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A critique on the water-scarcity weighted water footprint in LCA

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ABSTRACT

The water footprint (WF) has been developed within the water resources research community as a volumetric measure of freshwater appropriation. The concept is used to assess water use along supply chains, sustainability of water use within river basins, efficiency of water use, equitability of water allocation and dependency on water in the supply chain. With the purpose of integrating the WF in life cycle assessment of products, LCA scholars have proposed to weight the original volumetric WF by the water scarcity in the catchment where the WF is located, thus obtaining a water-scarcity weighted WF that reflects the potential local environmental impact of water consumption. This paper provides an elaborate critique on this proposal. The main points are: (1) counting litres of water use differently based on the level of local water scarcity obscures the actual debate about water scarcity, which is about allocating water resources to competing uses and depletion at a global scale; (2) the neglect of green water consumption ignores the fact that green water is scarce as well; (3) since water scarcity in a catchment increases with growing overall water consumption in the catchment, multiplication of the consumptive water use of a specific process or activity with water scarcity implies that the resultant weighted WF of a process or activity will be affected by the WFs of other processes or activities, which cannot be the purpose of an environmental performance indicator; (4) the LCA treatment of the WF is inconsistent with how other environmental footprints are defined; and (5) the Water Stress Index, the most cited water scarcity metric in the LCA community, lacks meaningful physical interpretation. It is proposed to incorporate the topic of freshwater scarcity in LCA as a “natural resource depletion” category, considering depletion from a global perspective. Since global freshwater demand is growing while global freshwater availability is limited, it is key to measure the comparative claim of different products on the globe’s limited accessible and usable freshwater flows.

1. Introduction

The water footprint (WF) concept was first presented at an international expert meeting on virtual water trade in December 2002 in Delft, the Netherlands (Hoekstra, 2003). The increasing amount of work on water use and scarcity in relation to consumption and trade has led to the emergence of the field of Water Footprint Assessment (WFA). Methodological advances over the past decade include the development of the four-step WFA methodology (setting scope of analysis, accounting, sustainability assessment and response formulation; Hoekstra et al., 2009a, 2011), the development of grey WF guidelines (Franke et al., 2013), the estimation of WFs at high spatial and temporal resolution (Mekonnen and Hoekstra, 2010), the exploration of the evolution of the global virtual water trade network (Dalin et al., 2012), the development of WF benchmarks for crops (Mekonnen and Hoekstra, 2014), the estimation of blue water scarcity in river basins based on blue WFs (Hoekstra et al., 2012), the assessment of water pollution levels in river basins based on nitrogen and phosphorus-related grey WFs (Liu et al., 2012; Mekonnen and Hoekstra, 2015), studying inter-annual variability of WFs (Sun et al., 2013), assessing WF uncertainties (Zhuo et al., 2014), the exploration of the use of remote sensing (Romaguera et al., 2010) and the development of future WF scenarios (Ercin and Hoekstra, 2014). Applications of the WF vary widely, from product assessments (Chapagain et al., 2006), sector studies (Mekonnen et al., 2015), diet assessments (Vanham et al., 2013), national studies (Ercin et al., 2013), catchment studies (Zeng et al., 2012) to global assessments (Hoekstra and Mekonnen, 2012a).

Since 2009 the life cycle assessment (LCA) community has shown interest in the WF concept, because of its relevance in comparing the environmental performance of products. The WF as developed and applied within the water resources research community has received criticism from the LCA community for not appropriately accounting for differences in potential

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environmental impact of water use given regional differences in water scarcity. There has been some exchange of letters between both communities (letter from Pfister and Hellweg, 2009; reply from Hoekstra et al., 2009b; letter from Ridoutt and Huang, 2012; reply from Hoekstra and Mekonnen, 2012b) and there have been efforts to come to fruitful exchange (e.g. Boulay et al., 2013), which however have not been very successful in bringing the two communities closer together, as apparent for instance from the letter by Pfister and Ridoutt (2014). They advise the water resources community to "update" the WFA methodology to bring it in line with their own LCA work. The problem, however, is that the LCA community has taken its own path of development, in a direction that can contribute little to improve water management and is therefore not very interesting for the water resources community. There is a vast body of literature in the field of WFA, which has evolved over time, but shows consistency and coherence. The water resources community has been little responsive to the demands from the LCA community to change direction and adapt methods to perceived LCA needs. There is a good reason for that: the so-called "better" approach to water footprinting as proposed in the LCA community is not better. It is narrowly focussed on assessing potential environmental impacts of products, while the broader issue of sustainable, efficient and equitable allocation of limited freshwater resources from catchment to global level remains out of scope. There is no reason to change a WFA method that is consistent with other environmental footprint methods (Giljum et al., 2011; Galli et al., 2012; Hoekstra and Wiedmann, 2014; Fang et al., 2014, 2015) and suitable to address big questions on water resources allocation (Hoekstra, 2013, 2014) merely to fit the specific goal of LCA. It should be the other way around: the incorporation of freshwater into LCA could better be based on insights as developed within the water resources science community. The current paper aims to supply a critique on the water-scarcity weighted WF approach, which is at the core of what LCA authors propose and at the basis of the dispute with the water resources research community. The need for this critique has gained urgency now that the idea of the water-scarcity weighted WF has been adopted in ISO's LCA-based WF standard (ISO, 2014).

Section 2 explains the concepts and definitions applied by the two communities. Section 3 contains the actual critique on the water-scarcity weighted WF approach as promoted by LCA authors. Section 4 concludes with a reflection on the way forward.

2. Concepts and definitions

2.1. The volumetric WF in water resources studies

The WF is a measure of consumptive and degradative freshwater use. The consumptive WF includes a green component, which refers to the consumption of rainwater, and a blue component, which refers to the consumption of surface- or groundwater (Hoekstra et al., 2011). The inclusion of the green WF enables the broadening of perspective on water resources beyond the historical focus of water engineers on blue water (Falkenmark and Rockström, 2004). The degradative WF, the so-called grey WF, represents the volume of water required to assimilate pollutants entering freshwater bodies (Hoekstra et al., 2011), an idea that builds on the concept of dilution water requirement earlier applied by Postel et al. (1996).

The WF of one single process step (a unit process) is the basic building block of all WF accounts (Hoekstra et al., 2011). The WF of a product is the sum of the WFs of the process steps taken to produce the product. The WF of a business is the sum of the WFs of the final products produced by the business, which includes the operational WF of the business as well as its supply-chain WF. The WF of a consumer is the sum of the WFs of all products consumed. The WF of national consumption is the sum of the WFs of the country's inhabitants, which includes an internal component (the WF within the national territory for making products that are consumed within the country) and an external component (the WF in other countries for making products imported by and consumed within the country considered). The external WF of national consumption is made possible by import of water-intensive commodities. This trade implies so-called virtual water flows between exporting and importing countries (Hoekstra, 2003). Finally, the total WF within a certain area (e.g. a municipality, province or state, or a hydrological unit like a catchment area) is sum of the WFs of all processes taking place within the area.

The WF concept introduced supply-chain thinking in the field of water management and is helpful in analysing the link between human consumption and the appropriation of freshwater (Hoekstra et al., 2011). The concept is used to assess water use along supply chains, sustainability of water use within river basins, efficiency of water use, equitability of water allocation and reliance on external water supplies or dependence on water in the supply chain. The sustainability of water use can be evaluated by comparing the WF within an area to the maximum sustainable WF in that area (Hoekstra, 2014). Efficiency of water use can be assessed by comparing the WF of a specific process or product to a WF benchmark for that process or product, which can be based on best available technology and practice (Mekonnen and Hoekstra, 2014; Chukalla et al., 2015). Equitability of water use can be discussed by comparing the WFs related to the consumption levels and patterns of different communities (Seekell, 2011; Hoekstra, 2014). Water dependency and security can be assessed by analysing the extent to which companies or communities depend on unsustainable water use in their supply chain (Ercin et al., 2013). Common in the various types of analysis is the study of how water volumes are allocated to competing demands. Counting water volumes is a key element, which explains the unease in the water resources community to talk in terms of "weighted cubic metres" of water, which is seen as key in the LCA community.

2.2. The water-scarcity weighted WF in LCA

Critique on the volumetric WF and the idea to weight consumed water volumes based on local water scarcity emerged in 2009 (Ridoutt et al., 2009; Pfister et al., 2009; Ridoutt and Pfister, 2010). The proposal had enormous traction within the LCA community, which had just started to ask how water use could be incorporated into LCA (Koehler, 2008; Milà i Canals et al., 2009). The rationale is straightforward. The purpose of LCA studies is to estimate the different sorts of potential environmental impact attributable to the life cycle of a product, from cradle to grave (Hellweg and Milà i Canals, 2014). An LCA is a *comparative* analysis of potential environmental impacts of alternative processes or products, for instance when using alternative materials or designs (Rebitzer et al., 2004). A life cycle inventory (LCI), which compiles natural resources use and emissions for each process in the life cycle of a product, is followed by a life cycle impact assessment (LCIA), which includes a selection of the "environmental impact categories" of interest, a calculation of "impact category indicators" based on inventory data using "characterisation factors" (characterisation) and optionally a calculation of "impact category indicator results" relative to reference values (normalisation) and a grouping and/or weighting of the results (Pennington et al., 2004).

The carbon footprint (CF) is one of the popular "impact category indicators", for the impact category of climate change. The emissions of different greenhouse gases are weighted based on their "global warming potential" (GWP) relative to carbon dioxide (e.g. one kg of methane has a much greater GWP than one kg of carbon dioxide). The weighting is technically called "characterisation" of the inventory results, and the GWPs of different greenhouse gases are the characterisation factors. The resultant CF is expressed in terms of tonnes CO_2-equivalents.

The idea to calculate a water-scarcity weighted WF in the LCIA stage has been inspired by the way the CF has been incorporated into LCA procedures (Pfister and Hellweg, 2009; Ridoutt and Pfister, 2010). An implicit choice made – which is an important point to which I will come back in the next section – was that water use itself is not interesting, but that the focus should be on the potential environmental impact of water use. Given that focus, the question was how to "characterize" water consumption, i.e. what characterization factor to use. The logic followed was that one litre of water consumption in a water-scarce basin is worse than the same water consumption in a water-abundant basin, hence the proposal to use water scarcity as the characterization factor of water consumption. Using water scarcity as a characterization factor for water consumption was presented as an analogy to using GWP as a characterization factor of greenhouse gas emissions – which isn't a proper analogy though, as I will show in the next section.

The above was all about the blue WF. The green WF was put aside as irrelevant, because the green WF doesn't affect runoff and therefore not (blue) water scarcity (Pfister and Hellweg, 2009; Ridoutt and Pfister, 2010). The grey WF has received mixed responses, because on the one hand the grey WF has been regarded as already including proper characterization (because loads of different chemicals are made comparable by calculating the water volume needed to assimilate them based on the difference between maximum allowable and natural concentration of the chemical), while on the other hand there is some unease with the grey WF because it overlaps with some other existing environmental impact categories (like eutrophication).

3. A critique on the water-scarcity weighted WF

3.1. History and background

The points of critique that will be elaborated in the next sections follow from a core assumption made by LCA authors regarding what constitutes the essence of the water use problem. The LCA method aims to consider both input-related impact categories (natural resource depletion) and output-related impact categories (pollution) (Udo de Haes, 2002), but recent LCA literature doesn't make this distinction and rather speaks generally about "environmental impact categories" (Hellweg and Milà i Canals, 2014). The issue of water use is treated as a category of environmental impact, whereby insufficient thought has been on what actually is the environmental issue. Obviously, the issue covers both water depletion and pollution, which makes it a heterogeneous category. But the LCA community has primarily jumped on the issue of consumptive water use (given that water pollution is already partially covered through other environmental impact categories like eutrophication) and thereby focussed on "impact of water consumption". The latter implies that water use in itself isn't regarded as the environmental issue, but rather "environmental impact of water use". The issue of water depletion (which fractions of the available green and blue water resources and which part of the available waste assimilation capacity are already appropriated) is thus ignored.

A logic consequence was that the LCA community wasn't satisfied with a metric of just water use (the volumetric green and blue WFs) and started searching for a metric that can represent the environmental impact of water use. Confusingly, LCA scholars took the existing WF, which had been defined as a *water use* indicator, and started to criticise it for not being a *water use impact* indicator. The reasoning was that the WF concept had to be transformed into another metric, to serve the purpose of a water use impact indicator. This development has been unfortunate, because it would have been better if another term had been chosen, e.g. a WF impact index, as proposed by the Water Footprint Network (Hoekstra et al., 2011). It would have prevented the current dispute over terminology. There was an important reason for the water resources research community to stay with the WF as an indicator of water use: the WF had been developed and used to feed discussions about sustainable, efficient and equitable allocation of limited freshwater resources and about resource security, given that many countries depend on water resources outside their territory. The essence of allocating limited water resources is about allocating litres among competing human purposes and about allocating within sustainability limits, respecting environmental water needs. If properly allocated, within sustainability limits, respecting environmental water needs, the environmental impacts will remain within acceptable limits. The issue of environmental impacts is thus part of the larger theme of sustainable, efficient and equitable allocation of limited water resources. Redefining the WF concept to refer to environmental impact of water use is not instrumental to this larger theme. Allocation is about allocating litres, not about allocating scarcity-weighted litres.

3.2. The environmental relevance of water productivity and WFs in water-rich areas

The proponents of the water-scarcity weighted approach have persistently pointed at the need for "environmental relevance" of the WF indicator (Pfister and Hellweg, 2009; Ridoutt and Huang, 2012; Berger and Finkbeiner, 2013). This has been interpreted as: it should reflect environmental impact of water use. Blue water consumption in a water-scarce catchment is regarded to have potential environmental impact, because it reduces runoff and may affect downstream ecosystems and livelihoods. A similar amount of blue water consumption in a water-rich catchment has less impact and would therefore have smaller environmental relevance. Green water consumption does hardly change runoff (since evaporation from a farmland or production forest is in the same order of magnitude as evaporation from natural vegetation) and would therefore have no environmental relevance at all. The problem with this reasoning, however, is that the term "environmental relevance" is interpreted in a too narrow sense.

A substantial component of the solution to overexploitation of blue water resources in water-scarce regions is to use (green and blue) water resources in water-rich regions more productively, because producing more water-intensive products where water is sufficient takes away the need to produce those products in places where water is scarce. Improving land and water productivity in rain-fed agriculture in all those regions with sufficient rain would reduce the need for irrigated agriculture in regions that are basically unsuitable for crop production given the limited availability of water (Rockström et al., 2009). By focussing on blue water consumption in water-scarce basins, one overlooks two important features of water: (1) water is a global resource: water-intensive commodities can be traded from water-rich to water-poor river basins, which means that where in the world water is being used and how much is partly subject to the working of the global economy (Hoekstra and Hung, 2005; Hoff, 2009; Vörösmarty et al., 2015); and (2) blue water use cannot be considered independently from green water use. Since water is a global resource, water depletion has a global character as well. Global water availability is the sum of the water availability in the various basins in the world; some of them contribute a lot to overall availability, others only a little. Every litre of water consumption – whether it's in a water-rich or water-poor river basin and whether it's green or blue water – will reduce the water volume remaining for other uses and thus has equal environmental relevance. The fact that green and blue water resources in many relatively water-rich regions are inefficiently used (i.e. low water productivity in terms of production units per m^3 or large WF in terms of m^3 per production unit), is highly environmentally relevant, because here lies part of the solution to the problems in water-poor areas: producing more crops with the water in water-rich basins reduces the need to produce in water-poor basins and thus helps to reduce the water consumption and scarcity in those water-poor basins. Looking at the contribution of a product to local water depletion shouldn't be the mere focus in a product-LCA. Comparing the environmental performance of two cotton shirts, for instance, requires to look at both the total (green and blue) water consumption underlying each shirt *and* the fraction of the total taking place in river basins where overall water consumption levels are so high that minimum environmental water needs are no longer met.

The theme of water consumption can be compared to that of land use. The environmental issue around land use is twofold as well. The first concern is that overall land use keeps on rising, causing global land scarcity; remind the 1.5 Earths we need to sustain our current global economy (Borucke et al., 2013). The second concern is that some forms of land use (e.g. urban land) have large local environmental impact (larger than other forms of land use, like e.g. production forest). The issue of different local environmental impacts of different forms of land use is no reason to ignore the concern of total land use. A product with larger land requirement to produce it is of greater environmental concern than a similar product with smaller land requirement. The same is true for water: a product with larger (volumetric) green and blue WF to produce it is of greater environmental concern than a similar product with smaller WF.

I will illustrate the insufficiency of the geographic focus on water-scarce basins with a simple example. Suppose the hypothetical case of two river basins, with the same surface (Table 1). Basin A is relatively dry, with a water availability of 50 water units per year. Farmers in the basin consume 100 water units per year to produce 100 crop units. The WF (100) thus exceeds the maximum sustainable level (50). Basin B has more water: 250 water units per year. Farmers in this basin consume 200 water units per year, to produce 100 crop units, the same amount as in basin A, but using two times more water per crop unit. In basin B, the WF (200) remains below the maximum level (250), so this is sustainable. According to the logic of LCA authors, the environmental performance is good for the crops originating from basin B and bad for those from basin A. From a geographic perspective, this is true: the WF of crop production in basin A needs to be reduced, that seems to be the crux. From a product perspective, however, we observe that the WF per crop unit in basin B is two times larger than in basin A. If the farmers in basin B would achieve the same water productivity as in basin A, they would produce twice as many crops without increasing the total WF in the basin. If farmers in basin A cannot easily further increase their water productivity, the only solution – in order to maintain global production – is to bring down the WF in basin A to a sustainable level by cutting production by half, while enlarging production in basin B by increasing the water productivity. When in basin B the same water productivity is achieved as in basin A, global production would increase while halving the total WF in basin A and keeping it at the same level in basin B. The fact that crops in basin B had a volumetric WF of twice that in basin A was thus highly environmentally relevant information.

3.3. The neglect of green water use

The LCA community has thus far neglected green water consumption as a relevant resource use metric. According to Pfister and Hellweg (2009), green water consumption in agricultural production can be neglected if green water consumption in the crop field is comparable to that by the original natural vegetation, which is generally the case. Ridoutt and Pfister (2010) argue that green water consumption doesn't contribute to water scarcity and that, due to the inseparability of green water and land, the consumption of green water is better considered in the context of land use impacts. This, however, reflects a limited view on the issue of sustainable water resources use. It is true that runoff (blue water) will not change significantly as a result of green water consumption and that green water resources are inseparably linked to land. It is not right, though, to say that green water resources are not scarce. It is very common that farmers structurally suffer from shortage of rain. Conflicts over blue water allocation among farmers occur precisely for the reason that green water resources are insufficient. Green water shortage in agriculture is in fact the reason for agriculture's blue water demand and therefore the driver of blue water scarcity.

Much of the trouble around blue water scarcity relates to the historical focus of engineers and policy makers on blue water resources exploitation and the neglect of green water use. The insight that green and blue water resources use should be considered in combination emerged in the water resources community in the second half of the 1990s (Falkenmark, 1997) and has received increasing attention since (Falkenmark and Rockström, 2004, 2006). When emphasizing that green water consumption can be ignored in an LCA because it cannot be considered as an additional loss to the watershed, Pfister and Hellweg (2009) actually argue that green water consumption is not blue water consumption, which is right of course, but which betrays their preoccupation with the idea that blue water consumption is the only relevant thing. Ridoutt and Pfister (2010) are explicit in this respect by arguing that *green* water resources consumption is not relevant because it doesn't contribute to *blue* water scarcity.

Green water resources are often not perceived as scarce, because rain comes for free, but actually they are (Savenije, 2000; Falkenmark, 2013). There are alternative competing uses for green water (e.g. production of food crops, feed for animals, energy crops, fibre crops or trees for timber and paper) and there is a conflict between appropriating green water resources for the economy versus leaving them for natural vegetation (Schyns et al., 2015). Competing demands for a limited resource defines the resource as scarce. When all available green water resources are fully used we can say that the resource is depleted. This is the case in many regions of the world, where hardly any land and associated green water is left for natural vegetation. It's not sufficient to focus on land

Table 1
Example of how overexploitation in a water-stressed river basin (A) can be solved by increasing water productivity in a water-abundant basin (B).

Parameter	Unit	Current situation		Possible solution	
		Basin A	Basin B	Basin A	Basin B
Maximum sustainable WF	Water units per unit of time	50	250	50	250
(Volumetric) WF	Water units per unit of time	100	200	50	200
Production	Product units per unit of time	100	100	50	200
WF per product unit	Water units per product unit	1	2	1	1
Water productivity	Product units per water unit	1	0.5	1	1

Source: Hoekstra (2014).

appropriation and neglect green water consumption as proposed by Ridoutt and Pfister (2010), because we cannot disconnect green and blue water resources, ignore the former and focus on the latter. The bigger issue is freshwater scarcity in general, i.e. competition over precipitation, the undifferentiated form of freshwater, which will partition in a green flow (evaporation) and blue flow (groundwater recharge/surface runoff) (Falkenmark and Rockström, 2006). The world's largest consumer of blue water, i.e. irrigated agriculture, uses a lot of green water as well. Green and blue water scarcity and depletion in a catchment are strongly connected. The reason why crops are irrigated is that the rain is insufficient to give a good crop yield. In all catchments with significant blue water scarcity as a result of blue water consumption in irrigated agriculture, green water resources are scarce as well, otherwise there hadn't been the demand for irrigation. One cannot get a good picture of water scarcity if the focus is on blue water resources alone. If rain-fed agriculture produces more (closing the so-called yield gap and increasing green water productivity), there is less need for irrigated agriculture, thus reducing blue water scarcity. An essential component in solving the overconsumption of blue water resources and associated environmental impacts in water-scarce areas is to use green water resources more productively in water abundant areas, because if water-intensive products are produced in areas where sufficient water is available, there is no further need to produce those products in areas where insufficient water is available (Hoekstra, 2014). A large green WF of a crop (in litre/kg) represents low green water productivity (kg/litre) and should therefore be counted in LCA as worse than a small green WF. Ignoring this aspect means that an essential element in (indirect) environmental impact is overlooked.

3.4. Squaring the footprint and being charged for the footprint of others

The idea of a water-scarcity weighted WF leads to the surprising and undesirable situation in which the WF of a specific water consumer or company will inherently be a function of the WF of others. We will thus face the confusing situation in which an increasing or decreasing WF of a specific activity, product, consumer or company may tell little about the changed environmental performance of that activity, product, consumer or company but rather about the changed environmental performance of others. This strange implication of the water-scarcity weighted WF can easily be illustrated.

The water-scarcity weighted WF of an activity or production process i in a certain catchment (WF_i^*) can be calculated by multiplying the volumetric WF of that activity or production process (WF_i) by the water scarcity (WS) in the catchment:

$$WF_i^* = WF_i \times WS = WF_i \times \frac{WF_t}{WA} = WF_i \times \frac{\sum_{i=1}^{n} WF_i}{WA}$$

whereby WS is the ratio of the total volumetric water footprint (WF_t) in the catchment to the water availability (WA). WF_t is equal to the aggregate volumetric WFs of all activities n in the catchment.

This approach has two odd implications. The first is that the overall WS-weighted water footprint in a catchment (WF_t^*) will be defined as:

$$WF_t^* = WF_t \times WS = WF_t \times \frac{WF_t}{WA} = \frac{(WF_t)^2}{WA}$$

There is no logic in defining the WF within a catchment as the square of the total water consumption in the catchment divided by water availability. If this footprint-square approach were copied to the carbon footprint (CF) concept we would get a CF defined as something that increases with the square of the volume of greenhouse gas emissions, which is obviously an odd approach. It's equally odd to do this for water.

The second odd implication is that, when WS-weighted, the WF of a consumer or company will not only go up if a consumer or company increases its *own* water consumption, but also if *other* consumers or companies increase *their* water consumption. Imagine an analogous CF definition whereby the CF of a company goes up while the company factually reduces its greenhouse gas emissions because the increasing greenhouse gas emissions of others count in the CF of this company as well. This would make the CF useless for the company as an indicator of its contribution to global warming. Exactly the same is the case for water: the WF becomes useless for a company as an indicator of its contribution to WS if the indicator is affected by the contributions of others to WS. Nonetheless, the ISO standard for WF prescribes companies to calculate their WF based on a WS-weighted approach (ISO, 2014). We thus have got a standard whereby *the WF of an activity in a catchment will depend on the WF of other activities in the catchment.* Ironically, the WF of a company will inevitably increase if the WFs of *other* companies increase, punished for the bad environmental performance of others.

If we want a proxy for potential environmental impact of water consumption on runoff in a catchment, water scarcity (or "relative water scarcity" if WS metrics for different catchments are normalized based on the WS in one specific river basin or country as proposed by Pfister and Hellweg, 2009) is not a proper characterization factor. The "runoff impact potential" of one litre of water consumption is larger in a catchment with relatively small natural water availability (WA) than in a catchment with relatively large natural WA. Therefore, "relative water availability" is a better metric for "runoff impact potential" than "relative water scarcity". In LCA terminology: if volumetric WFs are to be interpreted in terms of their potential local environmental impact, then they better be multiplied with a characterization factor that reflects relative WA than with a factor that reflects WS or relative WS.

It is proposed here to abandon the idea of weighting based on WS as proposed by Ridoutt and Pfister (2010) and other LCA authors and as prescribed by ISO (2014), because the idea is based on a fundamental error in logic. Weighting volumetric WFs more heavily if WS increases is similar to following a logic of weighting one tonne of greenhouse gas emissions more heavily if global warming progresses or weighting one hectare of land use more heavily if land becomes scarcer. A more sound way of getting a proxy for potential environmental impact of water consumption in different catchments is to weight volumetric WFs by dividing them by relative WA in the catchments considered (instead of multiplying them with relative WS).

The water-availability weighted water footprint (WF_i^{**}) of an activity i in a catchment can be defined as:

$$WF_i^{**} = \frac{WF_i}{WA/WA_{ref}}$$

whereby WA represents the water availability in the catchment and WA_{ref} the water availability in a reference catchment. Dividing by WA_{ref} is done in order to normalize the value of WA.

The water-availability weighted WF of all water-consuming activities in a catchment is:

$$WF_t^{**} = \frac{WF_t}{WA/WA_{ref}}$$

The differences between the volumetric, WS-weighted and WA-weighted WFs are illustrated in Table 2, which includes calculation examples for three basins, X to Z, at two points in time. We consider one specific activity A, taking place in each basin, everywhere with a volumetric WF of 1 water unit per unit of time. In all three basins, the volumetric WF of activity A is assumed to decrease by 10% from t to t + 1. We also consider the total WF of all activities within the basin. In all three basins, the total volumetric WF in the

Table 2
Calculation of the volumetric, water-scarcity weighted and water-availability weighted blue WF in three hypothetic river basins at two points in time.

		Basin X	Basin X	Basin Y	Basin Y	Basin Z	Basin Z
		Time *t*	Time *t* + 1	Time *t*	Time *t* + 1	Time *t*	Time *t* + 1
Water availability	Water availability (WA)	100	100	100	100	200	200
	Relative WA[a]	1	1	1	1	2	2
Water scarcity	Water scarcity (WS)	0.5	0.6	0.25	0.3	0.25	0.3
	Relative WS[a]	1	1.2	0.5	0.6	0.5	0.6
WF of activity A	Volumetric WF	1	0.9	1	0.9	1	0.9
	WS-weighted WF[b]	1	1.08	0.5	0.54	0.5	0.54
	WA-weighted WF[b]	1	0.9	1	0.9	0.5	0.45
Total WF in the basin	Volumetric WF	50	60	25	30	50	60
	WS-weighted WF[b]	50	72	12.5	18	25	36
	WA-weighted WF[b]	50	60	25	30	25	30

[a] WA and WS in basin X at time *t* are chosen as the reference.
[b] Expressed in terms of Basin X water equivalents as at time *t*.

basin is assumed to increase by 20% from t to t + 1. We can make four observations from the numerical examples in the table. First, in all three basins the WS-weighted WF of activity A increases over time while actual water consumption of activity A decreases, which illustrates the inappropriateness of the metric as an indicator of the individual contribution of an activity to potential environmental impact. Second, in all three basins the total WS-weighted WF doesn't increase linearly with increasing water consumption in the basin (factor 1.2) but exponentially (factor 1.44), which lacks any logic. Third, if we compare basins X and Y, which are similar basins but only differ in terms of the fraction of the available water already consumed (the total volumetric WF in basin X is two times bigger than in basin Y), we see that the WS-weighted WF of activity A in basin Y is half of that in basin X, while we talk about the same activity with the same water consumption in two basins naturally endowed with the same amounts of water. More logically, the WA-weighted WF of activity A is the same in both basins. Finally, when comparing basins X and Z we see that water availability in Z is two times the water availability in X, while the total volumetric WFs in both basins are the same and increasing over time at the same rate. Over time, the relative WA in Z (compared to X) remains constant, while the relative WS in Z increases. As a result, the WS-weighted WF of activity A in Z increases even though the actual water consumption of activity A decreases. The WA-weighted WF of activity A in basin Z decreases with the same rate as the volumetric water consumption of the activity. In the WA-weighted case, 1 unit of water consumption in Z is equivalent to half a unit of water consumption in X, because water availability in Z is two times bigger than in X.

It should be noted here that the LCA literature refers to various alternative WS indicators that could be used to weight consumed water volumes (Jeswani and Azapagic, 2011; Kounina et al., 2013; Boulay et al., 2015a, 2015b, 2015d). The above argument has been built on the assumption that WS is defined as the total volumetric WF divided by the water availability in the catchment. Many WS indicators that have been proposed within the LCA community look different, including for instance the Water Stress Index (WSI) of Pfister et al. (2009) or the recently proposed inverse of the Available Water Remaining (AWaRe) per m^2, with the available water remaining being measured as the total water availability in a catchment minus the human and environmental water demands (Boulay et al., 2015d). One may wonder whether the argument against the WS-weighted WF holds if these various other definitions of WS are applied. This is certainly the case, since any metric of WS will increase if the volumetric WF in a basin increases. This is also the case for Pfister's WSI, although the effect here is obscured by the complexity of that index (see Section 3.6), or Boulay's inverse of AWaRe per m^2. Whatever WS indicator is used, it will positively relate to the volumetric WF in the catchment, with the inevitable effect that the WS-weighted WF of a specific activity or process will increase if *other* activities or processes consume more water.

Another note is to be made on the measurement of water availability (WA). One can measure total runoff (Vörösmarty et al., 2000) or natural runoff minus environmental flow requirements (Hoekstra et al., 2011, 2012), whereby the latter is better but requiring more data. All variables – WA, WF and WS – can be measured on annual or monthly basis. Obviously, measurement per month will capture the intra-annual variability in the three variables, which will be lost in case of measurement on annual basis. Therefore, both the water resources (Hoekstra et al., 2011, 2012; Wada et al., 2011) and LCA community (Pfister and Bayer, 2014) will easily agree that monthly measurement is to be preferred over annual measurement.

3.5. Inconsistency with other footprint definitions

Several LCA scholars have pointed at the need to weight water consumption based on local WS with the argument that this is consistent with carbon footprint (CF) accounting, where emissions of greenhouse gases are weighted based on their global warming potential (Pfister and Hellweg, 2009; Ridoutt and Pfister, 2010; Kounina et al., 2013; Boulay et al., 2015b). By multiplying each consumed litre of water by a local WS factor between zero and 1, water consumption can be expressed in litres of H_2O-equivalents (Boulay et al., 2015b). One litre of water consumed in an area with a Water Stress Index of 0.5, which refers to the threshold between 'moderate' and 'severe' water stress, would thus count as 0.5 litre of H_2O-eq. Apart from the fact that these H_2O-equivalents have no meaningful physical interpretation (unlike CO_2-equivalents that do have a meaning) and the fact that the use of different, alternative WS indicators leads to different weightings and thus different and incomparable sorts of H_2O-equivalents (Boulay et al., 2015a), there is a fundamental error in reasoning here. The water-scarcity weighted WF is not consistent with the general footprint concept at all.

Common to all environmental footprints is that they quantify the human appropriation of natural capital as a source or a sink (Hoekstra and Wiedmann, 2014). The footprint that was first introduced is the ecological footprint and measures the appropriation of land as a resource and the land needed for waste uptake (CO_2 absorption) (Wackernagel and Rees, 1996). The (volumetric) WF measures both the consumption of fresh water as a resource (the green and blue WF) and the use of fresh water to assimilate waste (the grey WF) (Hoekstra and Mekonnen, 2012a). The CF measures emission of greenhouse gases to the atmosphere (Hertwich and Peters, 2009). The material footprint measures raw material extraction (Wiedmann et al., 2015). In all cases, footprints measure the volume of resource use and/or a volume of emission, and as such represent a certain *pressure* exerted by humans on the environment. In none of the cases the footprints tell something about the resultant *impact*. Footprints become meaningful when evaluated against maximum sustainable levels, which relate to the carrying or assimilation capacity of the environment.

The CF has been adopted in LCA studies as a *proxy* for impact (as a mid-point impact indicator), which has been a major source of confusion, since many scholars have started to consider CF as an indicator of impact and expect the WF to fulfil that role as well (Pfister and Hellweg, 2009; Ridoutt and Pfister, 2010). However, the CF can only be interpreted as a pressure indicator, because it simply measures greenhouse gas emissions – indeed in CO_2-equivalents to bring the different types of emission under one common denominator – and tells nothing about the resultant impacts, such as changing spatial patterns of temperature, evaporation and precipitation or about melting glaciers and icecaps or sea level rise, let alone something about final impacts on human well-being or ecosystem integrity. The idea of CF as an impact indicator, however, has taken hold and has led to the claim that WF should show impact as well. The step towards weighting water consumption based on local WS then seemed a logical step. As a consequence, however, the WF would become inconsistent with the general idea of footprints as measures of resource use and/or waste generation.

Comparing the footprints of two alternative products makes always sense, because the size of a footprint tells the amount of resources use or emission per unit of product. From a global point of view, one can always say that the smaller the footprint the better (under the condition of other circumstances remaining equal; if that is not the case, inevitable trade-offs may be involved). This is true for the amount of land use behind a product, the amount of greenhouse gas emissions, and also for the volume of green en blue water consumption and the size of the grey WF. Footprints represent the overall pressure on the global environment. Impacts will become manifest locally and may differ across regions. A large land footprint per unit of a product in a big, thinly populated country may matter little from a local environmental perspective. Similarly a large WF per unit of product in a water-abundant catchment may matter little from a local point of view. One may even ask whether a CF and the resultant global warming matters a lot for a region that happens to be better off through climate change instead of worse. The issue is that we need to differentiate between global and local environmental relevance. LCA authors have made the implicit choice in the case of water use to fully ignore the global pressure exercised by increasing volumetric water demands. In this way, for example, biofuels produced in water-abundant areas completely disappear from the radar of environmental concern, while actually the quickly increasing demand for biofuels may be one of the most important drivers of water shortages in the future (Gerbens-Leenes et al., 2012). The same can be said for the production of animal products in water-abundant areas or in regions where livestock mainly depends on rain-fed grass or feed crops. The quickly increasing demand for meat and dairy per capita is a significant driver behind the increasing WF of humanity (Liu and Savenije, 2008; Ercin and Hoekstra, 2014), with various localized problems as a result. It is a major error in LCA to omit the volumetric WF of products, because it whitewashes products that are causing an increasing pressure on the world's scarce freshwater resources and should be a major environmental concern.

3.6. The lack of physical interpretation of the Water Stress Index

Whereas the previous sections include fundamental critique on the weighting of WFs, there are also some problems around the practical proposals that have been made on which weighting factor to use, i.e. how to measure water scarcity (WS). The most cited method to estimate WS in LCA is the Water Stress Index (WSI) by Pfister et al. (2009). They define *WSI* per catchment as follows:

$$WSI = \frac{1}{1 + e^{-6.4 \times VF^p \times WTA} \times (1/0.01 - 1)}$$

in which *WTA* represents the annual withdrawal-to-availability ratio in the catchment (calculated as the total annual gross water withdrawal divided by the annual freshwater availability), *VF* a fixed variation factor reflecting monthly and annual temporal variability of water availability in order to account for increased scarcity in watersheds with irregular water availability, and *p* an exponent equalling 0.5 for catchments with strongly regulated flows and 1 for catchments without strongly regulated flows. The factor *VF* is defined as follows:

$$VF = \frac{\sum_{i=1}^{n} e^{\sqrt{\ln(s^*_{m,i})^2 + \ln(s^*_{y,i})^2} \times P_i}}{\sum_{i=1}^{n} P_i}$$

whereby P_i represents mean annual precipitation in grid cell i within the catchment (which is supposedly schematized into n grid cells), $s^*_{m,i}$ the standard deviation of monthly precipitation in grid cell i, and $s^*_{y,i}$ the standard deviation of annual precipitation over a 30-yr period in grid cell i. This may look impressive and advanced, but in essence the WSI is a metric without possible meaningful interpretation. The fact that the argument of the exponential function is not dimensionless inhibits a physical interpretation of the construct. In addition, the two standard deviations have different units: one is in mm/month, whereas the other is in mm/year. It's impossible to meaningfully add them. Somehow the metric captures the effect of temporal variability, which has been used as an argument that WSI is a better scarcity indicator than the annual withdrawal-to-availability ratio (*WTA*), which has been widely used as a WS indicator in water resources literature (e.g. Vörösmarty et al., 2000). On the other hand, the WSI equation is calibrated such that a WSI of 0.5 is obtained for a WTA ratio of 0.4, which has in the past often been (arbitrarily) used as the threshold between moderate and severe water stress in a catchment. Through its definition, WSI will lie between 0.01 and 1.

The WSI has been embraced by the LCA community as a useful metric to be used as a weighting factor in the calculation of water-scarcity weighted WFs. Recently, Pfister and Bayer (2014) published an improved version of the WSI, which however is similar as the above one, though calculated now on a monthly rather than annual basis, with the suggestion that it thus captures WS even better. It is difficult to criticize the WSI because it has no pretended physical meaning, so there is no way of checking whether it makes sense. It is difficult, though, to see why we would rely on a metric that is essentially a meaningless construct. It would be more useful if the LCA community would rely on advanced water stress and WS indicators being developed within the water resources community. Wada et al. (2011), for instance, computed water stress at a high spatial resolution on a monthly basis as water consumption over water availability. Hoekstra et al. (2012) took a similar approach, but also accounted for environmental flow requirements when estimating water availability.

Ridoutt and Pfister (2013) presented a new WF calculation method integrating consumptive and degradative water use into a single stand-alone weighted indicator that makes things even more obscure than the WSI. In the new method they proposed to calculate consumptive water use by multiplying water consumption by WSI (with a result in H_2O equivalents), compute degradative water use by converting "ReCiPe points" earned based on emissions to water into H_2O equivalents, and to finally add the two incomparable sorts of H_2O equivalents into an aggregated WF (again in H_2O

equivalents). This all is problematic particularly because there is no way to validate whether the resulting figure correctly represents "potential environmental impact". Since the metric cannot be interpreted in any physical way and since the outcomes cannot be empirically tested, nothing else remains than a meaningless construct. A similar critique holds for the Water Impact Index by Bayart et al. (2014), another effort to express the environmental impact of consumptive and degradative water use in a single metric.

The fact that the resulting "impact category indicators" from Pfister et al. (2009), Ridoutt and Pfister (2013) and others have no empirical interpretation becomes even worse given the ambition of the LCA community to translate the H_2O equivalent WF into its impact on human health (considering disability adjusted life years) and biodiversity or ecosystem quality (Pfister et al., 2009; Bayart et al., 2010; Berger and Finkbeiner, 2010). It is already impossible to know how reduced groundwater levels and river flows affect humans and ecosystems, given the multitude of contextual factors that play a role, it is complete madness to establish a relation between the meaningless H_2O equivalent WF in a catchment and its impacts on human health and biodiversity. And this, though, is precisely what several LCA authors propose to do (Boulay et al., 2011, 2015a, 2015b). The pretention is to assess the cost of water consumption in terms of disability adjusted life years per unit of H_2O equivalents per catchment. This makes no sense at all, because it is impossible to isolate the impact of local water depletion on local human health (given e.g. coping capacity, the possibility to import), let alone that one can establish a relation using non-empirical metrics. There has been no study ever showing empirical evidence of some generalized relation between WS and human health in catchments, which is to be expected, because drinking water requirements are generally relatively small and thus difficult to be affected by local WS. Even if local WS affects public water supplies, people may still be able to cope if they can afford to buy imported water. Furthermore, even though WS in a catchment can easily affect food harvests, this doesn't necessarily lead to malnutrition of local populations, since people may still be able to get food from elsewhere. There are too many pathways between WS and human health, with too many other variables in between, to find a single equation that relates both factors.

Whereas the LCA community tries to build consensus on the development of a stress-based indicator for LCA-based impact assessment of water consumption (Boulay et al., 2015c), it is probably better to take a step back, so as to first thoroughly reconsider the soundness of the idea of a water-scarcity weighted WF indicator, and to assess the feasibility to develop generalized relationships between water use, WS and water pollution versus human health and ecosystem quality that can be empirically tested. One must admit that expressing the environmental impact of products in terms of human health damage and ecosystem degradation in the form of single metrics as aimed for in the LCA methodology may run against the limits of what is possible, given the complexity of the socio-ecological system.

4. The way forward

The fact that water scarcity is a major environmental concern is a reason to get the water scarcity issue well into LCA. As I have argued, the (volumetric) green, blue and grey WFs are all equally relevant from an allocation and depletion point of view. One litre of green or blue water allocated for consumption for one purpose is not available for a competing purpose, and one litre of green or blue water allocated to human use is not available for nature. Green and blue WFs thus subtract from the supply capacity left. Similarly, if one activity has a grey WF of one litre and thus consumes part of the total assimilation capacity of a water stream, this subtracts from the assimilation capacity left for other polluting activities. Since water is a global resource, every litre of water consumption or pollution counts. The essence of growing WS is not environmental impact, but increasing global resource use given limited global resource availability, with heterogeneously spread local environmental impacts as a by-product in places where local resource use exceeds local maximum sustainable levels. When considering the contribution to water scarcity or water depletion, it is key to consider how much water units are used per product, wherever that happens. If only 100 units of green and blue water are sustainably available, 80 may be available in water-abundant areas and 20 in water-poor areas. There is no reason to not count certain types of water use (like green water) or count certain types of water use less (like blue water in water-abundant areas). Only by considering all forms of water use and all forms of water availability, it will be possible to get a picture of depletion.

It would be useful to incorporate the topic of freshwater scarcity in LCA as a "natural resource depletion" category. This is an unexplored direction as yet, see for instance the treatment of freshwater depletion in the review by Klinglmair et al. (2014). Freshwater depletion should be considered from a global perspective, since freshwater is a global resource, with growing global freshwater demand while global freshwater availability is limited. This limitation is determined by the limited global freshwater renewal rate (precipitation over land), the uneven spatial and temporal distribution of water availability, the limited transport and storage possibilities, the need to let part of the natural water flows untouched, and the impossibility to use part of the natural flows (e.g. as they flow in unaccessible areas or in times where there is too much rather than too little water). Given the limited accessible freshwater flows globally available for productive uses, it is important to measure (volumetric) WFs of products, to measure the comparative claim of different products on those limited freshwater flows.

When looking at the potential *local environmental impact* of water use in the full life cycle of a product, it makes sense to focus on the blue and grey WF, because the former may lead to ecosystem impacts as a result of runoff modification and the latter may impact on ecosystems if pollution levels get too high. Ridoutt and Pfister (2010) are right in their argument that the environmental impact of the green WF can as well be considered in the context of the land use impact category. The impact indicator representing the local environmental impact of a blue WF could be based on the idea of water-availability weighting as proposed in this paper. In other words, the blue WF per catchment is weighted based on the carrying capacity per catchment, which depends on blue water availability (runoff minus environmental flow requirements). The impact indicator representing the local environmental impact of a grey WF could be based on a similar approach, e.g. weighting the grey WF per catchment based on assimilation capacity per catchment, which depends on the amount of runoff to assimilate a grey WF.

I have argued that weighting water consumption based on relative WA per catchment gives a better proxy of potential local environmental impact of water consumption than weighting based on relative WS per catchment, but one may retain doubts about the usefulness of weighted metrics altogether, given the lack of physical meaning of such constructs. The difficulty remains that LCA aims to compare different sorts of potential environmental impacts – indeed comparing apples and pears, like the impact of water use in one basin to the impact of water use in another basin, or the impact of water consumption to the impact of water pollution. Weighted metrics may have their specific use within a product LCA, but one should be extremely careful in applying such metrics outside that context.

The critique in this paper does not concern LCA in itself, but the way some authors have proposed to account for water consumption and water scarcity within an LCA. Water Footprint Assessment (WFA) and LCA serve different purposes and employ different methods, but both can use the WF concept. It is confusing if the fields employ different definitions of the concept, and as argued here, the original volumetric definition is most useful and the only one consistent with the ecological (land) and carbon footprint concepts.

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