insecurity.

CONCLUSIONS

Many authors have calculated the water footprints of agricultural and industrial activities, and many have described the 'flows of virtual water' between countries and between sub-national units, such as provinces, states, and counties. Many of the authors promote reducing water footprints by re-locating production activities or investing in water conserving technologies, in the interest of improving water use efficiency. Some authors recommend revising international trade guidelines to account for the water used to produce goods and services, to save water globally. Others propose a product labelling or certification programme pertaining to water footprints, thus promoting comparison shopping by consumers. Many of these proposals might seem appropriate at first glance, yet they are based on a rationale that lacks conceptual and empirical support.

The notion of virtual water is not consistent with the economic concept of comparative advantage, and thus it is not an appropriate criterion for determining optimal trading strategies. Importers and exporters must consider many variables when engaging in trade, rather than focusing on a single input used in production. Water footprints also contain limited information and they represent an average, rather than incremental, calculation. Thus, water footprints are not appropriate for determining the optimal use of land and water resources, selecting investment strategies, or allocating water among competing uses. Additional information is required, regarding the opportunity costs of water and other inputs, and the potential impacts of any changes in land and water allocation on livelihoods.

In recent years, several authors have begun extending the notions of virtual water and water footprints to analyses involving sustainability, efficiency, and equity. However, it is not clear that the notions are suitable for such analysis. Sustainability is a complex idea that involves resilience, adaptability, and learning. Sustainability analysis must allow also for the pulsing of dynamic systems in response to changes in resource conditions and environmental stresses. The notions of virtual water and water footprints contain too little information to support such analysis. Efficiency requires that the incremental gains are equated with incremental costs. Water footprints represent the average amount of water per unit of output, and thus are not suitable for determining an efficient water allocation. Equitable outcomes must account for the impacts on livelihoods, which are not considered in estimates of virtual water or water footprints.

In sum, the notions of virtual water and water footprints are not suitable for enhancing understanding of sustainability, efficiency, or equity. The notions contain too little information and they represent averages, rather than increments, which are required for determining optimal strategies. Additional information regarding market demands and supplies, resource conditions, opportunity costs, and livelihood impacts in local and regional settings is needed to assess sustainability, efficiency, and equity. Meaningful assessments require the application of legitimate analytical frameworks involving a wide range of perspectives from the social and physical sciences, with due consideration of dynamics, uncertainty, and the impacts of policy choices on livelihoods and natural resources.

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