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Chapter 2

A review of water stress and water footprint accounting

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Abstract

Production and consumption activities deplete freshwater, generate water pollution, and may further lead to water stress. The accurate measurement of water stress is a precondition for sustainable water management. This paper reviews the literature on physical water stress induced by blue and green water use, and by water pollution. Specifically, we clarify several key concepts (i.e., water stress, scarcity, availability, withdrawal, consumption and water footprint) for water stress evaluation, and review physical water stress indicators in terms of quantity and quality. Furthermore, we identify research gaps in physical water stress assessment, related to environmental flow requirements, return flows, outsourcing of water pollution and standardization of terminology and approaches. These research gaps can serve as venues for further research dealing with the evaluation and reduction of water stress.

Keywords

water footprint; water stress; water scarcity; water quantity; water quality

2.1 Introduction

With rapidly increasing human population, economic growth, expansion of irrigated agriculture and changing consumption patterns, water shortages and pollution have become serious global issues (Ercin and Hoekstra, 2014; He et al., 2018; Liu et al., 2017; Vörösmarty et al., 2000) threatening human health, the living environment and sustainable development (Clarke, 2013; Vörösmarty et al., 2010). Around the world, two billion people live in countries under conditions of high-water stress (UN-Water, 2018), and four billion people experience severe water stress at least one month per year, and 1.8 billion people at least six months per year (Mekonnen and Hoekstra, 2016). Estimates show that by 2050, over half of the global population will live in water-stressed regions (UN, 2015). In addition, millions of people die from diseases associated with water shortage and water pollution every year (Prüss-Üstün et al., 2008; WHO, 2002; WHO and UNICEF, 2015). It is apparent that clean freshwater has become an increasingly limited natural resource for humans. Insufficient or unsuitable available freshwater resources already impact human and ecosystem health, energy supply, food security, and human livelihoods, globally (Bhaduri et al., 2016; UN, 2015; Vanham, 2016; Vörösmarty et al., 2010). Addressing water stress has been an urgent target for the United Nations Sustainable Development Goals (SDGs) (Bhaduri et al., 2016; UN, 2015), with a high level of interdependencies with other SDGs (Blanc, 2015). The World Economic Forum reported that water stress is one of the top risks of today (World Economic Forum, 2015). As such, there is an urgent need to solve water stress problems.

These needs and urgency have triggered a large increase in water stress related research (Liu et al., 2017). But most studies focus on stress induced by blue water use (Liu et al., 2016; Schyns et al., 2019) (refers to water use from surface freshwater and groundwater). Nevertheless, water pollution and green water use also exacerbate water stress (Ma et al., 2020; Schyns et al., 2015; van Vliet et al., 2017). Green water is “the precipitation on land that does not run off or recharge the groundwater but is stored in the soil or temporarily stays on top of the soil or vegetation” (p. 189) (Hoekstra et al., 2011). Green water is a major contributor in the production of food, feed, fibers, timber and bioenergy (Hoekstra and Mekonnen, 2012; Liu et al., 2009; Rost et al., 2008), as green water consumption accounted for 74% of global annual average freshwater consumption in the period 1996–2005, whereas blue water consumption and grey water footprint (freshwater consumed to dilute pollutants to satisfy specific water quality standards) accounted for only 11% and 15%, respectively (Hoekstra and Mekonnen, 2012). Given its importance to the production of agricultural products and the fact that many regions in Europe, Central America, the Middle East, and South Asia encounter green water stress (Schyns et al., 2019), green water cannot be ignored in water stress assessment. Another important component of water stress is induced by discharged polluted wastewater from industry, households and agriculture contributing to water quality deterioration and posing further constraints on water usability (UN-Water, 2011; van Vliet et al., 2017). But only few studies considered water stress induced by water pollution (Liu et al., 2020, 2016; Liu and Zhao, 2020; Ma et al., 2020; van Vliet et al., 2017; Zeng et al., 2013). As a result, water quantity stress induced by blue and green water use and water quality stress induced by pollution provide important and complementary information (Liu et al., 2016).

During the last four decades, there have been a few review papers on how to measure water stress. For example, Brown and Matlock (Brown and Matlock, 2011) reviewed primary water stress indicators and methodologies, providing a brief overview for blue water stress and characteristics for each blue water stress indicator, but without considering green water and water pollution. Liu et al. (Liu et al., 2017) reviewed nine water quantity stress indicators, providing a timeline for the introduction of each water stress indicator and presenting suggestions on incorporating green water, water quality and environmental flow requirements in water stress measurements. Rijsberman (Rijsberman, 2006) reviewed in detail strengths and weaknesses of four blue water stress indicators: per capita water availability, withdrawal-to-availability indicator, physical and economic scarcity indicators and water poverty index. The paper also discusses water stress at global and regional levels and implications for policy. Vanham et al. (Vanham et al., 2018) discussed whether the Sustainable Development Goal's water quantity stress indicator considers a number of essential elements and made suggestions on how to improve blue water stress assessment providing in-depth insights and principles for using and further developing water stress indicators. Damkjaer and Taylor (Damkjaer and Taylor, 2017) reviewed commonly employed blue water stress metrics. They highlighted weaknesses of these indicators and critically examined how to best measure water stress. But their review is mainly concerned with the characterization of water stress in developing countries in tropical areas. Schyns et al. (Schyns et al., 2015) provided a very detailed review on three green water scarcity indicators and 80 green water availability indicators. It was the first review article that focused on green water scarcity broadening the scope of water stress assessment (Schyns et al., 2015).

The aim of this review paper is to give an overview of the literature on how to measure physical water stress. Physical water stress means that there is not enough water (in terms of quantity and quality) to meet all demand (including requirements of ecosystems) (Kummu et al., 2010), and it does not include social, economic and political aspects of water stress. To review how to measure physical water stress, we clarify a few key concepts for water stress evaluation, and then review commonly used physical water stress indicators and summarize approaches to account for water footprints (blue, green and grey water footprints). Finally, issues and gaps are identified to improve water stress assessment and provide future directions for water stress studies. We performed the literature search using the keywords “water stress”, “water scarcity” and “water footprint” through Google Scholar. Not all articles have been incorporated in the text as that would have been too many, but we mainly refer to key contributions, highly cited or the first publication that introduced a certain topic. The literature review was stopped on a specific aspect when 'theoretical saturation' was achieved (Corbin and Strauss, 2014) , or in other words, when references on that topic did not add any new insights.

2.2 Key concepts for water stress measurement

Physical water stress and scarcity are often used interchangeably, or the exact nature of the underlying data is not clearly defined or justified (e.g., water use versus consumption versus withdrawal) (Nouri et al., 2019; Zhang et al., 2016; Zhang and Anadon, 2014). A number of authors have noted that there is no widely accepted definition for physical water stress and scarcity (Distefano and Kelly, 2017; Rijsberman, 2006; White, 2014). We summarize different definitions for physical water stress and scarcity in Table 1.1. It is broadly understood that water has a quantity and quality dimension, both providing two sides of the same coin. Water scarcity is either seen through a quantity (Distefano and Kelly, 2017; Falkenmark et al., 1989; Vanham et al., 2018) or quality point of view (van Vliet et al., 2017), and less frequently from both dimensions (Liu et al., 2017; Liu et al., 2016; Ma et al., 2020; Zeng et al., 2013). Likewise, the concept of water stress has been mainly used to refer to the quantity dimension (Alcamo et al., 2000; Feng et al., 2014a; Pfister et al., 2009; Vanham et al., 2018; White et al., 2015; Zhao et al., 2015), and to a lesser extent to both quantity and quality dimensions (CEO Water Mandate, 2014; Zhao et al., 2016).

Compared with different definitions of water stress and scarcity in Table 1.1, we find that they seem to be converging on the definition given by the CEO Water Mandate (CEO Water

Mandate, 2014), since the CEO Water Mandate compared many different water stress and scarcity definitions and had a lengthy expert consultation on definitions, aimed at reaching an agreement on terminology . CEO Water Mandate defines water scarcity as lack of physical abundance of freshwater resources without considering whether water is suitable for use, and water stress as lack of ability to meet human and ecological demand for freshwater, in terms of water quantity and quality and accessibility to water.

Water availability refers to both available green and blue water (Rockström et al., 2009; Schyns et al., 2015). However, researchers mainly consider blue water availability, almost by default. But green water availability is also important especially for crop production, grazing lands, forestry and terrestrial ecosystems (Hoekstra et al., 2011; Schyns et al., 2015). Thus, blue and green water availability need to be considered in their own right providing complementary information about the water situation. Green water availability is more complex to calculate and refers to "total evapotranspiration of rainwater from land minus environmental green water requirements (the evapotranspiration from land reserved for natural vegetation) minus the evapotranspiration from land that cannot be made productive" (p. 189) (Hoekstra et al., 2011).

While often treated as synonymous, there is an important distinction between water withdrawal (or water abstraction) and blue water consumption (Hoekstra et al., 2011; Wada et al., 2014; Weckström et al., 2020). Water withdrawal is defined as freshwater taken from surface or ground water sources without accounting for how much is returned to the freshwater sources after use, whereas blue water consumption subtracts return flows from water withdrawal (CEO Water Mandate, 2014). Blue water consumption refers to water that is (temporarily) lost from local hydrological systems due to evaporation or being incorporated into a product (CEO Water Mandate, 2014). More specifically, after withdrawal from surface water and groundwater, water is either incorporated into products (blue water consumption); consumed due to evaporation or transpiration (blue water consumption); directly discharged into the environment (return flows); or treated as wastewater and then discharged into the environment (return flows) (Weckström et al., 2020).

Hoekstra et al. (Hoekstra et al., 2011) defined human freshwater consumption in terms of water footprints, which are spatially and temporally explicit indicators, providing information on volume of water consumption, pollution, time and location. Specifically, the blue water footprint refers to volume of surface and groundwater consumed as a result of the production of goods and services (Hoekstra et al., 2011). The volume of rainwater consumed by agricultural and forestry products is referred to as the green water footprint, where it refers to “the total rainwater evapotranspiration (from fields and plantations) plus the water incorporated into the harvested crop or wood” (p 189) (Hoekstra et al., 2011). Hoekstra et al. (Hoekstra et al., 2011) developed a definition of the “grey water footprint” to measure water pollution, which is “the volume of freshwater that is required to assimilate the load of pollutants based on natural background concentrations and existing ambient water quality standards”(p. 190) (Hoekstra et al., 2011). The grey water footprint is essentially a hypothetical measure of water pollution which does not show actual water consumption nor provides any indication of pollution treatment costs (Chenoweth et al., 2013; Gawel and Bernsen, 2011), but it is seen as a good indicator to reflect the extent of water pollution (Hoekstra et al., 2011).To assess blue water stress, water withdrawal and blue water consumption are two very different quantities. To quantify blue water use and blue water stress along supply chains, blue water consumption is usually used rather than water withdrawal (Vanham et al., 2018). The blue water footprint has been introduced to quantify direct water consumption as well as indirect or upstream blue water consumption over the entire supply chain (Hoekstra et al., 2011). To add to the confusion, also blue water footprint based on withdrawal can be frequently found in the literature due to data limitations (Chen and Chen, 2013; Cohen and Ramaswami, 2014).

2.3 Physical water stress indicators

Many indicators have been developed to evaluate water stress since the 1980s when it had been recognized as being a significant issue (Liu et al., 2017). As there is no consensus on how to define and measure water stress, different indicators have been developed. In this section, we focus on reviewing and comparing physical water stress indicators and ignore to discuss socio-economic aspects of water stress, such as economic water scarcity (Seckler et al., 1998) and the water poverty index (Sullivan et al., 2003). For a summary of water quantity and quality stress indicators see Table 1.2.

Table 1.1 Definitions of physical water stress and scarcity

Water scarcity or water stress	Definition	Water quantity or quality dimension	Main References
Water scarcity	The fraction of the total annual run-off available for human use	Quantity	(Brown and Matlock, 2011; Falkenmark et al., 1989)
Water scarcity	“The volumetric abundance, or lack thereof, of freshwater resources” (p. 4)	Quantity	(CEO Water Mandate, 2014)
Water scarcity	“Water scarcity occurs where there are insufficient water resources to satisfy long-term average requirements”	Quantity	(European Environment Agency, 2020)
Water scarcity	“Extent to which demand for water compares to the replenishment of water in an area, e.g., a drainage basin, without taking into account the water quality”	Quantity	(ISO, 2014)
Water scarcity	“The ratio of sectoral water withdrawals of acceptable water quality to the water availability” (p. 1)	Quality	(van Vliet et al., 2017)
Blue water scarcity	“The ratio of blue water footprint to blue water availability” (p. 187)	Quantity	(Hoekstra et al., 2011)
Green water scarcity	“The ratio of green water footprint to green water availability” (p. 190)	Quantity	(Hoekstra et al., 2011)
Water stress	“The impact of high water use (either withdrawals or consumption) relative to water availability” (p. 2)	Quantity	(Kummu et al., 2016)
Water stress	“The symptoms of water scarcity or shortage, e.g., widespread, frequent and serious restrictions on use, growing conflict between users and competition for water, declining standards of reliability and service, harvest failures and food insecurity”	Quantity	(AQUASTAT)
Water stress	“Water stress occurs when the demand for water exceeds the available amount during a certain period or when poor quality restricts its use. Water stress causes deterioration of freshwater resources in terms of quantity (aquifer over-exploitation, dry rivers, etc.) and quality (eutrophication, organic matter pollution, saline intrusion, etc.)”	Quantity and quality	(European Environment Agency)
Water stress	A logistic function of the ratio of total annual freshwater withdrawals to hydrological availability, and water stress ranges from 0 to 1	Quantity	(Pfister et al., 2009)
Water stress	“The ability, or lack thereof, to meet human and ecological demand for fresh water” (p. 4)	Quantity and quality	(CEO Water Mandate, 2014)
Blue water stress	“The ratio of total fresh water withdrawn by all sectors to the water availability (total renewable freshwater resources minus environmental flow requirements) in a particular country or region” (p. 219)	Quantity	(Vanham et al., 2018)

2.3.1 Per capita water availability

The per capita water availability index was introduced by Falkenmark et al. (Falkenmark et al., 1989) to evaluate blue water stress, and is defined as the fraction of total annual run-off available for human use (Falkenmark et al., 1989). Falkenmark's indicator is simple and intuitive, and data on human population and annual run-off within regions are readily available. These merits make the Falkenmark indicator one of the most widely used blue water scarcity indicators (Rijsberman, 2006). As green water is important especially for agricultural products (Schyns et al., 2019), this indicator has also been applied in the evaluation of green water scarcity, expressed as the per capita available green water resources in a region (Rockström et al., 2009; Schyns et al., 2015). A combination of green and blue water scarcity has been developed to assess water stress caused by both green and blue water (Rockström et al., 2009). If green and blue water consumption in a certain area is less than the global average level of 1,300 m^3/cap/year, then the area is considered as being water scarce (Rockström et al., 2009). Similarly, a combination of green and blue water consumption was used as part of the planetary boundaries concept that quantifies environmental limits within which humans can safely operate, with global maximum water consumption of 4,000 km3/year (Gerten et al., 2015; Rockström et al., 2009; Steffen et al., 2015). Due to the simplicity of the indicator 'per capita water availability', there are a few inherent problems with this indicator: (i) this national and annual based indicator hides scarcity information at smaller spatial and temporal scales (Brown and Matlock, 2011; Rijsberman, 2006); (ii) the thresholds ignore variations in demand among countries because of factors such as technology, lifestyle, and climate (Brown and Matlock, 2011; Rijsberman, 2006; Savenije, 2000); (iii) the indicator treats per capita water availability as a fixed requirement per person (Rijsberman, 2006) and (iv) it ignores important variations in demand and consumption among countries.

2.3.2 Withdrawal-to-availability ratio

The withdrawal-to-availability ratio is also referred to as a criticality ratio and is defined as "the ratio of average annual water withdrawals to water availability" (p 21) (Alcamo et al., 2000). This ratio is a commonly used water quantity stress indicator, and has been widely applied to measure blue water stress in a given area (Liu et al., 2017; Vanham et al., 2018). A region can be categorized into severe water scarcity (> 40%), water scarcity (20%-40%), moderate water scarcity (10%-20%) and low water scarcity (< 10%) (Alcamo et al., 2000;

Raskin et al., 1997). The threshold values have been adopted by the United Nations (UN, 1997), European Environment Agency (EEA, 2009) and numerous studies (e.g. (Oki et al., 2001; Seckler et al., 1999; Vörösmarty et al., 2000)). The development of hydrological models in the past decades allows the modelling of blue water withdrawal and availability globally at a high spatial resolution (Flörke et al., 2013; Hanasaki et al., 2008; Wada et al., 2014), facilitating withdrawal-to-availability ratio-based assessments.

Pfister et al. (Pfister et al., 2009) modified the withdrawal-to-availability ratio by using a logistic function providing water quantity stress levels at continuous values between 0 and 1, in order to use the water quantity stress index as a characterization factor to calculate environmental impacts induced by water withdrawal in a life cycle assessment model. Pfister's water quantity stress index has also been linked to input-output models to convert water footprints into water stress footprints through weighing water footprints with water stress factors reflecting availability at the place of water extraction at high spatial resolution.

2.3.3 Water footprint-to-availability ratio

It has been frequently argued that it makes more sense to use blue water consumption rather than water withdrawal to assess blue water stress (Hoekstra, 2017; Hoekstra et al., 2012), as the majority of the extracted blue water becomes eventually return flows, with only a small proportion of the actual water withdrawal being consumed. For example, in agriculture, 40% of withdrawn water returns to rivers and lakes, and 90%-95% of water withdrawn by industry and households flows back to nature (Hoekstra et al., 2012). Thus consumption-to-availability is seen as better suited to assess blue water stress than the withdrawal-to-availability ratio (Hoekstra et al., 2011). The consumption-to-availability ratio is generally used to measure blue water scarcity, which is the ratio between blue water consumption and availability. But green water scarcity and water pollution level defined by Hoekstra et al. (Hoekstra et al., 2011) are also expressed as green/grey water footprint to availability ratio.

Limited green water aggravates water stress (Schyns et al., 2019, 2015). Even though green water has properties of both quantity and quality, the definition of green water just describes quantity. The quality of green water is determined by soil properties such as the concentration or retention capacity of nutrients and toxic substances (Schyns et al., 2015). At present, there is no relevant research on green water quality induced stress. The reason may be that green water quality in the soil is difficult to measure and the usefulness of green water quality assessment for practical purposes remains unclear. Green water scarcity specifically refers to green water quantity stress, and is defined as the ratio of the green water footprint to green water availability (Hoekstra et al., 2011). As measurements of green water availability and environmental green water requirements are difficult to obtain, the application of green water scarcity has received only limited attention in the literature (Hoekstra et al., 2011; Liu et al., 2020; Schyns et al., 2019, 2015). The lack of opportunity costs of green water, i.e., the lack of competing uses in other sectors, is often stated as a reason to ignore green water (Konar et al., 2012; Yang et al., 2006), which is thus considered as being less important for water management than blue water (Hess, 2010).

For water quality stress, there is one major indicator defined as the ratio of grey water footprint to the actual run-off, also referred to as water pollution level (Hoekstra et al., 2011). This indicator can be applied to assess water quality stress of freshwater (Ma et al., 2020; van Vliet et al., 2017) or discharged wastewater from primary, secondary and tertiary economic sectors (Liu et al., 2017; Wan et al., 2016; Zeng et al., 2013; Zhao et al., 2016). If water quality stress is over 100%, it indicates that available blue water resources cannot dilute polluted water to satisfy water quality standards (Hoekstra et al., 2011). As it is a hypothetical indicator, the usefulness of the grey water footprint has been frequently questioned (Chenoweth et al., 2013; Yang et al., 2013), yet these arguments have not abated the popularity of this indicator nor its frequent application (Guan and Hubacek, 2008; Liu et al., 2016; Mekonnen and Hoekstra, 2018, 2015; van Vliet et al., 2017; Wan et al., 2016; Zeng et al., 2013; Zhao et al., 2016).

Calculating the grey water footprint is a precondition to measure water quality stress. Traditional grey water footprint calculations are based on individual pollutants by selecting the highest grey water footprint of an individual pollutant. However, due to limited water quality data availability (Liu et al., 2017), it is difficult to calculate the grey water footprint for all relevant pollutants, then to select the highest one. To solve this problem, major representative chemical water pollutants such as COD (Chemical Oxygen Demand) for measuring organic matters and NH_3-N (Ammonia Nitrogen), TN (Total Nitrogen) and TP (Total Phosphorus) for nutrient pollution are the most frequently used pollutants to express water quality (Liu et al., 2017; Ma et al., 2020; van Vliet et al., 2017; Wan et al., 2016; Zeng et al., 2013; Zhao et al., 2016), as these pollutants are commonly monitored for freshwater (Ouyang et al., 2006; Shrestha and Kazama, 2007; Simeonov et al., 2003) or discharged wastewater

(Carey and Migliaccio, 2009; Sun et al., 2016). In addition, physical parameters such as temperature and salinity (or electric conductivity) sometimes are also employed to evaluate water quality stress of surface water(Ma et al., 2020; van Vliet et al., 2017).

Table 1.2 Summary of indicators to measure physical water stress

Indicators	Quantification	Input indicators	Main references
Water quality stress (water pollution level)	The ratio of grey water footprint to the actual run-off	(1) Grey water footprint (2) Blue water resources	(Hoekstra et al., 2011; Ma et al., 2020; van Vliet et al., 2017)
Falkenmark indicator	Per capita annual run-off	(1) Population (2) Blue water resources	(Falkenmark et al., 1989)
Criticality ratio	The ratio of withdrawal to blue water resources	(1) Water withdrawal (2) Blue water resources	(Alcamo et al., 2000)
Pfister's water quantity stress index	A logistic function of the ratio of water withdrawal to blue water resources	(1) Water withdrawal (2) Blue water resources	(Pfister et al., 2009)
Blue water scarcity	The ratio of blue water footprint to blue water availability (run-off minus environmental flow requirements)	(1) Blue water footprint (2) Run-off (3) Environmental flow requirements	(Zeng et al 2013; Falkenmark et al., 1989)
Green water scarcity	Per capita available green water resources	(1) Population (2) Green water resources	(Schyns et al., 2015)
Green water scarcity	The ratio of green water requirements for self-sufficiency to green water availability	(1) Green water requirements for self-sufficiency (2) Green water resources	(Schyns et al., 2015)
Green water scarcity	The ratio of green water footprint to green water availability (total green water resources minus environmental green water requirements)	(1) Green water footprint (2) Green water resources (3) Environmental green water requirements	(Hoekstra et al., 2011; Schyns et al., 2019)

However, only focusing on a representative and specific pollutant leaves other pollutants and their interactions unexamined (Martínez-Alcalá et al., 2018; Wöhler et al., 2020), which may lead to an underestimation of the grey water footprint and water quality stress. For instance, Vale et al. (Vale et al., 2019) pointed out that many studies on agricultural products, only considering fertilizers but ignoring pesticides, underestimate the grey water footprint. Similarly, when looking at industrial pollution, cadmium, copper and mercury are critical pollutants for steel production, whereas cadmium is a critical pollutant for cement, and for glass, a critical pollutant would be suspended solids (Gerbens-Leenes et al., 2018). In contrast, most studies on industrial sectors just chose the same conventional pollutants (i.e., COD and ammonia nitrogen) (Ma et al., 2020; Zhao et al., 2016). Thus, there is a need to investigate critical water pollutants for different processes or sectors providing a foundation for more accurate assessment of the grey water footprint and water quality stress.

To address this deficiency, cumulative effects of multiple pollutants have been the focus in recent studies (Li et al., 2019; Liu et al., 2017; Yu et al., 2020), by combining footprint accounting with other tools, such as commonly used water quality evaluation tools (Yu et al., 2020), mass-balance models and fuzzy synthetic evaluation models (Li et al., 2019). But an important limitation of modified grey water footprint methods lies in the selection of pollutants, which requires calculation and analysis of pollutant data in advance (Yu et al., 2020). For this reason, traditional grey water footprint accounting is still dominating water quality stress assessments.

2.4 Water footprint accounting

2.4.1 Bottom-up approaches

Water footprint assessment (WFA) (Hoekstra et al., 2011) and life cycle assessment (LCA) (Guinée and Lindeijer, 2002) are based on detailed data of individual processes and thus can be classed as bottom-up approaches (Feng et al., 2011). WFA and LCA share a generic framework: setting goals and scope; accounting phase; impact assessment phase and interpretation (Boulay et al., 2013). But at different stages, they are distinguished from each other. WFA and LCA serve different goals, as life cycle assessment is a product-focused method, aiming to achieve sustainability of products, while water footprint assessment is a water management approach with a focus on the sustainability of water resources (Boulay et al., 2013; Matuštík and Kočí, 2020). Water footprint assessment was developed with a focus on agricultural sectors and food production processes (Yang et al., 2013) at its early stage, then extended to industrial sectors, with currently focusing on agricultural and forestry-based production (such as paper, dairy and textile) and energy production (such as bioenergy and electricity) (Zhuo et al., 2020), whereas LCA had its starting point with a focus on industrial products and sectors. At the accounting stage of calculating the water footprint, the LCA and WFA communities have an ongoing debate about green and grey water footprint accounting. The WFA community measures blue, green and grey water footprints, but LCA only includes the blue water footprint (Boulay et al., 2013). The LCA community argues that only net green water (the difference between cropland and natural vegetation) should be counted (Matuštík and Kočí, 2020), but it would lead to net negative green water footprints, because evapotranspiration of natural vegetation might be larger than that of croplands (Matuštík and Kočí, 2020).Thus, it is unusual to account for green water footprints in LCA. The other issue is with regard to grey water footprint accounting, as the LCA community argues that different indicators such as acidification, eutrophication or toxicity potential are better suited to measure water pollution (Yang et al., 2013). Even though LCA does not measure the grey water footprint directly, LCA databases have been applied in water footprint assessments to estimate grey water footprints (Gerbens-Leenes et al., 2018). For instance, LCA databases such as GaBi, Ecoinvent and Quantis can provide water pollution data for WFA (Feng et al., 2014b; Gerbens-Leenes et al., 2018), and also provide information on up-stream processes (Kounina et al., 2013; Paterson et al., 2015).

2.4.2 Top-down approaches

Top-down approaches provide a framework for quantification of environmental burdens based on national accounts that allows linking the entire supply chain in a production web to final consumption using macro-level approaches and concepts to analyze footprints of individuals, companies, sectors or regions (Castellani et al., 2019). At its core is the input-output (IO) table depicting monetary flows of goods and services among different economic sectors through trade (Miller and Blair, 2009). IO analysis provides detailed flows between production and consumption at the level of economic sectors provided in national accounts of statistical offices. The input-output model with economic multipliers at its core allows not only the direct effects of environmental impacts through changes in final consumption but also the round-by-round or indirect effects of subsequent expenditures and inputs to each layer of production. This mechanism coupled with environmental extensions such as water consumption per sector enables the analyst to calculate the water footprint throughout the entire supply chain (Lenzen and Foran, 2001; Velázquez, 2006).Such top-down water footprint accounting has a long history in input-output analysis (Hartman, 1965; Isard, 1972), even before the term water footprint was coined in 2002. Environmentally extended input-output analysis (EE-IO) is the most widely used top-down approach to calculate water consumption (Feng et al., 2011; Hubacek et al., 2009; Zhao et al., 2015).

Multi-Regional Input-Output (MRIO) analysis (Acquaye et al., 2017; Ewing et al., 2012; Miller and Blair, 2009) and water embodied in bilateral trade (Feng et al., 2011; Peters et al., 2011; Zhao et al., 2015) are both input-output approaches that can be used to calculate water consumption. But the difference is that MRIO analysis traces global supply chains, while the method of water embodied in bilateral trade just traces domestic supply chains and imports. Thus, this difference may lead to inter-regional and international cut-off effects (Feng et al., 2011) and thus, wrong allocation of footprints (Hubacek and Feng, 2016).

IO and MRIO provide the most complete information about supply chains, whereas bottom-up approaches are not able to capture entire industrial supply chains, by focusing only at the most important processes, leading to inter-sector cut-off compared with top-down approaches (Feng et al., 2011; Hoekstra, 2017). System boundaries of bottom-up approaches can lead to double counting (Lenzen, 2008) as well as to excluding important flows (Daniels et al., 2011). But a major drawback of input-output analysis is that economic sectors are aggregated and cannot show detailed process information compared with bottom-up approaches (Crawford et al., 2018; Suh and Huppes, 2005).

So-called hybrid analysis, linking process-based LCA and IO, combines advantages of both LCA and IO. There are three major approaches for hybrid analysis: tiered hybrid analysis (Joshi, 1999; Lenzen, 2009; Strømman et al., 2009; Suh and Huppes, 2005, 2002), which essentially combines process and input-output data within a process analysis framework in order to reduce truncation error of a pure process analysis (Crawford et al., 2018), IO-based hybrid analysis (Dixit, 2017; Dixit et al., 2015; Joshi, 1999; Suh and Huppes, 2005), which is performed by disaggregating industry sectors in the IO table to a more useful resolution, and integrated hybrid analysis (Feng et al., 2014b; Suh and Huppes, 2005; Wiedmann et al., 2011), which is carried out by connecting an input-output table to a technology matrix based on process data. These hybrid approaches have found a range of applications, for instance to quantify water consumption for wind power (Li et al., 2012), shale gas (Gao and You, 2018), other electricity generation technologies (Feng et al., 2014b), transport fuels (Harto et al., 2010), and non-electric energy (transport and heating energy) (Liu et al., 2018).

2.5 Issues and gaps in water stress research

2.5.1 Lack of standardized approaches to estimate environmental flow requirements

There is a slowly growing recognition that environmental flow requirements, which are "the quality, quantity, and timing of water flows required to maintain the components, functions, processes, and resilience of aquatic ecosystems" (Poff et al., 2010), need to be considered in the evaluation of water quantity stress (Liu et al., 2017; Vanham et al., 2018). However, current water quality stress evaluation seldom considers environmental flow requirements (Mekonnen and Hoekstra, 2018, 2015; Wan et al., 2016; Zhao et al., 2016). Even though there are more than 200 methods to estimate environmental flow requirements, which can be classified into hydrological methods; hydraulic rating methods; habitat simulation methods; and holistic methods (Pastor et al., 2014), it is still common to just use a certain percentage of total water resources as a proxy (Hoekstra et al., 2012; Liu et al., 2016; Ma et al., 2020; Mekonnen and Hoekstra, 2016; Munia et al., 2020; Pastor et al., 2014; Richter et al., 2012), because calculating environmental flow requirements for a catchment is always an elaborate task (Hoekstra et al., 2011). However, there are huge uncertainties in these estimates (Liu et al., 2016) and they cannot easily be transferred to different geographic contexts. The choice of these proxies may have considerable effects on outcomes of water stress evaluation and associated recommendations. For example, Liu et al. (Liu et al., 2016) compared water quantity stress based on seven recommended proxies of environmental flow requirements which maintain different habitat quality of aquatic ecosystems and it turned out that water quantity stress estimates are quite different under seven recommended environmental flow requirements. In addition, even less consideration is given to the water quality of environmental flow requirements, ignoring the fact that ecosystems require not only a certain quantity of water but also a minimum water quality to maintain different purposes and functions of freshwater ecosystem.

2.5.2 Overlooked return flows

In most cases, the water quality of return flows deteriorates during water use in production and consumption processes, contributing to water quality stress in a region. Despite its importance for water quality assessment, return flows have not received much attention in the literature. To mitigate water stress, deteriorated return flows can be treated by wastewater treatment facilities (Voulvoulis, 2018), rather than discharging return flows directly into the water body. At the same time, highly developed wastewater reuse technologies make it possible to recycle wastewater and alleviate blue water quantity stress. However, contributions of wastewater treatment and wastewater reclamation to reduce water quantity and quality stress are usually outside the system boundary of grey water footprint or water stress assessments. It is necessary to identify how flows return to nature and how quantity and quality of return flow changes on the way. Understanding return flows would be helpful to reduce water quantity and quality stress. For agricultural diffuse pollution, it is even more of a challenge to trace return flows and measure their impacts (Hoekstra et al., 2011) despite the existence of models used to estimate the fraction of agricultural nitrogen pollutant that enter into water bodies (Hoekstra et al., 2011).

2.5.3 Lack of teleconnections and outsourcing of water pollution

Another important issue related to the concept of footprints and especially virtual flows of water use and pollution is the problem of system boundaries and outsourcing of water problems to other areas. Resource consumption and environmental emissions are embedded in global supply chains via complex trade networks (Hubacek et al., 2014). Therefore, local consumption activities may cause environmental impacts and resource depletion elsewhere, because of the role of teleconnections, which describe the remote linkages between local consumption and remote environmental impacts (Hubacek et al., 2014; Yu et al., 2013). Regions can shift water quantity and quality stress to other areas by importing water-intensive products and outsourcing water pollution. There are numerous studies on virtual water trade and reduction of domestic water quantity stress (Cai et al., 2019; Feng et al., 2014a; Munoz Castillo et al., 2019, 2017; White et al., 2015). But there are only a few studies on outsourced water pollution and tele-connected water quality stress (Wu and Ye, 2020), due to challenges in quantification of pollution loads, availability of water pollution data and lack of clarity on how to best link water quality stress to the IO approach. Furthermore, both water quality stress locally and in remote areas should be considered at the same time to avoid spatial spillover effects and other unintended side effects. A focus on only one of them will lead to potentially severe underestimation of environmental impacts. But studies on considering water quantity and water quality stress simultaneously are still rare (Liu et al., 2017; Ma et al., 2020; Zeng et al., 2013; Zhao et al., 2016). Making clear how water stress (both quantity and quality) is tele-connected would lay a foundation for policy making and implication for water stress alleviation from trans-regional and trans-basin perspectives.

2.5.4 Lack of standardization of definitions and approaches

There is still a lack of standardization in terms of definitions of basic concepts, objects and scope of the analysis and selected indicators. For instance, water quantity induced stress receives much attention while water quality stress is often overlooked (Ma et al., 2020; van

Vliet et al., 2017). This is partly due to the fact that many definitions of water stress or scarcity do not consider pollution induced water stress, which influences the subsequent selection of indicators (Liu et al., 2017). Another example is the indiscriminate use of the terms water withdrawal and water consumption, which may lead to quite different results in water stress assessment because of their treatment of return flows. This difference may also have an impact on subsequent method selection. These two cases serve as examples of the importance of a wider agreement on key aspects of water stress assessment. These are preconditions to achieve more accurate and comparable results and conclusions and would thus be better suited to support evidence-based policy making and implementation.

2.6 Conclusions

Water stress poses an important societal challenge. In this paper, we reviewed how to measure physical water stress, which is an important phase of water stress assessment. We defined and discussed a number of key concepts, i.e., water stress, scarcity, availability, consumption, withdrawal and the water footprint. Then we summarized three types of physical water stress indicators, covering water stress induced by quantity and quality. Subsequently, we reviewed bottom-up and top-down approaches to account for water footprints. Finally, we summarized issues and gaps in current water stress assessment: estimation of environmental flow requirements, outsourcing and teleconnections of water pollution, accounting for return flows, and standardization for water stress assessment. These problem areas can serve as venues for further research dealing with the evaluation and reduction of water stress.

References Chapter 2

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