CHAPTER

4

Water Footprint Assessment: towards water-wise food systems

Joep F. Schyns[a], *Rick J. Hogeboom*[a,b] *and*
Maarten S. Krol[a]

[a]**Multidisciplinary Water Management Group, Faculty of Engineering Technology, University of Twente, the Netherlands** [b]**Water Footprint Network, the Netherlands**

4.1 Introduction

Freshwater is both a vital agricultural input to grow crops and raise animals as well as a sink for agricultural outflows, such as nutrient or pesticide runoff, that affect water quality. As such, food systems put significant pressure on global freshwater resources, both in terms of water quantity and water quality (Hoekstra and Mekonnen, 2012; Mekonnen and Gerbens-Leenes, 2020).

Freshwater is a renewable resource, but its availability is limited. Freshwater availability stems from precipitation over land, which differentiates into a blue water flow – runoff via groundwater and surface water – and a green water flow – rainfall that infiltrates the soil or is intercepted by vegetation and eventually flows back to the atmosphere as evapo(transpi)ration (Schyns, Hoekstra, Booij, Hogeboom, and Mekonnen, 2019). Every year, people claim parts of these blue and green water flows for use at home (blue), in industry (blue), in agriculture (mostly green; also blue) and forestry (mostly green; also blue). These claims should not exceed local annual replenishment rates. After all, the use of blue and green water flows subtracts from the freshwater availability to ecosystems in water and on land, which also require minimum environmental flows for their subsistence. Therefore, there are limits to the sustainable freshwater use by humanity which are lower than the annual replenishments rates of blue (Vörösmarty et al., 2010) and green water (Rockström and Gordon, 2001; Schyns et al., 2019).

4.1.1 The water footprint concept

To measure humanity's pressure on freshwater resources, the water footprint (WF) concept was coined by the late professor Arjen Y. Hoekstra in 2002 (Hoekstra, 2003). The concept is

DOI: https://doi.org/10.1016/B978-0-12-822112-9.00006-0

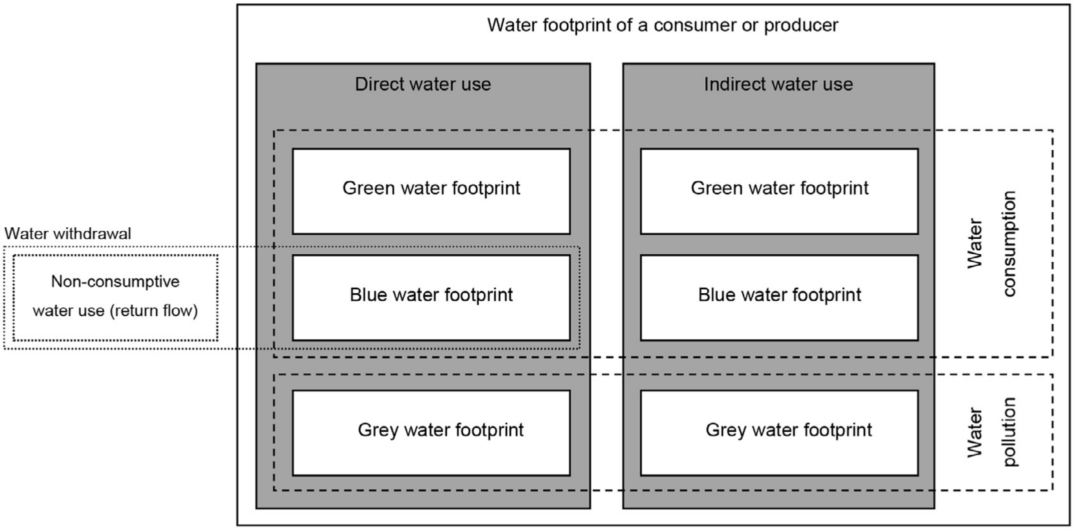

FIGURE 4.1 Schematic representation of the components of a water footprint. It shows that the non-consumptive part of water withdrawals (the return flow) is not part of the water footprint. It also shows that, contrary to the measure of 'water withdrawal', the 'water footprint' includes green and grey water and the indirect water-use component. Credit: From Hoekstra et al. (2011), Copyright © Water Footprint Network 2011. Reproduced with permission of Taylor & Francis Group through PLSclear.

inspired by the concept of 'ecological footprint' (Wackernagel & Rees, 1996) and is built upon a number of important notions (Hoekstra, 2017): (i) freshwater is a *global* resource, because goods produced with freshwater can be traded, resulting in virtual water trade, such that people can benefit from freshwater resources elsewhere (Allan, 1998); (ii) to comprehensively measure human pressure on freshwater systems, one must consider both *green* and *blue* water consumption (Falkenmark, 2000) and water *pollution* (Postel, Daily, and Ehrlich, 1996); (iii) *supply-chain thinking* is needed to link human consumption patterns to impacts on freshwater resources. Subsequently, the concept has been further developed into a multidimensional indicator of human appropriation of freshwater consumption and pollution in volumetric terms, which is embedded into a broader assessment framework: Water Footprint Assessment (WFA) (Hoekstra, Chapagain, Aldaya, and Mekonnen, 2011). The WF concept and WFA methodology have experienced wide uptake and led to the evolution of a new research field (Hoekstra, 2017).

The WF is an indicator of environmental pressure: it measures the human appropriation of freshwater. The WF has three components – green, blue, and grey – that are each specified geographically and temporally (Fig. 4.1). The green and blue WF measure consumptive use of green and blue water flows, respectively. Consumptive use in WF terms refers to water that is "lost" from the system, and that therefore cannot be used for other purposes at that particular time at that particular location (Hogeboom, 2020). These "losses" include the evaporation of water, water contained in products, or water that is transferred to another system, such as another river basin. The grey WF refers to the volume of water that is needed to assimilate

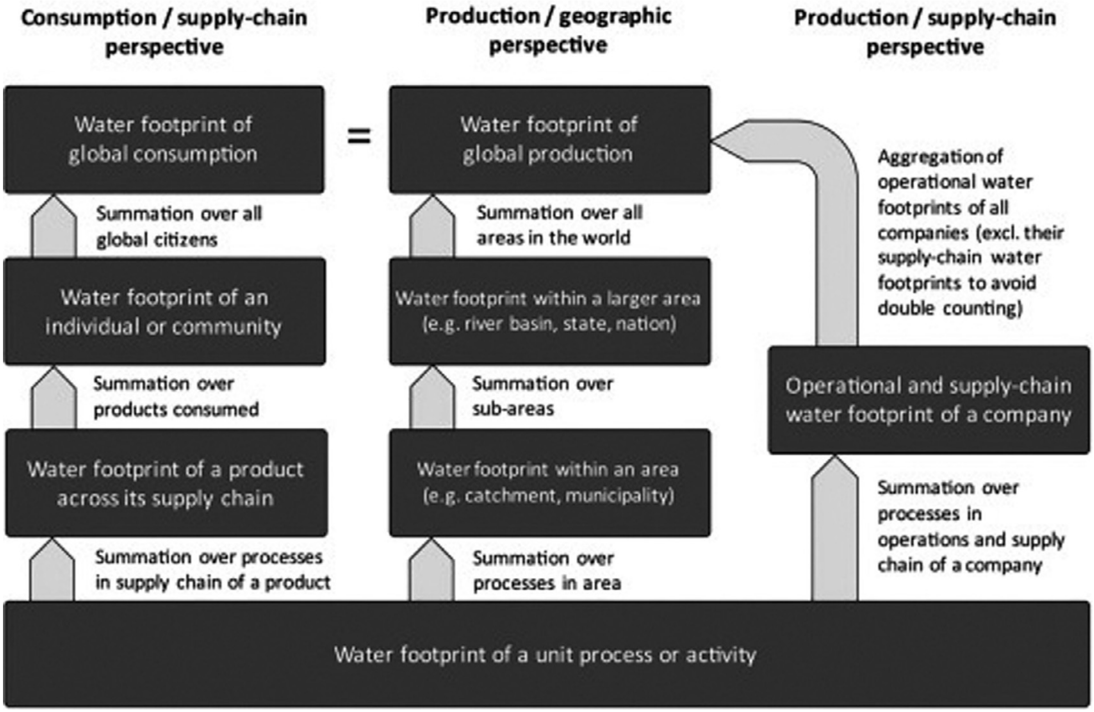

FIGURE 4.2 Water footprints of single processes or activities form the basic building blocks for the water footprint of a product, consumer, or producer or for the footprint within a certain geographical area. The footprint of global consumption is equal to the footprint of global production. Credit: Reprinted from Hoekstra (2017).

pollutants associated with a particular activity to meet ambient water-quality standards (more in Section 4.3.1.4). The WF thus measures both the appropriation of freshwater as a natural resource, via the green and blue WF, and the appropriation of freshwater as an agent to assimilate waste, via the grey WF (Hogeboom, 2020). The WF is not restricted to direct water use only, but also includes indirect water use in the supply chain. As such, the WF uncovers the link between the consumption of goods in one place and water use in another place.

WF accounts can be made for a range of different entities, such as the WF of a product, a consumer, a business, or within a certain geographic region. The building blocks of any WF account are the WFs of single processes or activities, which are mutually exclusive (see Fig. 4.2). WF accounts give spatiotemporally explicit information on how water is appropriated for various human purposes, which can feed the discussion on sustainable, efficient and equitable water use and allocation. To facilitate this, the WFA framework follows four main phases (Hoekstra et al., 2011): (i) setting the goals and scope of the assessment; (ii) WF accounting, in which data are collected and accounts are developed; (iii) WF sustainability assessment, in which the environmental sustainability, efficiency and equitability of the accounted WF are evaluated; and (iv) WF response formulation, in which response options, strategies or policies are developed.

4.1.2 The position of this chapter in WFA literature

A rich body of literature on WFA exists. The core consists of the WFA manual (Hoekstra et al., 2011), which is a technical reference containing the Global Water Footprint Standard, with definitions and calculation methods. The book *'Water Footprint of Modern Consumer Society'* (Hoekstra, 2020) is described by its author as a companion to the WFA manual. Hoekstra (2020) uses improved conceptual frameworks and insights from scientific publications to describe global risks of freshwater consumption and pollution and possible response options, taking a holistic perspective. For grey WF accounting, the flagship report by Franke, Boyacioglu, and Hoekstra (2013) provides additional guidelines to those laid out in the WFA manual. Next to these key references, there is a large and growing body of scientific literature that apply the methods and terminology of WFA (Hoekstra, 2017; Mubako, 2018; Zhang, Huang, Yu, and Yang, 2017).

This chapter focuses on the WF of food systems, which play a dominant role in the WF of many economies. Agricultural production contributes 92 percent to the WF of humanity (Hoekstra and Mekonnen, 2012). Trade in commodities, which have a WF associated with their production, results in virtual water flows. These international virtual water flows more than doubled from 1986 to 2008 (Carr, D'Odorico, Laio, and Ridolfi, 2012). Around the year 2000, 88 percent of these virtual flows were related to trade in agricultural commodities (Hoekstra and Mekonnen, 2012). The WF related to the consumption pattern of individuals is largely determined by the food they consume (Hoekstra and Mekonnen, 2012; Vanham, Mekonnen, and Hoekstra, 2013b). For the average consumer in the European Union, 74 percent of their consumptive WF lies in the food sector, 26 percent in the energy sector, and only a minor fraction in the water sector itself, relating to direct use of water at home (Vanham, Medarac, Schyns, Hogeboom, and Magagna, 2019).

This chapter is structured according to the major phases identified by the WFA framework (Section 4.1.1), so that Section 4.2 focuses on WF accounting, Section 4.3 on WF sustainability assessment, and Section 4.4 on WF response formulation. In Section 4.2, we describe how to account the green and blue WF of growing a crop. In Section 4.3, we summarize methods to assess the environmental sustainability, efficiency and equitability of the WF of food systems. Lastly, in Section 4.4, we discuss possible response strategies of different actors in the society to transform current food systems towards more water-wise food systems.

4.2 Accounting the consumptive water footprint of growing a crop

This section focuses on accounting the WF of growing a crop and omits the WF of other stages in the food chain, such as the processing, packaging and distribution. Studies on the WF of specific food and beverage products have shown that the majority of the WF is in the stage of crop production (Aldaya & Hoekstra, 2010; Ercin, Aldaya, & Hoekstra, 2011, 2012). Also for farm animal products (Mekonnen and Hoekstra, 2012) and fish from aquaculture production (Pahlow, van Oel, Mekonnen, and Hoekstra, 2015), the largest part of the WF relates to growing the crops which are used as animal and fish feed. Furthermore, the WF of growing a crop is an essential building block in calculating, for example, the WF of the agricultural sector in a nation (Hoekstra and Mekonnen, 2012) or the WF of alternative diets (Vanham, Comero, Gawlik, & Bidoglio, 2018; Vanham, Hoekstra, & Bidoglio, 2013a; Vanham, Mekonnen, & Hoekstra, 2013b).

The consumptive WF accounts for the water lost to the atmosphere in the process of growing a crop. It includes all (green plus blue) freshwater that evaporates from the crop field or transpires through the crop's leaves and is, therefore, no longer available to be consumed for other purposes within that area and part of the year. The WF of growing a crop differs largely from place to place, depending on differences in climate, agricultural practices, and the type of crop. For example, the global average green plus blue WF of growing a crop increases from sugar crops (182 m^3 of water per tonne of crop), vegetables (237 m^3/tonne), roots and tubers (343 m^3/tonne), fruits (874 m^3/tonne), cereals (1,460 m^3/tonne), oil crops (2,243 m^3/tonne), pulses (3,321 m^3/tonne), spices (6,616 m^3/tonne) to nuts (8,383 m^3/tonne) (Mekonnen and Hoekstra, 2011).

In layman's terms, the consumptive WF measures the inverse of "crop per drop." It is calculated as the sum of the green and blue crop water use (*CWU*, in m^3 of water per ha) divided by the harvestable crop yield (*Y*, in tonne of crop per ha) and measured in m^3 of water per tonne of crop (Hoekstra et al., 2011). Crop water use refers to the accumulation of daily actual evapotranspiration from the crop field (*ET*, in mm per day) over the length of the growing period (*lgp*, in days). See Eqs. (4.1) and (4.2) below, in which the factor 10 is used to convert mm per day into m^3 per ha (Hoekstra et al., 2011).

$$WF = \frac{CWU}{Y} \tag{4.1}$$

$$CWU = 10 \times \sum_{d=1}^{lgp} ET \tag{4.2}$$

Technically, the water content in the harvested crop is also part of the WF of a crop, but in practice this part is often neglected since its order of magnitude is typically less than one percent of the crop water use (Hoekstra et al., 2011).

The consumptive WF of a crop is often modeled rather than measured. Although *ET* from a crop field can be measured in the field this process is laborious and costly, and therefore unusual. This is especially true when the scope of interest is larger than a number of specific locations or time frames for which measuring equipment can be installed and monitored, which is the case for many research questions in WFA. For example, how does the WF of a crop vary across the globe? (Mekonnen and Hoekstra, 2011, 2014); how has the WF in a region changed over time? (Zhuo et al., 2016b; Zhuo, Mekonnen, Hoekstra, and Wada, 2016c); how does the WF of a crop respond to a range of different agricultural management practices? (Chukalla, Krol, and Hoekstra, 2017). Therefore, *ET* is generally estimated with models that account for variables and relations in the atmosphere-plant-soil continuum. Crop yields are commonly measured in the field and are reported on at the level of administrative units, such as the province or country. Still, modeling of crop yields is often desired, for example when exploring the effects of climate change or alternative agricultural management strategies.

We describe here how to estimate the consumptive WF of a crop with the use of soil water balance models (Section 4.2.1) or crop models (Section 4.2.2), which has become the new standard in WFA. Also, we explain why and how to distinguish between the green and blue WF of a crop (Section 4.2.3).

4.2.1 The use of soil water balance models for crop water footprint accounting

In the early stages of the development of the field of WFA (2002–14), studies estimated the green and blue WF of growing a crop according to the guidelines by Allen, Pereira, Raes, and Smith (1998) using a soil water balance model and crop coefficients (Hoekstra et al., 2011), and leveraging the strong empirical relationship between evapotranspiration and crop yield. The approach can be summarized in three steps. Step 1 estimates the potential *ET* of a crop given unlimited access to water. Step 2 determines the actual *ET* based on actual soil water availability during the growing season. Step 3 calculates crop yield based on the difference between actual and potential *ET* and the crop's yield response to water shortage (Doorenbos & Kassam, 1979). In this approach, the crop factor and the rooting depth vary over the growing season, but are pre-described inputs to the soil water balance model. Actual crop growth is not simulated by the model itself. The approach has been applied in the key publication on the WF of crops and derived crop products by Mekonnen and Hoekstra (2011), which has served as a basis for many subsequent studies.

4.2.2 The use of crop models for crop water footprint accounting

Alternatively, the consumptive WF of growing a crop can be estimated with crop models that simulate both crop growth and the soil water balance. Such crop models are more process-based, meaning that higher level responses (like crop transpiration and biomass growth) are determined by lower level simulated processes (like root growth and rate of photosynthesis). These processes represent biophysical relationships between the atmosphere, the crop and the soil, and often influence each other via feedback loops. Therefore, crop models more accurately represent reality and are better suited to analyze particular responses, like the yield response to irrigation water applied, compared to the soil water balance approach. Another advantage is that these models often include possibilities to simulate the effect of alternative irrigation (irrigation scheduling and application technique) and field (mulching, tillage, bunds) management options on the WF of a crop. It is for these reasons that in recent years, crop modeling has become the new standard in WFA (Chouchane, Krol, & Hoekstra, 2018b; Chukalla, Krol, & Hoekstra, 2015, 2017, 2018b; Gobin et al., 2017; Hogeboom & Hoekstra, 2017; Karandish, Hoekstra, & Hogeboom, 2018; Masud, McAllister, Cordeiro, & Faramarzi, 2018; Masud, Wada, Goss, & Faramarzi, 2019; Nouri, Stokvis, Galindo, Blatchford, & Hoekstra, 2019; Zhuo, Mekonnen, Hoekstra, & Wada, 2016c). The disadvantages of crop models are the higher input data requirements and computational demands, which makes their use for high spatial resolution modeling at large geographic scales (countries, river basins, global) challenging.

Crop models can be categorized in three groups, depending on the main mechanism that drives biomass accumulation (Steduto, 2006) namely: carbon-driven, solar-driven, and water-driven crop models. Carbon-driven models, for which growth is based on the carbon assimilation by the photosynthetic process, are highly mechanistic and simulate all the main biophysical processes and feedback loops that translate incoming solar radiation into biomass (Steduto, 2006). Solar-driven models simulate biomass accumulation directly from the intercepted solar radiation through a single coefficient, the radiation use-efficiency, which synthetically incorporates the underlying biophysical processes (Steduto, 2006). Water-driven models simulate biomass accumulation based on accumulated crop transpiration through a single coefficient, the water-use efficiency or biomass water productivity (Steduto, 2006). Although biomass is produced through photosynthesis, the relation between transpiration and photosynthesis is so close that it can be captured in a single coefficient when normalized for climatic conditions (Steduto, Hsiao, Raes, and Fereres, 2009).

The choice for a crop model to use will depend on one's objective and available means. Models differ in input data requirements and computational resource needs. In the field of WFA, the main interest is often the crop's response to water. For this purpose, the water-driven crop growth model type is particularly suited, because the crop's response to water is at the core of the model and the relationship between transpiration on biomass growth has proven to be robust, even under water stress conditions (Steduto, 2006). For this reason, the AquaCrop model (Steduto et al., 2009) has often been applied in WFA studies. The downside of the AquaCrop model is that it does not simulate a nutrient cycle, which impairs its use to assess the trade-offs between the consumptive (green and blue) WF and the grey WF related to fertilizer or pesticide application. For such assessments, other models such as the APEX model are needed (Chukalla et al., 2018b; Chukalla, Krol, and Hoekstra, 2018a).

4.2.3 Distinguishing between green and blue crop water use

Because green and blue water differ in terms of possibilities for storage and use, it has become common practice to distinguish between green and blue crop water use (Hoekstra, 2019a). Making this distinction is non-trivial, because it is not 'visible' which part of modeled *ET* originates from rainwater and which part from irrigation water or capillary rise. Hoekstra (2019a) describes a generic and physically-based method to differentiate green and blue soil evaporation (*E*) and crop transpiration (*T*) that is depicted in Fig. 4.3. The method demands a systematic accounting of the fractions of green and blue water in the different soil and vegetation (referring to water intercepted by vegetation) layers on a daily basis, as a basis to estimate green and blue water fractions in all fluxes leaving each layer. The input for this method – the daily water fluxes in each layer – is often the output of a crop model simulation, such that the green-blue accounting can be done in a post-processing step (Chukalla, Krol, & Hoekstra, 2015; Zhuo, Mekonnen, Hoekstra, & Wada, 2016c). Details of the method are provided by Hoekstra (2019a).

When the green-blue accounting method by Hoekstra (2019a) cannot be followed, an alternative, less accurate method exists to distinguish between green and blue crop water use. This method, which has been practiced in the past (Hoogeveen, Faurès, Peiser, Burke, & van de Giesen, 2015; Liu & Yang, 2010; Mekonnen & Hoekstra, 2010, 2011; Siebert & Döll, 2010), is to estimate blue *ET* as the difference between *ET* under irrigated conditions and *ET* under rainfed conditions. Note that this method thus requires two simulations for the same crop field: an irrigated and a rainfed case. Hoekstra (2019a) mentions a couple of drawbacks of this method. First, the rooting depth of crops is different under rainfed versus irrigated conditions, which affects the water uptake by plants and consequently the partitioning of green versus blue *E* and *T*. This issue can be partially resolved by simulating the rainfed case with a rooting depth as it would be under irrigated conditions. However, this approach is still inaccurate

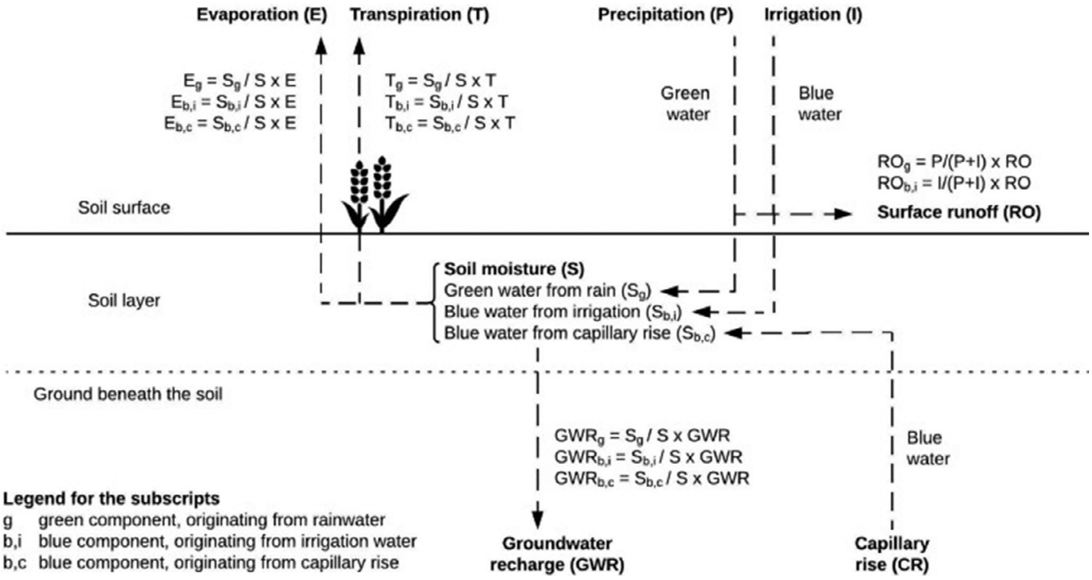

FIGURE 4.3 Green-blue water accounting of soil moisture and water fluxes entering and leaving the soil moisture in case of one soil layer. The 'colour codes' in the form of subscripts refer to the origin of the water. Credit: Reprinted from Hoekstra (2019a) with permission from Elsevier.

because irrigation affects the soil moisture dynamics over time, such that the partitioning of green versus blue *E* and *T* is different under irrigated versus rainfed conditions. Furthermore, a rainfed case is not always available, which is the case in many (arid) regions where rainfed cropping is not a viable alternative to irrigated agriculture. In these regions, a crop model will not be able to simulate crop development (the crop will hardly develop and/or die-off early in the season) and soil moisture dynamics under rainfed conditions that are realistically representative for the green *E* and *T*.

4.3 Environmental sustainability, efficiency and equitability of the water footprint of food systems

The purpose of WFA is to feed a debate on wise freshwater allocation. Relevant questions in the domain of food systems are for example: in which places is food production associated with unsustainable water abstractions or water pollution, and which consumers benefit from this food? Can we produce the same crop with a smaller water footprint? Or can we replace it for other foodstuffs with equivalent nutritional value but a smaller water footprint? Given that the global freshwater availability is limited, what are fair shares of water use per community to meet their (food) demand?

To shed light on such conversations, WFA goes beyond WF accounting to sustainability assessment. We follow the three pillars under wise freshwater allocation as postulated by

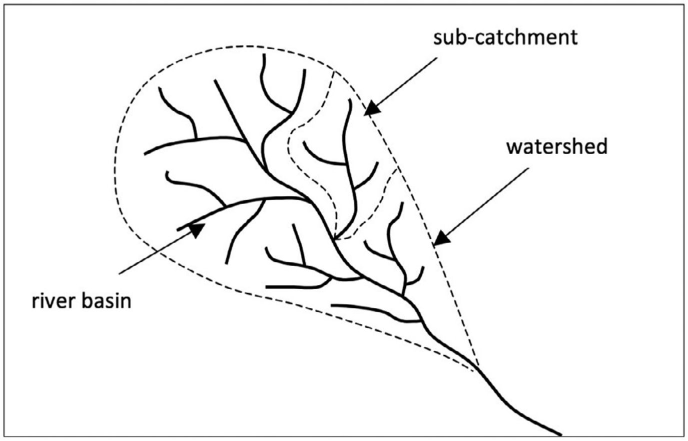

FIGURE 4.4 A river basin is a catchment area in which the precipitation is discharged by a network of streams, with one common outflow point. A river basin is a typical unit for analysis of freshwater availability and use. A river basin can be schematized into subcatchments. The watershed is the dividing line between different river basins. The watershed will generally follow the highest mountain ridges: precipitation on the one side of the ridge will run off in one direction and finally end up in one river, while precipitation on the other side of the ridge will run off in the other direction and finally end up in another river. Credit: Reprinted from Hoekstra (2019b) with permission from University of Twente.

Hoekstra (2014): assessing the environmental sustainability of a WF (Section 4.3.1), the efficiency of a WF (Section 4.3.2), and the equitability of a WF (Section 4.3.3). The methods we discuss in this section step away from the narrower focus on growing a crop (as per Section 4.2), to encompass food production, consumption and trade in a wider sense.

4.3.1 Environmental sustainability

In the field of WFA, the sustainability of a WF of a certain entity (an activity, a producer, or a consumer) is addressed by analyzing *total* WFs in the context of *maximum sustainable* WFs for a certain geographic area, often a river basin (see Fig. 4.4), and a certain period of time. The reasoning is that one cannot label a single WF in itself as 'sustainable' or 'unsustainable', but each WF contributes to a state at the system level which is sustainable or unsustainable, considering how the WF of all activities in the system altogether compare to the maximum sustainable WF of the system (Hoekstra, 2015). So, sustainability of an activity is always conditional on other activities present in its context. Note that this approach differs from the typical life-cycle analysis (see Chapter 3 of this book) approach to assessing the sustainability of an activity or product, which is to compare the potential environmental impacts of substitutable products. In other words, WFA tries to answer whether a WF is part of a system that is sustainable or unsustainable (absolute approach), while life-cycle analysis tries to answer whether a product is more sustainable than other substitutable products (comparative approach) (Hoekstra, 2015). In the following sub sections, we describe the concept of the maximum sustainable footprint and how this concept is used to assess the environmental sustainability of the blue, green, and grey components of a WF.

4.3.1.1 The maximum sustainable water footprint

Freshwater availability in a river basin stems from precipitation over land. Part of the precipitation evapotranspires from the land surface (green water flow), the rest generates runoff via surface and groundwater (blue water flow) towards the sea. Not all green and blue

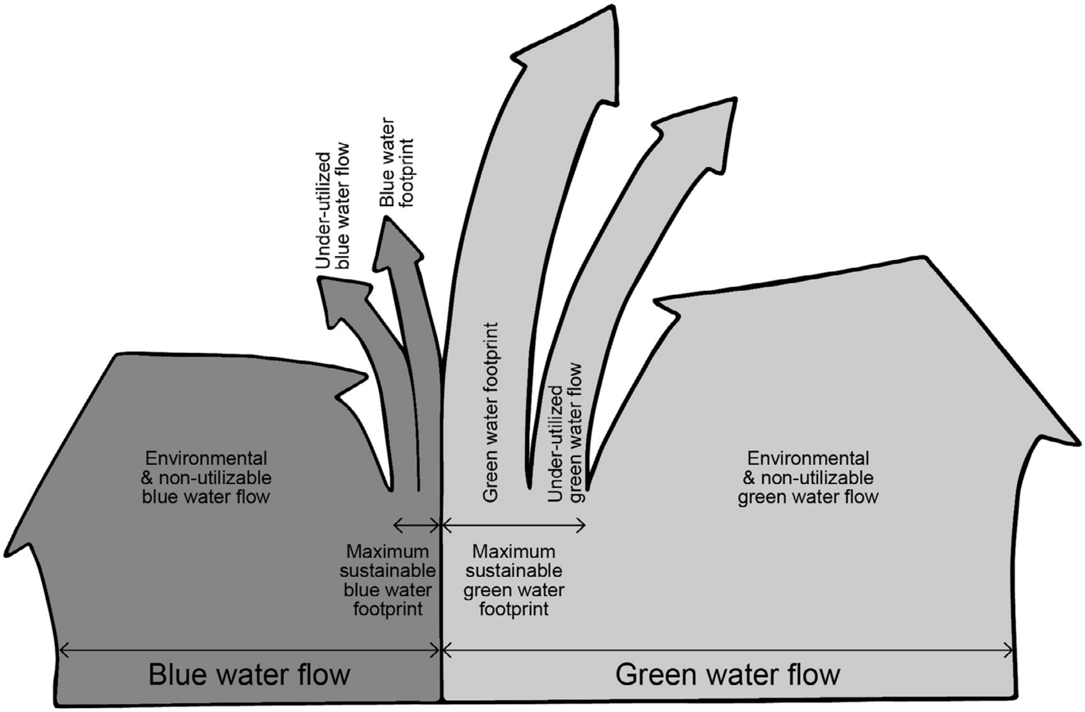

FIGURE 4.5 The partitioning of precipitation over land into blue and green water flows. Both flows further partition into environmental and non-utilizable (or non-accessible) flows, flows allocated to human activities (i.e. water footprint) and under-utilized flows below the maximum sustainable level. Credit: Reprinted from Schyns (2018).

water flows are available for consumption, and fractions of these flows need to be reserved for the functioning of terrestrial (for green water) and aquatic (for blue water) ecosystems, the so-called environmental flow requirements (see Fig. 4.5). The maximum sustainable green WF, or 'green water availability', is equal to the total green water flow minus environmental green water requirements. The maximum sustainable blue WF, or 'blue water availability', is equal to the total blue water flow minus environmental blue water requirements. The maximum sustainable green and blue WF reflect environmental limits to the consumptive use of green and blue water, respectively. The grey WF is defined in volumetric terms and also represents a pressure on blue water. The effect of the total grey water footprint in a river basin depends on the available blue water flow available to assimilate the waste (Hoekstra et al., 2011). The maximum sustainable grey WF, or 'waste assimilation capacity', is therefore equal to the actual runoff, which equals the natural runoff minus the blue WF.

4.3.1.2 *Sustainability of the blue water footprint*

To evaluate whether the blue WF of a particular activity, producer or consumer is sustainable or not requires placing that blue WF in the context of the degree of blue water scarcity experienced in the geographic area in which the WF is located. This is done using the blue

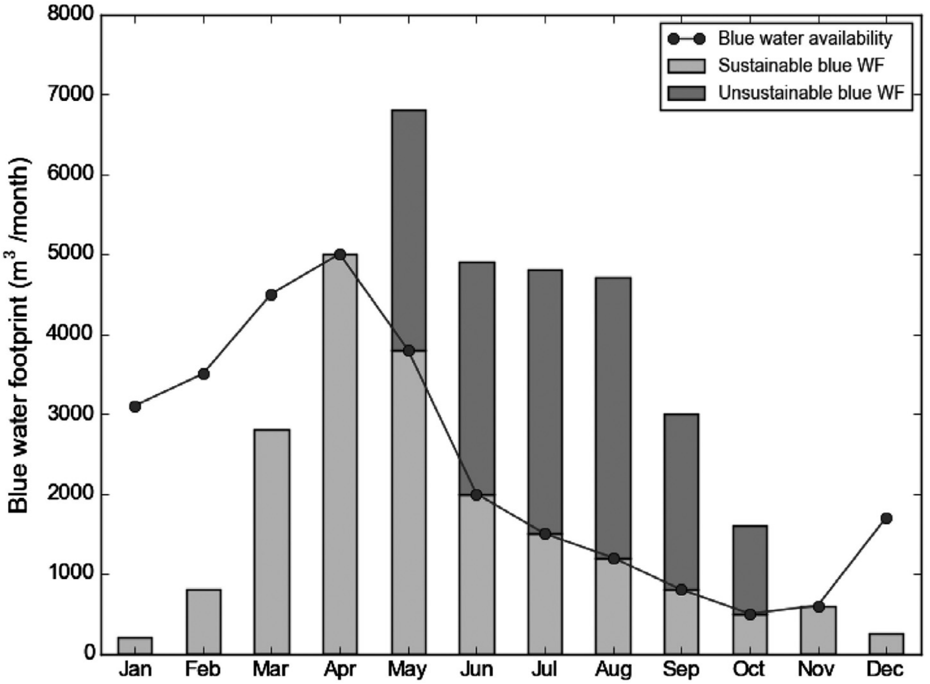

FIGURE 4.6 Theoretical example of the monthly sustainable (light gray) and unsustainable (dark gray) components of the blue water footprint. Credit: Reprinted from Mekonnen and Hoekstra (2020b) with permission from Elsevier.

water scarcity index, defined as the ratio of the total blue WF over the blue water availability (Hoekstra et al., 2011). A blue WF located in a place in which the total blue WF exceeds blue water availability (blue water scarcity index > 1) is labelled as unsustainable, because it contributes to the infringement of environmental flows (Fig. 4.6). This method has for example been used to assess the sustainability of the blue WF of crop production (Mekonnen and Hoekstra, 2020b) and of national consumption (Mekonnen and Hoekstra, 2020a).

Blue water scarcity typically manifests within particular parts of the year – because both blue water availability and blue WF vary strongly within the year (Fig. 4.6) – in particular parts of a river basin (often downstream). Therefore, the assessment of the environmental sustainability of the blue WF is ideally done on a month-by-month basis on the level of the river basin or sub-catchments (Fig. 4.4) (Hoekstra, Mekonnen, Chapagain, Mathews, & Richter, 2012; Mekonnen & Hoekstra, 2016).

4.3.1.3 *Sustainability of the green water footprint*

Like the blue WF, evaluating the sustainability of a particular green WF requires placing that green WF in the context of the degree of local green water scarcity. Green water scarcity is a bit harder to grasp than blue water scarcity. It refers to the competition over limited green water flows, which can either support a natural ecosystem or the production of biomass for various purposes in the human economy (Schyns, Hoekstra, and Booij, 2015). A green water

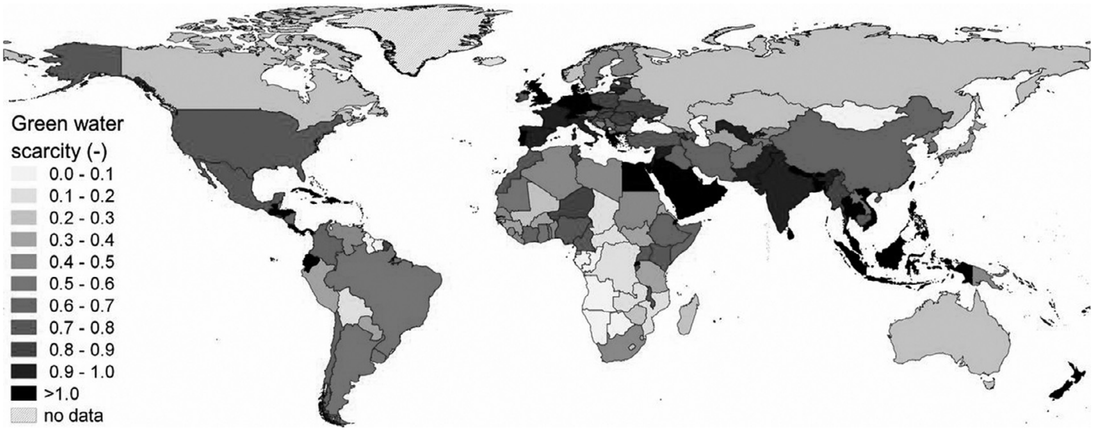

FIGURE 4.7 Green water scarcity per country, expressed as the ratio of the national aggregate green water footprint to the national aggregate green water availability. Credit: Reprinted from Schyns et al. (2019) with permission from the author.

scarcity index has been proposed (Hoekstra, Chapagain, Aldaya, & Mekonnen, 2011; Schyns, Hoekstra, & Booij, 2015) and recently developed (Schyns et al., 2019) that measures the ratio of the total green WF to green water availability (see Fig. 4.7) similar to the blue water scarcity index. Green water availability considers the agroecological suitability and accessibility of land, biophysical constraints to intensifying land use, and the need to set aside part of the land and green water flows for biodiversity conservation (Schyns et al., 2019). Countries with green water scarcity equal to 1 have fully allocated their sustainably available green water flow to human activities (or overshoot of the maximum sustainable level is canceled out by remaining potential in another part of the country).

In contrast to blue water scarcity, which is best estimated per month per sub-catchment, green water scarcity should be assessed on a (multi-)annual scale and larger spatial entities, such as countries, large river basins, or ecoregions. Green water is indirectly allocated through land-use decisions, which are generally made over long planning horizons and do not change on a monthly basis. Land use decisions tend to be made by local land owners, but they impact the biodiversity of the larger ecosystems in which the land resides. Thus, to estimate green water availability, one should consider a fairly large spatial region to determine which places within that region have the most biodiversity value and are most in need of conservation.

4.3.1.4 Sustainability of the grey water footprint

While the blue and green WF are measures of water use, the grey WF is a measure of water pollution. The grey WF of an activity is not a measure of the actual polluted volume (which can never exceed the actual volume of water), but an indicator of the severity of water pollution, expressed in terms of the water volume required to assimilate the load of pollutants that the activity adds to a freshwater body (Hoekstra et al., 2011). To evaluate whether a particular grey WF is sustainable or not requires assessing whether or not the waste assimilation capacity in the geographic area in which the WF is located is already fully consumed. For this, the water pollution level is used as indicator. The water pollution level is defined as the ratio of the

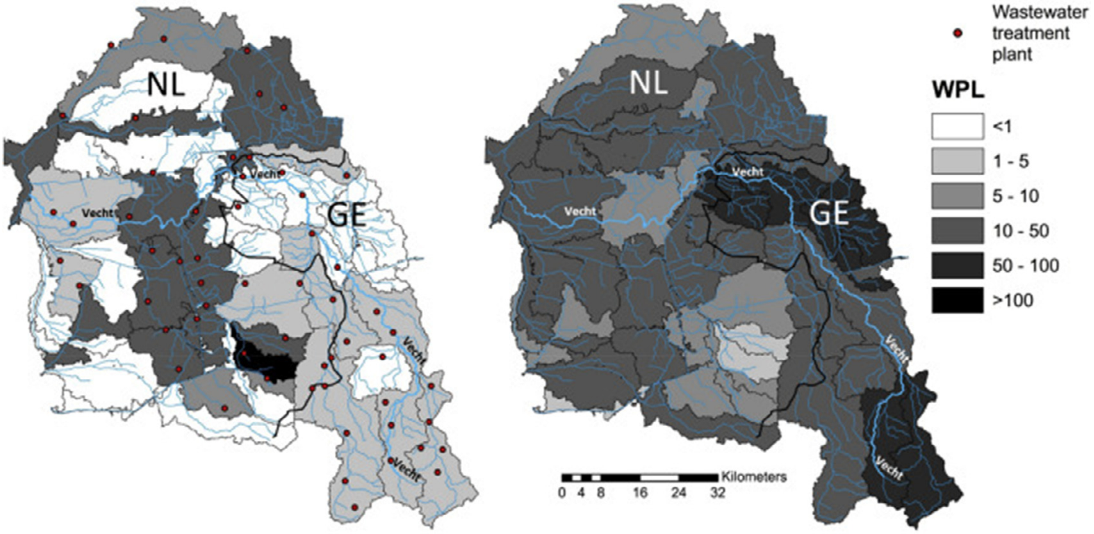

FIGURE 4.8 Annual average water pollution level (WPL) in the Vecht catchment – located in Germany (GE) and the Netherlands (NL) – resulting from the maximum grey water footprint of human (left) and veterinary (right) pharmaceutical use, resulting from the substances ethinylestradiol and amoxicillin, respectively. Credit: Reprinted from Wöhler et al. (2020b).

total grey WF in an area to the waste assimilation capacity. The total grey WF in an area is determined by the pollutant that is most critical, i.e. the pollutant with the largest associated grey WF.

Since the waste assimilation capacity depends on the blue water runoff, the assessment of the sustainability of the grey WF is (similar to the blue WF, see Section 4.3.1.2) best determined for each month of the year separately, and at the level of the river basin or sub-catchments (Fig. 4.4).

An example of an assessment of the environmental sustainability of the grey WF is provided in Fig. 4.8. This study by (Wöhler et al., 2020b) shows the water pollution level within the Vecht catchment as a result of the grey WF of pharmaceutical use by livestock and humans in the catchment. Another notable example is by Aldaya et al. (2020), who assessed the grey WF of diffuse nitrogen pollution in an agricultural region in Spain. Just like blue and green water scarcity (Mekonnen & Hoekstra, 2016; Schyns et al., 2019), the water pollution levels related to anthropogenic loads of nitrogen (Mekonnen and Hoekstra, 2015) and phosphorus (Mekonnen and Hoekstra, 2018) have also been mapped with global coverage.

4.3.2 Efficiency

The efficiency of a WF can be addressed from three perspectives (Hoekstra, 2020): a production (Section 4.3.2.1), a geographic (Section 4.3.2.2), and a consumption perspective (Section 4.3.2.3).

4.3.2.1 Efficiency from a production perspective

A production perspective on the efficiency of a WF addresses the question of how we can produce a certain process or product with less water (Hoekstra, 2020). In other words, how can we reduce the WF (volume per unit of product) or increase the water productivity (unit of product per volume of water, also called the water-use efficiency). This question can be answered by comparing the actual WF of a crop in a certain location and time period to a benchmark value. Such a benchmark represents a reasonable WF per unit of crop. There are two main methods that have been applied to derive a benchmark for the WF of growing a crop.

One method is to look at the spatial differences in the WF per unit of crop and derive a benchmark by ranking the WF achieved by producers under similar circumstances. A benchmark can then be set at the WF achieved by the top-10 percent or top-25 percent most efficient producers, for example. This method thus compares and ranks producers that operate under similar circumstances. So the key question is: what producers can we compare? Recent studies have grouped and compared producers that operate in similar climates (Karandish et al., 2018; Mekonnen, Hoekstra, Neale, Ray, and Yang, 2020; Zhuo, Mekonnen, and Hoekstra, 2016a) and/or on similar soils (Zhuo et al., 2016a), and by separating rainfed and irrigated agricultural systems (Mekonnen et al., 2020). However, no matter what spatial scope one takes in grouping producers, within that scope there will still be variability from place to place. Rainfall, for example, shows strong spatial variability over short distances, such that a few producers in a larger area simply had more favourable local circumstances. Therefore, one can always question the comparability of producers that operate in different locations and the WFs they achieve.

Alternatively, to avoid this drawback, one can derive a benchmark for the WF of a crop in a certain location by estimating the WF of the crop in the *same* location under a best practice scenario (Chukalla et al., 2015, 2017). These studies modeled crop WFs under the same conditions (location, year) under alternative management packages, to evaluate which package (the best practice) results in the smallest WF. Management packages are combinations of irrigation techniques (application methods, such as flooding or sprinkler systems), irrigation strategies (when and how much water to apply) and mulching practices that cover part of the soil to reduce soil evaporation. This method does not suffer from the drawback of comparability, but it can be laborious. One needs to have a crop model (or field measurements, which is arguably even more laborious) that can simulate agricultural management packages. Moreover, deciding on the exact setup of alternative potential ‘best management packages’ is not straightforward, because options are practically unlimited.

A final note on benchmarking is that comparing a WF in a particular location against a benchmark indicates whether or not the WF of that process or product can be reduced (and by how much). However, it does not indicate whether using the water in that location for that purpose is a wise decision. Better alternative uses of the water may exist in that location or upstream of that location. For example, consider a case in which a river flows through a fertile landscape into marginal land. A farmer on marginal land can use the water to irrigate crops in an efficient way such that the farmer achieves the local benchmark. However, it is probably wiser to not grow crops on the marginal land, but instead on the fertile lands upstream where more food can be produced with the same amount of water due to the more favourable circumstances.

4.3.2.2 *Efficiency from the geographic perspective*

The geographic perspective on the efficiency of a WF addresses the question of where we can best produce what to save water and reduce water scarcity (Hoekstra, 2020). Across the globe, there is large spatial variability in the WF of growing a crop (Liu and Yang, 2010; Mekonnen and Hoekstra, 2011), indicating potential to save water by changing spatial cropping patterns and international trade in agricultural goods, along with their accompanying virtual water flows. Indeed, a number of studies have assessed changes in national, regional or global cropping patterns and associated water savings (Chouchane, Krol, & Hoekstra, 2020; Davis, Rulli, Seveso, & D'Odorico, 2017a; Davis, Seveso, Rulli, & D'Odorico, 2017b; Schyns & Hoekstra, 2014; Ye et al., 2018). Recently, Chouchane et al. (2020) optimized global cropping patterns to minimize national blue water scarcity. They found that with a minimum allowance (≤10 percent) of expansion of irrigated and rainfed harvested area per crop, it is possible to decrease the global blue water footprint of crop production by 21 percent and decrease the global total harvested and irrigated areas by 2 percent and 10 percent, respectively. Consequently, the blue water scarcity in the world's most water-scarce countries can be greatly reduced (Chouchane et al., 2020).

4.3.2.3 *Efficiency from the consumption perspective*

The consumption perspective on the efficiency of a WF addresses the question of how we can best fulfill certain consumer needs with less water (Hoekstra, 2020). Studies taking this perspective on the efficiency of the WF of food systems have focused on reducing food losses (Jalava et al., 2016; Karandish, Hoekstra, and Hogeboom, 2020; Kummu et al., 2012; Mekonnen and Fulton, 2018) and changing diets (Harris et al., 2020; Jalava, Kummu, Porkka, Siebert, & Varis, 2014; Kim et al., 2020; Mekonnen & Fulton, 2018; Vanham, Comero, Gawlik, & Bidoglio, 2018; Vanham, Hoekstra, & Bidoglio, 2013a; Vanham, Mekonnen, & Hoekstra, 2013b). The question is how we can fulfill energy and protein needs in a water efficient manner. Reductions in food waste directly translate into reduced freshwater needs to grow the food in the first place. Meat and dairy generally have significantly larger WFs per kcal of energy and per gram of protein compared to plant-based alternatives (Mekonnen and Hoekstra, 2012). Therefore, the water efficiency of certain diets can be increased by replacing (part of) meat and/or dairy by suitable amounts of plant-based alternatives, while maintaining the overall nutritional value of the diet. A recent global study has shown that, for countries around the world, changes in diets (to diets with less red meat or pescetarian, lacto-ovo vegetarian or vegan diets) have significant effects on the both the blue and total consumptive (green plus blue) WFs per capita (Kim et al., 2020). They found the largest potential reductions in per capita consumptive WFs from shifting to vegan diets in countries where animal products account for a large share of the current (baseline) consumption pattern.

4.3.3 Equitability

The central question when assessing the equitability of a WF is: which communities finally benefit from the water used (Hoekstra, 2020)? This question is at the root of the WFA methodology which – by incorporating supply-chain thinking into water resources management – provides insight into what end-purposes and communities benefit from freshwater use. While a WFA can reveal what end-purposes benefit from freshwater used (cf. Zhuo et al. (2019)), it does not provide quantitative measures to answer the question whether that freshwater allocation is fair or equitable. Rather, indicators on the sustainability and efficiency of a WF – as discussed in previous sections – feed into a debate on what is equitable water sharing.

Hoekstra (2020) sheds light on the topic of equitable water use from three perspectives: the consumption, production and geographic perspective. The consumption perspective addresses the question of how we can fairly share the world's limited freshwater resources. Approaches to assess the efficiency of a WF from the consumption perspective feed into the debate on fair shares of water use per community. The production perspective raises the question of how we can produce more commodities to be shared amongst communities, without increasing the pressure on freshwater resources. Solutions to increase the water-use efficiency of food production are part of this debate. The geographic perspective is concerned with how we can fairly share water given large differences in the water availability per capita across nations. This relates to trade in water-intensive commodities (which encompasses practically all agricultural goods) in solidarity with people in water-poor countries. Research in this direction has for example assessed the need for increased export of staple crops to countries with decreasing water availability per capita (Chouchane, Krol, and Hoekstra, 2018a).

Compared to assessing the environmental sustainability and efficiency of WFs, the topic of assessing equitable water sharing is still relatively underdeveloped. As such, it remains an area for further research.

4.4 Towards water-wise food systems

It is difficult enough to make sure that complex and geographically dispersed food systems reach goals for food security, let alone achieving goals for water security (Hogeboom et al., 2021). The usual suspects to prevent harmful water outcomes and ignite positive change are farmers and governments: the former produce food and use water directly, while the latter typically oversee water allocation to users. However, other actors that may not be readily associated with water-for-food discourses can and should play a role too. Think of investors, civil society groups, or consumers. While it is true that such actors mainly carry an indirect or supply chain responsibility, their power to influence food and water outcomes should not be underestimated. To the contrary, there is not one actor that can transform current food systems into water-wise food systems by itself. This transformation is a shared responsibility, one that requires inclusion of a broad spectrum of actors (UN-WWAP, 2006).

So what can or should these actors do? It is clear there is no silver bullet, but what set of measures might lead us in the right direction? Without holding to the illusion of being complete and conclusive, this section highlights possible responses across the landscape of actors: governments, citizens, food sector companies, investors, international organizations, civil society, media, and academia.

A summary overview of response actions for each of these actor groups is provided in Table 4.1. Note that these examples are not exclusively derived from the WF concept,

TABLE 4.1 Examples of response options per actor to transform food systems into water-wise food systems.

Actor	Response options
Governments	*Regulatory instruments*
	Set WF caps
	Formulate WF benchmarks
	Mandatory product WF labeling
	Certification
	Institute a WF permit system
	Ensure minimum water rights
	Enforcement and monitoring
	Water neutral spatial development
	Economic instruments
	(Full cost) water pricing
	Taxing pollution
	Taxing water-intensive commodities
	Subsidizing Best Available Technologies
	Stop subsidizing ill-performing practices and technologies
	Institutional arrangements
	Harmonize policies across policy domains
	Take responsibility for external WF of consumption
	Make companies take responsibility for their supply chains
	International agreements
	Water-sensitive trade
	Fair shares of WF of consumption
	Standards
	WF disclosure
	WF performance
	Awareness raising
	Campaigning
	Water in educational programmes
	Promoting water transparency
	Promoting water-sustainable diets
Citizens	*As consumer*
	Substitute water-intensive foodstuffs in diet
	Reduce overall caloric intake
	Reduce meat and animal product intake
	Buy local and organic
	Reduce/avoid food waste
	Avoid water-intensive non-food products that compete with foodstuffs over water (e.g., biofuels, cotton)
	As voter
	Vote for parties that are mindful of water resources
	Vote not for parties that harm or neglect water resources
	As investor
	Invest in companies/pension funds that are mindful of water resources
	Invest not in companies/pension funds that harm or neglect water resources
	Choose a water-aware bank
	As activist
	Hold opinions on water issues
	Demand product transparency from companies
	Demand government regulation concerning water issues

(*continued on next page*)

TABLE 4.1 Examples of response options per actor to transform food systems into water-wise food systems—cont'd

Actor	**Response options**
(Food sector) Companies	*Reduce operational WF* Substitute water-intensive for water-extensive crops, products, and processes Use Best Available Technologies and Best Available Practices for water Set and meet WF reduction targets *Reduce supply chain WF* Supplier WF agreements Change (non-compliant) suppliers Set and meet WF reduction targets *Corporate water stewardship* Measure and monitor WFs in operations Measure and monitor WFs in supply chain Certify sites for water performance Comply with (WFA) standard Actively engage with (basin) communities Label products Disclose corporate water performance
Investors	*Investment water policy* Have one Require WF disclosure/compliance of investees Adjusted interest rates for water-wise investment activities Increase share of water-wise companies/products in portfolio Screen companies on water performance Divest from water harming companies or water-scarce locations Set portfolio WF reduction targets *Corporate water stewardship* Measure and monitor WFs in operations Measure and monitor WFs in portfolio Disclose portfolio water dependency and risk Adopt responsible investment principles Share best investment practices
International organisations	Agenda setting Leverage stakeholder convening power Invest in WF reduction programs Water-informed trade agreements Advocate a 'Kyoto Protocol' for water Advocate minimum water rights Develop an international water pricing protocol Develop international nutrient housekeeping agreements Develop international water standards and taxonomies
Civil society groups	Awareness raising Agenda setting Promoting transparency Lobbying for water sustainable food systems
Media	Investigative journalism Maintaining private and public sector accountability Monitoring transparency
Academia	Providing water solutions (technical and non-technical) Evaluating water policies

nor is their applicability limited to water considerations in food systems alone. Rather, we deduce insights that emerge from the conceptual framework of WFA to explore perhaps unconventional measures and solution pathways (Hoekstra, 2020).

4.4.1 What can governments do?

Governments come in many shapes and sizes. They are a collective of many different governmental bodies that each has its own policy mandates and jurisdictions. The most direct responsibility to manage water resources wisely in their local context lies with local authorities, such as municipalities, utilities, and waterboards. These are the bodies that can, for instance, set a WF cap at the catchment level. Capping WFs entails earmarking sufficient water for nature, so that what remains can be considered sustainably allocable to human activities; the sum of all WF permits issued in that catchment should remain below that agreed upon WF cap (Hogeboom et al., 2020).

More challenging is the fact that other governmental bodies without direct mandates related to water or food hold important pieces of the water-for-food puzzle as well. For example, Ministries of Agriculture can promote installing efficient irrigation technologies and growing drought-resistant crop varieties. Ministries of Environment can define ecological boundary conditions within which total water consumption should remain (viz. define a 'safe operating space', (cf Hogeboom, de Bruin, Schyns, Krol, & Hoekstra (2020)). Ministries of Infrastructure can maintain and cover irrigation canals and ponds to prevent leakage or evaporation (cf. Haghighi, Madani, and Hoekstra (2018)). Ministries of Energy should steer clear of solving the carbon crisis at the expense of worsening water crises, which may happen through for instance the production of biofuels that compete with food systems over water resources (cf. Holmatov, Hoekstra, and Krol (2019), Vanham et al. (2019)). Ministries of Health can enforce protocols for disposing of pharmaceuticals to avoid water pollution, rendering this water useless for further use (cf. Wöhler, Hoekstra, Hogeboom, Brugnach, and Krol (2020)). Ministries of Economic Affairs can stop subsidizing water-wasting farming practices and water-polluting industries and many more examples could be mentioned.

For some solutions, multilateral cooperation between governments is needed. For example, to reduce their internal WF, national governments can save water domestically by importing more food (and thus virtual water) from other countries. The risk with such a trade strategy is that it solves water issues in the importing country at the cost of the exporting countries to which the water issues are being externalized. This need not be the case, though. If WFs per unit of product in the exporting countries are lower than domestic unit WFs, there will be a net saving associated with this trade flow. National governments could therefore investigate differences in unit WFs across countries, and select trade partners on the basis of their comparative advantage in terms of water productivities (Hoekstra and Chapagain, 2008). Such trade strategies create win-win situations in which not only national, but also global water savings are possible.

4.4.2 What can citizens do?

For all governments can do, they cannot legislate virtue nor force consumers to make water-wise food decisions. Hence, citizens have a responsibility too. Given there are almost 8 billion of us on this planet, citizens in the role of consumers are arguably the most powerful actor group to solve our collective water issues. Adopting a vegetarian diet, for example, reduces the typical WF by 40 percent (Vanham et al., 2018). Reducing food waste goes a long way too (Karandish et al., 2020). But if you are standing in the grocery store, contemplating tonight's dinner, how will you know if you are choosing the best product from a water perspective? A major obstacle for consumers to making an informed decision is that many products do not come with the necessary transparency on water performance (nor on other sustainability aspects, for that matter). If you are of the activist type, pressuring companies to provide this information might thus be a sensible thing to do. You might be surprised to see how sensitive companies can be to feedback that may harm their reputation. But beyond your own diet and idealistic activism, have you thought about promoting water sustainability through the ballot box? And how about your personal investments? Do you hold stocks in water-wasting companies? Is your bank mindful about water sustainability or will they lend your savings to water-polluting businesses? Where does your pension fund invest your retirement savings?

4.4.3 What can companies do?

Actions to reduce WFs that can be taken by food sector companies, including fishers, fertilizer producers, processors, traders, retailers, transporters, markets, tech companies, and others, again cover a wide spectrum of solutions. In general, though, we postulate that each can and should adopt Best Available Practices in terms of water for their respective business processes and products. What is particularly important to note is that, although reducing water use and pollution in their direct operations is imperative, the bulk of the WF is often in the supply chain, especially in agricultural products (Linneman, Hoekstra, and Berkhout, 2015). WFs in supply chains thus need to be addressed too. Fortunately, there are increasingly better water stewardship programmes and tools that help guide companies on their water journey. The WFA Global Standard (Hoekstra et al., 2011), for example, provides a systematic, science-based methodology to measure WFs along corporate value chains and provides guidelines to help formulate quantitative WF reduction targets within and beyond the company fence.

4.4.4 What can investors do?

Financial institutions, including asset owners (banks, pension funds, insurance companies), asset managers, rating agencies, and regulators, are a powerful yet enigmatic actor group that to date has hardly wielded their influence in water-for-food discourses. Lagging behind companies, investors by and large turn a blind eye to the effect that the activities they finance may have on (the future state of) water resources. This is unfortunate, as investing in, say, a large-scale avocado plantation in a water-scarce area today will have implications for consumption of local water resources for decades to come. A recent assessment by (Hogeboom, Kamphuis, & Hoekstra (2018)) confirmed that there is still a lot to be done to build investor awareness on the reality and potential impact of water crises, their role therein, and the development of sound, water-informed investment policies.

4.4.5 What can international organizations do?

The example above in Section 4.4.1 on national and global water saving strategies already illustrated that there is a global dimension to water management beside the traditional local one (Vörösmarty, Hoekstra, Bunn, Conway, and Gupta, 2015). In fact, one could argue that the global market dictates to a large extent which crops and food products are produced. Therefore, international organizations, such as the United Nations, multilateral development banks, and regional collaborations such as the European Union, seem logical actors to address water considerations associated with supra-national trade flows. Trade agreements through the World Trade Organization, for example, include a so-called non-discrimination clause, meaning countries cannot differentiate between similar products in trade (Hoekstra, 2020). Consequently, countries cannot favor trade partners on the basis of differences in water performance of products either: according to WTO rules, a tomato is a tomato, regardless the WF or impact on water resources that the cultivation of that tomato has. Trade agreements could thus be renegotiated to accommodate for differentiation in products based on water and/or other sustainability criteria.

Another potential action by international organizations is to negotiate a water version of the Kyoto Protocol. Such a protocol for water could define fair shares for WFs of consumption per capita, coupled to an international WF pricing scheme (Hoekstra, 2020).

4.4.6 What can civil society, media, and academia do?

A final, broad category of actors that can take meaningful water action in the food domain are civil society groups, the media, and academia. Civil society groups can be particularly effective in putting water-for-food discussions higher on political and corporate agendas. Media, in turn, have an important role to play in unearthing detrimental water performance by other actors, for example by diving deeply into often hidden supply chains of companies or investigative journalism of water impact of governmental policies. Lastly, academics can contemplate, research and develop both technical and non-technical (behavioural, governance, political) solutions to transforming food systems into water-wise food systems.

References

Aldaya, M.M., Hoekstra, A.Y., 2010. The water needed for Italians to eat pasta and pizza. Agricultural Systems 103, 351–360.

Aldaya, M.M., Rodriguez, C.I., Fernandez-Poulussen, A., Merchan, D., Beriain, M.J., Llamas, R., 2020. Grey water footprint as an indicator for diffuse nitrogen pollution: the case of Navarra, Spain. The Science of the Total Environment 698, 134338.

Allan, J.A., 1998. Virtual water: a strategic resource global solutions to regional deficits. Groundwater 36, 545–546.

Allen, R.G., Pereira, L.S., Raes, D., Smith, M., 1998. Crop evapotranspiration: Guidelines for computing crop water requirements. FAO Irrigation and drainage paper no, 56. FAO, Rome, Italy.

Carr, J.A., D'Odorico, P., Laio, F., Ridolfi, L., 2012. On the temporal variability of the virtual water network. Geophysical Research Letters 39, L06404.

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

Chouchane, H., Krol, M.S., Hoekstra, A.Y., 2018b. Virtual water trade patterns in relation to environmental and socioeconomic factors: a case study for Tunisia. The Science of the Total Environment 613-614, 287–297.

Chouchane, H., Krol, M.S., Hoekstra, A.Y., 2020. Changing global cropping patterns to minimize national blue water scarcity. Hydrology and Earth System Sciences 24, 3015–3031.

Chukalla, A.D., Krol, M.S., Hoekstra, A.Y., 2015. Green and blue water footprint reduction in irrigated agriculture: effect of irrigation techniques, irrigation strategies and mulching. Hydrology and Earth System Sciences 19, 4877–4891.

Chukalla, A.D., Krol, M.S., Hoekstra, A.Y., 2017. Marginal cost curves for water footprint reduction in irrigated agriculture: guiding a cost-effective reduction of crop water consumption to a permit or benchmark level. Hydrology and Earth System Sciences 21, 3507–3524.

Chukalla, A.D., Krol, M.S., Hoekstra, A.Y., 2018a. Grey water footprint reduction in irrigated crop production: effect of nitrogen application rate, nitrogen form, tillage practice and irrigation strategy. Hydrology and Earth System Sciences 22, 3245–3259.

Chukalla, A.D., Krol, M.S., Hoekstra, A.Y., 2018b. Trade-off between blue and grey water footprint of crop production at different nitrogen application rates under various field management practices. The Science of the Total Environment 626, 962–970.

Davis, K.F., Rulli, M.C., Seveso, A., D'Odorico, P., 2017a. Increased food production and reduced water use through optimized crop distribution. Nature Geoscience 10, 919–924.

Davis, K.F., Seveso, A., Rulli, M.C., D'Odorico, P., 2017b. Water savings of crop redistribution in the United States. Water (Basel) 9, 83.

Doorenbos, J., Kassam, A.H., 1979. Yield response to water. *FAO Irrigation and Drainage Paper 33*. Food and Agriculture Organization of the United Nations, Rome, Italy.

Ercin, A.E., Aldaya, M.M., Hoekstra, A.Y., 2011. Corporate water footprint accounting and impact assessment: the case of the water footprint of a sugar-containing carbonated beverage. Water Resources Management 25, 721–741.

Ercin, A.E., Aldaya, M.M., Hoekstra, A.Y., 2012. The water footprint of soy milk and soy burger and equivalent animal products. Ecological Indicators 18, 392–402.

Falkenmark, M., 2000. Competing freshwater and ecological services in the river basin perspective. Water International 25, 172–177.

Franke, N.A., Boyacioglu, H., Hoekstra, A.Y., 2013. Grey water footprint accounting: tier 1 supporting guidelines. Value of Water Research Report Series No. 65. UNESCO-IHE, Delft, the Netherlands.

Gobin, A., Kersebaum, K.C., Eitzinger, J., Trnka, M., Hlavinka, P., Takáč, J., et al., 2017. Variability in the water footprint of arable crop production across European regions. Water (Basel) 9, 93.

Haghighi, E., Madani, K., Hoekstra, A.Y., 2018. The water footprint of water conservation using shade balls in California. Nature Sustainability 1, 358–360.

Harris, F., Moss, C., Joy, E.J.M., Quinn, R., Scheelbeek, P.F.D., Dangour, A.D., et al., 2020. The water footprint of diets: a global systematic review and meta-analysis. Advances in nutrition 11, 375–386.

Hoekstra, A.Y., 2003. Virtual water: An introduction. In: HOEKSTRA, A.Y. (Ed.), Virtual water trade: proceedings of the international expert meeting on virtual water trade, Value of Water Research Report Series No.12. UNESCO-IHE, Delft, the Netherlands, pp. 13–23.

Hoekstra, A.Y., 2014. Sustainable, efficient, and equitable water use: the three pillars under wise freshwater allocation. Wiley Interdisciplinary Reviews: Water 1, 31–40.

Hoekstra, A.Y., 2015. The sustainability of a single activity, production process or product. Ecological Indicators 57, 82–84.

Hoekstra, A.Y., 2017. Water footprint assessment: evolvement of a new research field. Water Resources Management 31, 3061–3081.

Hoekstra, A.Y., 2019a. Green-blue water accounting in a soil water balance. Advances in water resources 129, 112–117.

Hoekstra, A.Y., 2019b. *Lecture notes 'Water'*. Faculty of Engineering Technology. Enschede. University of Twente, the Netherlands.

Hoekstra, A.Y., 2020. The water footprint of modern consumer society, Second edition Routledge, London, UK.

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

Hoekstra, A.Y., Chapagain, A.K., Aldaya, M.M., Mekonnen, M.M., 2011. The water footprint assessment manual: Setting the global standard. Earthscan, London.

Hoekstra, A.Y., Mekonnen, M.M., 2012. The water footprint of humanity. Proceedings of the National Academy of Sciences of the United States of America 109, 3232–3237.

Hoekstra, A.Y., Mekonnen, M.M., Chapagain, A.K., Mathews, R.E., Richter, B.D., 2012. Global monthly water scarcity: blue water footprints versus blue water availability. Plos One 7, e32688.

Hogeboom, R.J., 2020. The water footprint concept and water's grand environmental challenges. One Earth 2, 218–222.

Hogeboom, R.J., Borsje, B.W., Deribe, M.M., van der Meer, F.D., Mehvar, S., Meyer, M.A., et al., 2021. Resilience meets the water–energy–food nexus: mapping the research landscape. Frontiers in Environmental Science 9, 630395.

Hogeboom, R.J., de Bruin, D., Schyns, J.F., Krol, M.S., Hoekstra, A.Y., 2020. Capping human water footprints in the world's river basins. Earth's Future 8, e2019EF001363.

Hogeboom, R.J., Hoekstra, A.Y., 2017. Water and land footprints and economic productivity as factors in local crop choice: the case of silk in Malawi. Water (Basel) 9, 802.

Hogeboom, R.J., Kamphuis, I., Hoekstra, A.Y., 2018. Water sustainability of investors: Development and application of an assessment framework. Journal of Cleaner Production 202, 642–648.

Holmatov, B., Hoekstra, A.Y., Krol, M.S., 2019. Land, water and carbon footprints of circular bioenergy production systems. Renewable & Sustainable Energy Reviews 111, 224–235.

Hoogeveen, J., Faurès, J.M., Peiser, L., Burke, J., van de Giesen, N., 2015. GlobWat – a global water balance model to assess water use in irrigated agriculture. Hydrology and Earth System Sciences 19, 3829–3844.

Jalava, M., Guillaume, J.H.A., Kummu, M., Porkka, M., Siebert, S., Varis, O., 2016. Diet change and food loss reduction: what is their combined impact on global water use and scarcity? Earth's Future 4, 62–78.

Jalava, M., Kummu, M., Porkka, M., Siebert, S., Varis, O., 2014. Diet change—A solution to reduce water use? Environmental Research Letters 9, 074016.

Karandish, F., Hoekstra, A.Y., Hogeboom, R.J., 2018. Groundwater saving and quality improvement by reducing water footprints of crops to benchmarks levels. Advances in water resources 121, 480–491.

Karandish, F., Hoekstra, A.Y., Hogeboom, R.J., 2020. Reducing food waste and changing cropping patterns to reduce water consumption and pollution in cereal production in Iran. Journal of hydrology 586, 124881.

Kim, B.F., Santo, R.E., Scatterday, A.P., Fry, J.P., Synk, C.M., Cebron, S.R., et al., 2020. Country-specific dietary shifts to mitigate climate and water crises. Global Environmental Change 62, 101926.

Kummu, M., de Moel, H., Porkka, M., Siebert, S., Varis, O., Ward, P.J., 2012. Lost food, wasted resources: global food supply chain losses and their impacts on freshwater, cropland, and fertiliser use. The Science of the Total Environment 438, 477–489.

Linneman, M.H., Hoekstra, A.Y., Berkhout, W., 2015. Ranking water transparency of Dutch stock-listed companies. Sustainability 7, 4341–4359.

Liu, J., Yang, H., 2010. Spatially explicit assessment of global consumptive water uses in cropland: green and blue water. Journal of hydrology 384, 187–197.

Masud, M.B., McAllister, T., Cordeiro, M.R.C., Faramarzi, M., 2018. Modeling future water footprint of barley production in Alberta, Canada: implications for water use and yields to 2064. The Science of the Total Environment 616-617, 208–222.

Masud, M.B., Wada, Y., Goss, G., Faramarzi, M., 2019. Global implications of regional grain production through virtual water trade. The Science of the Total Environment 659, 807–820.

Mekonnen, M.M., Fulton, J., 2018. The effect of diet changes and food loss reduction in reducing the water footprint of an average American. Water International 43, 860–870.

Mekonnen, M.M., Gerbens-Leenes, W., 2020. The water footprint of global food production. Water (Basel) 12, 2696.

Mekonnen, M.M., Hoekstra, A.Y., 2010. A global and high-resolution assessment of the green, blue and grey water footprint of wheat. Hydrology and Earth System Sciences 14, 1259–1276.

Mekonnen, M.M., Hoekstra, A.Y., 2011. The green, blue and grey water footprint of crops and derived crop products. Hydrology and Earth System Sciences 15, 1577–1600.

Mekonnen, M.M., Hoekstra, A.Y., 2012. A global assessment of the water footprint of farm animal products. Ecosystems 15, 401–415.

Mekonnen, M.M., Hoekstra, A.Y., 2014. Water footprint benchmarks for crop production: a first global assessment. Ecological Indicators 46, 214–223.

Mekonnen, M.M., Hoekstra, A.Y., 2015. Global gray water footprint and water pollution levels related to anthropogenic nitrogen loads to fresh water. Environmental Science & Technology 49, 12860–12868.

Mekonnen, M.M., Hoekstra, A.Y., 2016. Four billion people facing severe water scarcity. Science Advances 2, e1500323.

Mekonnen, M.M., Hoekstra, A.Y., 2018. Global anthropogenic phosphorus loads to freshwater and associated grey water footprints and water pollution levels: a high-resolution global study. Water Resources Research 54, 345–358.

Mekonnen, M.M., Hoekstra, A.Y., 2020a. Blue water footprint linked to national consumption and international trade is unsustainable. Nature Food 1, 792–800.

Mekonnen, M.M., Hoekstra, A.Y., 2020b. Sustainability of the blue water footprint of crops. Advances in water resources 143, 103679.

Mekonnen, M.M., Hoekstra, A.Y., Neale, C.M.U., Ray, C., Yang, H.S., 2020. Water productivity benchmarks: the case of maize and soybean in Nebraska. Agricultural Water Management 234, 106122.

Mubako, S.T., 2018. Blue, green, and grey water quantification approaches: a bibliometric and literature review. Journal of Contemporary Water Research & Education 165, 4–19.

Nouri, H., Stokvis, B., Galindo, A., Blatchford, M., Hoekstra, A.Y., 2019. Water scarcity alleviation through water footprint reduction in agriculture: the effect of soil mulching and drip irrigation. The Science of the Total Environment 653, 241–252.

Pahlow, M., van Oel, P.R., Mekonnen, M.M., Hoekstra, A.Y., 2015. Increasing pressure on freshwater resources due to terrestrial feed ingredients for aquaculture production. The Science of the Total Environment 536, 847–857.

Postel, S.L., Daily, G.C., Ehrlich, P.R., 1996. Human appropriation of renewable fresh water. Science 271, 785–788.

Rockström, J., Gordon, L., 2001. Assessment of green water flows to sustain major biomes of the world: implications for future ecohydrological landscape management. Physics and Chemistry of the Earth, Part B: Hydrology, Oceans and Atmosphere 26, 843–851.

Schyns, J.F., 2018. Sustainable and efficient allocation of limited blue and green water resources [PhD thesis]. University of Twente, Enschede, the Netherlands.

Schyns, J.F., Hoekstra, A.Y., 2014. The added value of water footprint assessment for national water policy: a case study for Morocco. Plos One 9, e99705.

Schyns, J.F., Hoekstra, A.Y., Booij, M.J., 2015. Review and classification of indicators of green water availability and scarcity. Hydrology and Earth System Sciences 19, 4581–4608.

Schyns, J.F., Hoekstra, A.Y., Booij, M.J., Hogeboom, R.J., Mekonnen, M.M., 2019. Limits to the world's green water resources for food, feed, fiber, timber, and bioenergy. Proceedings of the National Academy of Sciences of the United States of America 116, 4893–4898.

Siebert, S., 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, 198–217.

Steduto, P., 2006. Biomass water-productivity: Comparing the growth-engines of crop models. WUEMED training course "Integrated approaches to improve drought tolerance in crops". Bologna, Italy.

Steduto, P., Hsiao, T.C., Raes, D., Fereres, E., 2009. AquaCrop-the FAO crop model to simulate yield response to water: I. concepts and underlying principles. Agronomy Journal 101, 426–437.

UN-WWAP, 2006. The United Nations World Water Development Report 2006 - Water: A shared responsibility. United Nations Educational, Scientific and Cultural Organization (UNESCO), Paris, France.

Vanham, D., Comero, S., Gawlik, B.M., Bidoglio, G., 2018. The water footprint of different diets within European sub-national geographical entities. Nature Sustainability 1, 518–525.

Vanham, D., Hoekstra, A.Y., Bidoglio, G., 2013a. Potential water saving through changes in European diets. Environment International 61, 45–56.

Vanham, D., Medarac, H., Schyns, J.F., Hogeboom, R.J., Magagna, D., 2019. The consumptive water footprint of the European Union energy sector. Environmental Research Letters 14, 104016.

Vanham, D., Mekonnen, M.M., Hoekstra, A.Y., 2013b. The water footprint of the EU for different diets. Ecological Indicators 32, 1–8.

Vörösmarty, C.J., Hoekstra, A.Y., Bunn, S.E., Conway, D., Gupta, J., 2015. Fresh water goes global. Science 349, 478–479.

Vörösmarty, C.J., McIntyre, P.B., Gessner, M.O., Dudgeon, D., Prusevich, A., Green, P., et al., 2010. Global threats to human water security and river biodiversity. Nature 467, 555–561.

Wackernagel, M., Rees, W., 1996. Our ecological footprint: reducing human impact on the Earth, Gabriola Island. New Society Publishers, Gabriola Island, Canada, 176p.

Wöhler, L., Hoekstra, A.Y., Hogeboom, R.J., Brugnach, M., Krol, M.S., 2020a. Alternative societal solutions to pharmaceuticals in the aquatic environment. Journal of Cleaner Production 277, 124350.

Wöhler, L., Niebaum, G., Krol, M., Hoekstra, A.Y., 2020b. The grey water footprint of human and veterinary pharmaceuticals. Water Research X 7, 100044.

Ye, Q., Li, Y., Zhuo, L., Zhang, W., Xiong, W., Wang, C., et al., 2018. Optimal allocation of physical water resources integrated with virtual water trade in water scarce regions: a case study for Beijing, China. Water Research 129, 264–276.

Zhang, Y., Huang, K., Yu, Y., Yang, B., 2017. Mapping of water footprint research: a bibliometric analysis during 2006–2015. Journal of Cleaner Production 149, 70–79.

Zhuo, L., Liu, Y., Yang, H., Hoekstra, A.Y., Liu, W., Cao, X., et al., 2019. Water for maize for pigs for pork: an analysis of inter-provincial trade in China. Water Research 166, 115074.

Zhuo, L., Mekonnen, M.M., Hoekstra, A.Y., 2016a. Benchmark levels for the consumptive water footprint of crop production for different environmental conditions: a case study for winter wheat in China. Hydrology and Earth System Sciences 20, 4547–4559.

Zhuo, L., Mekonnen, M.M., Hoekstra, A.Y., 2016b. The effect of inter-annual variability of consumption, production, trade and climate on crop-related green and blue water footprints and inter-regional virtual water trade: a study for China (1978–2008). Water Research 94, 73–85.

Zhuo, L., Mekonnen, M.M., Hoekstra, A.Y., Wada, Y., 2016c. 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.