Assessing the Environmental Impacts of Freshwater Consumption in LCA

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A method for assessing the environmental impacts of freshwater consumption was developed. This method considers damages to three areas of protection: human health, ecosystem quality, and resources. The method can be used within most existing life-cycle impact assessment (LCIA) methods. The relative importance of water consumption was analyzed by integrating the method into the Eco-indicator-99 LCIA method. The relative impact of water consumption in LCIA was analyzed with a case study on worldwide cotton production. The importance of regionalized characterization factors for water use was also examined in the case study. In arid regions, water consumption may dominate the aggregated life-cycle impacts of cotton-textile production. Therefore, the consideration of water consumption is crucial in life-cycle assessment (LCA) studies that include water-intensive products, such as agricultural goods. A regionalized assessment is necessary, since the impacts of water use vary greatly as a function of location. The presented method is useful for environmental decision-support in the production of water-intensive products as well as for environmentally responsible value-chain management.

Introduction

In many regions, human well-being and ecosystem health are being seriously affected by changes in the global water cycle, caused largely by human activities (*1*). Despite the relevance of freshwater to human health and ecosystem quality, the life cycle assessment (LCA) methodology is lacking comprehensive approaches to evaluate the environmental impacts associated with water use (e.g., ref *2*). Water use is often reported in the life-cycle inventory phase, where resource use and energy and material consumption are recorded, but little differentiation is made between the types of water use (see, e.g., ref *3*). Even less attention is paid to water use in life-cycle impact assessment (LCIA), in which emissions and resource uses are grouped and compared according to their environmental impacts (e.g., global warming or resource depletion). So far, impacts on water resources have only been described qualitatively (*4*), with the exception of the Ecological Scarcity 2006 method (UBP06). The Ecological Scarcity method quantifies eco-factors on the basis of defined environmental targets (*5*) without addressing specific damages to human health and ecosystems. Due to the spreading application of LCA worldwide and increasingly refined LCIA-methods, regional differentiation of damage characterization factors has gained in importance (*6*). For an appropriate assessment of water use, regionalization is crucial to capture hydrological conditions.

Several hydrological assessments of global water resources exist (e.g., ref *7*), and different models of global water availability and resulting water stress have been developed (*8, 9*) allowing for the assessment of water shortage in climate-change and varying population-dynamics scenarios. However, none of these methods explicitly considers cause−effect relationships between water use and environmental impacts.

The goals of this paper are to (1) set up a framework for life cycle inventory analysis; (2) to describe and model the impacts of water use on the safeguard objects, human health, ecosystems, and resources; (3) to present a regionalized approach for assessing water-use related environmental impacts within existing LCIA methods; and (4) to illustrate the application of this method and the relevance in a LCA case study, using the example of worldwide cotton production.

Materials and Methods

Water Use in Life-Cycle Inventory Analysis. In the life-cycle inventory (LCI) phase, quantities of water used are often reported (e.g., ref *3*). Ideally, the type of use, source of water, and geographical location should also be documented. As recognized in earlier research (*4*), in-stream and off-stream use need to be distinguished. Water withdrawals represent off-stream use, while in-stream use denotes the use of water in the natural water body, e.g., hydropower production and transport on waterways. Further, we suggest differentiating between consumptive and degradative use (see Supporting Information Figure S1 for an illustration of the framework proposed). Consumptive use (water consumption) represents freshwater withdrawals which are evaporated, incorporated in products and waste, transferred into different watersheds, or disposed into the sea after usage (*10*). Degradative use describes a quality change in water used and released back to the same watershed, and requires a description of inputs and outputs in the inventory analysis. While the effect of emissions to the aquatic environment is assessed in conventional LCA, e.g., regarding ecotoxicity, the loss of water quality still needs to be quantified as a loss of freshwater resources. The water source, either groundwater, surface, brackish, or seawater provides a rough indicator of quality, and is differentiated in some LCI databases (*3*). Additionally, water should be characterized by a set of quality classes, considering, for instance, organic pollution and temperature increase. Degradative use would consequently be assessed as loss of high quality and gain of lower quality water.

In the method outlined here, we focus on the assessment of consumptive water use ($WU_{consumptive}$), as it is the crucial use type from a hydrological perspective (*10*). To generate a regionalized inventory for water consumption in agriculture, we use the "virtual water" database, which covers many crops for most countries (*11*). Virtual water defines the amount of water evaporated in the production of, and incorporation into, agricultural products, neglecting runoff. It consists of "green" and "blue" water flows (*10*). Green water is precipitation and soil moisture consumed on-site by vegetation. Blue water, in contrast, denotes consumption of any surface and groundwater, and in the case of agricultural production particularly, irrigation water. In the framework proposed in this paper, only the amount of blue virtual water consumption is considered. However, virtual water currently has been separated into green and blue water for some crops only. Such data gaps were abridged with calculation routines

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that quantify blue water as a function of crop-property and climate data (*12*).

Regionalization. The ecological impacts of water use depend on many spatial factors, such as freshwater availability and use patterns at the specific location under study. For the regionalized impact assessment of water use, we utilized a geographic information system (GIS) allowing data processing and statistical evaluation on different spatial resolutions. Impact factors were calculated on country and major watershed levels (e.g., the Rhine; bigger rivers as Mississippi or Murray-Darling are divided into subcatchments, as defined by ref *9*). The watershed level is a more appropriate choice for the assessment, as hydrological processes are connected within watersheds, but in many cases only country-level inventory data is available (*3*).

Screening Assessment with the Water Stress Index (WSI). Water stress is commonly defined by the ratio of total annual freshwater withdrawals to hydrological availability (WTA, eq 1). Moderate and severe water stress occur above a threshold of 20 and 40%, respectively (*8, 13*). These figures are expert judgments and thresholds for severe water stress might vary from 20 to 60% (*13*). We advance this concept to calculate a water stress index (WSI), ranging from 0 to 1, which serves as a characterization factor for a suggested midpoint category "water deprivation" in LCIA. The WSI indicates the portion of $WU_{consumptive}$ that deprives other users of freshwater.

To calculate WSI, the WaterGAP2 global model (*9*) was used, describing the WTA ratio of more than 10 000 individual watersheds. The model consists of both a hydrological and a socio-economic part, quantifying annual freshwater availability (WA_i) and withdrawals for different users j (WU_{ij}), respectively, for each watershed i:

$$WTA_i = \frac{\sum_j WU_{ij}}{WA_i} \tag{1}$$

where WTA_i is WTA in watershed i and user groups j are industry, agriculture, and households.

Hydrological water availability modeled in WaterGAP2 is an annual average based on data from the so-called *climate normal period* (1961−1990) (*9*). However, both monthly and annual variability of precipitation may lead to increased water stress during specific periods, if only insufficient water storage capacities are available (e.g., dams) or if much of the stored water is evaporated. Such increased stress cannot be fully compensated by periods of low water stress (*13*). To correct for increased effective water stress, we introduced a variation factor (VF) to calculate a modified WTA (WTA*, eq 2), which differentiates watersheds with strongly regulated flows (SRF) as defined by Nilsson et al. (*14*). For SRF's, storage structures weaken the effect of variable precipitation significantly, but may cause increased evaporation. For a conservative assessment a reduced correction factor was applied (square-root of VF):

$$WTA^* = \begin{cases} \sqrt{VF} \times WTA & \text{for SRF} \\ VF \times WTA & \text{for non} - \text{SRF} \end{cases} \tag{2}$$

VF was derived from the standard deviation of the precipitation distribution. We analyzed the monthly and annual precipitation provided in the global climate data set CRU TS2.0 (*15*) and tested the fitting of these data for normal and log-normal distribution. For monthly precipitation, the Kolmogorov−Smirnov test favored the log-normal over the normal distribution for 61% of all grid cells and for 90% of grid cells with a coefficient of variation >0.85. For annual variability of precipitation, McMahon et al. (*16*) investigated 1221 unimpacted stream flows worldwide and found that the log-normal distribution generally fits better than the normal distribution. Consequently, we defined VF as the aggregated measure of dispersion of the multiplicative standard deviation of monthly (s^*_{month}) and annual precipitation (s^*_{year}), assuming a log-normal distribution and considering precipitation data from 1961−1990 (*15*):

$$VF = e^{\sqrt{\ln(s^*_{month})^2 + \ln(s^*_{year})^2}} \tag{3}$$

To arrive at WTA*, we calculated variation factors for each grid cell i (VF_i) and aggregated all VF_i on watershed-level (VF_{ws}), weighted by the mean annual precipitation P_i (m) in grid cell i:

$$VF_{WS} = \frac{1}{\sum P_i} \sum_{i=1}^{n} VF_i \cdot P_i \tag{4}$$

Obviously, water stress is not linear with regards to WTA* as indicated by the water stress definitions (see above). We adjusted the water stress index (WSI) to a logistic function to achieve continuous values between 0.01 and 1:

$$WSI = \frac{1}{1 + e^{-6.4 \cdot WTA^*}\left(\frac{1}{0.01} - 1\right)} \tag{5}$$

WSI has a minimal water stress of 0.01 as any water consumption has at least marginal local impact. The curve is tuned to result in a WSI of 0.5 for a WTA of 0.4, which is the threshold between moderate and severe water stress, when applying the median variation factor of all watersheds (VF_{median} = 1.8, WTA* = 0.72). Accordingly, WTA of 0.2 and 0.6 result in WSI of 0.09 and 0.91, respectively (Figure 1a).

The expanded WSI index can serve as general screening indicator or characterization factor for water consumption in LCIA, e.g., as a separate impact category in methods such as CML2001 (*17*). It will also be used in the assessment of damages to human health.

Damage Assessment. In the present work, damage assessment is performed according to the framework of the Eco-indicator-99 assessment methodology (EI99 (*18*)), but it can also be adapted to similar methods, such as LIME (*19*) or IMPACT2002+ (*20*). The potential environmental damages of water use are assessed for three areas of protection (AoP): human health, ecosystem quality, and resources.

Damage to Human Health. Two major water-scarcity related impact pathways for human health are generally observed (*21*): lack of freshwater for hygiene and ingestion, resulting in the spread of communicable diseases, and water shortages for irrigation, resulting in malnutrition. Both pathways are mainly relevant in developing countries. We disregard water shortages for pure drinking purposes as they are a problem of disasters (e.g., extreme droughts or war) (*21*), and such extraordinary events are generally excluded from LCA. Damages to human health from malnutrition and poor hygiene are often linked (Supporting Information Figure S4) but can also contribute independently in different settings. Here, we focus on the food production effects of water deprivation because competition in water scarce areas ultimately affects irrigation. Damage arising from reduced water availability for hygiene depends on local circumstances, e.g., the distance of the population to the next well, and is therefore difficult to assess in LCA.

In developing countries, freshwater shortage is associated on a regional scale with numerous influencing factors (e.g., lack of wastewater-treatment infrastructure), in addition to physical water scarcity (*22*). Socio-economic parameters are also relevant for mitigation measures of potential health damages. To cope with this high complexity, we evaluated three steps in the cause-effect chain from water consumption to human health effects: (1) Quantifying the lack of freshwater for human needs, (2) Assessing vulnerability, and (3) Estimating quantitative health damages related to water deficiency. The damage ($\Delta HH_{malnutrition,i}$), induced by water consumption in a watershed or country i ($WU_{consumptive,i}$, (m^3)), is measured in disability adjusted life years (DALY), as in the Eco-indicator-99 method for assessment of human health effects:

$$\Delta HH_{malnutrition,i} = \underbrace{\underbrace{WSI_i \cdot WU_{\%,agriculture,i}}_{WDF_i} \times \underbrace{HDF_{malnutrition,i} \cdot WR_{malnutrition}^{-1}}_{EF_i} \cdot DF_{malnutrition}}_{CF_{malnutrition,i}} \cdot WU_{consumptive,i} \quad (6)$$

where $CF_{malnutrition,i}$ ($DALY/m^3_{consumed}$) is the expected specific damage per unit of water consumed (as specified in the LCI-phase). The water deprivation factor WDF_i ($m^3_{deprived}/m^3_{consumed}$) uses the physical water stress index WSI_i (eq 5) and the fraction of agricultural water use $WU_{\%,agriculture,i}$ which was calculated for each watershed i based on 0.5° grid-data (*8*). The effect factor EF_i quantifies the annual number of malnourished people per water quantity deprived ($capita \cdot yr/m^3_{deprived}$). It incorporates the per-capita water requirements $WR_{malnutrition}$ to prevent malnutrition ($m^3/(yr \cdot capita)$) and the human development factor $HDF_{malnutrition,i}$ (eq 7) which relates the human development index (HDI) to malnutrition vulnerability. The damage factor $DF_{malnutrition}$ denotes the damage caused by malnutrition ($DALY/(yr \cdot capita)$). $WR_{malnutrition}$ and $DF_{malnutrition}$ are independent of location.

We applied the national HDI reports for all countries (*23*) except for India, Brazil, China, and Russia, for which HDI-values of the main regions within these countries are applied (see the Supporting Information). $HDF_{malnutrition}$ is derived from a polynomial fit of DALY values for malnutrition per 100 000 people in 2002 ($DALY_{malnutrition,rate}$ (*24*)) with corresponding HDI data (Figure 1b):

$$HDF_{malnutrition} = \begin{cases} 1 & \text{for} \quad HDI < 0.30 \\ 2.03\ HDI^2 - 4.09\ HDI + 2.04 & \text{for} \quad 0.30 \le HDI \le 0.88 \\ 0 & \text{for} \quad HDI > 0.88 \end{cases} \quad (7)$$

We set $WR_{malnutrition}$ equal to 1,350 $m^3/(yr \cdot capita)$, the minimum direct human dietary requirement, including blue and green water (*10*). This value matches modeled water resource thresholds for food security (*25*). $DF_{malnutrition}$ is derived on a country level from linear regression of the malnutrition rate ($MN_{\%}$ (*26*)) and $DALY_{malnutrition,rate}$ (Figure 1c) resulting in a per-capita malnutrition damage factor of 1.84×10^{-2} $DALY/(yr \cdot capita)$.

Damage to Ecosystem Quality. In places where plant growth is water-limited, withdrawals of blue water ($WU_{consumptive}$) may eventually reduce the availability of green water and thus diminish vegetation and plant diversity. Riparian and groundwater-dependent vegetation is often crucial for ecosystems (*27, 28*) including birds and insects which are again important for the whole ecosystem, e.g., for pollination and dissemination of plants (*29*). Especially in semiarid and arid regions, terrestrial ecosystems are, to a large extent, runoff dependent (*30, 31*), and biodiversity in these areas contributes significantly to the overall ecosystem quality within a watershed.

The effects of freshwater consumption on terrestrial ecosystem quality (ΔEQ ($m^2 \cdot yr$)) are assessed following the Eco-indicator-99-method (*18*), with units of potentially disappeared fraction of species (PDF). PDF values are generally assessed as vulnerability of vascular plant species biodiversity (VPBD) (*18*). Global spatially explicit data for assessing water-shortage related vegetation damage are only available for net primary production (NPP), which we considered to be a proxy for ecosystem quality. We tested the global relation between VPBD and NPP and found a significant correlation (see Supporting Information Figure S9). Nemani et al. (*32*) describe the "potential climatic constraints to plant growth" based on long-term climate statistics applied to climate models: they developed indices from 0 to 1 quantifying growth constraints due to limited temperature, radiation, and water availability. We adopted these results (see the Supporting Information) and included limitations from soil conditions setting the index to 0 for barren lands (*33*) as there is no natural vegetation that can be affected. The resulting fraction of NPP which is limited by water availability ($NPP_{wat-lim}$) represents the water-shortage vulnerability of an ecosystem, and is used as a proxy for PDF, due to the high correlation with VPBD (Supporting Information Figure S9).

ΔEQ is described similarly to the assessment of land-use impacts within EI99: where CF_{EQ} is the ecosystem damage

$$\Delta EQ = CF_{EQ} \cdot WU_{consumptive} = \underbrace{NPP_{wat\text{-}lim}}_{PDF} \cdot \underbrace{\frac{WU_{consumptive}}{P}}_{A \cdot t} \quad (8)$$

factor ($m^2 \cdot yr/m^3$), and P the mean annual precipitation (m/yr). The ratio of $WU_{consumptive}$ and P denotes the theoretical area-time equivalent which would be needed to recover the amount of consumed water by natural precipitation. We calculated CF_{EQ} for each grid cell i on the global 0.5°-grid ($CF_{EQ,i}$), before aggregating these damage factors by each watershed and country j, respectively, using grid-cell specific precipitation (P_i) as a weighting factor:

$$CF_{EQ,j} = \frac{\sum_{i=1}^{n} NPP_{wat-lim,i}}{\sum_{i=1}^{n} P_i} \quad (9)$$

Damage to Resources. In many locations, precipitation has an annual cycle, so this is the minimum time-step for evaluating whether water resource depletion has occurred. Water stock exhaustion can be caused by the extraction of fossil groundwater or the overuse of other water bodies, as in the case of the Aral Sea. The backup-technology concept (*34*) as used to assess abiotic resource depletion in EI99 (*18*), expressed in "surplus energy" (MJ) to make the resource available in the future, is employed here for assessing the damage to freshwater resources (ΔR, eq 10). Desalination of seawater may be applied as a backup technology to compensate for water resource depletion (*34*). Note that the backup-technology concept does not imply that the amount of all water depleted will be desalinated. It merely serves as a theoretical indicator to make water use comparable to other types of resource use (for a detailed discussion, see ref *34*).

$$\Delta R = E_{desalination} \cdot F_{depletion} \cdot WU_{consumptive} \quad (10)$$

where $E_{desalination}$ is the energy required for seawater desalination (MJ/m^3) and $F_{depletion}$ is the fraction of freshwater consumption that contributes to depletion (−). $F_{depletion}$ serves also as characterization factor for the midpoint indicator "freshwater depletion".

We set $E_{desalination}$ to 11 MJ/m^3 based on state-of-the-art energy demand of seawater desalination technologies for potable water (*35*). This matches the lower energy demand of operating desalination plants at 11−72 MJ/m^3 (*36*). $F_{depletion}$ per water consumed in each watershed i ($F_{depletion,i}$) is derived from the WTA ratio as follows:

$$F_{depletion,i} = \begin{cases} \frac{WTA - 1}{WTA} & \text{for} \quad WTA > 1 \\ 0 & \text{for} \quad WTA \le 1 \end{cases} \quad (11)$$

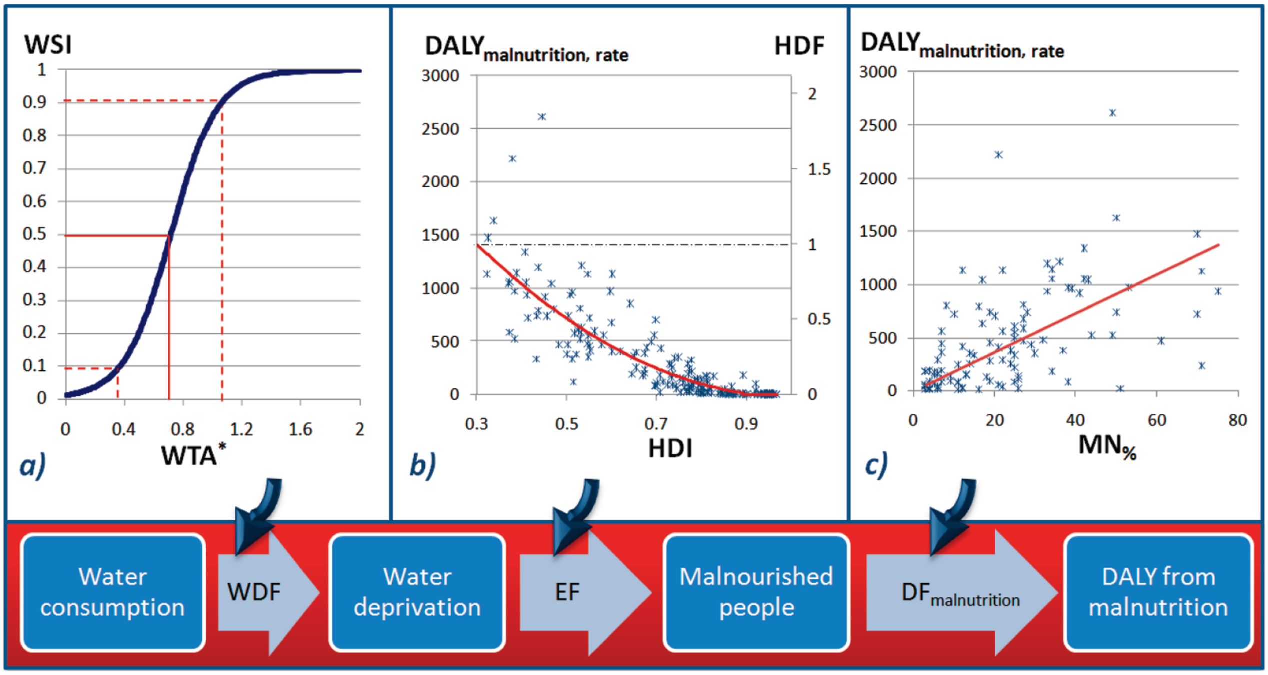

FIGURE 1. Inputs to the impact pathway: (a) relation between WSI and WTA* (blue line, logistic function), (b) $DALY_{malnutrition,rate}$ for each country (blue stars) and HDF modeled (red line, $R^2 = 0.71$) based on HDI, (c) $DALY_{malnutrition,rate}$ for each country (blue stars) against corresponding $MN_{\%}$ and linear regression (red line, $R^2 = 0.26$).

We calculated $F_{depletion}$ for countries ($F_{depletion,country}$) by aggregating the values for $F_{depletion,i}$ of all watersheds in the country, using total annual withdrawal within the watershed i ($WU_{total,i}$ (m^3)) as a weighting factor (eq 12). $WU_{total,i}$ of cross-boundary watersheds located in several countries are assigned to these countries according to the area share of watershed i within the specific country ($a_{i,country}$):

$$F_{depletion,country} = \frac{1}{\sum_{i=1}^{n} WU_{total,i} \cdot a_{i,country}} \sum_{i=1}^{n} F_{depletion,i} \cdot WU_{total,i} \cdot a_{i,country} \qquad (12)$$

Integration into LCIA Methods. To integrate the above methodology into the existing EI99 method, default normalization and weighting factors ("hierarchist perspective" (*18*)) were used to calculate single-scores, denoted hereafter as EI99HA.

Cotton Textiles Case Study. We illustrate our method with an example of cotton textile production. Cotton cultivation occupies 2.4% of total arable land (*37*) and is a globally traded commodity. We used country-specific virtual water content data for cotton cultivation, yarn and textile production (*38*) for water-related LCIs. For fiber processing, 20% of water used was assumed to be consumptive. The impact assessment was performed on a country level. As purchasers are often unable to determine the national origin of cotton materials, the average worldwide impacts are also calculated based on production shares. Water related impacts were compared to other impacts from cotton production, assessing existing inventory data (*3*) with the EI99-method (*18*).

To test the relevance of a regionalized assessment, we calculated water demand and related environmental damages on watershed level for the U.S., applying county-specific production data (*39*) and regional irrigation requirements (IR). We calculated IR using the CROPWAT-model (*12*) ($US_{CROPWAT}$) and, in addition, an approach based on reported U.S. water-use data ($US_{estimate}$), as depicted in the Supporting Information.

Results

LCIA Characterization and Damage Factors. Global characterization and damage factors for watershed-level consumptive water use are shown in Figure 2. For complete lists of all watersheds and countries, see the Supporting Information.

The correlation on watershed level between WSI and aggregated Eco-indicator-99-scores is $R = 0.51$. The correlation is poorer in less developed countries and regions with low population-density. On global average, the damage categories human health, ecosystem quality, and resources contribute to aggregated, water-use related Eco-indicator-99-scores by 3, 69, and 28%, respectively.

Cotton Textiles Case Study. The irrigation requirements, yields, and the environmental impacts from water consumption of cotton production vary greatly from nation to nation (Table 1). For cotton textiles at the factory gate, the fraction of total damage attributable to water consumption ranges from less than one percent (Brazil) to 77% (Egypt) using the augmented Eco-indicator-99 assessment method. On global average, this fraction is 17%. The overall environmental impact of cotton textile production depends largely on the country of cultivation (Figure 2f).

For the United States the modeled national average based on the two approaches $US_{CROPWAT}$ and $US_{estimate}$ lead to considerably higher water consumption than the value reported on country level (*38*), as shown in Table 1 and described in the Supporting Information.

Discussion

Inventorying Water Use. Current inventory databases only contain limited information about water use and therefore need to be complemented with further information on amounts and sources of water used. The application of national virtual water content data (*11*) is convenient to estimate the amount of water used for agricultural products,

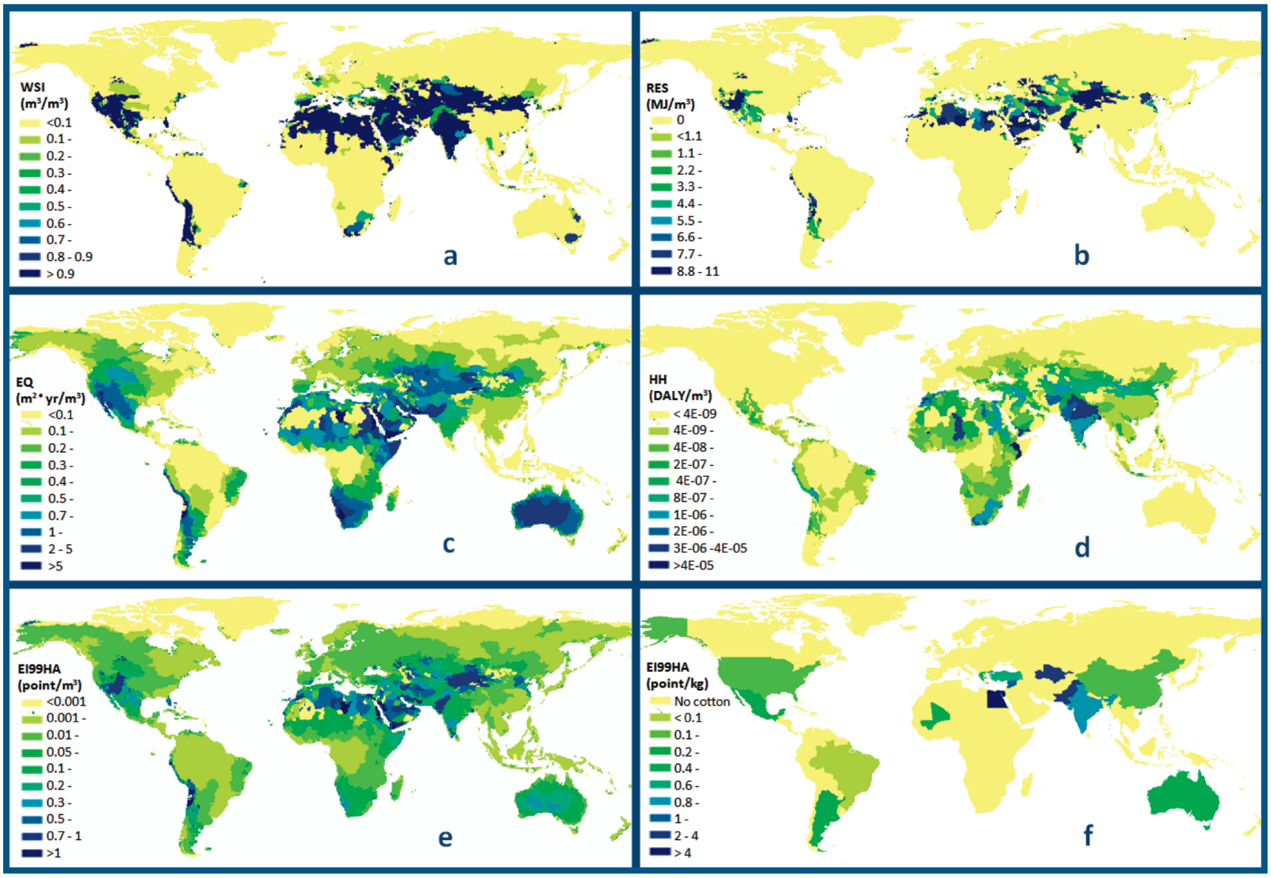

FIGURE 2. Characterization and damage factors on the watershed level per m^3 water consumed: (a) Water stress index, (b) damage on resources, (c) damage on ecosystem quality, (d) damage on human health, and (e) aggregated Eco-indicator-99 damage factor. Map (f) shows the aggregated Eco-indicator-99 damage per kg cotton textile on the country level.

TABLE 1. Inventory Data and Environmental Impacts Per kg Cotton Textile

	global production share	consumptive use (blue water reported in ref *38*) (m^3/kg)	Water deprivation (m^3/kg)	ecosystem quality (PDF·m^2·yr/kg)	human health (10^{-6} DALY/kg)	resources (MJ/kg)	fraction of total damage (Eco-indicator-99) caused by water consumption[a]
Argentina	0.7%	6.11	2.01	2.71	0.206	5.45	12%
Australia	1.4%	3.92	1.42	5.10	0	1.07	14%
Brazil	5.6%	0.61	0.01	0.0188	0.004	0.00946	0%
China	27.2%	2.35	0.93	0.449	0.61	3.97	5%
Egypt	0.8%	10.79	10.15	87.1	18.36	53.9	77%
Greece	1.8%	4.89	3.20	0.806	0.126	7.41	9%
India	19.9%	5.73	5.16	2.12	11.93	15.0	24%
Mali	0.6%	4.07	0.99	3.29	5.681	0	14%
Mexico	0.6%	4.52	3.12	2.62	0.695	7.07	13%
Pakistan	8.5%	9.88	9.17	15.7	20.68	41.6	52%
Syria	0.9%	8.41	8.00	8.23	7.752	39.1	41%
Turkey	3.3%	7.34	5.40	3.65	3.741	13.6	21%
Turkmenistan	1.1%	14.12	13.66	13.6	12.27	65.3	53%
United States	16.4%	1.90	0.75	0.465	0.003	2.80	4%
Uzbekistan	4.4%	11.14	10.58	10.8	11.71	39.6	45%
average	**93.4%**	**8.54**	**3.48**	**3.88**	**5.71**	**12.8**	**17%**
$US_{CROPWAT}$[b]	*16.4%*	*8.91*	*3.72*	*4.91*	*0.0274*	*16.7*	*23%*
$US_{estimate}$[b]	*16.4%*	*3.27*	*2.48*	*3.61*	*0.0237*	*13.6*	*19%*

[a] Total damage includes state-of-the-art LCA results for final cotton textile at plant: 2.58 points/kg, without the damages of water consumption (*3, 18*). [b] For the United States, we derived national average impacts of water consumption from data on watershed-level, applying two approaches ($US_{CROPWAT}$ and $US_{estimate}$).

but it has deficiencies in spatial resolution and reliability. In addition, the virtual water content considers only a few industrial products, and water use in supply chains is neglected. Particularly for large countries, further characterization of the specific watersheds may be necessary to appropriately assess water-use related impacts, as illustrated by the U.S. cotton case study in this paper. The two different approaches applied to derive the irrigation water requirements for U.S. cotton show the difficulty of assessing specific water consumption. Theoretical water consumption based on CROPWAT lead to overestimation for cultivations that are subject to deficit irrigation. Reported data, on the other hand, usually represents a lower bound, because irrigation water is often unmetered or comes from possibly illegal private pumping. These aspects are crucial for analyzing water consumption data properly in LCI. Furthermore, consumptive losses from distribution and storage systems need to be considered. Irrigation efficiency, on the other hand, is less relevant for water consumption, as excess irrigation water, possibly contaminated, is recharging groundwater, although there may be additional evaporation.

We have focused on blue water use, and assumed that green water consumption does not change as a function of the activities assessed in LCA. However, this is a simplification, and the related effects of potential changes in green water flows should be addressed in future research.

For most industrial processes, water-use data is scarce and the available data is heterogeneous. To obtain this information in a consistent format will be a major challenge in further studies. In particular, water quality degradation needs to be reported and assessed, as this is a particular concern for industrial production.

Water-Consumption Impact-Assessment Method. The water stress index (WSI) may serve as a simple screening indicator for the assessment of water use, accounting for water availability and withdrawals. The consideration of temporal variability of water availability in WSI allows for assessing increased impacts in specific periods. While infrastructure may enable sufficient water storage and hence mitigate water stress for human needs, ecosystems will still be affected and additional water will be evaporated from the surfaces of water-storage systems. Therefore, we diminished the variation factor, which accounts for temporal variability in water availability, only for strongly regulated flows (eq 2), for which large dams allow efficient storage. The environmental effects of such major dams, however, should be additionally assessed in LCA as in-stream storage (see Supporting Information Figure S1).

Human-health damages are highly dependent on socio-economic factors and local conditions. Generally, where water scarcity is affecting health, data availability is extremely poor, and the uncertainties are therefore high. The empirical approach applied here copes with incomplete data by assessing the cause−effect chain stepwise, which is similar to the emission-to-impact modeling for toxic emissions (*18*). While we abstained from characterizing impacts due to lack of water for hygiene, it might become relevant for industrial water use in certain countries, as the purchase power of industries may dominate competition with households.

Terrestrial ecosystem damages are assessed in accordance with existing methods (*18*), using the fraction of water-limited net-primary production as a proxy for the number of vascular-plant species. However, only part of vascular plants are ground- or surface water dependent, and aquatic ecosystems, birds, vertebrates, and unique wetlands are also affected by water-use. We partially addressed this limitation by implicitly assuming that all precipitation water contributes to ecosystem quality (eq 9), being either evapo-transpired by vegetation on-site (green water) or leading to runoff (blue water). The blue water portion might only partially be consumed by vegetation, and aquatic ecosystems (including groundwater habitats and coastal zones) may benefit instead. However, in some cases all blue water is evaporated and any consumption leads to diminished vegetation and ecosystem quality, such as in the case of the Okavango Delta (*40*).

Impacts on aquatic ecosystems depend only to some extent on water quantity. Often, infrastructure (channeling rivers, dams) and water-quality deterioration seem to be more important to ecosystem-quality loss than reduced water quantity (*41*). These aspects are an open research question which should be tackled in future studies.

Damage to resources is assessed with surplus energy potentially required by future users (*18*). The concept of using theoretical backup technologies to assess the impacts of resource use (*34*) is controversial, but does allow for a combined assessment of stock and flow resources. It should be interpreted as a conceptual approach of quantifying environmental impacts for future generations in terms of additional energy consumption eventually needed to compensate for water scarcity. As an approximation, the surplus-energy cost ($E_{desalination}$) is considered constant for all regions, although seawater desalination is not possible everywhere. Long-distance transportation of water can have significantly higher impacts than desalination alone (see the Supporting Information), but water-intensive production of future generations could be moved closer to sea or available water resources.

The method proposed in this paper could also be adjusted to other impact assessment methods than Eco-indicator 99, e.g., resource depletion could also be assessed in terms of monetary units, as done in LIME (*19*) and ReCiPe 2008 (*42*).

In addition to water consumption, degradative use can make resources unavailable for relevant users and consequently have an impact on human health or even resource depletion, as the use of fossil aquifers might be favored over water purification. Furthermore, local depletion of water stocks may also occur if blue water regionally available in the watershed is not completely used. On the other hand, watershed-WTA above 100% (eq 11) might not lead to depletion if degraded water is reused downstream. Such aspects are not considered in the method proposed here.

Water-Consumption Damage Scores. On global watershed average, the impact of water consumption on ecosystem quality is larger than the impact to human health and resources, using default Eco-indicator 99 normalization and weighting factors (*18*). One reason is that ecosystem-quality impacts occur in many parts of the world, whereas human-health and resources effects are restricted to specific regions. Freshwater-resource depletion is also important, contributing on average 28% of the aggregated freshwater-use related damage score, using the Eco-indicator-99 weighting scheme, as shown by decreasing groundwater levels around the world (*1*).

Cotton Textile Case Study. Several studies reported water use as one of the major environmental issues in cotton production (e.g., ref *37*), which is confirmed by the regionalized results of our study. Both needed water quantities and impact factors vary strongly as a function of location, leading to high differences in the total damages from water consumption for cotton production in different countries (Table 1). However, environmental damages from water consumption through irrigation might be outweighed by fertilizer applications and land-use impacts if virgin land is occupied for cotton cultivation (e.g., deforestation in Brazil). Thus, the importance of water use will depend on the local conditions and the alternatives for cotton cultivation or textile production that are available.

Regionalization on a country level proved not to be specific enough for large countries, as illustrated for the U.S. Therefore, large countries with varying climatic zones, such as the U.S., India, and China should be assessed on a watershed level. The case-study results further highlight the importance of optimizing water consumption, e.g. with technological measures in water-scarce regions, and demand caution against shifting production to high-yielding arid areas.

Application in LCA. The method presented in this paper describes the impact of freshwater consumption in the life cycle of products or processes. However, similar to the assessment of other impact categories in LCA, the uncertainties are large. Therefore, if an LCA study points to a potential environmental problem from water consumption, a more detailed assessment could be done. This assessment could take into account the local conditions and identify appropriate mitigation measures. Despite uncertainties, the proposed method represents a rigorous and comprehensive accounting of the environmental damages of freshwater use, an area of impact which has been missing in LCIA so far. For example, more and more products are assessed by greenhouse-gas emissions and land use, but there are many products like food and biofuels where water consumption presents an additional important ecological dimension that needs to be considered to provide a more complete basis for environmental decision making and to avoid burden shifting.

Increasing interest in regionalized LCA should improve inventory-data quality in future and reduce uncertainties in regionalized LCIA. In addition to data, enhanced modeling tools facilitating regionalized LCIA are required for researchers and practitioners. Such tools are currently under development. Integration of our method into existing assessment schemes will improve LCA results, especially for agricultural products. In addition to the assessment of water consumption, methodologies depicting the loss of freshwater resource due to degradative freshwater use need to be developed, as this issue is particularly important for industrial production.

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Supporting Information Available

An acronym list, further method descriptions, characterization and damage factors, and raster data. This material is available free of charge via the Internet at http://pubs.acs.org.

Literature Cited

(1) UNEP. *Global Environment Outlook GEO 4: Environment for Development*; UNEP: Malta, 2007.
(2) Koehler. Water Use in LCA: Managing the planet's freshwater resources. *Int. J. Life Cycle Assess.* **2008**, *13* (6), 477–486.
(3) Ecoinvent Centre. Ecoinvent data v2.01. Available at: http://www.ecoinvent.org.
(4) Owens, J. W. Water resources in life-cycle impact assessment: Considerations in choosing category indicators. *J. Ind. Ecol.* **2001**, *5* (2), 37–54.
(5) Frischknecht, R.; Steiner, R.; Jungbluth, N. *Ökobilanzen: Methode der ökologischen Knappheit−Ökofaktoren 2006*; Öbu−Netzwerk für nachhaltiges Wirtschaften: Zurich, 2008.
(6) Bare, J.; Gloria, T.; Norris, G. Development of the method and U.S. normalization database for life cycle impact assessment and sustainability metrics. *Environ. Sci. Technol.* **2006**, *40* (16), 5108–5115.
(7) Shiklomanov, I. A. *World Water Resources at the Beginning of the 21st Century*, International Hydrology Series; Shiklomanov, I. A., Rodda, J. C. Eds.; Cambridge University Press: Cambridge, 2003.
(8) Vorosmarty, C. J.; Green, P.; Salisbury, J.; Lammers, R. B. Global water resources: Vulnerability from climate change and population growth. *Science* **2000**, *289* (5477), 284–288.
(9) Alcamo, J.; Doll, P.; Henrichs, T.; Kaspar, F.; Lehner, B.; Rosch, T.; Siebert, S. Development and testing of the WaterGAP 2 global model of water use and availability. *Hydrol. Sci. J.* **2003**, *48* (3), 317–337.
(10) Falkenmark, M.; Rockstrom, J. *Balancing Water for Humans and Nature: The New Approach in Ecohydrology.* Earthscan: London, 2004.
(11) Chapagain, A. K.; Hoekstra, A. Y. *Water Footprints of Nations: Volume 1: Main Report*, Research Report Series No. 16.; UNESCO-IHE: Delft, 2004.
(12) FAO. *CROPWAT for WINDOWS*; FAO: Rome, Italy. 1999.
(13) Alcamo, J.; Henrichs, T.; Rösch, T. World Water in 2025: Global Modeling and Scenario Analysis. In *World Water Scenarios*; Rijsberman, F., Ed.; Earthscan Publications: London. 2000; 243−281.
(14) Nilsson, C.; Reidy, C. A.; Dynesius, M.; Revenga, C. Fragmentation and flow regulation of the world's large river systems. *Science* **2005**, *308* (5720), 405–408.
(15) Mitchell, T. D.; Jones, P. D. An improved method of constructing a database of monthly climate observations and associated high-resolution grids. *Int. J. Climatology.* **2005**, *25* (6), 693–712.
(16) McMahon, T. A.; Vogel, R. M.; Peel, M. C.; Pegram, G. G. S. Global streamflows - Part 1: Characteristics of annual streamflows. *J. Hydrol.* **2007**, *347*, 243–259.
(17) Guinée, J. B. *Life Cycle Assessment: An Operational Guide to the ISO Standards*; Centre for Environmental Science, Leiden University: Leiden, 2001.
(18) Goedkoop, M.; Spriensma, R. *The Eco-Indicator 99: a Damage Oriented Method for Life Cycle Impact Assessment: Methodology Report*, Publikatiereeks produktenbeleid; nr. 36A; Ministerie van Volkshiusvesting, Ruimtelijke Ordening en Milieubeheer: Den Haag, 2001.
(19) Itsubo, N.; Sakagami, M.; Washida, T.; Kokubu, K.; Inaba, A. Weighting across safeguard subjects for LCIA through the application of conjoint analysis. *Int. J. Life Cycle Assess.* **2004**, *9* (3), 196–205.
(20) Jolliet, O.; Margni, M.; Charles, R.; Humbert, S.; Payet, J.; Rebitzer, G.; Rosenbaum, R. IMPACT 2002+: A new life cycle impact assessment methodology. *Int. J. Life Cycle Assess.* **2003**, *8* (6), 324–330.
(21) UNESCO. Water for people - water for life. *The United Nations World Water Development Report*. UNESCO, and Berghahn Books: Paris, 2003.
(22) IWMI. *Water for Food, Water for Life: A Comprehensive Assessment of Water Management in Agriculture*; Molden, D. Ed.; Earthscan and IWMI: London 2007.
(23) UNDP. Human Development statistical tools. Available at: http://hdr.undp.org/en/statistics/.
(24) WHO. Death and DALY estimates for 2002 by cause for WHO Member States. Available at: http://www.who.int/healthinfo/bodestimates/en/index.html.
(25) Yang, H.; Reichert, P.; Abbaspour, K. C.; Zehnder, A. J. B. A water resources threshold and its implications for food security. *Environ. Sci. Technol.* **2003**, *37* (14), 3048–3054.
(26) FAO. *The State of Food Insecurity in the World (SOFI) 2003*; FAO: Rome, 2003.
(27) Nilsson, C.; Svedmark, M. Basic principles and ecological consequences of changing water regimes: Riparian plant communities. *Environ. Manage.* **2002**, *30* (4), 468–480.
(28) Olson, D. M.; Dinerstein, E. The global 200: A representation approach to conserving the Earth's most biologically valuable ecoregions. *Conserv. Biol.* **1998**, *12* (3), 502–515.
(29) Falkenmark, M. The greatest water problem: The inability to link environmental security, water security and food security. *Int. J. Water Resour. Dev.* **2001**, *17* (4), 539–554.
(30) Harvey, F. E. Ground water dependence of endangered ecosystems: Nebraska's eastern saline wetlands. *Ground Water.* **2007**, *45*, 736–752.
(31) Fetter, C. W. *Applied Hydrogeology*; Macmillan: New York, 1994.
(32) Nemani, R. R.; Keeling, C. D.; Hashimoto, H.; Jolly, W. M.; Piper, S. C.; Tucker, C. J.; Myneni, R. B.; Running, S. W. Climate-driven increases in global terrestrial net primary production from 1982 to 1999. *Science* **2003**, *300* (5625), 1560–1563.
(33) Defries, R. S.; Townshend, J. R. G. Ndvi-derived land-cover classifications at a global-scale. *Int. J. Remote Sensing* **1994**, *15* (17), 3567–3586.
(34) Stewart, M.; Weidema, B. A consistent framework for assessing the impacts from resource use - A focus on resource functionality. *Int. J. Life Cycle Assess.* **2005**, *10* (4), 240–247.
(35) Fritzmann, C.; Lowenberg, J.; Wintgens, T.; Melin, T. State-of-the-art of reverse osmosis desalination. *Desalination* **2007**, *216* (1−3), 1–76.
(36) von Medeazza, M. G. L. "Direct" and socially-induced environmental impacts of desalination. *Desalination* **2005**, *185* (1−3), 57–70.
(37) Kooistra, K.; Termorshuizen, A.; Pyburn, R. *the Sustainability of Cotton*; Wageningen University: Wageningen, The Netherlands, 2006.
(38) Chapagain, A. K.; Hoekstra, A. Y.; Savenije, H. H. G.; Gautam, R. The water footprint of cotton consumption: An assessment of the impact of worldwide consumption of cotton products on the water resources in the cotton producing countries. *Ecol. Econ.* **2006**, *60* (1), 186–203.
(39) National Cotton Council of America. Cotton Crop Databases. Available at: http://www.cotton.org/econ/cropinfo/cropdata/.
(40) Milzow, C. L. K.; Bauer-Gottwein, P.; Meier, P.; Kinzelbach, W. Regional review: The hydrology of the Okavango Delta, Botswana—processes, data and modelling, *Hydrogeol. J.* **2009**, In Press.
(41) Millennium Ecosystem Assessment. *Ecosystems and Human Well-Being*; Island Press: Washington, DC, 2005.
(42) Goedkoop, M.; Heijungs, R.; Huijbregts, M.; De Schryver, A.; Struijs, J.; van Zelm, R. ReCiPe 2008 - A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. Available at: http://lcia.wik.is.

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