LCA Methodology

Assessing Freshwater Use Impacts in LCA Part I: Inventory Modelling and Characterisation Factors for the Main Impact Pathways

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Preamble: In this series of two papers the methodological aspects related to the assessment of freshwater resources use in LCA are discussed (Part I) and the operational method and characterisation factors suggested are illustrated for a case study of broccoli produced in the UK and Spain (Part II).

Abstract

Background, Aim and Scope. Fresh water is a basic resource for humans; however, its link to human health is seldom related to lack of physical access to sufficient fresh water, but rather to poor distribution and access to safe water supplies. On the other hand, fresh water availability for aquatic ecosystems is often reduced due to competition with human uses, potentially leading to impacts on ecosystem quality. This paper summarises how this specific resource use can be dealt with in LCA.

Main Features. The main quantifiable impact pathways linking freshwater use to the available supply are identified, leading to definition of the flows requiring quantification in the LCI.

Results. The LCI needs to distinguish between and quantify evaporative and non-evaporative uses of 'blue' and 'green' water, along with land use changes leading to changes in the availability of fresh water. Suitable indicators are suggested for the two main impact pathways (namely freshwater ecosystem impact, FEI, and freshwater depletion, FD) and operational characterisation factors are provided for a range of countries and situations. For FEI, indicators relating current freshwater use to the available freshwater resources (with and without specific consideration of water ecosystem requirements) are suggested. For FD, the parameters required for evaluation of the commonly used Abiotic Depletion Potentials (ADP) are explored.

Discussion. An important value judgement when dealing with water use impacts is the omission or consideration of non-evaporative uses of water as impacting ecosystems. We suggest considering only evaporative uses as a default procedure, although more precautionary approaches (e.g. an 'Egalitarian' approach) may also include non-evaporative uses. Variation in seasonal river flows is not captured in the approach suggested for FEI, even though abstractions during droughts may have dramatic consequences for ecosystems; this has been considered beyond the scope of LCA.

Conclusions. The approach suggested here improves the representation of impacts associated with freshwater use in LCA. The information required by the approach is generally available to LCA practitioners

Recommendations and Perspectives. The widespread use of the approach suggested here will require some development (and consensus) by LCI database developers. Linking the suggested midpoint indicators for FEI to a damage approach will require further analysis of the relationship between FEI indicators and ecosystem health.

Keywords: virtual water; water footprint; water resource; freshwater ecosystem impact; LCA; LCI; LCIA; evaporative use; ecosystem; FEI; FD

Introduction

Water is a precious and increasingly scarce resource. It is critical for ecosystem functions (as both habitat and resource) and equally essential for humans. Water abstracted for human purposes can have significant impacts on water systems. Over 100,000 species (almost 6% of all described species) live in fresh water and countless others depend on fresh water for survival (Dudgeon *et al*. 2005). Freshwater species and habitats are more imperilled globally than their terrestrial or marine counterparts (WWF 2006). In the most extreme cases, water scarcity has resulted in complete ecosystem collapse (Micklin 1988). Similarly, some major rivers have periodically completely dried up, including the Rio Grande/Bravo in Mexico and the Great Ruaha River in Tanzania (WWF 2007).

A lack of adequate access to safe water supplies is dire from a human health point of view. Globally nearly two million people die from diarrhoeal disease every year, with 88 percent of cases attributed to unsafe water and inadequate sanitation or hygiene (WHO, 2004). However, the water needed to feed humanity requires significantly higher amounts than daily drinking and cleaning water. In general it takes about 3400 litres per person per day to support a global average consumption pattern. There is a wide variation in this amount: from more than 6000 litres per person per day in many western countries (e.g. USA, Canada, Greece, Italy, Spain, etc.) to much smaller amounts in developing countries e.g. India (2700) and China (1900) litres per person per day) (Chapagain and Hoekstra 2004). This is due mainly to differences in diets and, while eradicating malnutrition by 2025 will require a doubling of water used for agriculture (Rockstrom *et al*. 2006), rising affluence in many emerging countries will raise their averages toward the western norm.

Measurement of water use therefore provides important information for attributing responsibility, assessing impact and developing solutions to water use by companies, communities and individuals. Different environmental system analysis tools such as Life Cycle Analysis (LCA) and Virtual Water (VW) are able to measure amounts of water used in the production of various products. However, both methods currently lack a proper assessment of the relative scarcity and opportunity cost of water at the point of production.

Water use in LCA

Water use impacts have been underrepresented since the start of LCA methodology in the late 1960s, probably due to LCA being developed for industrial systems (usually less dependent on water resources than agricultural ones) in water-abundant countries. Basically, LCA studies report the total amount of water used by the production system, from cradle (raw material acquisition) to grave (waste management). In general, such studies do not even distinguish the source from which water is obtained nor the way or condition in which water leaves the product system.

Early examples explicitly addressing water use in LCA included water-intensive products such as nappies (Johnsson 1994; Sauer *et al*. 1994). Other references quantified the total amount of water used in industrial (e.g. Milà i Canals *et al*. 2002; Muñoz *et al*. 2004; 2006) and agricultural systems (e.g. Antón *et al*. 2005; Milà i Canals *et al*. 2006; Coltro *et al*. 2006; Hospido et al. forthcoming; Muñoz *et al*. accepted). Lundie *et al*. (2004) assess water supply as a function, and quantify the water flow associated to different scenarios for water provision in Sydney (Australia), but do not address the impacts related to water use. Recently, some authors have suggested ways to progress beyond merely quantifying the volume of water used per functional unit. Owens (2002) clarifies the definitions of different water inputs to (sources) and outputs from (dispositions) the system, key to a proper inventory of freshwater use, and suggests the qualitative aspects that should be factored into water assessment in LCA. Brent (2004) stresses the need for spatial differentiation, but suggests simply adding a kg of water used with no further characterisation along the life cycle stages. Other authors have explored ways to include water use in agricultural systems through evapo-transpiration in the inventory (e.g. Antón *et al*. 2005), and/or to incorporate water use in the impact assessment (LCIA) phase (e.g. Heuvelmans *et al*. 2005). Bauer and Zapp (2004) offer background information to relate potential effects of mining on water resources, but do not provide a means to use this information in the characterisation stage. In summary, the incorporation of such improvements in LCA practice has been extremely limited to date. An obvious reason is that the main LCIA methods do not provide characterisation factors for water as an abiotic resource (EDIP 1997: Hauschild and Wenzel 1998; EI99: Goedkoop and Spriensma 1999; CML 2001: Guinée *et al*. 2002). Guinée *et al*. (2002, part 3, p. 184) conclude that there is no satisfactory method to address the habitat aspect of freshwater (related to what they call "desiccation") in LCA. Even though impacts resulting from freshwater use are certainly an environmental issue in many production systems, and ISO 14044 requires that LCIA reflects 'a comprehensive list of environmental issues related to the product system studied', no satisfactory method yet exists to include them in LCA.

The concept of Virtual Water (VW)

The concept of Virtual Water (VW) has evolved since the early 1990s and refers to the amount of water required to produce a certain product. VW was introduced by Allan (1998; 2001), who investigated VW imports through the trade of water-intensive crops as a partial solution to problems of local water scarcity in the Middle East. Allan suggests that such trade relieved the need for importing countries to use their own, often scarce, water resources to produce the same product. Water is termed as 'virtual' because the amount of water physically contained in the final product is negligible compared to the amount that went into its production. VW studies have taken on more precise and practical applications since Hoekstra and Hung (2002); Chapagain and Hoekstra (2003; 2004); Chapagain and Orr (in press), began to quantify and calculate VW flows and related water footprints.

Originally, Hoekstra and Hung (2002) estimated the 'blue' water footprint by excluding the 'green' water use (use of effective rainfall to produce crops) from domestic production (see below for definitions). Subsequently, Chapagain and Hoekstra (2004) included the 'green' water footprint related to the consumption of domestic production, and Chapagain *et al*. (2006b) included the 'grey' component of VW, accounting for the water volumes needed to dilute waste flows to agreed water quality standards.

Clearly, there is a need for a more systematic assessment to characterise the sustainability of freshwater use by production systems in LCA. Chapagain and Orr (in press) argue that such an indicator could also be useful for VW. In this paper we first introduce some terms drawn from LCA and VW literature, in order to avoid confusion by establishing a common terminology (section 1). The main impact pathways resulting from freshwater use are described in section 2; this leads us to the definition of the relevant water flows that need quantification and assessment in LCA. Section 3 offers guidance to quantify such flows, and in section 4 we review some of the existing indicators currently used to characterise the sustainability of water uses. Characterisation factors for the most promising indicators are provided in the appendix in a format ready to use in LCA studies. Finally, discussion, conclusions and recommendations are provided in sections 5, 6 and 7 respectively. A practical application to illustrate the suggested methodology is offered in Milà i Canals *et al*. (forthcoming).

1. Definitions

One possible reason why water has not yet been properly assessed in LCA is the plethora of forms and routes in which water enters and exits production systems. This section suggests some terminology referring to how water enters and leaves the system.

1.1. Water as an input to the production system

From a resource point of view, water may be a flow (e.g. rivers, rain), a fund (groundwater) and even a deposit or stock (fossil water) (Bauer and Zapp 2004). Water may enter the production system from surface bodies such as rivers and lakes (flow); rainfall (flow); groundwater bodies (fund or stock); or the sea (e.g. used for cooling or as input to desalination plants). Owens (2002) further details whether water is used *in-stream* (e.g. in a power dam) or is withdrawn *off-stream*.
Water occurs in the form of *green water* (stored as soil moisture and available for evaporation through crops and terrestrial vegetation) and *blue water* (surface or groundwater). Blue water is the volume of water in ground (aquifer) and surface water bodies available for abstraction. The distinction between blue water and green water is important as green water is only available for use by plants at the precise location where it occurs, whereas blue water is available generally for use in a wide range of human managed systems, including but not limited to use by plants. Another possible way for rainwater to enter a system is through rainwater harvesting. This represents a special case of storing this form of blue water directly in human infrastructures and therefore avoiding abstraction from a natural body although, as for other forms of land use, it diverts water from replenishing a natural body.

1.2. Water as an output from the production system

As crucial as where water comes from is how water is returned to nature. Two main paths must be distinguished here:

- *non-evaporative water use* ('water use' according to Owens 2002: water is returned to the original basin and may be used by other users after leaving the system) and
- *evaporative water use* ('water consumption' according to Owens 2002: water is dissipated and not immediately available after use).

Obviously, non-evaporative water may return to a different system, which requires a proper definition of the temporal and spatial system boundaries: when and where do we consider that water leaves a system and/or becomes available to other users? The main difference between water and many other mineral resources is that even dissipated (evaporated) water will eventually become useful in a relatively short period of time through rainfall (and then become renewable surface or groundwater), although this water cycle generally occurs over vast geographic areas. Some references (e.g. Mohamed 2005) suggest that only 10-20% of evaporated water falls as precipitation on land, the rest being "lost" as rainfall on oceans, and thus not immediately available to other users. This reinforces the importance of the distinction between evaporative and non-evaporative uses.
Water transfers between river basins (or, less frequently, between countries) are a special case of water input/output. Owens (2002) suggests considering water transfers as an evaporative use ('consumption'). However, we see no reason why this should be the case unless the transferred water is actually evaporated. We suggest treating transfers (including embodied water) from a resource availability perspective; i.e. the transfer increases the resource in the receiving basin and decreases it in the providing one. Nevertheless, global databases of water resources do not subtract water exports from a country's available water reserves, because water is actually available before being exported. In effect, we are not considering any particular environmental impact related to the resource aspect of transferred water. Indeed, the transfer itself is associated with environmental impacts due to energy use, infrastructure construction, etc., but local effects e.g. on freshwater ecosystems, are deemed too spatially-specific for LCA.

2. Main impact pathways from effects of freshwater use on availability

The use of freshwater resources may lead to undesired impacts such as reduced availability of water for other users, locally lowering the level of water courses and lakes with effects on aquatic ecosystems, and ultimately impacts on human health due to insufficient water availability and poor water quality. As mentioned previously, in this paper we focus on impacts related to the availability of sufficient freshwater and only suggest research needs for those quality aspects not yet sufficiently covered in LCA. Two main aspects of water need to be addressed here: *water as a resource* for humans; and *water as a habitat* (resource for ecosystems; environmental water requirements as defined by Smakhtin *et al*. 2004). Related to these two aspects, the following four main impact pathways may be distinguished and merit attention in LCA; they are illustrated in **Figure 1**:

1. Direct water use leading to changes in freshwater availability for humans leading to changes in human health;
2. Direct water use leading to changes in freshwater availability for ecosystems leading to effects on ecosystem quality (Freshwater Ecosystem Impact, FEI);
3. Direct groundwater use causing reduced long-term (fund and stock) freshwater availability (Freshwater Depletion, FD);
4. Land use changes leading to changes in the water cycle (infiltration and runoff) leading to changes in freshwater availability for ecosystems leading to effects on ecosystem quality (FEI).

Fig. 1

2.1. Effects of Freshwater Use on Human Health

Despite the critical nature of water as a resource for humans, the relationship between natural water resources availability and human health is not straightforward. When naturally available renewable water resources per capita on a national level are compared with basic indicators of economic development, human health or well-being, no statistically significant correlations result (Chenoweth 2008 a; b). Besides, the dramatic amount of human deaths linked to water is mainly caused by poor water quality and/or sanitation and not by the physical amount available. We suggest that these aspects should be properly included in LCA methodology where this is not currently the case (e.g. water pollution with faecal bacteria).
This supports the assumption that the link between human health and water quantity *per se* is not a significant issue. As such we suggest omitting this aspect from LCA. However, it may be necessary to include it in specific cases, when information about human deaths caused by lack of adequate water resources is available.

2.2. Change of water quantity affecting ecosystem health (Freshwater Ecosystem Impact, FEI)

While critical use of water for human health is generally guaranteed (see above), this is not the case for ecosystems. Indeed, ecosystems may be damaged due to excessive water abstraction for human purposes, through for example, changes in the groundwater table (with effects to wetlands) or changes in the environmental flows of rivers. We suggest calling this impact "freshwater ecosystem impact" (FEI).

In the case of flow and fund water resources, only *evaporative use* leads to reduced (temporal and/or spatial) availability for other users (humans or ecosystems). It is assumed that the amount of *non-evaporative* water *used* which is subsequently returned to the water source does not lead to relevant environmental impacts from a resource perspective, and should be disregarded in the LCIA phase. This is a big assumption, and water extracted from a river to provide drinking water for a city might actually not return to the same river directly, thus reducing its flow (potentially below the defined environmental flow). However, such a localised perspective is beyond the conventional scope of LCA (see below), and we consider that water returns to "the environment" after a non-evaporative use, rather than to a specific ecosystem. This is a default (baseline) suggestion, and the option is left open for practitioners to characterise non-evaporative uses with a characterisation factor >0 for FEI if deemed appropriate (see section 4.2.2.).

Because fossil water (stock water resources) would not occur in an ecosystem if it was not abstracted by humans, its use does not affect ecosystem health. Besides, for ecosystems to feel a positive effect from abstracted fossil water there would have to be a continuous abstraction and return to ecosystems, which is a highly unlikely scenario. We thus recommend not considering any positive effect from fossil water use on FEI.

Obviously, changes in the quality of the returned water may create important impacts and these need to be considered, but perhaps in other impact categories such as eutrophication or toxicity. Some flows are not normally included in LCIA but may be important for some water uses; an example is heat in the case of cooling water. These flows should be further studied in the relevant impact categories, but are not considered in this paper.

From a quantitative point of view, though, it needs to be stressed that only potential impacts due to the amount of dissipated resource may be assessed in LCA. Site-specific local effects, e.g. on flow in a watercourse (see above) or on wetlands, are usually not accountable in LCA; rather, they should be addressed during the assessment of specific projects in the framework of Environmental Impact Assessment.

2.3. Depletion of freshwater resources (Freshwater Depletion, FD)

As stated above, water can be a flow, fund or stock resource. In general, a flow resource such as river water cannot be depleted but there can only be competition over its use, whereas depletion may be an issue for fund and stock resources (Guinée *et al*. 2002, p. 154). Competition amongst human users is outside the scope of LCA, whereas competition for water flows with ecosystems is addressed in the freshwater ecosystem impact potential. Conversely, using groundwater may reduce its availability for future generations, when aquifers are over-abstracted or fossil water is used, and so needs to be included as an impact on natural resources.

A special case related to the availability of freshwater resources is desalination of sea water. Sea water is so abundant that its use will not cause resource concern in the foreseeable future. In fact, desalination of sea water may be considered as a way to increase freshwater resources, and treated as a beneficial effect on the freshwater depletion potential. Desalinisation brings with it a raft of other environmental issues (e.g. carbon emissions, energy use, brine discharge, disturbance to marine ecosystems around water intakes and brine returns) but these are not related to water as a resource. In general, a positive feedback with FEI is not expected because desalinated water is usually used close to the sea, and any non-evaporative use returned to the environment will not benefit freshwater ecosystems. This may however be contemplated in specific cases if the practitioner has evidence of benefit to freshwater systems.

A different case is rainwater harvesting: even though some might consider it a way of increasing freshwater resources, it should not be considered in the same way as suggested for desalinated sea water, because rainwater would be available as green or blue water if it was not harvested. The benefits of a system using harvested rainwater will appear as a smaller use of green or blue water.

2.4. Changes in the water cycle caused by land use related to production system (FEI)

Accounting for water stored as soil moisture (green water) is essential for VW in order to show the total water use of a crop, to calculate the amount of blue water abstracted, and to show where that water came from in the hydrological cycle. However, LCA does not account for issues not affected by the production system. For example, the amount of solar energy used to grow crops is not accounted because solar radiation is independent of the crop or production system; i.e. the land will receive the same insolation regardless of the type of crop established[1]. In the same way, a similar amount of soil moisture (stored rain water) will be used by different crops and/or natural ecosystems regardless of the production system; therefore the use of rainwater does not change the environmental effects that would occur if the studied system was not established. However, knowledge of soil moisture (green water) used by plants is needed to assess the amount of blue water required to grow those plants. Thus, green water use should be included to estimate the blue water impacts through evaporation of irrigation water. The LCI results thus calculated will be compatible with VW quantifications, but the amount of green water will subsequently be disregarded in the LCIA phase because green water use leads to no environmental impacts.

An issue to consider however is that production systems may significantly change the amount of rainwater available to other users through changes in the fractions of rainwater that follow each one of the three basic paths: Infiltration (I);

[1] Differences in albedo between different crops may be relevant for global modeling, but are outside the scope of LCA.

Evapo-transpiration (ET); and Runoff (R). In general, ET is the amount "lost" from the system (evaporative use), whereas I and R return to the system. Owens (2002) suggests that for in-stream water consumption "evaporative losses from reservoirs and canals in excess of unrestricted river losses" are to be accounted (i.e. the unrestricted river is considered as the reference system). Likewise, it is here suggested that changes in evaporation are most significant in 'aquatic land uses' such as reservoirs and canals (see 3.2.1). Bauer and Zapp (2004) suggest using the difference between monthly precipitation and evapo-transpiration as an estimate of surface and sub-surface runoff (i.e. I+R) and provide values as a world map. However, such an approach would suggest, rather counter-intuitively, that sealed land (with largely reduced ET) has a positive effect on the water cycle due to the increased R. Actually, and particularly with heavy rainfalls, such rapid returns in the form of R may have little effect in replenishing aquifers (see Section 3.3), can lead to increased flooding and be of no use to ecosystems. Heuvelmans *et al.* (2005) suggest using changes in (I – ET) alongside other indicators for land use impacts on water (namely change in surface runoff and precipitation surplus). However, the feasibility of their approach in all life cycle stages is doubtful due to the large amount of data and modelling required for each land use along a product's life cycle.

Therefore, we suggest assessing I+R (as done by Bauer and Zapp) for "non-sealed" land uses, and only the potential changes in I for all other uses. The argument is that land acts as a buffer of the water cycle through I, and this is the "ecological quality" of land which should be protected (Milà i Canals *et al.* 2007). Note that in specific situations increases in I may lead to adverse effects (e.g. by raising saline groundwater table) but such site-specific situations are beyond the scope of LCA and therefore not considered here. Changes in I are assessed against the reference system in the land use impact assessment (Milà i Canals *et al.* 2007).

This change in groundwater recharge may be linked to the freshwater ecosystem impact potential (2.2). Land use may also be linked to freshwater depletion potential (2.3) through groundwater recharge, if land use affects the recharge of overexploited aquifers; we suggest not including this pathway as a default because it is site-specific. In both cases, the land use effect may be beneficial (if water infiltration is increased) or damaging (if infiltration is reduced).

2.5. Water flows[2] to be quantified in LCI

In summary, the following water flows need to be accounted in LCI:

From a *freshwater ecosystem impact potential* point of view:

- **Surface and groundwater evaporative uses**: in-stream evaporation in reservoirs and power dams and off-stream evaporation of abstracted water through e.g. irrigation; in cooling towers; etc. In VW terms: evaporative blue water.
- Any type of **land use occupation and transformation** processes.

From a *freshwater depletion potential* point of view:

- **Water stocks (groundwater - fossil water) and over abstracted water funds (groundwater - aquifers)**: **both evaporative and non-evaporative uses** need to be quantified. This is consistent with existing methods for abiotic resource depletion such as CML2001, but in 'consequence-oriented LCIA methods' ('Type 3' methodologies according to Lindeijer *et al.* 2002) only the amount of resource dissipated (evaporated) may merit being accounted for (Stewart and Weidema 2005). Our suggestion at this point is to record both evaporative and non-evaporative use flows separately so they can then be treated appropriately in LCIA.

Many LCI databases contain flows and datasets for provision of "tap water"; this paper is only concerned with the elementary flows (coming from/going to nature). As with energy resources, it will be a matter for public LCI databases to define the water resource flows related to "national water grids"; the impacts of such grids will in turn depend on the mix of water sources, much as the impacts of power grids depend on the mix of energy sources and technologies.

3. Life Cycle Inventory (LCI) modelling

3.1. Calculation of water evaporated from irrigation

The water withdrawal and subsequent evaporation from a crop field can be calculated following the methodology used by Chapagain and Orr (in press). In brief, the various steps from their study are as follows:

First, the virtual water content of the primary crop is calculated as the ratio of water used for crop production to the yield per unit area. The volume of water used for production is made up of two components, evaporative and non-evaporative water. Following Chapagain and Orr (in press), non-evaporative water use is a function of irrigation losses (implicitly including all losses in storage, conveyance systems, and field application) and volume of water rendered unsuitable for use further downstream as a result of polluted return flows (grey water). They suggest that environmental effects of grey water are more suitably addressed in other impact categories.

The evaporative demand is met from soil moisture that is within the root zone depth of the plant and immediately available for plant uptake. Depending upon the source of water maintaining the soil moisture, Chapagain (2006) distinguishes between *green water use* (WU_g) originating from rainfall on crop land and *blue water use* (WU_b) originating from irrigation water supply.

The components WU_g and WU_b depend on the specific crop evaporation requirement and soil moisture availability in the field. The crop evaporation requirement ($ET_c[t]$) is calculated using the classical Penman-Monteith equation to estimate reference crop evaporation following the methodology recommended by FAO (Allen *et al.* 1998). This can easily be done with e.g. the publicly available CROPWAT model (FAO, 1992) or alternative models. The USDA SCS (USDA Soil Conservation Service) approach has been used to estimate the effective rainfall ($p_{eff}[t]$), as it is one of the most widely used methods in estimating effective rainfall in agricultural water management (Cuenca, 1989; Jensen *et al.* 1990). Climate data

[2] Flow refers here to the element listed in a life cycle inventory (LCI), i.e. "elementary flow", and not to the type of resource as in "flow, fund or stock".

for representative climate stations may be obtained from the study inventory or alternatively from CLIMWAT (FAO, 1993). Green water use, u_g[t], is equal to the minimum of effective rainfall and the crop evaporation requirement at that time step, and total green water use (WU_g) in crop production is calculated by summing-up green water use for each time step over the entire cropping period, l (day). Green water use is independent of irrigation water supply and solely depends on the effective rainfall and crop evaporation requirements, whereas blue water use depends on crop evaporation requirement, green water availability and irrigation water supply. The fraction of ET_c[t] not met by u_g[t] is the irrigation requirement (I_r[t]). The blue water use (u_b[t]) is the minimum of irrigation requirement, I_r[t], and the effective irrigation water supply, I_{eff}[t]. The effective irrigation supply is the part of the irrigation water supply that is stored as soil moisture and available for crop evaporation. Blue water use is zero if the entire crop evaporation requirement is met by the effective rainfall. Total blue water use (WU_b) in crop production is calculated by summing-up blue water use for each time step over the entire cropping period, l (day).

There are inevitable irrigation losses from the local system, as it is hard to match the blue water demand with irrigation water supply in time and space. Irrigation losses (I_{loss} [t]) are calculated by subtracting blue water use u_b[t] from irrigation water supply I_s[t] if known. Total irrigation losses from the field over the crop period are calculated by summing the losses in each time step over the entire crop growth period, l (day).

3.2. Calculation of water evaporated from other processes

LCA studies include water use in a variety of processes apart from irrigation (discussed above), many of which will be essentially non-evaporative. In some, though, part of the water used is evaporated, and this needs to be estimated in the LCI in order to provide relevant information for the LCIA. The following paragraphs suggest ways in which losses can be estimated for the main industrial processes causing water evaporation as a proportion of the total water input to the process.

3.2.1. Evaporation from reservoirs and canals (‘aquatic land uses’)

Basic data to calculate evaporation from a reservoir per m^3 of water used are its area, volume abstracted per year, and potential evaporation in the region (in mm, or litres per m^2, common meteorological parameter). As a specific example, a 10,000 m^2 reservoir for irrigation providing 900,000 m^3/year of water for irrigation in a region with a mean annual evaporation potential of 1,400 mm has an evaporation loss of 1,400*10,000/ 900,000 = 15.5 litres/m^3 of water delivered. This calculation may be modified to account for devices used to reduce the evaporation rate, months when the reservoir is kept empty, etc. when such information is available.

3.2.2. Cooling water

Industrial plants, particularly thermal plants generating electrical power from fossil or nuclear fuel, are major water users. Water is used for two essential functions: as the “working fluid” driving steam turbines and as coolant in condensers and other heat exchangers. The water in a turbine cycle is highly purified and used on a closed-loop basis: water make-up is therefore small and dependent on details of plant design and operation. Coolant water is sometimes abstracted from a water body or river, used once and then returned at a higher temperature; net coolant water use is then small. More commonly, the coolant water is used in a circuit in which it is heated in the condensers and heat exchangers, to be cooled and re-used. Where the plant is operated in the Combined Heat and Power (CHP) mode, exporting heat by distributing hot water, the hot water is circulated to the heat user and returned to the plant; net water use is then again small. Where the plant exports steam, the net use depends on whether the steam is vented, or condensed and reused. Where there is no heat output, the cooling water is used in a circuit with the heat dissipated by evaporation in cooling towers. There is then a net water loss, corresponding to water evaporated.

To a first approximation, the cooling load in a generating plant using evaporative cooling corresponds to the latent heat (strictly, enthalpy change) of evaporation of the water lost. For example, a plant operating with 35% efficiency (i.e. electrical output energy is 35% of the thermal energy released from the fuel so that 65% must be dissipated as “waste heat”) requires a cooling load of 3600x65/35 kJ per kWh of electrical output; i.e. about 6700 kJ/kWh. The latent heat of evaporation of water is about 2400 kJ/kg, so that the theoretical evaporative loss is 6700/2400; i.e. about 2.8 kg of real water loss per kWh. Actual figures may be somewhat larger or smaller, depending on details of plant design and operation (including whether any of the waste heat can be used – CHP is more efficient than electricity-only generation on water use as well as energy use). The average figure for nuclear thermal power production of 9.01 kg water/kWh(e) given by the Ecoinvent database represents water abstracted; net loss by evaporation must be established as primary data for any specific plant but should be around one third of this figure even for energetically wasteful evaporative cooling.

3.2.3. Textile drying

The washing stage is one of the most important sources of water usage in textile products. Most of the water used in washing is returned as wastewater, and part remains in the clothes and is partly evaporated through drying. On average, water content in wet (after centrifugation) clothes is 0.7 kg/kg clothes (70%), and this goes down to residual moisture of ca. 0.03 kg/kg clothes (3%) after conventional drying (Group for efficient appliances 1995). These values thus yield an evaporative use of ca. 0.67 kg water evaporated per kg dried clothes; considering a value of about 1.8 m^3 evaporated blue water per kg seed cotton in the cropping stage (Chapagain *et al.* 2006b), washing clothes about 100 times in their lifetime represents about 3-5% of the evaporative water use in growing cotton.

3.3. Estimation of land use effects on rainwater infiltration

Table 1 suggests some values for the % of water “lost” due to different land occupation processes. The land uses listed (derived from the Ecoinvent classification) have been treated in two different ways:

- In systems generally allowing infiltration (non-sealed land), both infiltration and runoff (I+R) have been considered as useful paths for ecosystems; they are marked as "N" land uses. Lost water is ET. Most values for such uses are derived from a large review of worldwide water catchments studies (Zhang *et al.* 1999).
- In systems that are heavily transformed and generally sealed, only I has been considered as useful for ecosystems, and water "lost" is ET+R. They are marked as "S" land uses in **Table 1**. All these values are estimated based on the fact that fully sealed land has negligible infiltration, and assuming an infiltration rate of 1% on land that is not fully sealed (e.g. construction site).

This table has to be seen as a gross approximation, which may be useful as a default set of values but needs refinement. The assumptions made have been clear in some cases (e.g. sealed soil reduces infiltration to 0% by definition), while in some others the justification is less clear. The table thus offers plausible, rather than accurate, values.

Table 1

Table 1 provides values for low and high precipitation areas (< and > 600mm year^{-1}); in general, though, values for only one gross precipitation group will be needed for LCA databases. As an example, the last column in **Table 1** suggests values of rainfall "lost" per m^{2}year of different land occupations in Europe, assuming average precipitation of 734mm (Gleick *et al.* 1993) with forest as the reference (potential) land use. As a specific example, the estimated reference rainwater lost from forest is 67% whereas in arable land this is 73%; therefore, the extra loss due to using arable land is 6% of rainwater, or 44 litres per m^2 per year for average precipitation of 734mm.

4. Characterisation factors for Life Cycle Impact Assessment (LCIA)

4.1. Indicators for freshwater ecosystem impact (FEI)

Several indicators are used to compare the sustainability of water supplies in different countries. Falkenmark (1986) proposed an indicator based upon Water Resources (WR) Per Capita (WRPC=WR/population) with defined threshold values for *water stress* – less than 1667 (usually rounded to 1700) m^3 per capita, *water scarcity* – less than 1000 m^3 per capita, and *absolute water scarcity* – less than 500 m^3 per capita year. While this indicator has since been adopted as the standard indicator of water scarcity, it fails to consider the ability of nations to adapt to reduced per capita water availability through means such as VW, and does not consider differences in water use patterns between countries or multiple in-stream uses (Raskin *et al.* 1997). Besides, WRPC is intended to apply to human direct (domestic) use (drinking + sanitary) but this is seldom the problem: most water is used in agriculture and then by industry. This indicator has been used in LCA by Antón *et al.* (2005).

Feitelson and Chenoweth (2002) suggest an index of water sustainability based around the affordability of water supplies (index of structural water poverty). Its major shortcoming is that there is no simple or transparent way to determine real water supply costs in a country, and thus data for the index are lacking. Besides, this is an index for social impacts related to water, rather than environmental impacts as sought by LCA.

A more useful index for determining the environmental sustainability of water supply and water stress is the Water Use Per Resource (WUPR=WU/WR) indicator put forward by Raskin *et al.* (1997). This index compares the percentage of available water resources being withdrawn from natural water bodies. This index does not address water quality issues but it highlights the water remaining for in-stream usage or further development and/or ecosystems, a factor disregarded by the standard WRPC indicator. Therefore WUPR is a good indicator for potential impact on aquatic ecosystems. A high WUPR indicates serious water stress as most available water is being used. Additionally, because of climate variability, the higher the water exploitation ratio, the greater the chances of water shortages during dry years. The WUPR ratio thus indicates the "marginal impacts of water usage": the impacts of providing 1 extra unit of water increase as the proportion of resources already used increases. This is true both from an ecosystems perspective and from a resource provision perspective (e.g. desalination becomes an option only after most of the easily available resources are used). Data for this indicator are readily available for most of the world at a national level. River basin level data may be more relevant from a FEI perspective, and LCA practitioners are encouraged to find and use an appropriate level of precision for the foreground system. The WUPR indicator may be used directly as a characterisation factor:. multiplying by the % of water used gives bigger weight to water used in countries/regions where a bigger proportion is used. Alternatively, more sophisticated Characterisation Factors (CF) may be constructed to describe step changes using "threshold values" and/or non-linear relationships (see section 4.1.2.).

An alternative indicator for environmental water stress is explored by Smakhtin *et al.* (2004), who make a first attempt at estimating the Environmental Water Requirements (EWR) for all world river basins. They then combine EWR with the water resources available and their use (i.e. WUPR defined per river basin), by subtracting EWR from the available resources to derive a Water Stress Indicator (WSI=WU/(WR-EWR)). A more accurate indication of the water resources available for further human use after "reserving" the necessary resource for ecosystems (EWR) can thus be obtained. However, the use of this indicator is at an early stage lack of data might hamper its use in LCA.

4.1.1. Developing factors for the suggested indicators

Table A-1 in the Appendix provides values for the WRPC and WUPR indicators for most countries. It has been compiled using data from the FAO Aquastat database (FAO 2004) and the UNDP human development indicators (United Nations Development Programme 2006). The basic parameters required to construct the indicators (population, water resources and use) should be found for smaller geographical areas when available and appropriate for CF calculation.

Table A-2 in the Appendix provides values for WSI for the world's main river basins (Smakhtin *et al.* 2004).

4.1.2. Possible further sophistication: thresholds and damage approach

As noted above, the WRPC, WUPR or WSI ratios may be used directly as a characterisation factor for the amount of water evaporated. However, it may be argued that the potential for impacts is unlikely to follow a linear relationship with such ratios; for example, a double-exponential relationship may be more appropriate, with low effects for low WUPR and a maximum effect before the 100% value to reflect the likelihood that most ecosystems will be damaged even before all water is used by humans.

The construction of such relationships requires extensive research on the effects on aquatic ecosystems of increasing appropriation of freshwater for human uses. Alcamo *et al*. (2000) note that there is no objective basis for setting stress thresholds for the water exploitation index (WUPR); however, they suggest that a withdrawal to availability (WUPR) ratio exceeding 0.8 indicates very high stress; a ratio between 0.4 and 0.8 indicates high stress, 0.2 to 0.4 medium stress, 0.1 to 0.2 low stress, and below 0.1 no stress. The European Environment Agency has essentially adopted these thresholds but combines the medium, high and very high stress categories into a single “stressed” category (European Environment Agency, 2003). Smakhtin *et al*. (2004) discuss the definition and meaning of such thresholds and suggest that 20-50% of the available river water volume should be left for ecosystems (stressing that this value is arbitrary and ecosystem- / river basin-dependent). Further development of the WUPR- or WSI-effect on ecosystem health (i.e. the ‘dose-response curve’) is required to link freshwater use to damage-level indicators such as PAF (Potentially Affected Fraction of species). Smakhtin *et al*. (2004) review on-going initiatives to enlarge the knowledge base on aquatic species diversity, which would be necessary to derive a relationship between WUPR or WSI and PAF. However they suggest that total water abstraction, rather than only the evaporative uses, should be considered as impacting ecosystems, arguing that it is normally unclear how much of the withdrawn water actually returns to the abstracted system (and in what condition). As explained in section 2.5, we propose considering evaporative use flows only as a baseline approach; nevertheless, alternative world views (e.g. an “egalitarian view” according to the Cultural Theory adopted in the EcoIndicator 99, see Goedkoop and Spriensma 1999) might prefer taking a more conservative attitude, including also non-evaporative uses for FEI.

Table 2 provides a description of FEI as a new impact category following the format suggested by Guinée *et al*. (2002), with WSI as the recommended characterisation model.

Table 2

4.2. Characterisation factors for freshwater depletion (FD)

Given that water is an abiotic resource and that it may, in some circumstances, be at least temporally and spatially depleted, the Abiotic Depletion Potential (ADP) (Guinée and Heijungs 1995) suggested as a baseline method for abiotic resources depletion in the CML 2001 guide (Guinée *et al*. 2002) seems the most appropriate approach.

4.2.1. Developing ADP factors for freshwater

Adapting the ADP formula (Guinée *et al*. 2002, p. 544) with the possibility of regeneration of water funds (in a similar way as with biotic resources, see p. 546), we get:

$$ADP_i = \frac{ER_i - RR_i}{(R_i)^2} \times \frac{(R_{Sb})^2}{DR_{Sb}} \qquad \text{[Eq. 1]}$$

where ADP_i is the Abiotic Depletion Potential of resource *i* (e.g. groundwater from aquifer x); ER_i is the Extraction Rate of resource *i*; RR_i is the Regeneration Rate of resource *i*; R_i is the ultimate reserve of resource *i*; DR_{Sb} is the Deaccumulation Rate of the reference resource for ADP (Sb, Antimony); R_{Sb} is the ultimate reserve of the reference resource for ADP (Sb, Antimony). Note that underexploited groundwater bodies (i.e. with RR>ER) would yield a negative characterisation factor: such cases would not lead to depletion of freshwater resources and therefore should be neglected in the assessment.

The problem with this approach is that groundwater reserves are seldom quantified in terms of their relative abundance compared to potential use, with the exception of small aquifers (Prof R. Llamas, personal communication, November 2007). When assessed on a country level, values tend to be very uncertain, and in any case RR is often bigger than ER. For example, Hernández-Mora *et al*. (2007) provide a very detailed assessment of groundwater resources in Spain, but give only a rough estimate of the country’s reserves at 150,000-300,000 Mm^3 and highlight that some specific aquifers are known to be overexploited. If there is knowledge that the relevant aquifer is being over-abstracted, or that fossil water is being used, then the LCA practitioner should find the necessary values to develop ADP factors for the specific water bodies in question. For instance, Custodio (2002) provides a detailed examination of many cases of reported overexploited aquifers around the world. As an illustration, he cites the case of California, where an overexploitation (ER-RR) of $2.5*10^9$ m^3year^{-1} and estimated total reserves of $1{,}600*10^9$ m^3 are reported (of which only $140*10^9$ m^3 are assumed usable). In Almeria (Spain), the same author reports a depletion rate (ER-RR) of $50*10^6$ m^3year^{-1} and total reserves of $1{,}100*10^6$ m^3 (of which $750*10^6$ m^3 are usable; ca. 15 years to depletion). Applying the ADP formula above, and using as a reference Antimony’s R^2/DR of $5.69*10^{-24}$, yields the following ADP for groundwater (gw):

$ADP_{gw,California} = 1.72*10^5$ kg Sb eq/kg

$ADP_{gw,Almeria} = 7.26*10^9$ kg Sb eq/kg

These ADP factors are much higher than ADP for most other abiotic resources listed in Guinée *et al*. (2002). This indicates that water may be comparatively more vulnerable to depletion locally than other resources. Thus, when water from overexploited aquifers is involved, it may dominate the ADP category. However, in most cases groundwater will be found to be renewable (i.e. RR>ER) and neglected from the abiotic resource depletion impact category.

4.2.2. Possible further sophistication: damage approach

”Depleted” (dissipated) water actually returns to being naturally available in a short period of time. However, if one acknowledges that water may be temporally and locally depleted, consequence-oriented approaches such as the surplus energy to obtain the resource when the natural source has been exhausted may be used. In the case of water, desalination may be considered the ultimate backup technology (Stewart and Weidema 2005). In the 1970s, early reverse osmosis plants used as much as 20 kWh of electricity per cubic metre but this was reduced to as little as 3.5 kWh per cubic metre by the

end of the 1990s (Fritzmann *et al.* 2007). The theoretical amount of energy required for desalination of seawater, regardless of the technology used, has been estimated to be less than 1kWh per cubic metre (Avlonitis *et al.* 2003).
Desalination technologies are continuing to evolve and it is hard to predict how far efficiency might improve in the future. The Ammonia-Carbon Dioxide Forward Osmosis process, which has been demonstrated in the lab and for which a pilot scale plant is currently under construction (Elimelech 2007) has an estimated energy requirement of 0.84 kWh of energy per cubic metre of water desalinated (McGinnis and Elimelech 2007). Much of this energy may be sourced as low temperature heat (as low as 40 degrees), with only 0.25kWh of energy required as electrical power.
From the above discussion, the lower and upper limits for the energy requirements (Stewart and Weidema 2005) for water ultimate backup technology (desalination) are 0.84 and 3.5 kWh/m^3.

5. Discussion

It is crucial for Life Cycle Inventory (LCI) to distinguish between evaporative and non-evaporative uses of freshwater. The paper provides guidance to quantify both types of uses for the main processes commonly assessed with LCA.
Some value judgements have been identified through the paper: one of the main issues is whether non-evaporative use of water should be addressed at all in impact assessment (LCIA). Although we suggest considering only evaporative losses as affecting freshwater ecosystems, a more precautionary (egalitarian) approach might call for inclusion of non-evaporative use as well. Indeed, considering that non-evaporative uses of water represent no impact on FEI may yield an underestimation of local effects (e.g. when water abstracted from a river does not return to the same river): locally, the effects on river flows might be devastating.
A new midpoint impact category, Freshwater Ecosystem Impact (FEI) has been suggested, which could be linked to ecosystem health at a damage level. Three different indicators have been assessed for FEI, of which WSI (water stress indicator) appears as the most useful at the present state of understanding. However, information on water use for different river basins is not usually available in LCA (except for the foreground system). Average values for larger regions (e.g. Europe) may be necessary, although possibly not so representative from an ecosystem point of view.
It needs to be highlighted that variation in seasonal flows is not considered in the FEI indicators suggested (which are based on annual resource estimates): during drought periods any abstraction may have dramatic consequences for ecosystems, but this is not yet implemented in the approach. On the other hand, time-dependency is implicitly included in the VW equations for evaporative use of water in agriculture.
The uncertainty and inherent variability in most parameters for the calculation of water characterisation factors are big, and the stakes are particularly high when inter-country comparisons leading to sensitive commercial decisions are to be made. For the US, for example, estimates of internal renewable water resources range from 1,890 km^3 to 3,760 km^3 depending upon the source (FAO 2003). The Aquastat database (FAO 2004) lists the US as having 2818.4 km^3. This closely links to the spatial-dependency issue, as pointed out by Owens (2002). This paper suggests a simple yet plausible set of LCI models and LCIA characterisation factors that may be used as a default. It is left to the practitioner's responsibility to provide more detailed data for the foreground level if relevant, following the framework defined in this paper to derive new characterisation factors for smaller reference areas such as river basins.
The approach suggested in this paper has important requirements for LCI databases with respect to the number of elementary flows that need to be recorded. We argue that this requirement is justified by the relevance of impacts on freshwater ecosystems, while we recognise that some decisions will need to be made on where the spatial differentiation is justified. Besides, it needs to be decided whether water flows are recorded on a national level or a river basin level; information currently exists for both, and the latter has more environmental meaning. In any case, "average" characterisation factors will be required for regions such as Europe, North America or Asia, because site-generic LCI databases will not have more precise information for specific manufacturing sites. Recent methodological developments based on the WUPR suggest grouping water flows in six classes according to scarcity (Frischknecht 2008); this would significantly increase the possibilities of applying such an approach, with a small compromise on resolution of relative FEI. In the case of FD, the impact is so localised that it will probably affect only known cases of aquifer over-abstraction in the foreground system; modelling such cases should not cause a problem for LCI databases.

6. Conclusions

Current work to calculate Virtual Water (VW) associated with delivery of goods and services, particularly for biotic production systems, can help to compile LCI information. In particular, calculations of blue and green water use are valuable for both VW calculations and LCI. However, the requirements for LCA differ from VW estimation. E.g., green water, essential in VW calculations to show the total water use of a crop, receives a characterisation factor of zero in LCIA.
This paper addresses primarily quantitative aspects related to fresh water use, but highlights some qualitative issues that are not yet properly assessed in LCIA, namely impacts on aquatic ecosystems due to changes in water temperature (e.g. from cooling water) and impacts on human health from microbiological pollution (usually in less developed countries).
The information required for systematic assessment of freshwater resources in LCA and VW is generally available, although it will require some degree of consensus on the level of detail and extensive effort from LCA database developers.
This paper has proposed indicators that will improve the representation in LCA of impacts arising from use of freshwater resources, and will be useful to assess the comparative merits/threats posed by water-intensive products such as food or feedstock for bioenergy sourced from different regions. Such characterisation may also be useful for VW studies aiming to highlight the potential consequences of trade on source countries, in terms of impacts on freshwater ecosystems and long-term freshwater availability.
One new midpoint-level impact category has been suggested: Freshwater Ecosystem Impact, which addresses the potential effects on aquatic ecosystems caused by changes in freshwater availability. Characterisation Factors (CF) have been proposed for the midpoint level approach as well as some hints to derive related factors for a damage level approach.

In addition, the use of groundwater resources should be considered alongside other resources in categories for abiotic resource depletion, following current methods to derive CF from use and replenishment rates. Preliminary indications are that groundwater depletion can be much more significant than depletion of other abiotic resources when aquifers are over abstracted.

7. Outlook/ Needs for further research

Future development of the Water Stress Indicator (WSI) to measure Freshwater Ecosystem Impact (FEI) will require measurements of WSI at the scale of individual river basins, along with data on all principal groundwater bodies at least at the scale of main geographical regions and preferably at a more localised scale. To provide threshold values for this impact category, better understanding is needed of the relationship between WSI and other indicators such as the Potentially Affected Fraction (PAF) of freshwater species.

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Fig. 1: Main impact pathways related to freshwater use. All the pathways are discussed in this paper but only those depicted with solid arrows are considered for LCA. The concepts in circles denote common denominations in the Water Footprint field. The numbers refer to the impact pathways defined in sections 2.1-2.4
Table 1: Effects of land occupation on usable proportion of precipitation, considering Ecoinvent's land occupation flows
Table 2: Description of the necessary components of FEI according to ISO 14044 (point 4.4.2.2.2.)

Table A-1: Values for WRPC and WUPR for world countries
Table A-2: Values for WSI for the main world river basins (kindly provided by V. Smakhtin)

Table 1

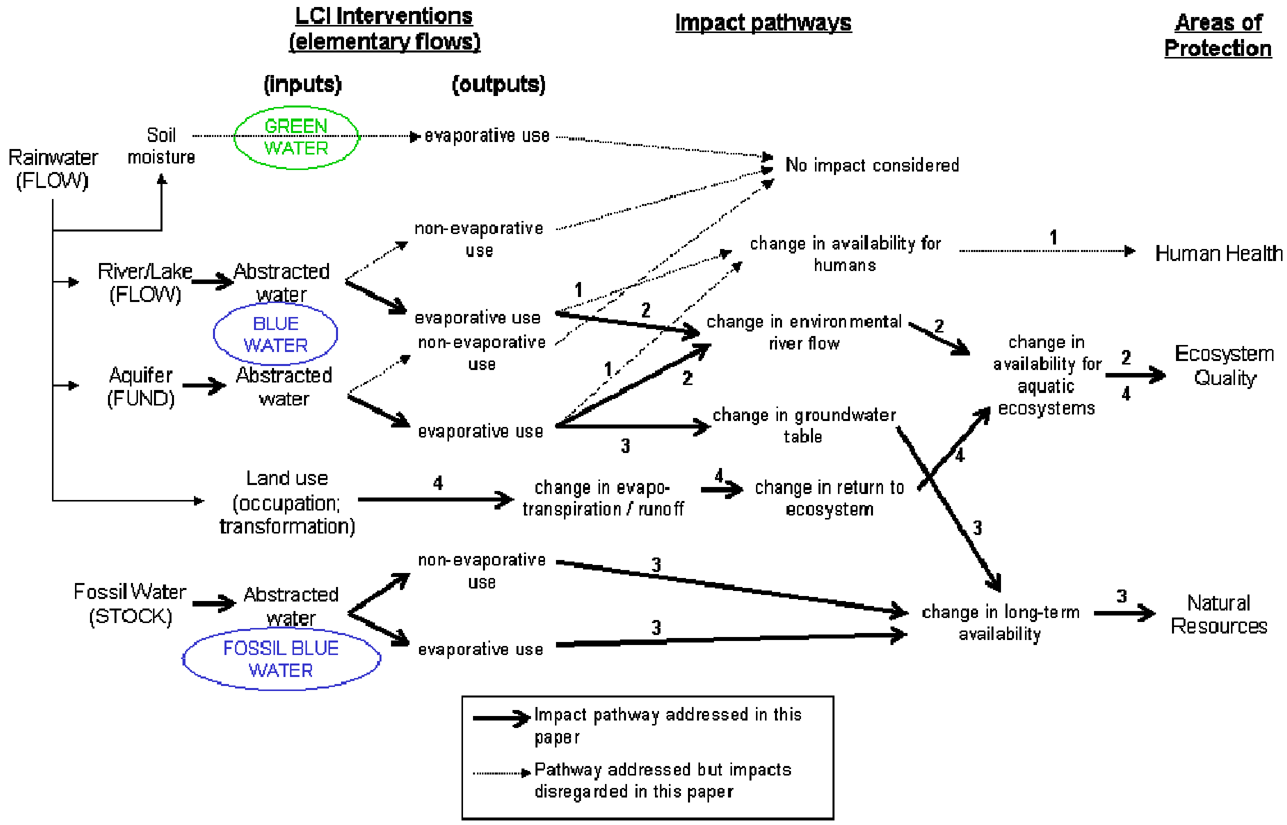
Ecoinvent land occupation flows	Land use Type	Rainfall <600 mm/year: Percentage of "lost" precipitation	Rainfall <600 mm/year: Sample size	Rainfall >600 mm/year: Percentage of "lost" precipitation	Rainfall >600 mm/year: Sample size	"Lost precipitation" mm/m^{2}year assuming 734 mm/year rainfall (Gleick 1993) and forest as reference land use
Occupation, arable, non-irrigated [a]	N	93	10	73	19	44
Occupation, construction site	S	99	Est.	99	Est.	235
Occupation, dump site	S	99	Est.	99	Est.	235
Occupation, dump site, benthos	- [b]	-		-		-
Occupation, forest, intensive [a]	N	83	4	67	36	0
Occupation, forest, intensive, normal [a]	N	83	4	67	36	0
Occupation, industrial area [c]	S	99	Est.	98	Est.	228
Occupation, industrial area, benthos	- [b]	-		-		-
Occupation, industrial area, built up	S	100	Est.	100	Est.	242
Occupation, industrial area, vegetation [d]	N	94	Est.	73	Est.	44
Occupation, mineral extraction site	S	100	Est.	100	Est.	242
Occupation, pasture and meadow,	N	94	15	73	35	44
Occupation, pasture and meadow,	N	94	15	73	35	44
Occupation, permanent crop, fruit,	N	95	19	67	51	0
Occupation, shrub land, sclerophyllous [a]	N	92	3	64	7	-22
Occupation, traffic area, rail embankment	N	95	Est.	95	Est.	206
Occupation, traffic area, rail network	N	95	Est.	95	Est.	206
Occupation, traffic area, road	N	95	Est.	95	Est.	206
Occupation, traffic area, road network	S	100	Est.	100	Est.	242
Occupation, urban, discontinuously built [c]	S	99	Est.	98	Est.	228
Occupation, water bodies, artificial	- [b]	-		-		-
Occupation, water courses, artificial	- [b]	-		-		-

[a] Data derived from Zhang *et al.* (1999).
[b] "Aquatic land uses" have been considered in a different way (see text)
[c] Assumed to have 5% vegetated area.
[d] Considered as pasture.

Table 2

Component	Description
LCI results assigned to Impact category	Described in section 2.5
Characterisation model	3 options suggested. Recommended model: WSI $WSI = \frac{WU}{WR - EWR}$ where WSI is the Water Stress Indicator (Smakhtin *et al.* 2004); WU (Water Use) is the amount of water abstracted for human uses; WR are the renewable water resources; and EWR are the Environmental Water Requirements (Smakhtin *et al.* 2004)
Characterisation factors	WSI for main world river basins are provided in Table A-2 in the Appendix
Category indicator	m^3 of "ecosystem-equivalent" water, referring to the volume of water likely to be affecting freshwater ecosystems. It does not seem appropriate to utilise a reference ecosystem such as "m^3 Amazon-water-equivalents". From a damage approach, the Potentially Affected Fraction of species (PAF) may be used once a relationship is worked out between WSI and PAF.
Category endpoint	Freshwater ecosystems
Environmental relevance	There is a good linkage between the category indicator results and the endpoint

Fig. 1



LCI Interventions (elementary flows)
Impact pathways
Areas of Protection
(inputs)
(outputs)
Rainwater (FLOW)
Soil moisture
GREEN WATER
evaporative use
No impact considered
non-evaporative use
change in availability for humans
Human Health
River/Lake (FLOW)
Abstracted water
BLUE WATER
evaporative use
non-evaporative use
change in environmental river flow
change in availability for aquatic ecosystems
Ecosystem Quality
Aquifer (FUND)
Abstracted water
evaporative use
change in groundwater table
Land use (occupation; transformation)
change in evapo-transpiration / runoff
change in return to ecosystem
Fossil Water (STOCK)
Abstracted water
FOSSIL BLUE WATER
non-evaporative use
evaporative use
change in long-term availability
Natural Resources
1
2
3
4
Impact pathway addressed in this paper
Pathway addressed but impacts disregarded in this paper

Table A-1

	Total renewable water resources (m^3/capita per year) (WRPC)	Percentage of actual renewable water resources being used (WUPR)
Albania	13452	4.1
Algeria	442	42.4
Angola	11871	0.2
Antigua and Barbuda	520	n.a.
Argentina	21198	3.6
Armenia	3510	28.1
Australia	24724	4.9
Austria	9476	2.7
Azerbaijan	3604	57.0
Bahamas	67	n.a.
Bahrain	166	258.0
Bangladesh	8697	6.6
Barbados	268	104.9
Belarus	5918	4.8
Belgium	1760	n.a.
Belize	61850	0.7
Benin	3024	1.0
Bhutan	45238	0.4
Bolivia	69170	0.2
Botswana	8000	1.0
Brazil	44769	0.7
Brunei Darussalam	21250	n.a.
Bulgaria	2731	49.3
Burkina Faso	977	6.2
Burundi	493	6.5
Cambodia	34501	0.9
Cameroon	17844	0.3
Canada	90688	1.6
Cape Verde	600	9.3
Central African Republic	36100	0.0
Chad	4574	0.5
Chile	57267	1.4
China	2215	21.8
Colombia	47483	0.5
Comoros	1500	n.a.
Congo	213333	0.0
Congo, Dem. Rep. of	22952	0.0
Costa Rica	26140	2.4
Côte d'Ivoire	4525	1.1
Croatia	23444	n.a.
Cuba	3404	21.5
Cyprus	975	31.3
Czech Republic	1289	19.5
Denmark	1111	21.1
Djibouti	375	2.6
Dominican Republic	2386	16.1
Ecuador	33231	3.9
Egypt	803	117.8
El Salvador	3710	5.0
Equatorial Guinea	52000	0.4
Eritrea	1500	4.8
Estonia	9852	1.3
Ethiopia	1455	2.4
Fiji	35688	0.2
Finland	21154	2.3
France	3378	19.6
Gabon	117143	0.1
Gambia	5333	0.4
Georgia	14073	5.7
Germany	1864	30.6
Ghana	2452	1.0
Greece	6689	10.4

Guatemala	9046	1.8
Guinea	24565	0.7
Guinea-Bissau	20667	0.4
Guyana	301250	0.7
Haiti	1670	7.0
Honduras	13704	0.9
Hungary	10297	7.3
Iceland	566667	0.1
India	1745	34.1
Indonesia	12894	2.9
Iran, Islamic Rep. of	1999	53.0
Ireland	12683	2.2
Israel	253	122.2
Italy	3298	23.2
Jamaica	3617	4.4
Japan	3362	20.6
Jordan	157	115.4
Kazakhstan	7406	31.9
Kenya	901	5.2
Korea, Rep. of	1464	26.7
Kuwait	8	2227.1
Kyrgyzstan	3958	49.0
Lao People's Dem. Rep.	57509	0.9
Latvia	15413	0.8
Lebanon	1259	31.1
Lesotho	1679	1.8
Libyan Arab Jamahiriya	105	801.9
Lithuania	7324	1.1
Luxembourg	6200	n.a.
Macedonia, TFYR	3200	n.a.
Madagascar	18619	4.4
Malawi	1371	5.8
Malaysia	23293	1.6
Maldives	100	n.a.
Mali	7634	6.9
Malta	126	109.6
Mauritania	3800	14.9
Mauritius	1842	27.7
Mexico	4326	17.1
Moldova, Rep. of	2774	19.8
Mongolia	13385	1.3
Morocco	935	44.0
Mozambique	11140	0.3
Myanmar	20912	3.2
Namibia	8970	1.5
Nepal	7902	4.8
Netherlands	5617	8.7
New Zealand	81750	0.6
Nicaragua	36424	0.7
Niger	2493	6.5
Nigeria	2224	2.8
Norway	83043	0.6
Oman	394	137.1
Pakistan	1438	76.1
Panama	46244	0.6
Papua New Guinea	138103	0.0
Paraguay	56000	0.1
Peru	69312	1.1
Philippines	5870	6.0
Poland	1596	26.3
Portugal	6606	16.4
Qatar	66	554.2
Romania	9722	10.9
Russian Federation	31322	1.7
Rwanda	584	1.5
São Tomé and Principe	10900	n.a.
Saudi Arabia	100	721.7

Senegal	3456	4.0
Sierra Leone	30189	0.2
Singapore	140	n.a.
Slovakia	9278	n.a.
Slovenia	15935	n.a.
Solomon Islands	89400	n.a.
South Africa	1059	30.6
Spain	2617	32.0
Sri Lanka	2427	25.2
Sudan	1817	57.9
Suriname	305000	0.5
Swaziland	4510	18.4
Sweden	19333	1.7
Switzerland	7431	4.8
Syrian Arab Republic	1412	76.0
Tajikistan	2497	74.9
Tanzania, U. Rep. of	2420	2.2
Thailand	6436	21.2
Togo	2450	1.1
Trinidad and Tobago	2954	8.0
Tunisia	456	59.8
Turkey	3176	16.4
Turkmenistan	5150	99.7
Uganda	2374	0.4
Ukraine	2969	26.9
United Arab Emirates	35	1537.5
United Kingdom	2471	6.5
United States	10391	15.6
Uruguay	40882	2.3
Uzbekistan	1924	115.7
Venezuela	46889	0.7
Viet Nam	10725	8.0
Yemen	202	161.7
Zambia	9148	1.7
Zimbabwe	1550	13.1

n.a.: Data on country's water use are not available.

Table A-2

Basin	WSI
Alabama	14.9%
Amazon	0.1%
Amu Darya	144.1%
Amur	5.3%
Balsas	11.5%
Brahmaputra	37.2%
Brazos	220.7%
Chao Phrya	53.3%
Chubut	9.0%
Colorado	2080.9%
Columbia	18.2%
Congo	0.1%
Dalalven	1.9%
Danube	49.1%
Dnieper	91.0%
Dniester	37.9%
Don	60.6%
Ebro	88.8%
Elbe	101.3%
Fly	0.0%
Fraser	1.5%
Ganges	37.2%
Garonne	39.5%
Glama	2.6%
Godavari	63.8%
Guadalquivir	177.4%
Hudson	56.2%
Hwang Ho	175.5%
Indigirka	0.0%

Indus	407.8%
Irrawaddy	1.6%
Jubba	40.5%
Kapuas	0.1%
Kemijoki	1.1%
Kolyma	0.0%
Krishna	191.3%
Kura	152.1%
Lake Balkhash	96.1%
Lake Chad	20.9%
Lake Turkana	1.6%
Lakes Titicaca and	733.4%
Lena	0.1%
Limpopo	62.9%
Loire	16.9%
Mackenzie	0.8%
Magdalena	3.1%
Mahakam	0.0%
Mahanadi	43.2%
Mangoky	4.9%
Mania	1.5%
Mekong	4.3%
Mississippi	69.3%
Murray	41.6%
N. Dvina	0.5%
Narmada	106.4%
Nelson	13952.9%
Lake Ladoga	5.5%
Niger	7.4%
Nile	29.6%
Ob	50.4%
Oder	48.2%
Ogooue	0.1%
Okavango	1.2%
Orange	43.9%
Orinoco	0.3%
Oued Draa	1142.7%
Parana	3.4%
Parinaba	2.8%
Po	54.1%
Rhine-Maas	80.2%
Rhone	35.7%
Rio Colorado	157.1%
Rio Grande	275.4%
Rio Grande de	81.1%
Rio San Pedro &	0.2%
Sacramento	98.0%
Salween	3.7%
Sao Francisco	9.1%
Seine	53.0%
Senegal	14.2%
Sepik	0.0%
Shaballe	40.5%
Song Hong	8.1%
St. Lawrence	45.6%
Susquehanna	26.4%
Syr Darya	288.2%
Tagus	92.3%
Tapti	167.0%
Tarim	6005.8%
Thelon	0.0%
Tigris & Euphrates	252.4%
Tocantins	0.3%
Ural	33.4%
Uruguay	2.1%
Vistula	47.2%
Volga	9.7%

Volta	3.2%
W. Dvina	4.7%
Weser	90.9%
Xun Jiang	8.0%
Yalu Jiang	9.1%
Yangtze	13.5%
Yenisey	0.3%
Yukon	0.2%
Zambezi	1.5%
Yaqui	87.8%
Negro	10.1%
Rufiji	0.5%
Cunene	2.6%
Cuanza	0.2%
Duero	55.4%
Kizil	98654.3%
Pecora	0.1%
Belyando	20.0%
Dawson	75.3%