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Water footprint: methodologies and a case study for assessing the impacts of water use

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ABSTRACT

The methodologies, approaches and indicators for assessing the impacts of freshwater usage are still evolving. The development of the water footprint concept has been an important step in this direction but the existing methodologies mainly assess the quantity of water used rather than the related impacts. Although there is a recognised need to consider the latter, particularly on a life cycle basis, the difficulty is that there are little or no reliable data on water usage in life cycle databases; furthermore, there is no agreed life cycle impact assessment method for estimating impacts related to freshwater use. However, there have been some methodological developments which propose methods for inventory modelling and impact assessment for water use in life cycle assessment. This paper reviews some of these approaches and discusses their strengths and limitations through a case study, which considers the impacts of freshwater consumption from corn-derived ethanol produced in 12 different countries. The results show a huge variation in the results between different methods and demonstrate the need for a standardised methodology for assessing the impacts of water use on a life cycle basis. Specific recommendations for further research in this field have been made accordingly.

1. Introduction

Freshwater is essential for humans and ecosystems but as shown in Fig. 1, its distribution across the globe is uneven. Currently, a fifth of the world's population or around 1.2 billion people live in areas of physical water scarcity and a further 500 million people are approaching this situation (CAWMA, 2007). In many places, water is overexploited for economic development, especially for agricultural and industrial activities as well as for meeting the needs of the growing population (WWAP, 2009). The pressure on the freshwater resources is expected to increase significantly with the climate change as well as with some measures for reducing the emissions of greenhouse gases, e.g. cultivation of biofuel crops. This could cause a wide range of social and environmental problems (Falkenmark, 2008).

Measuring water use and assessing its environmental impacts, particularly on a life cycle basis, are therefore important steps towards minimising these impacts. One of the methods for assessing water use on a life cycle basis that is probably most-widely used is the water footprint approach developed by Hoekstra et al. (2009). Conceptually, it is similar to the carbon footprint used in Life Cycle Assessment (LCA) (Carbon Trust, 2007; CCaLC, 2010) but unlike it, the water footprint concept has evolved independently from LCA and has so far focused on the quantification of water use. Even though the water footprint approach provides a method for assessing the impacts, the developers of the method argue that aggregated index is not the intention of the approach (Hoekstra et al., 2009). Therefore, the water footprints are expressed on a volume basis (WFN, 2010). Although the volumetric water footprint indicator is useful from the water-resource management perspective, it does not reflect the potential environmental (and social) impacts of the water use, which are important from the LCA perspective.

Currently, water use is not systematically recorded in life cycle inventory (LCI) databases and the life cycle impact assessment (LCIA) methods do not consider impacts related to freshwater use (Koehler, 2008). This is probably because LCA is largely time and location independent, while the impacts and issues related to water are seasonal and very local in nature, within the confines of the watersheds and river basins.[1] However, recently there have been some attempts to incorporate water use in LCA, including the

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[1] A river basin is the area of land where water from rain and melting snow drains into a river and its tributaries. A watershed represents a smaller area of land that drains to a smaller stream, lake or wetland. There are several watersheds within a river basin.

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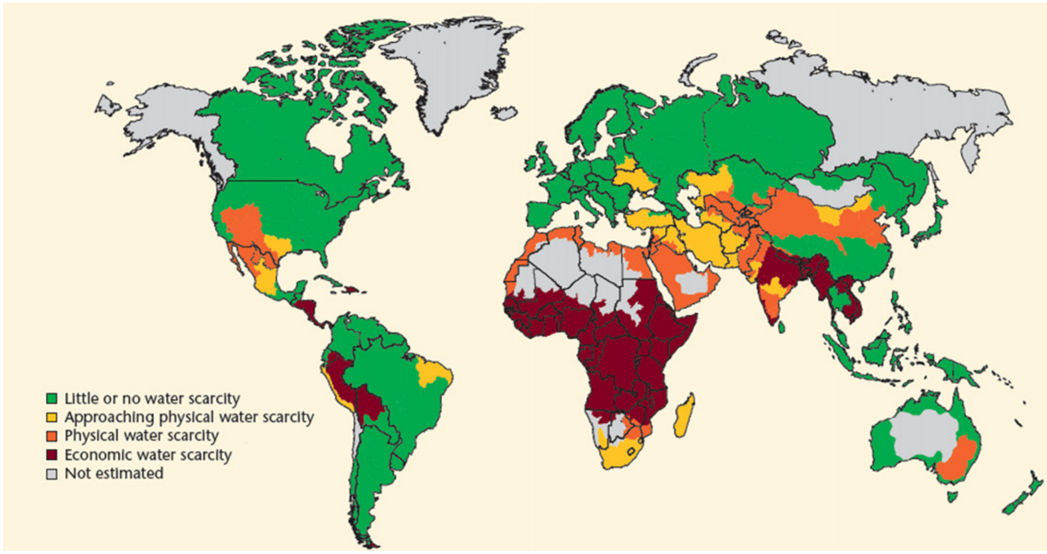

Fig. 1. Areas of water scarcity (CAWMA, 2007).

eco-scarcity method (Frischknecht et al., 2009a), Milà i Canals et al. (2009) approach and Pfister et al. (2009) approach. This paper reviews some of the approaches for inventory modelling and assessing the impacts of freshwater consumption. The aim is to explore their strengths and limitations and to identify further research needs in this field. For these purposes, a case study of ethanol produced from corn grown in 12 different countries has been developed. The case study illustrates how the results vary between different methods and demonstrates the need for a standardised methodology for assessing the impacts of water use on a life cycle basis.

2. Review of approaches for assessing the impacts of water use

This section reviews some approaches for quantifying water use and the related impacts proposed for use in LCI and LCIA, respectively. The terminology used in these approaches is summarised in Table 1.

2.1. Methods for quantifying water use in life cycle inventory

As shown in Table 2, LCI databases contain limited information on water use. In the Gabi and ELCD databases, input water flows are only differentiated with respect to the basic water source (e.g. river, groundwater, sea), while the SimaPro and Ecoinvent databases also include some additional flows based on the intended use (e.g. turbine use water in the Ecoinvent and process, cooling, drinking water in the SimaPro). It is apparent that not only are the terminology and categorisation in the databases inconsistent, but also that the key parameters for assessing water use impacts (such as geographic location) are missing. Furthermore, most LCA studies that report water use simply report the total input without differentiating between the in-stream (e.g. turbine use) and off-stream uses or between freshwater and seawater. Table 2 also shows that most of the databases do not consider systematically wastewater discharges and hence do not report consistently water outputs.

The distinction between the input and output water flows is essential for an adequate assessment of the potential impacts of water use. Because of these inadequacies at the inventory level, LCA is not generally considered suitable for quantifying water use. In an attempt to address these issues, several approaches have been proposed for water use modelling in LCI (Milà i Canals et al., 2009; Pfister et al., 2009). In addition to the LCA-related methodologies, the water footprint approach, developed in the context of water resource management, also provides a potentially useful methodology for quantifying water use for LCI (Hoekstra et al., 2009). These methods are discussed below.

2.1.1. The Hoekstra et al. approach

Building on the concept of virtual water (Allan, 1998), Hoekstra (2003) introduced the concept of water footprint which has been subsequently developed and refined as a method for quantifying water use by a product, service or nation (Hoekstra and Chapagain, 2008; Hoekstra et al., 2009). The water footprint represents the sum of all the water used in a supply chain, comprising blue, green and grey water. Blue water is defined as the volume of freshwater abstracted from rivers, lakes and aquifers. The amount of rainwater (stored in the soil as soil moisture) used by plants is referred to as

Table 1
Terminology used in different water use assessment approaches.

Term	Explanation
Blue water	Freshwater available in surface water bodies (rivers, lakes) and aquifers for abstraction.
Green water	Rainwater (stored in the soil as soil moisture) used by plants and vegetation.
Grey water	The volume of freshwater required to dilute pollutants so that the quality of water remains above water quality standards set by regulations.
Evaporative use	Water which is evaporated during its use hence not immediately available for further use.
Non-evaporative use	Water returned to any freshwater source after its use and available for further use.
Water consumption	Freshwater withdrawals which are evaporated, discharged into different watersheds or the sea after use and embodied in products and waste.
Water degradation	Water which is discharged in the same watershed after the quality of water has been altered.
Irrigation water	The blue water consumed in agricultural activities.

Table 2
Types of water included in current LCA databases.

	Ecoinvent (Ecoinvent, 2007)	Gabi (PE International, 2007)	SimaPro (PRé Consultants, 2009)	ELCD (European Commission, 2010)
Input flows	River water Lake water Groundwater Seawater Salt sole water Turbine use water	Feed water Groundwater Seawater Surface water Water with river silt Lake water	Water, barrage Water, cooling, salt, ocean Water, cooling, surface Water, cooling, unspecified natural origin Water, cooling, well, in ground Water, fresh Water, lake Water, process and cooling, unspecified natural origin Water, process, drinking Water, process, salt, ocean Water, process, surface Water, process, unspecified natural origin Water, process, unspecified natural origin Water, process, well, in ground Water, river Water, salt, ocean Water, salt, sole Water, turbine use, unspecified natural origin Water, unspecified natural origin Water, unspecified natural origin Water, well, in ground	Feed water River water Seawater Well water
Output flows	–	Wastewater processing residue Water (river water)	Water, wastewater	–

green water. Finally, grey water accounts for the impact of pollution on water resources and represents the volume of freshwater needed to dilute pollution so that the quality of the water remains above water quality standards set by regulations. This approach has been used for calculating the water footprints of various agricultural products; for example, the water footprint of beef has been estimated at 15,500 L/kg, sugar at 1500 L/kg, wine at 120 L/glass and bioethanol (from corn) at 110 m^3/GJ (WFN, 2010). It has also been used as a tool for developing corporate water reduction strategies (Ridoutt et al., 2009) and for water-footprint labelling of products (Sabmiller and WWF, 2009).

However, there are concerns that this approach could provide misleading results (Ridoutt et al., 2010; Ridoutt and Pfister, 2010). The main concern relates to the fact that, unlike the carbon footprint, the water footprint represents just the quantity of the water used without an estimation of the related environmental impacts, such as due to water scarcity. Even the quantification of water use is controversial due to the inclusion of green water (rainwater as moisture in soils) which does not affect availability of blue water and therefore should not be accounted. Recently, some companies have adopted the concept of "net green" water – the difference between the water evaporated from crops and the water that would have evaporated from natural vegetation (Sabmiller and WWF, 2009). Furthermore, water abstraction – rather than consumption – is often used in quantifying the blue water footprints (Hoekstra and Chapagain, 2008). This could be problematic, especially in the case of industrial water use where only a small part of the abstracted water is actually consumed (e.g. evaporated in cooling towers or embodied in the product) and the remainder is often discharged back to the water bodies (e.g. cooling water or industrial effluents from wastewater treatment plants). With respect to grey water footprint, it is argued that environmental impacts of grey water are more suitably addressed in other impact categories such as eutrophication or toxicity (Milà i Canals et al., 2009). Moreover, in the absence of an agreed method for the quantification of dilution volumes for assimilation, the estimation of grey water footprint is subjective.

2.1.2. The Milà i Canals et al. approach

This approach considers water use at the level of a river basin. According to this method, both the source of water and type of use of freshwater should be included in LCI (Milà i Canals et al., 2009). With respect to the source, it follows the Hoekstra et al. approach by classifying water sources as blue and green water. The blue water category is further differentiated into three types: flow (river/lake), fund (aquifer) and stock (fossil). The water use is split into two categories: evaporative and non-evaporative use. The latter is defined as water returned to the freshwater source after its use and available for further use. It is further suggested that the green water and the non-evaporative use of river, lake and aquifer water should be disregarded in LCIA because their use does not lead to relevant environmental impacts from a resource perspective (i.e. reduced availability of water for other users and effects on freshwater ecosystem). Instead, it is proposed to assess the land use effects on rainwater availability which accounts for changes in infiltration and evapo-transpiration in the production system relative to a reference land use. It is suggested that for high precipitation areas (rainfall > 600 mm/year), rainwater lost from arable land is 73%, where as with forest as the reference (potential) land use, this is 67%; therefore the additional loss due to arable land use is 6% of rainwater (Milà i Canals et al., 2009). For low precipitation areas (rainfall < 600 mm/year), the extra 10% of rainwater is lost due to similar change in land occupation.

2.1.3. The Pfister et al. approach

This approach (Pfister et al., 2009) considers water usage on a smaller scale than the Milà i Canals et al. method, taking watershed[1] as the area of focus. Unlike the previous two approaches, this method considers only blue water. This method differentiates three categories of water use: in-stream water use, water consumption (where the water is no longer available in the watershed) and water-quality degradation (where the water is still available after use but with diminished quality). The main difference between the Milà i Canals et al. method and this approach is that the water discharged to another watershed is treated here as consumed while the Milà i Canals et al. approach considers the water discharged to any freshwater source as a non-evaporative use. Furthermore, unlike the Milà i Canals et al. approach, this method suggests that the wastewater discharge should be assessed for the loss of the water quality. However, Pfister et al. do not elaborate on how this could be done.

2.2. *Life cycle impact assessment (LCIA) methods for assessing the impacts of water use*

Changes in the freshwater availability due to the withdrawal of freshwater and degradation of the quality caused by the discharge of wastewater could have significant impacts on ecosystems and human health. Although impacts related to the discharge of wastewater are considered in LCA in the impact categories such as eutrophication or toxicity, LCIA methods currently do not consider impacts from water consumption. Recently, methodological frameworks have been proposed by Frischknecht et al. (2009a), Milà i Canals et al. (2009) and Pfister et al. (2009) to integrate such impacts in LCIA. These methods are discussed below.

2.2.1. *The eco-scarcity method*

This method, which is based on the distance-to-target principle, provides eco-factors for various environmental impacts including water use (Frischknecht et al., 2009a,b). Here, water use is defined as the total input of freshwater abstracted for production or consumption. As shown in Table 3, water use is grouped into six water-scarcity categories from low (using less than 10% of the available freshwater resources) to extreme (using more than 100% of the available freshwater resources). For assessing the impacts of water scarcity, each category is then assigned an individual eco-factor based on the average water withdrawal-to-availability (WTA) values. The eco-factor is calculated as follows:

$$\begin{aligned}\text{eco}-\text{factor} &= \frac{1\times EP}{\text{Normalisation}}\times \text{Weighting}\times \text{Constant } t\\ &= \frac{1\times EP}{Fn}\times (WTA)^2\left(\frac{1}{20\%}\right)^2\times C\left(\text{EP/m}^3\right) \qquad (1)\end{aligned}$$

where: EP – Eco-point (the unit of assessed impact); Fn – normalisation factor for water consumption (km^3/yr; with Switzerland as a reference region; Fn = 2.57 km^3/yr); WTA – ratio of water use to available resources (scarcity ratio) (–); 20% – percentage of tolerable stress on water resources (20% according to OECD, 2004); C – constant (10^{12}/yr) for obtaining presentable numerical quantities.

Table 3 shows eco-scarcity factors for several countries.

The method can be applied at the country, region or watershed level. However, national eco-factors provide little insight into local water scarcity for heterogeneous countries such as the US, China, Australia and India. Since the method uses WTA which is based on the annual data, it does not capture the seasonal variations. For example, many regions with WTA < 0.2 often face severe shortage in summer months. Furthermore, the use of WTA can overestimate water stress. For instance, Table 3 shows that the weighting factors for the 'extreme' category (WTA ≥ 1) are almost 1000 times higher than for the 'low' category. Assigning very high weighting factors for countries in the extreme category could be inappropriate especially for countries such as United Arab Emirates and Israel which meet their freshwater requirements using seawater desalination and the use of desalinated seawater does not result in the reduced availability of freshwater for aquatic ecosystem.

2.2.2. *The Milà i Canals et al. approach*

This method considers two primary pathways through which the use of freshwater can impact the available supply: (1) freshwater ecosystem impact (FEI) and (2) freshwater depletion (Milà i Canals et al., 2009). FEI is defined as the volume of 'ecosystem-equivalent' water, referring to the volume of water likely to affect ecosystems. For assessing the FEI, the method proposes the use of a water stress indicator as a mid-point characterisation factor. The water stress indicator is defined as a ratio of water withdrawal from a river basin to the water available for human use (Smakhtin et al., 2004). The water available for human use is a difference between the total amount of water available in the basin and the estimated environmental water demand needed for sustaining the basin's ecosystem. The river basins are categorised as follows: slightly exploited (water stress indicator < 0.3); moderately exploited ($0.3 \leq$ water stress indicator < 0.6); heavily exploited ($0.6 \leq$ water stress indicator < 1) and overexploited (water stress indicator > 1) (Smakhtin et al., 2004). Fig. 2 shows the map of water stress indicators for major river basins in the world.

Although calculating the water stress indicator at a river basin scale provides additional detail, such analysis still struggles to reflect fully the level of stress upon the local water resources. For example, many rivers exhibit extreme seasonal variations in flow distribution but the water stress indicator, which is based on the annual average, does not capture such fluctuations. Also the method cannot be applied for regions which do not belong to any river basin. Furthermore, as recognised by the proponents, the use of water stress indicator as the characterisation factor would mean that the water use impacts increase linearly with water usage, which would be highly unlikely (Milà i Canals et al., 2009) – arguably, the impacts of water use would increase more rapidly with incremental changes in the water stress values.

Milà i Canals et al. (2009) also suggest to categorise the use of groundwater (where its over-abstraction may reduce its availability for future generations) as depletion of abiotic resources. However,

Table 3
Eco-scarcity weighting factors and eco-factors for assessing water use impacts (Frischknecht et al., 2009a).

Water pressure category	WTA range	WTA used for weighting calculation	Weighting factor $(WTA)^2\left(\frac{1}{20\%}\right)^2$	Normalisation (km^3/yr)	Eco-factor (EP/m^3)	Countries
Low	<0.1	0.05	0.0625	2.57	24	Argentina, Madagascar, Russia, Switzerland, UK
Moderate	0.1 to <0.2	0.15	0.563	2.57	220	France, Greece, Mexico, USA
Medium	0.2 to <0.4	0.3	2.25	2.57	880	Japan, Thailand, China, Germany, Spain
High	0.4 to <0.6	0.5	6.25	2.57	2400	Algeria, Morocco, Sudan, Tunisia
Very high	0.6 to <1.0	0.8	16.0	2.57	6200	Pakistan, Syria, Tajikistan, Turkmenistan
Extreme	≥1	1.5	56.3	2.57	22,000	Israel, Yemen, Kuwait, Saudi-Arabia, Egypt

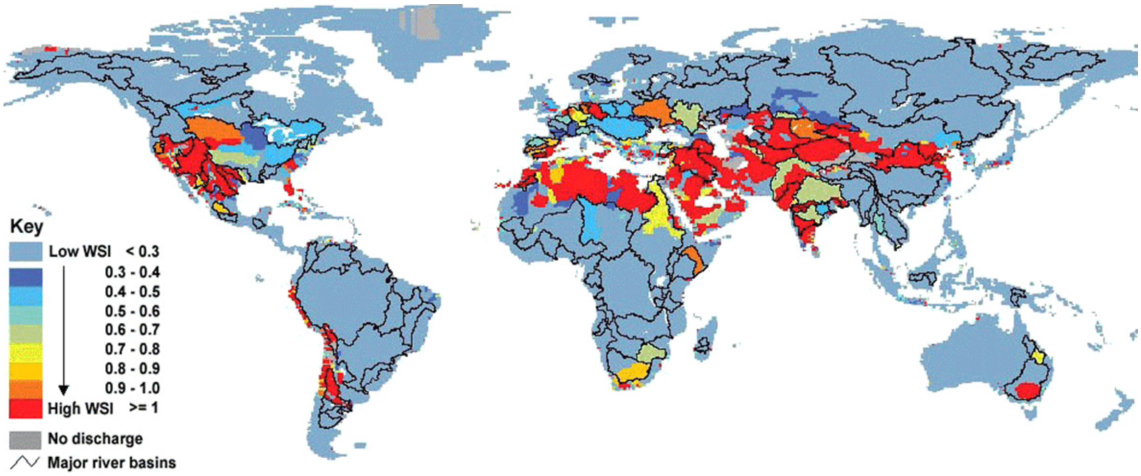

Fig. 2. A map of water stress indicators for major river basins in the world (Smakhtin et al., 2004).

the calculation of abiotic depletion potential (ADP) factors for groundwater is challenging as most of the groundwater reserves are seldom quantified in terms of their relative abundance compared to their potential use. Moreover, in many cases, the groundwater is renewable hence including this impact into the abiotic resource depletion impact category could be problematic.

2.2.3. The Pfister et al. approach

This approach considers both mid-point and the end-point characterisation factors for assessing the environmental impacts of freshwater consumption (Pfister et al., 2009). A Water Stress Index (WSI), which indicates the water consumption impacts in relation to the water scarcity, is proposed as a mid-point characterisation factor. Like the eco-scarcity method, the index is also based on the WTA and can be applied to any spatial scale. However, Pfister et al. (2009) recommend that water use impacts should be assessed at a watershed level. It also accounts for monthly and annual variability of precipitation as well as watersheds with strongly regulated flows. The WSI values, which range from 0.01 to 1 as shown in Fig. 3, are derived using the following logistic function:

$$WSI = \frac{1}{1 + e^{-6.4WTA^{*}\left(\frac{1}{0.01} - 1\right)}} \tag{2}$$

where WTA* is a modified WTA to account for monthly and annual variability of precipitation.

The severity of water scarcity of watersheds is ranked as follows: WSI < 0.1 low; $0.1 \leq$ WSI < 0.5 moderate; $0.5 \leq$ water stress indicator < 0.9 severe and WSI > 0.9 extreme (Pfister et al., 2009; Ridoutt and Pfister, 2010).

The end-point impact category focuses on three areas of protection related to water consumption: human health, ecosystem quality and depletion of freshwater resources. These are calculated according to the following three formulae, respectively:

i) Human health

$$\begin{aligned}\Delta HH_{\text{malnutrition}} = {} & WSI \times WU_{\text{agriculture}} \times HDF_{\text{malnutrition}} \\ & \times WR^{-1}_{\text{malnutrition}} \times DF_{\text{malnutrition}} \\ & \times WU_{\text{consumptive}}\ (\text{DALY})\end{aligned} \tag{3}$$

where: $\Delta HH_{\text{malnutrition}}$ – damage to human health due to malnutrition in disability adjusted life years (DALY); WSI – water stress index (–); $WU_{\text{agriculture}}$ – fraction of agricultural water use at the watershed level (–); $HDF_{\text{malnutrition}}$ – human development factor linking human development index to malnutrition vulnerability (–); $WR_{\text{malnutrition}}$ – water requirement to prevent malnutrition

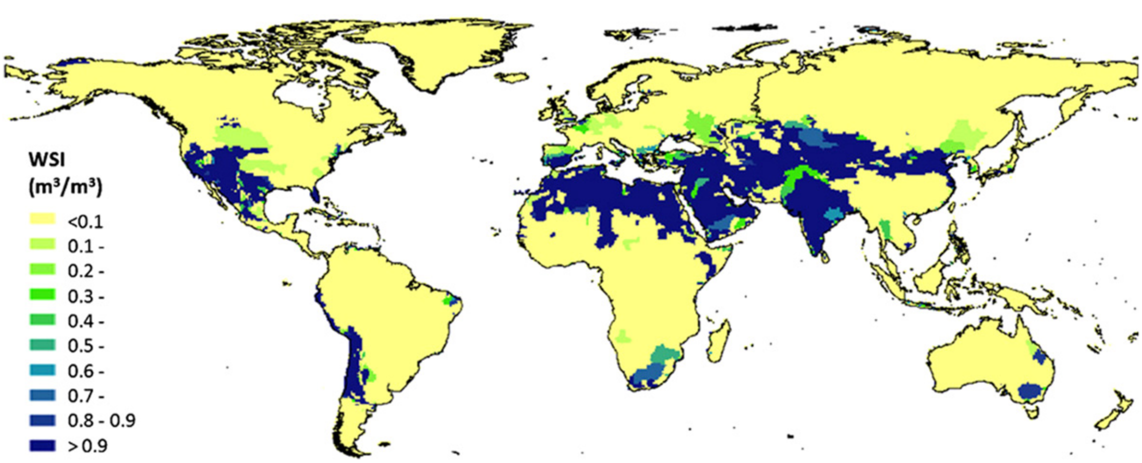

Fig. 3. A map of water stress index at the watershed level (Pfister et al., 2009).

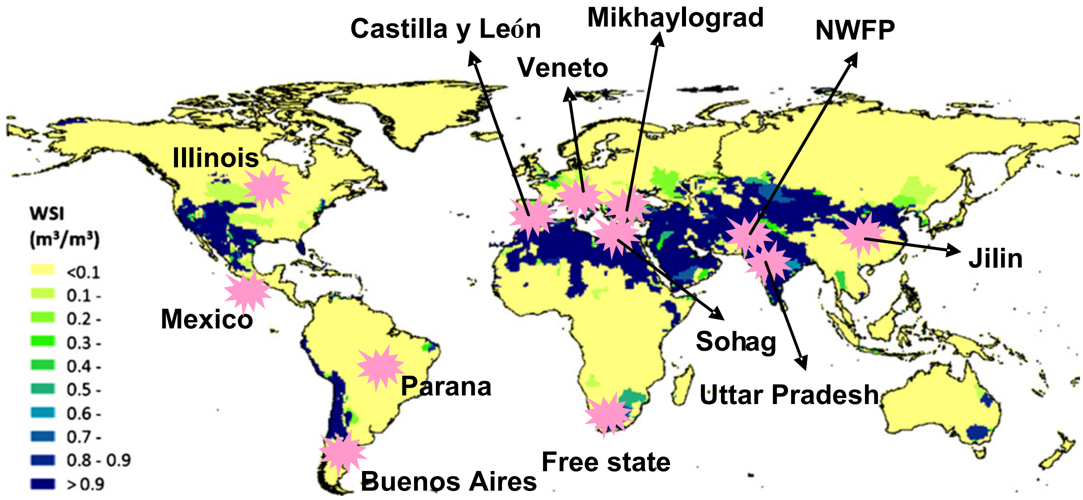

Fig. 4. Location of the regions considered in the case study (see Table 4 for more detail).

(m^3/year person); $DF_{malnutrition}$ – damages due to malnutrition (DALY/year person); $WU_{consumptive}$ – blue water consumption (m^3).

ii) Ecosystem quality

$$\Delta EQ = \frac{NPP_{wat-lim}}{P} \times WU_{consumptive} \left(m^2 \cdot year\right) \quad (4)$$

where: ΔEQ – damage to ecosystem quality (m^2 year); $NPP_{wat\text{-}lim}$ – net primary production limited by water availability representing vulnerability of an ecosystem due to water shortage (–); P mean annual rainfall (m/year).

iii) Depletion of freshwater resources

$$\Delta R = E_{desalination} \times F_{depletion} \times WU_{consumptive}\ (MJ) \quad (5)$$

where: ΔR – damage to freshwater resources (MJ); $E_{desalination}$ – energy required for seawater desalination (MJ/m^3); $F_{depletion}$ – fraction of freshwater consumption that contributes to depletion.

The above impacts categories are aggregated into a single score indicator as per the Eco-indictor 99 (hierarchist perspective) method (Goedkoop and Spriensma, 2001). The impact due to water consumption is expressed as EI99HA points/m^3.

Although these measures are devised to suit the available data, they represent an attempt to develop an indicator that reflects local impacts. The method does not model any ecological impacts arising from the 'degraded' water (water released back to the watershed after use).

The next section illustrates how the above-discussed methods can be used for assessing the impacts of water use and how the results might differ. The case study of bioethanol from corn is used for these purposes.

Prior to that, it is worth noting that in addition to the above-discussed methods, a few other methods have been proposed for assessing the impacts of water use (Berger and Finkbeiner, 2010). These include: Brent (2004) method for assessing site-specific impacts for South Africa; Motoshita et al. (2008) approach for quantification of damages to human health due to undernourishment related to agricultural water scarcity; Van Zelm et al. (2009) approach for assessing the ecological damage of groundwater extraction; and Motoshita et al. (2011) model for assessing damages to human health due to infections caused by domestic water scarcity. Although these methods are suitable for assessing a wide range of water uses, their focus is limited either to a particular country, a specific type of water use or a specific type of damage. This therefore limits their application and they are not considered further in this paper.

Table 4
Water stress levels in the regions considered in the case study.

Country	Selected region/state	River basin	Water stress level			
			Country level (eco-scarcity method) (Frischknecht et al., 2009a)	River basin (Smakhtin et al., 2004)	Watershed (Pfister et al., 2009)	Watershed (eco-scarcity method) (Frischknecht et al., 2009b)
USA	Illinois	Mississippi (Illinois river basin)	Moderate	Heavily exploited	Low	Low
China	Jilin	Amur (Songhua River)	Medium	Slightly exploited	Moderate	Very high
Brazil	Parana	Parana	Low	Slightly exploited	Low	Low
Mexico	Mexico	Balsas	Moderate	Slightly exploited	Moderate	Extreme
Argentina	Buenos Aires	Parana	Low	Slightly exploited	Low	Extreme
India	Uttar Pradesh	Ganges	Medium	Moderately exploited	Extreme	Extreme
Italy	Veneto	Po	Medium	Moderately exploited	Low	Moderate
South Africa	Free state	Orange	Medium	Moderately exploited	Severe	Extreme
Egypt	Sohag	Nile	Extreme	Slightly exploited	Extreme	Extreme
Pakistan	North West Frontier Province (NWFP)	Indus	Very high	Overexploited	Moderate	High
Spain	Castilla y León	Ebro	Medium	Overexploited	Moderate	High
Bulgaria	Mikhaylograd	Danube	High	Moderately exploited	Moderate	Very high

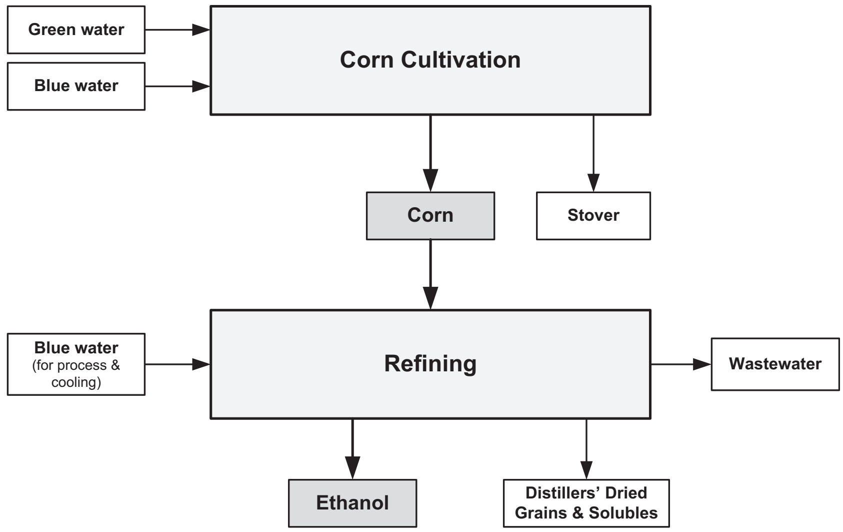

Fig. 5. Ethanol production from corn.

3. Case study: assessing water use impacts of bioethanol from corn

The purpose of this case study is to compare the suitability of the above-discussed methods for assessing the impacts of water use. Corn is most-widely used as a cattle feed grain and is also processed into multitude of food and industrial products including bioethanol. The availability of freshwater is prerequisite during the cultivation of corn and its subsequent processing into industrial products. Recent increase in production of corn-based bioethanol, especially in the USA, has highlighted the tensions between energy, food and water security (Dominguez-Faus et al., 2009). This study assesses the impacts of water use in the production of ethanol from corn produced in 12 different countries.

These countries (see Fig. 4) have been selected for study here because they produce 80% of the global corn production (USDA, 2009). Moreover, they differ significantly with respect to the water stress levels, from no water stress to a very high stress (Table 4) and are therefore appropriate for assessing the suitability of different methods discussed in the previous sections. Table 4 also shows that the Pfister et al. (2009) and the eco-scarcity (Frischknecht et al., 2009b) methods classify stress level differently for some watersheds. The selected states and regions for each country considered in the case study are given in Fig. 4; these have the highest corn production in their respective countries (USDA, 2009). It is important to note that most of these countries have not yet started production of corn-ethanol at a large scale.

3.1. Life cycle inventory of water use

The life cycle of ethanol production from corn is shown in Fig. 5. With respect to water use, two main production stages can be differentiated in the life cycle: corn cultivation and bioethanol production (refining). The type of water used in each stage is also indicated in the figure. The analysis is based on the functional unit of 1 GJ of energy contained in bioethanol.

As shown in Fig. 5, both corn cultivation and refining produce co-products: stover in the former and distillers' dried grain and solubles (DDGS) in the latter stage. Therefore, it is necessary to allocate the water use between the respective co-products. For the purposes of this study, allocation based on economic value has been applied; allocation factors are summarised in Table 5.

The following sections outline the methodology and discuss the results of water use estimation for each life cycle stage in turn.

3.1.1. Water use in agricultural production

Table 6 summarises the results for water use in this stage. The water use (m^3/GJ of ethanol) in the agricultural stage has been calculated by dividing crop water requirement (CWR) by the ethanol yield (Gerbens-Leenes et al., 2009). CROPWAT software has been used to calculate the CWR for corn cultivation in the 12 countries (FAO, 2009a). CROPWAT calculates total as well as freshwater (blue) requirements based on evapo-transpiration, using climatic data including availability of green water (rainwater available as a soil moisture). The data for the main corn production areas in these countries and the cropping seasons have been derived from the USDA (2009). The climate data for these locations required for CROPWAT have been collected from CLIMWAT (FAO, 2009b).

The ethanol yield (GJ/ha) has been calculated by multiplying corn yield (tonne/ha) by ethanol production (litres/tonne of corn) and energy content of ethanol (GJ/litre) (Gerbens-Leenes et al., 2009). Data on corn yield for each country, obtained from USDA (2009), are given in Table 6. It has been assumed that one tonne of corn produces about 400 L of ethanol and that 1 L of ethanol has the energy content equivalent to 0.0236 GJ (National Academy of Sciences, 2008).

Table 5
Allocation of water use between co-products.

Production stage	Products	Mass ratio (Kim and Dale, 2002; Luo et al., 2009)	Price (US$/kg) (Ethanol Market, 2008; Tiffany, 2008)	Economic value (%)
Corn cultivation	Corn[a]	0.63	0.21	81.6
	Stover	0.37	0.08	18.4
Refining	Ethanol	0.52	0.6	86.7
	DDGS	0.48	0.01	13.3

[a] The agricultural water allocated for corn is further allocated between ethanol and DDGS in the ratio of 86.7:13.3.

Table 6
Calculated water use in the cultivation of corn.

Region (Country)	Rainwater use m^3/ha	Irrigation water required m^3/ha	Total CWR m^3/ha	Corn yield tonne/ha (2007–2008) (USDA, 2009)	Ethanol yield GJ/ha	Green water use m^3/GJ	Blue water use m^3/GJ	Total water use m^3/GJ
Illinois (USA)	2242	2354	4596	9.46	89.3	25.1	26.4	51.5
Jilin (China)	2426	826	3253	5.17	48.8	49.7	16.9	66.7
Parana (Brazil)	2690	336	3026	3.78	35.7	75.4	9.4	84.8
Mexico (Mexico)	2644	13	2658	3.22	30.4	87.0	0.4	87.4
Buenos Aires Province (Argentina)	2342	1228	3570	7.67	72.4	32.4	17.0	49.3
Uttar Pradesh (India)	2224	546	2770	2.44	23.0	96.5	23.7	120.3
Veneto (Italy)	1430	1991	3421	9.67	91.3	15.7	21.8	37.5
Free state (South Africa)	2476	951	3427	4.00	37.8	65.6	25.2	90.8
Sohag (Egypt)	0	6667	6667	8.05	76.0	0.0	87.7	87.7
NWFP (Pakistan)	826	3145	3971	3.43	32.3	25.5	97.2	122.8
Castilla y León (Spain)	920	3438	4358	9.92	93.6	9.8	36.7	46.6
Mikhaylograd (Bulgaria)	1304	2250	3553	1.46	13.8	94.6	163.3	257.9

Table 7
Freshwater use in a corn-ethanol refinery (National Academy of Sciences, 2008).

Water (blue) input			Water output	
Process water (m^3/GJ)	Cooling water (m^3/GJ)	Total (m^3/GJ)	Wastewater discharge (m^3/GJ)	Water consumed (m^3/GJ)
0.05	0.08	0.13	0.03	0.10

3.1.2. Water use for ethanol production

Freshwater requirements in a corn-ethanol refinery consist of process water and cooling water. Part of the water intake is lost via evaporation in the cooling tower and the dryer system in the refinery and the remaining is discharged as wastewater. Table 7 provides a typical water balance for a large refinery (National Academy of Sciences, 2008). It also shows that in comparison to the agricultural water consumption, the industrial water consumption is negligible.

3.1.3. Comparison of water use results

Fig. 6 shows the results for water use obtained using the water footprint, Milà i Canals et al. (2009) and Pfister et al. (2009) approaches. The results have been calculated using the water footprinting tool CCaLC V2.0 (CCaLC, 2011). As discussed in Section 2.1, the water footprint approach considers blue, green and grey water; the Milà i Canals et al. method accounts for blue and green water while the Pfister et al. method considers only blue water. Fig. 6 shows the results for both the blue and green water as well as the total water use; grey water has not been considered due to the lack of reliable and consistent data, as discussed previously.

According to the water footprint and Milà i Canals et al. approaches, Mikhaylograd (Bulgaria) has the highest water usage (258 m^3/GJ) and Veneto (Italy) the lowest (37 m^3/GJ). If only the blue water is considered, as is the case in the Pfister et al. approach, then Mikhaylograd still has the highest usage (163 m^3/GJ) but corn from Mexico has the lowest water consumption (0.54 m^3/GJ) as the cultivation relies almost entirely on green water. Similarly, the majority of water requirements for corn cultivation in Parana (Brazil), Uttar Pradesh (India), Free state (South Africa), Jilin (China) and Buenos Aires province (Argentina) are met by green water, which is in contrast to Mikhaylograd (Bulgaria), Sohag (Egypt), NWFP (Pakistan) and Castilla y León (Spain), where the corn cultivation relies heavily on the blue water.

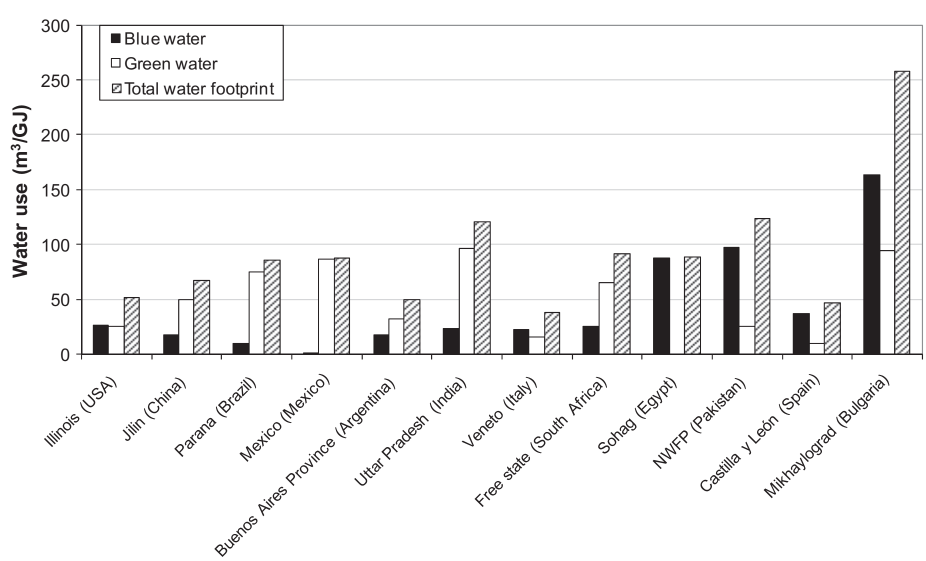

Fig. 6. Case study results based on different water methods for estimating water use in LCI.

While these results are informative in terms of the different types of water used, they do not tell us anything about the impacts that this has on the country or the region. For example, as shown in Table 4, the water stress in Mikhaylograd at both the river basin and watershed levels is 'moderate' while that in Uttar Pradesh is 'extreme' at the watershed level. The water footprint for the latter is the third largest in this case study (120 m^3/GJ) so, while only a half of that in Mikhaylograd, its impact would arguably be higher than suggested merely by the amount of water used. As discussed previously, this information cannot be captured at the inventory level and impact assessment methods should be used instead. This is discussed in the following sections.

3.2. Impacts of water consumption

The three approaches for assessing the impacts of water consumption discussed in Section 2.2 (eco-scarcity, Milà i Canals et al. and Pfister et al. methods) provide different characterisation factors for assessing the impacts of water consumption either at a country level, river basin level or watershed level. The eco-scarcity and Pfister et al. approaches consider the impacts of blue water only while Milà i Canals et al. method also includes land use effects on water. The results of applying these approaches to the case study are presented in Table 8 and Fig. 7 and discussed below.

3.2.1. The eco-scarcity method

Table 8 shows the water use impacts of ethanol for each country based on the eco-scarcity method (Frischknecht et al., 2009a). The eco-factors for assessing the impacts of water consumption are based on WTA at national level. The results at country level show that the water consumption impact for Egypt, which is under the 'extreme' water-scarcity category, is thousand times higher than the water consumption impact for Brazil, and Argentina where the water stress level is 'low'. The results also indicate that the impact in Pakistan and Bulgaria, which are in the 'very high' and 'high' water stress categories, respectively (Table 4), are very high. Since the use of green water is not considered in this method, the water use impact for Mexico, where most of the crop water requirements are met through rainfall, is negligible. Argentina and Brazil also have negligible impacts due to the very small eco-factors.

Fig. 8 compares the results for the eco-scarcity method at the country level with the watershed levels. As can be seen, the results at the watershed level are significantly different from those at the country level in several cases because of the different water stress values at the national and watershed levels (see Table 4). This applies for example to the case of the Buenos Aires province (Argentina), Uttar Pradesh (India), Free state (South Africa), Jilin (China) and Illinois (USA).

3.2.2. The Milà i Canals et al. approach

As discussed in Section 2.2.2, this method proposes the use of water stress indicator (the ratio of water withdrawal from a river basin to the water available for human use) as the mid-point characterisation factor. The water stress indicators for the river basins for the selected areas for each country are shown in Table 8; also given are the land use effects on water which are calculated as discussed in Section 2.1.2. The results show that the highest water consumption impact is for NWFP (Pakistan) because of the very high water stress indicator for the river Indus. According to this approach, Mikhaylograd (Bulgaria), Castilla y León (Spain), Sohag (Egypt) and Illinois (USA) also have a relatively high impact, while Brazil, Argentina, Mexico and China have a low impact. This is partly due to the water stress indicator of the relevant river basins and partly due to the quantity of blue water consumed.

Table 8
Impacts of water use from ethanol production based on different methods.

Region (Country)	Total blue water consumed (m^3/GJ)	Eco-scarcity approach (country level) (Frischknecht et al., 2009a,b)		Milà i Canals et al. (2009) method (river basin)			Pfister et al. (2009) method (WSI – watershed)		Pfister et al. (2009) method (EI99HA – watershed)	
		Characterisation factors (EP/m^3) (Frischknecht et al., 2009a)	Water use impacts (EP/GJ)[a]	Land use effects on water (m^3/GJ)	Characterisation factors (Smakhtin et al., 2004)	Water use impacts (ecosyst-eq. water m^3/GJ)[b]	Characterisation factors (Pfister et al., 2009)	Water use impacts (m^3/GJ)[c]	Characterisation factors (Pfister et al., 2009)	Water use impacts (EI99HA points/GJ)[c]
Illinois (USA)	26.45	220	5820	1.9	0.69	19.65	0.01	0.29	0.02	0.44
Jilin (China)	17.03	880	14,987	6.1	0.05	1.16	0.12	2.12	0.02	0.35
Parana (Brazil)	9.51	24	228	5.9	0.03	0.52	0.01	0.11	0.01	0.05
Mexico (Mexico)	0.54	220	118	9.4	0.12	1.14	0.27	0.14	0.11	0.06
Buenos Aires Province (Argentina)	17.07	24	410	2.6	0.03	0.67	0.01	0.18	0.03	0.46
Uttar Pradesh (India)	23.80	880	20,948	9.2	0.40	13.04	1.00	23.80	0.11	2.73
Veneto (Italy)	21.91	880	19,277	1.1	0.54	12.45	0.02	0.49	0.01	0.12
Free state (South Africa)	25.28	880	22,246	5.7	0.44	13.60	0.54	13.67	0.14	3.64
Sohag (Egypt)	87.84	22,000	1,932,427	0	0.30	26.00	1.00	87.84	0.21	18.65
NWFP (Pakistan)	97.33	6200	603,449	3.5	4.08	411.18	0.40	38.86	0.07	6.37
Castilla y León (Spain)	36.82	880	32,404	1.1	1.01	38.41	0.17	6.30	0.02	0.78
Mikhaylograd (Bulgaria)	163.39	2400	392,142	11	0.49	85.63	0.11	17.71	0.02	2.70

[a] Impacts of water consumption = Total blue water abstracted × characterisation factor.
[b] Impacts of water consumption = {Total blue water consumed + land use effects on water} × characterisation factor.
[c] Impacts of water consumption = Total blue water consumed × characterisation factor.

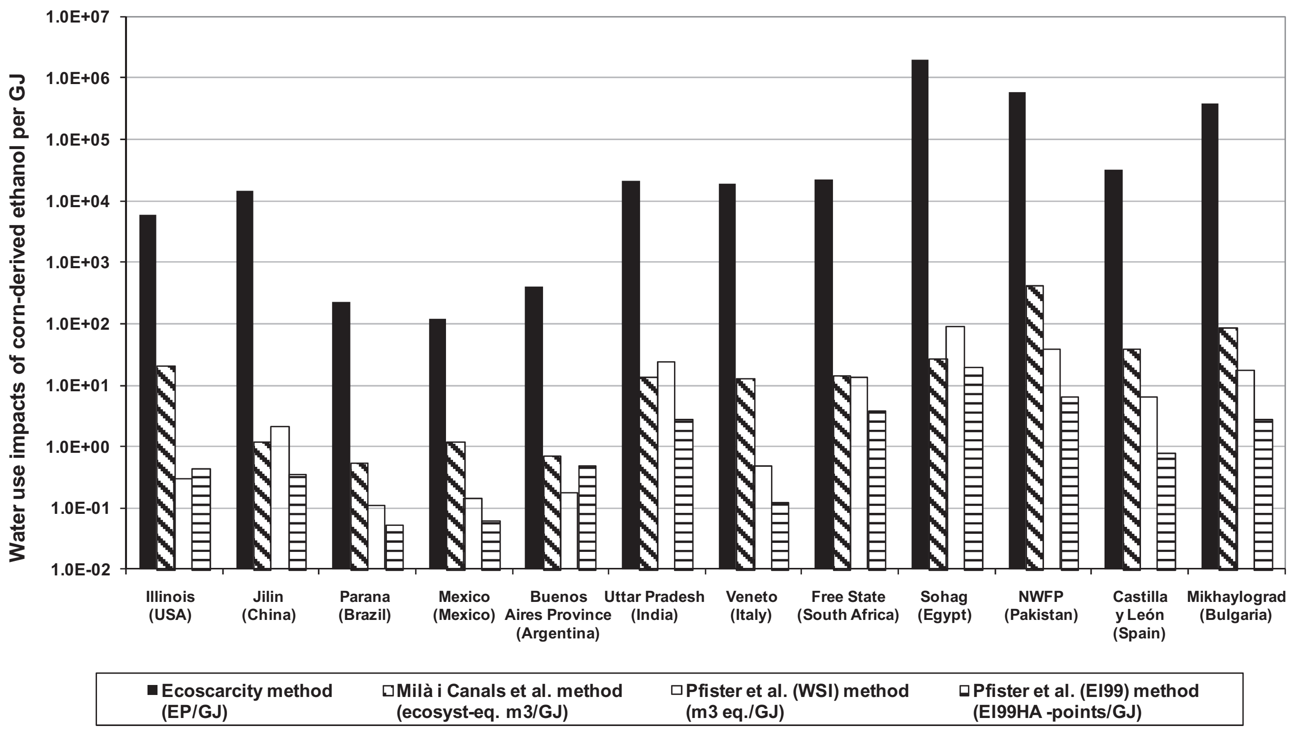

Fig. 7. Case study results based on different impact assessment approaches.

This approach includes two methods (WSI and EI99HA) for assessing the impacts of water consumption. The characterisation factors used in the WSI approach are based on the ratio of water withdrawals to availability at a watershed level (Section 2.2.3). Table 8 provides these factors (water stress index) for the watersheds for the selected areas for each country considered in this case study. According to this approach, the water stress index for the selected regions in Uttar Pradesh (India) and Sohag (Egypt) is categorised as 'extreme' (Table 4) resulting in the highest characterisation factors. Consequently, the production of bioethanol in Sohag (Egypt) has the highest water use impact (Fig. 7). On the other hand, the impact in Uttar Pradesh (India) is about one quarter of that in Sohag (Egypt) because of high availability of green water during the cropping season, leading to a lower requirement for blue water. Although Mikhaylograd (Bulgaria) and NWFP (Pakistan) have the highest blue water consumption, owing to the 'moderate' stress level of the relevant watersheds (Table 4) the impact of water consumption is much lower than for Sohag (Egypt). Similarly, the equivalent impact

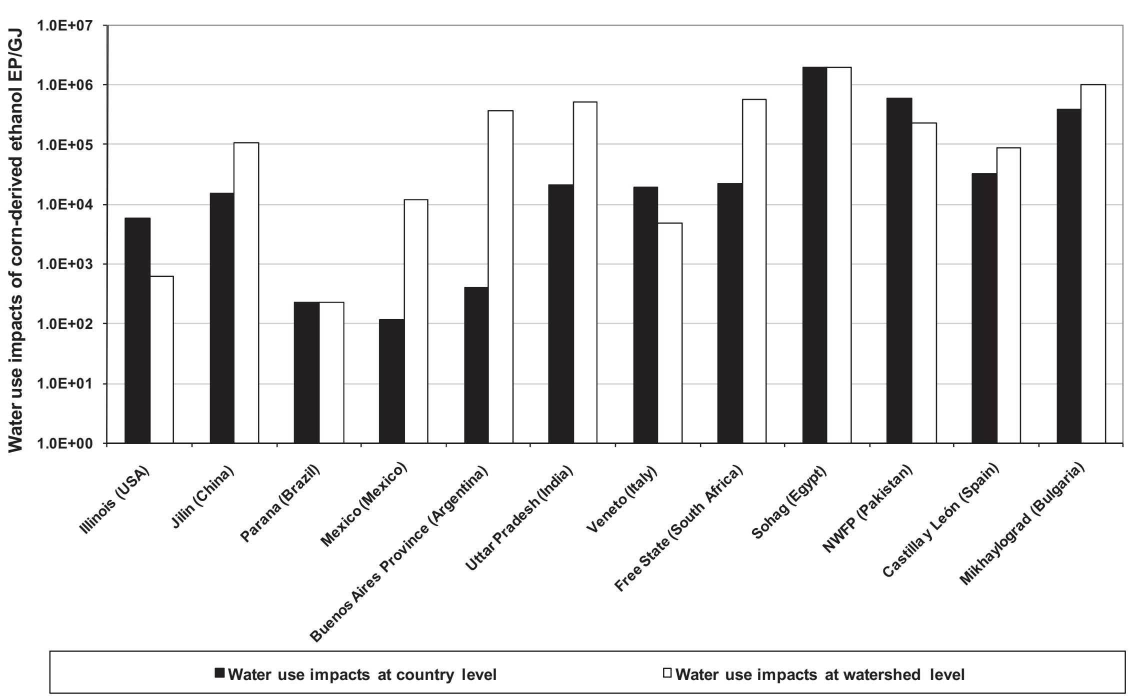

Fig. 8. Comparison of results based on the eco-scarcity method at the country and watershed levels.

in Illinois (USA), Parana (Brazil), Mexico (Mexico), Buenos Aires province (Argentina), Veneto (Italy), Jilin (China) and Castilla y León (Spain) is negligible to very low because of the very low characterisation factors for the concerned watersheds.

According to the EI99HA method, the highest water consumption impact is for Sohag (Egypt) because of the very high characterisation factor (Table 8 and Fig. 7). Fig. 7 also shows that NWFP (Pakistan), Uttar Pradesh (India), Mikhaylograd (Bulgaria) and Free state (South Africa), have a relatively high impact partly due to the characterisation factors for the relevant watersheds and partly due to the quantity of blue water consumed.

3.3. Discussion

These results show that the demand for the freshwater for corn cultivation varies among countries depending on the climatic conditions, seasonal variations and amount of rainfall in the area during the cropping season. The use of freshwater for other agricultural products would also vary significantly from one part of the country to the other, for the same reasons. Therefore, quantification of water use for the agricultural activities would need to be carried out for the specific location under study as the national level data would not truly reflect the actual water use (Pfister et al., 2009).

The results of the above case study also show that in comparison to the agricultural water consumption, the industrial water consumption is negligible. Nevertheless, the industrial water consumption is often withdrawn from the local point sources and can have localised impacts on water availability (Dominguez-Faus et al., 2009).

With respect to the suitability of the different methods for assessing the impacts of water use, arguably the approaches which only assess the quantity of the water used provide only part of the information. The environmental impacts of water consumption will be different depending on the level of water scarcity of the area even if the quantity used is the same for a particular product. Therefore, the volumetric water footprint could be misleading as it does not account for the environmental impacts of water consumption (Ridoutt et al., 2010; Ridoutt and Pfister, 2010).

Similar to the quantification of water use, impacts of water consumption also need to be assessed for the specific location under study (Pfister et al., 2009). This means that the national level characterisation factors are not appropriate for assessing the water use impacts. The case study results show that the characterisation factors based on the river basin data are also not suitable as they do not differentiate between the upstream and downstream water use. For example, in this approach river Nile is categorised as having 'slightly exploited' water stress, even in Egypt, where the availability of water is limited. Similarly, river Indus is categorised as 'highly exploited' regardless of location, even though upstream (in NWFP Pakistan, in this case study) water availability is not an issue.

Therefore, the impact assessment methods based on watersheds appear to be most appropriate. However, the watershed based approaches discussed here also have some limitations; for example, although the Pfister et al. (2009) method takes into account seasonal water variations, these are averaged across all the seasons thus obscuring the specific variations. As shown in this case study, the watershed for Uttar Pradesh (India) is categorised under 'extreme' stress level because it is based on the annual data, even though the corn cropping is carried out during the monsoon season when, owing to the heavy rains, water availability is generally not an issue. In fact, during that time it is often the monsoon flooding which causes problems rather than the water scarcity.

In order to assess the water use impacts using LCA, ideally, LCI should contain data on quantities of water abstracted, sources of water, geographical location, timing of water abstraction and water discharges. The data on water abstraction, sources and discharge would help in estimating the quantities of freshwater consumed, while the spatial and temporal data on water use are needed for assessing the impacts. However, considering LCA as a site-generic assessment tool, incorporation of location specific and time specific data is a considerable challenge.

4. Conclusions and recommendations for future work

The methodologies, approaches and indicators for assessing the impacts of freshwater usage are still evolving. The concept of water footprint has been an important step in this direction. However, volumetric water footprints do not address the issue of water availability, or scarcity, which is often of concern in assessing the impacts of freshwater consumption. This has been illustrated in the case study presented in this paper, showing that the impacts of water consumption are better portrayed when the scarcity factors are taken into account.

Different types of characterisation factors have been proposed for assessing water stress, based on water-use-to-availability ratios at the national, river basin or watershed levels. This paper has argued that the water stress indicators at the national or river basin levels inevitably mask large variations, especially for geographically diverse countries and large rivers. Ideally, the characterisation factors should be based on site-specific information on water availability and its usage. However, availability of such type of data is an issue. Characterisation factors at the watershed level provide a useful alternative but this case study reveals that they also have some limitations. Like the national and river basin level indicators, they are also based on the annual data and hence cannot account for the seasonal variations with regard to the water availability and abstraction. Furthermore, the environmental impacts of water consumption depend not only on water scarcity, but also on the magnitude and frequency of shocks like floods and droughts, resilience and vulnerability of ecosystems. Currently, damage to the ecosystems due to the water is considered only in one of the methods (Pfister et al.) while other approaches do not take these parameters into account. Therefore, further methodological developments are needed to provide a more robust basis for considering these issues. Further research should also examine if LCA is an appropriate tool to address these issues or other tools should be used instead.

Availability of quality data on water use for various purposes is the limiting factor for assessing the water use impacts. For most agricultural and industrial processes, actual water use data are scarce because few farmers and companies collect or report water usage. Moreover, in the absence of a uniform reporting format, data that do exist are not collected and reported in a consistent way. Other issues such as lack of measurement of groundwater usage and wastewater discharges make it difficult to estimate correctly water consumption. Obtaining this information in a consistent format would be a further necessary step in the development of appropriate methods for estimating the impacts of water use.

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