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Life Cycle-based Water Assessment of a Hand Dishwashing Product: Opportunities and Limitations

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ABSTRACT

It is only recently that life cycle-based indicators have been used to evaluate products from a water use impact perspective. The applicability of some of these methods has been primarily demonstrated on agricultural materials or products, because irrigation requirements in food production can be water-intensive. In view of an increasing interest on life cycle-based water indicators from different products, we ran a study on a hand dishwashing product. A number of water assessment methods were applied with the purpose of identifying both product improvement opportunities, as well as understanding the potential for underlying database and methodological improvements. The study covered the entire life cycle of the product and focused on environmental issues related to water use, looking in-depth at inventory, midpoint, and endpoint methods. "Traditional" water emission driven methods, such as freshwater eutrophication, were excluded from the analysis. The use of a single formula with the same global supply chain, manufactured in 1 location was evaluated in 2 countries with different water scarcity conditions. The study shows differences ranging up to 4 orders in magnitude for indicators with similar units associated with different water use types (inventory methods) and different cause–effect chain models (midpoint and endpoint impact categories). No uncertainty information was available on the impact assessment methods, whereas uncertainty from stochastic variability was not available at the time of study. For the majority of the indicators studied, the contribution from the consumer use stage is the most important (>90%), driven by both direct water use (dishwashing process) as well as indirect water use (electricity generation to heat the water). Creating consumer awareness on how the product is used, particularly in water-scarce areas, is the largest improvement opportunity for a hand dishwashing product. However, spatial differentiation in the inventory and impact assessment model may lead to very different results for the product used under exactly the same consumer use conditions, making the communication of results a real challenge. From a practitioner's perspective, the data collection step in relation to the goal and scope of the study sets high requirements for both foreground and background data. In particular, databases covering a broad spectrum of inventory data with spatially differentiated water use information are lacking. For some impact methods, it is unknown as to whether or not characterization factors should be spatially differentiated, which creates uncertainty in their interpretation and applicability. Finally, broad application of life cycle-based water assessment will require further development of commercial life cycle assessment software. *Integr Environ Assess Manag* 2013;9:633–644. © 2013 SETAC

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Life Cycle & Sustainability

INTRODUCTION

With a growing interest in sustainable production and consumption, products and services are being evaluated using a life cycle thinking (LCT) approach. A methodological framework was first established by the Society of Environmental Toxicology and Chemistry (SETAC) (SETAC 1993) and soon after life cycle assessment (LCA) methods were standardized by the International Standards Organization (ISO 14040 2006; ISO 14044 2006). LCA aims to comprehensively assess products and services through evaluation on multiple indicators across the entire life cycle (cradle-to-grave approach). Tracking water inputs has been included in early LCA studies, but because models on water availability impacts only recently became available, the information was not used to assess the potential impact from water use, as it has been the case for other indicators.

In 2002, a novel method to quantify water flows within the supply chain was developed by the Water Footprint Network (Hoekstra and Hung 2002), which was recently updated (Hoekstra et al. 2011). This method classifies water in different categories by quantifying water consumption and degradation. Sustainable management of water systems includes protection of water availability as well as avoidance of pollution leading to degradation of water quality. The latter could be seen as making water unavailable for future use. The Hoekstra method is not capable of quantifying any potential impact, because no cause–effect chain is elaborated. Within the LCA framework, quantification of potential impacts is done at the life cycle impact assessment (LCIA) phase. Within the classification step of this phase, elementary flows are first grouped into a common environmental impact category (e.g., global warming effect, aquatic eutrophication). The contribution of each elementary flow within the impact category is subsequently quantified (characterization), usually by taking a reference flow and comparing the contribution relative to this reference. Modeling environmental effects of water use and change in quality is

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extremely complex, as local conditions play a major role. Because traditional LCA approaches aggregate elementary flows across geographic boundaries, critical information of the local conditions is not included. For effects occurring at a global scale such as global warming or stratospheric ozone depletion, this approach is sufficient. However, most effects for water are observed at the local level, which means that the local information is of utmost importance to adequately model potential environmental impacts. It took until 2001 for a framework to be created (Owens 2001) outlining how to develop indicators describing potential environmental impacts of water use. This framework or some aspects thereof was further refined by Bayart et al. (2010) and Boulay, Bouchard et al. (2011). Many water indicators have been developed because, quantifying the cause–effect chains of water use on 3 different areas of protection (AoP): human health, ecosystem quality, and resources (Berger and Finkbeiner 2010; Kounina et al. 2012). Their applicability has primarily been demonstrated for agricultural systems, because water related issues are most prevalent within this sector (World Water Assessment Programme UN 2009). Typically, these case studies are geographically well-defined and most of these studies are only cradle to gate assessments.

To increase our understanding on LCT-based water assessment methods, Procter & Gamble (P&G) commissioned Quantis in 2010 to inventory and evaluate various water impact assessment methods. We focused on methods describing the ecosystem quality AoP and based it on the framework developed by Kounina et al. (2012). Traditional methods with water in scope and dependent on inventory emission data, such as aquatic ecotoxicity and freshwater eutrophication, are excluded. In general, we applied methods using water elementary flow information as input and describing consumptive and degradative use aspects.

Methods were selected on the basis of practical applicability, data availability, level of spatial differentiation, different types of water use covered (water consumption, in-stream water use, etc.) and scientific review.

MATERIALS AND METHODS

Hand dishwashing case study

Within Europe, there exist different habits of dishwashing. In the full sink method (FS), consumers fill their sink with water and then add product. Dishes are cleaned in the solution and rinsed separately. The concentration of the product remains constant during the entire dishwashing process. In the direct application method (DA), consumers first add product to the sponge or wash towel, clean the dishes, and then rinse under running tapwater. The product therefore is gradually rinsed out during the entire dishwashing process. In some instances, consumers will use concentrated minisolutions, a habit that is more frequently applied in regions where water is not easily available. After reviewing the broad range of dishwashing practices that exist in Europe, the DA and FS habits were selected for this case study, because they are quite different in total water use for the dishwashing process. Once the habits to be used for the study were defined, a selection of water assessment methods were applied to a P&G-developed hand dishwashing cradle-to-grave LCA study with a functional unit of hand washing 10 plates. The reference flow used was 2.4 or 4.8 g of product per use, depending on the dishwashing habit (FS or DA method respectively). The reference flow is higher for the DA method, because the product is gradually rinsed out.

The system boundaries include: the production of raw materials (both for the product as well as its packaging), the formulation of raw materials into a hand dishwashing formula, the package-making operation, distribution of the packed product, the use of the product by consumers, and the final disposal of the packaging as well as the wastewater treatment of the down-the-drain emissions (Tables S1–S3 in Supplemental Data). Primary data include: formula and packaging information and specific removal of chemicals in sewage treatment. Secondary data are from Ecoinvent v2.2 (2010). The geographical scope is the production of the ingredients at the suppliers' manufacturing locations, the manufacturing of the product in London (UK), and the use and end-of-life of the hand dishwashing product in Germany and Spain. These countries were selected for 2 reasons. First, the dishwashing product sold in these countries is sourced from the same location. Within the context of a water assessment study, the identification of the location of all water uses in the supply chain is important. Therefore, a single supply chain limits the data collection effort. Second, Spain and Germany are very different in water scarcity index (WSI), an important parameter for the spatial differentiation within some of the applied methods.

For the purpose of the method comparison, a baseline scenario was selected, representing the cleaning of 10 plates with a hand dishwashing product used in Spain under the conditions of the DA method with wash water heated by an electrical boiler (Figure 1).

Water assessment methods

Building on a review of methods by Kounina et al. (2012), selected methods were categorized into 3 classes: 1) inventory methods, 2) midpoint, and 3) endpoint methods. Table 1 lists the methods evaluated within each of the 3 classes.

Inventory methods. Inventory methods classify elementary water flows according to their type (origin of water resource, intake water quality, etc.) and their use (off-stream, in-stream, consumptive/degradative use) into water inventory categories. These can be used as input for impact assessment methods to develop water impact categories at midpoint or endpoint. All inventory methods use volumetric units. The inventory methods used in this study classify water use into the following inventory categories: water withdrawal, renewable groundwater withdrawal, blue water consumption, green water consumption, thermally polluted water, chemically polluted water, turbined water, and Quantis gray water.

Some inventory categories consider withdrawal, representing the total volume withdrawn from a water body, whereas others measure consumption. Consumption is defined as the difference between the amount of water withdrawn from a water body and the amount released back to the same body. The difference may be due to water being evaporated (e.g., cooling systems), returned to a different water body (groundwater returned to a river) or included in a finished product. Another observation is that some inventory categories deal with quantitative aspects (withdrawal, consumption), whereas others focus on qualitative aspects (thermal or chemical changes in quality). Blue water includes freshwater uses from surface or groundwater, whereas green water relates to precipitation water stored in or on the soil or in the vegetation.

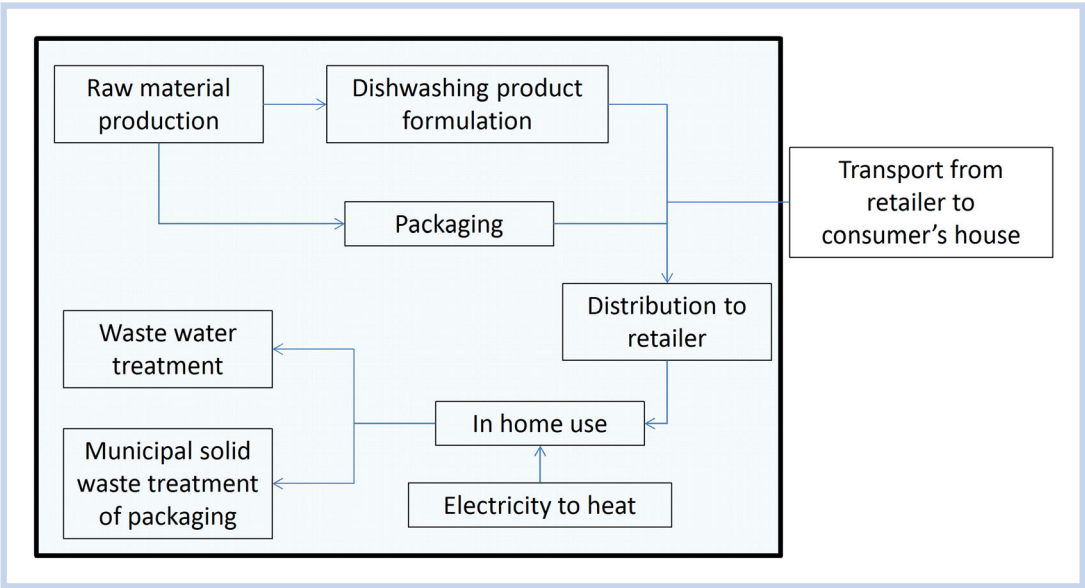

Figure 1. System boundaries for the hand dishwashing baseline scenario. The scenario is for Spain assuming a direct application practice of cleaning the dishes. Processes outside the shaded area are excluded.

Table 2 provides an overview of the elementary water flows used for the calculation of water inventory categories and an overview of the water inventory categories used by the water midpoint and endpoint impact categories in this study.

In a typical LCA study, technical flows are converted into elementary flows, using background databases (e.g., Ecoinvent). Commercial LCA software aggregates all elementary flows in the inventory list, which is used to apply impact assessment methods. During this aggregation process, all site-specific information is lost. A similar procedure is followed when water inventory methods are applied. For each technical flow, the Quantis water database (Quantis 2010), beta version November 2010, is checked to determine whether or not water inventory category information is available (e.g., renewable groundwater withdrawal). If information is unavailable in the Quantis water database, the relevant elementary water flows in Ecoinvent are used to convert the information into water inventory categories, a process for which water inventory methods are necessary. Similar to the inventory list in traditional LCA studies, this process results in a list of water inventory categories for each technical flow in the system. The information, however, is not aggregated, because local information must be stored for each individual water inventory flow. Because commercial LCA software is not designed for this, calculations for this study were done in Excel.

The Quantis water database uses the elementary water flows from Ecoinvent combined with a large literature review, to build a database with processes and materials. Sometimes assumptions are necessary, given that the source of some elementary water flows is unspecified in Ecoinvent (e.g., "water, unspecified natural origin"). In those cases, a conversion factor is defined to quantify how much water from the unspecified source originates from groundwater, lakes, etc. As an example, if a unit process includes the elementary flow

Table 1. Overview of water assessment methods applied on the hand dishwashing water assessment study

Type	Method	Reference
Inventory	WFN method (green water footprint)	Hoekstra et al. 2011
	Quantis water databases methods (water withdrawal, renewable groundwater withdrawal, blue water consumption, green water consumption, turbined, thermally and chemically polluted water)	Quantis 2011
Midpoint	Swiss ecological scarcity	Frischknecht et al. 2006
	Blue water consumption	Boulay et al. 2011
	Blue water consumption	Pfister et al. 2009
	Blue and gray water consumption	Ridoutt and Pfister 2010
	Method evaluating blue and green water	Milà-i-Canals et al. 2009
Endpoint	Terrestrial species diversity from blue water consumption	Pfister et al. 2009
	Terrestrial species diversity from renewable groundwater consumption	Van Zelm et al. 2011
	Aquatic species diversity from thermally polluted water	Verones et al. 2010

WFN = Water Footprint Network.

Table 2. Overview of the water elementary flows and water inventory categories used in this study

		Midpoint					Endpoint		
Water elementary flow (from Ecoinvent v2.2)	Water inventory category	Swiss Ecological Scarcity (2006)	Pfister et al. (2009)	Ridoutt and Pfister (2010)	Boulay et al. (2011)	Milà-i-Canals et al. (2009)	Pfister et al. (2009)	Van Zelm et al. (2011)	Verones et al. (2010)
Water, well, in ground	Water withdrawal								
Water, lake	Renewable groundwater withdrawal							x	
Water, river	Blue water consumed	x	x	x	x	x	x		
Water, salt, ocean	Green water consumed								
Water, salt, sole	Thermally polluted water			x					x
Water, cooling, unspecified natural origin	Chemically polluted water			x					
Water, turbine use, unspecified natural origin	Turbined water								
Water, unspecified natural origin	Quantis–gray water footprint								

Water inventory methods, specified in the Quantis water database, define how water elementary flows are grouped into water inventory categories. The various midpoint and endpoint methods use different water inventory categories as input, combined with geographical information (if spatially differentiated).

"Water cooling, unspecified natural origin," then an assumption is made that 70% of the cooling water is sourced from rivers, 29% is sourced from lakes, and 1% from groundwater. Information about the evaporation rate is necessary for cooling operations. The Quantis water database assumes this to be 5% for river water. This information is necessary to close the water balance on a unit process basis. In general, assumptions are based on available statistics and on expert judgment. Water databases should therefore clearly document the inventory methods applied, as well as the assumptions and data used to convert elementary water flows into water inventory categories.

The inventory methods included in this study differ significantly in their objective. The water inventory methods used in the Quantis database aim at including all relevant water uses to apply the various water impact assessment methods. The Water Footprint Network (WFN) method is primarily designed as a water accounting tool for organizations or nations and does not accommodate specific water related uses.

Midpoint methods. All of the midpoint methods included in this research characterize water consumption by using a WSI. Water scarcity indexes applied in this study are based on water withdrawal to availability ratios for the Swiss ecological scarcity method (Frischknecht et al. 2006), the Pfister et al. (2009) method, and the Ridoutt and Pfister (2010) method and on water consumption to availability ratios for the Boulay, Bulle et al. (2011) method. The Milà-i-Canals et al. (2009) method uses 3 different characterization methods. One is based on WSI developed by Smakhtin et al. (2004), another is based on a Falkenmark index (Falkenmark et al. 1989), and the third is based on a Raskin index (Raskin et al. 1997). The Falkenmark index is based on a water use per capita ratio (WRPC), whereas the Raskin index is based on water use per resource ratio (WUPR). Because there is no global coverage of the Smakhtin WSIs, this characterization factor could not be applied to the Milà-i-Canals et al. (2009) method.

The function of the WSI is to convert the volumetric unit of the water use into a water stress equivalent unit. The units of the midpoint methods based on WSI are always expressed in m^3-equivalents (a unitless ratio is applied to the water inventory flow). The Swiss ecological scarcity method uses Swiss ecopoints (UBP). The water impact of 1 m^3-eq includes information on the location of the water consumption and could be the consumption of 10 m^3 in a region with low water stress (WSI = 0.1) or could be the consumption of 1 m^3 in a region with very high water stress (WSI = 1). All methods, except the one by Ridoutt and Pfister (2010), use the inventory category blue water consumption as an input. Ridoutt and Pfister (2010) use the sum of both blue and gray water, with gray water being the sum of thermally and chemically polluted water.

Endpoint methods. The endpoint methods assessed in this study are focused on the ecosystem quality AoP, reflecting changes in species diversity, expressed as the potentially disappeared fraction of species (PDF) integrated over space and time. Although the units of all selected endpoint methods are similar, they all represent a different type of potential impact from water use. From the methods used in this study, we can distinguish 1 endpoint method that models diversity impacts on aquatic species and 2 methods that model diversity impacts on terrestrial plant species. Within the terrestrial plant species methodologies, Pfister et al. (2009) models blue water

consumption into potential impacts on net primary production, using it as a proxy for ecosystem quality. The potential impact of the use of shallow groundwater on terrestrial plant species, resulting in a lower water table, is modeled by Van Zelm et al. (2011). The impacts on aquatic species attributed to thermally polluted water is modeled by Verones et al. (2010). The water inventory input used for this method is the amount of thermally polluted water released to watersheds. In this case study, all 3 endpoints were evaluated, which are in fact all complementary to each other as they model different cause–effect chains.

RESULTS

In the following sections, results for the inventory, midpoint, and endpoint methods are shown and broken down across life cycle stage contribution. All results are discussed for the baseline scenario, assuming DA practice for cleaning dishes in Spain. Tables with absolute values are in the Supplemental Data (Tables S4–S6).

Inventory methods

Water inventory results are between 0.3 and 29 L per 10 plates cleaned (Figure 2). Turbined water use is 1500 L per 10 plates cleaned, but not included in Figure 2 due to the large difference with all other water inventory categories. The different water use types, modeled by the various water inventory categories, explain the large differences observed. As uncertainty information was not available in the beta version of the Quantis water database used at the time of the study, no information is presented in this study on the uncertainty distribution associated with elementary water flows as well as resulting uncertainty distributions for inventory and impact results. However, the data from the Quantis water database are used in the Ecoinvent v3 database. Consistent with the Ecoinvent methodology, the uncertainty for each water elementary flow is described by a pedigree matrix, allowing us to define the stochastic variability. Despite being based on a large literature review, no information is provided on the uncertainty associated with the modeling of elementary water flows into water inventory categories. Results are therefore presented without uncertainty information.

Except for green water, water inventory results are primarily driven by the use stage. This is observed both for inventory categories describing water availability (bottom 3 bars in Figure 2) and water quality (top 3 bars in Figure 2) aspects. The contribution at the use stage can be further separated into direct and indirect water use. Direct water use is tapwater used for the cleaning of the plates. Indirect water use is associated with the heating of the tapwater. Because the baseline scenario assumes heating with an electrical boiler, indirect water represents water use for electricity production used to heat the water.

Turbined water, thermally polluted water, and blue water consumption are primarily associated with indirect water use (contribution in total is 95%, 96% and 73%, respectively), as this is related to electricity production. In the case of blue water consumption, this is due to water evaporation in power plants (cooling towers or water reservoirs in hydropower installations). Contributions from indirect water use into other water inventory categories range between 6% (chemically polluted water) and 40% (renewable groundwater withdrawal).

Contributions from direct water use are driven by tapwater production and range between 58% (renewable groundwater withdrawal) and 94% (chemically polluted water). The high contribution of tapwater production into chemically polluted water is explained by an assumption in the Quantis water

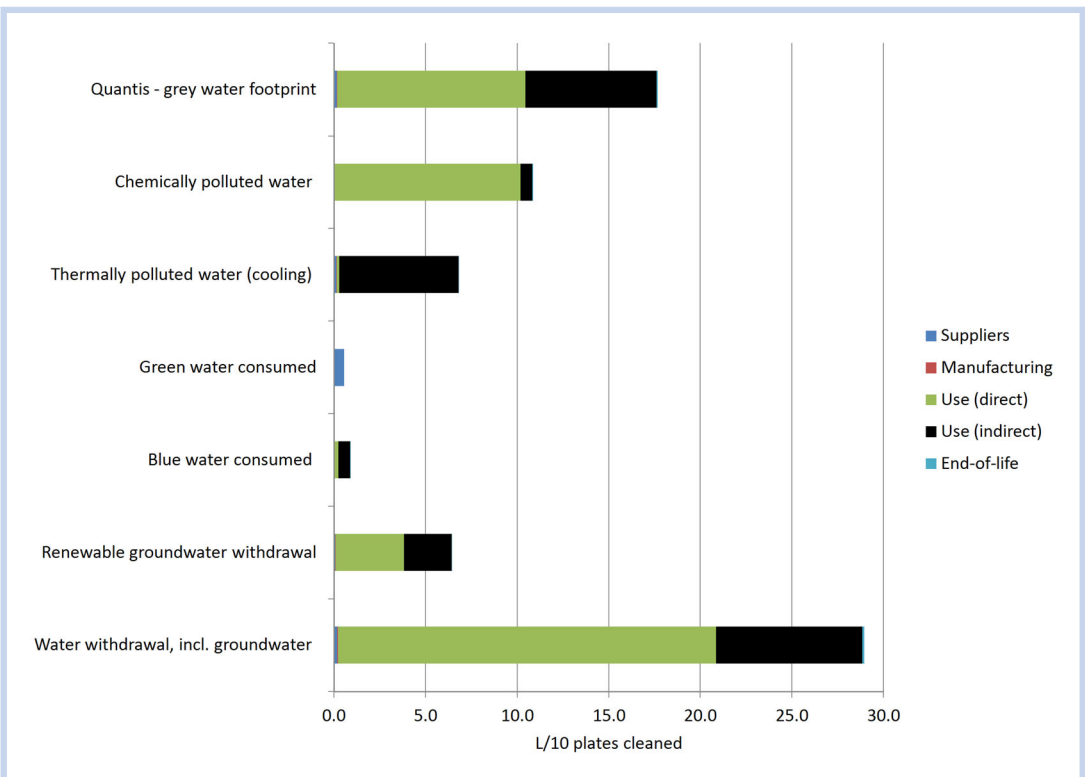

Figure 2. Water inventory category results for the cleaning of 10 plates using a hand dishwashing detergent in Spain with the direct application method and using an electrical boiler to heat the water. Results are broken down by their contribution across the different life cycle stages. Direct water use at the use stage refers to tapwater used for the cleaning; indirect water use at the use stage is associated with the heating and represents upstream water use from electricity production. Turbined water use is not shown on the graph because it is orders of magnitude larger.

database, beta version November 2010 (half of the volume released to water bodies is below the receiving water body quality standard).

Green water use is associated with the water use in crops being used as feedstock material for ethanol production through fermentation. Ethanol is 1 of the ingredients in the dishwashing product (Table S1).

Midpoint methods

Midpoint results range from 0.16 to 2400 L-eq with the lowest result for the Boulay, Bulle et al. (2011) method and the highest for the Milà-i-Canals et al. (2009) method based on the Falkenmark index. The Swiss ecological scarcity method result is 870 kUBP. Consistent with the inventory methods, the use stage is the primary contributing source to the midpoint results ranging from 97%–99% (Figure 3). Results are shown on a relative scale, because different units are presented (L-eq or kUBP). The Boulay country method in Figure 3 refers to the spatial differentiation at the country level (i.e., a single WSI is used per country). The Milà-i-Canals method is specified according to the WSI applied (WRPC applies the Falkenmark index, WUPR applies the Raskin index). Except for the Ridoutt and Pfister (2010) method, all midpoint methods use blue water consumption as the water inventory category input. With the blue water consumption from electricity production in Spain being approximately 3 times higher than the blue water consumption for tapwater production (Figure 2), indirect water use is the most important (72%–74%) for all midpoint methods, except for Ridoutt and Pfister (2010).

The Ridoutt and Pfister (2010) method uses both blue water consumption and thermally and chemically polluted water as water inventory category input. As can be seen from Figure 2, direct water use is the primary driver for chemically polluted water. Indirect water use is primarily responsible for blue water consumption and thermally polluted water. This explains the smaller contribution (42%) of the indirect water use in the Ridoutt and Pfister (2010) method.

Endpoint methods

Endpoint results range over 3 orders of magnitude (Figure 4), with the lowest result seen using the Verones et al. (2010) method (1.4E-6 PDF.m^2.yr) and the highest result for the Van Zelm et al. (2011) method (1.4E-3 PDF.m^2.yr). Considering the methods evaluating the terrestrial species diversity (bottom 2 bars in Figure 4) and without uncertainty information, the Van Zelm et al. (2011) method shows the highest impact, although results only differ a factor of 3.

Consistent with the inventory and midpoint methods, the use stage is the most important (Figure 5). The use of indirect water (associated with electricity use) contributes 96% of the total impact for the Verones et al. (2010) method, as seen in Figure 5. Indeed, electricity production takes a prominent place

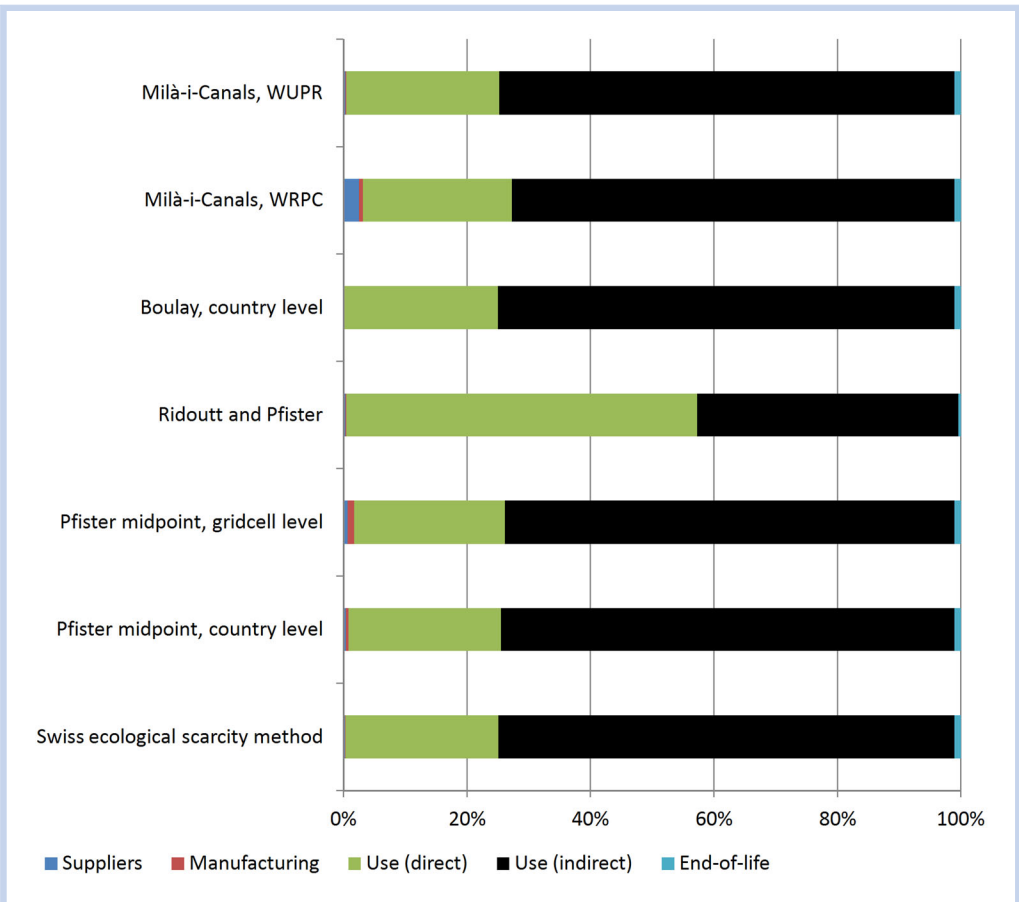

Figure 3. Water impact results at the midpoint level for the cleaning of 10 plates using a hand dishwashing detergent in Spain with the direct application method and using an electrical boiler to heat the water. Results are in percents of the total indicator score and broken down by their contribution across the different life cycle stages. Direct water use at the use stage refers to tapwater used for the cleaning; indirect water use at the use stage is associated with the heating and represents upstream water use from electricity production.

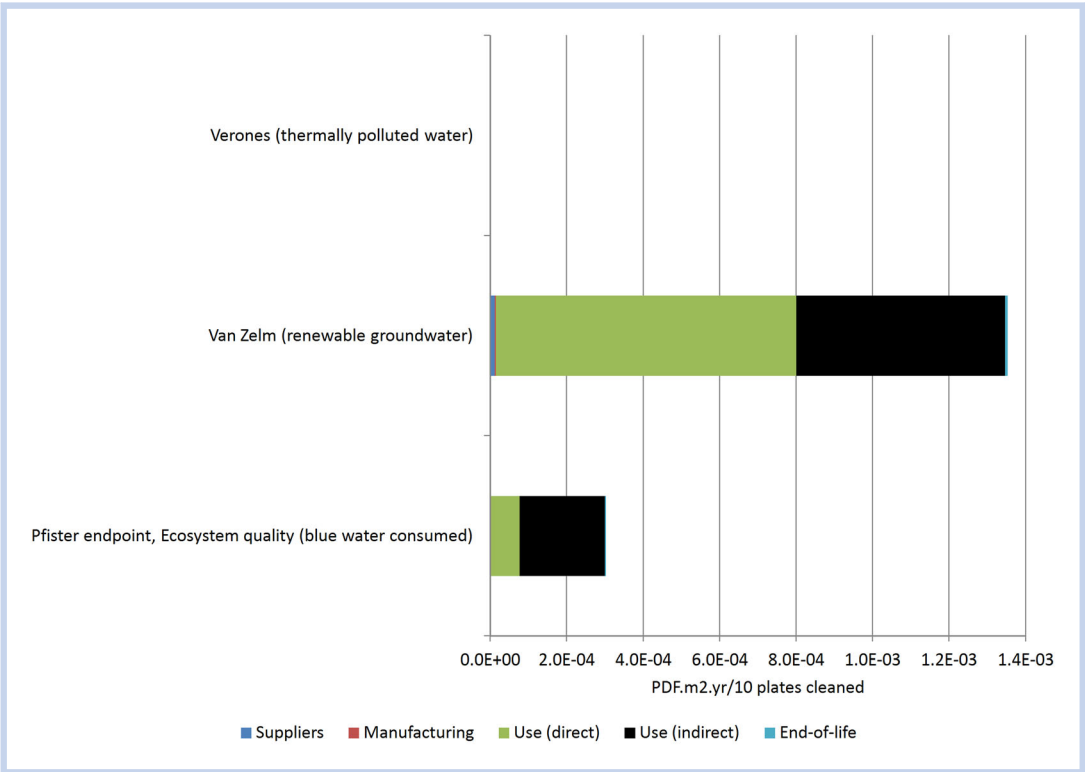

Figure 4. Water impact results at the endpoint level for the cleaning of 10 plates using a hand dishwashing detergent in Spain with the direct application method and using an electrical boiler to heat the water. Results are broken down by their contribution across the different life cycle stages. Direct water use at the use stage refers to tapwater used for the cleaning; indirect water use at the use stage is associated with the heating and represents upstream water use from electricity production.

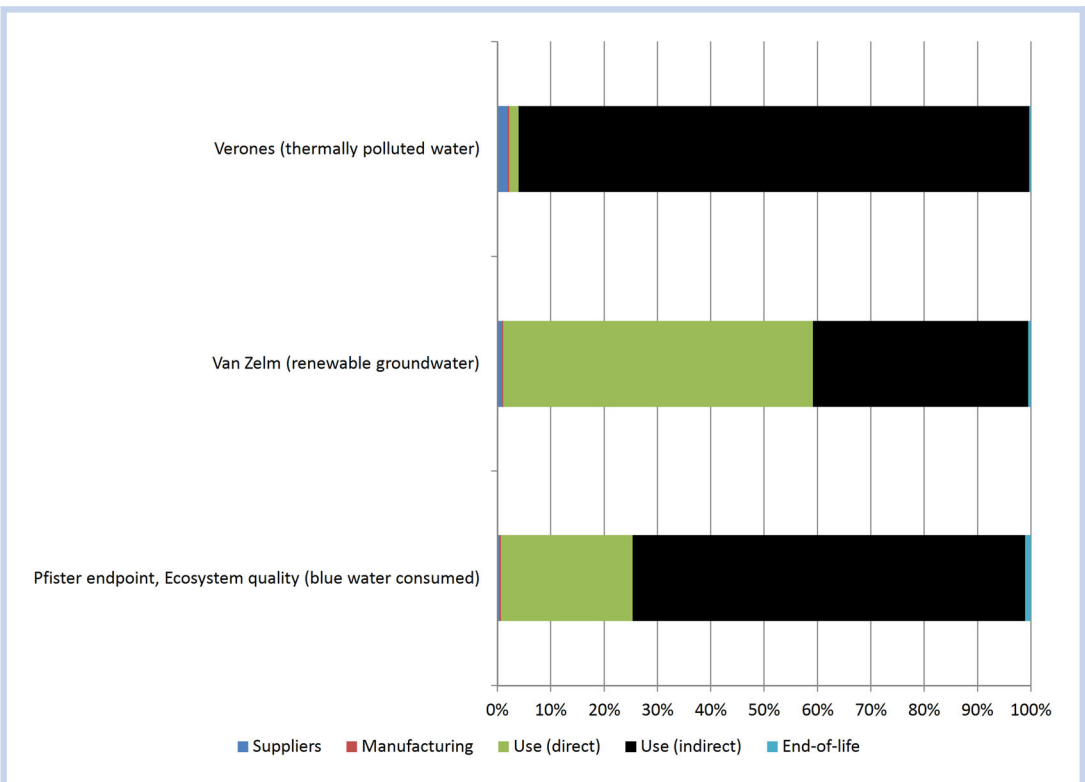

Figure 5. Water impact results at the endpoint level for the cleaning of 10 plates using a hand dishwashing detergent in Spain with the direct application method and using an electrical boiler to heat the water. Results are in percents of the total indicator score and are broken down by their contribution across the different life cycle stages. Direct water use at the use stage refers to tapwater used for the cleaning; indirect water use at the use stage is associated with the heating and represents upstream water use from electricity production.

in the modeling pathways for this method. Also the Pfister et al. (2009) method concludes that indirect water use (electricity production) has the biggest impact (74%).

Modeling water inventory categories with midpoint and endpoint methods introduces additional uncertainty, even beyond the uncertainty described in the water inventory methods section. Unfortunately, uncertainty on water impact assessment methods is not well described in the literature. Pfister and Hellweg (2011) provide an uncertainty assessment on 2 methods describing human health impacts due to lack of water for human use. They observed uncertainty on midpoint methods within 1 order of magnitude and for endpoints sometimes stretching over 4 orders of magnitude. Uncertainty assessment solely related to the scope of the cause–effect chain is not addressed. Spatial aggregation at the country level leads to increased uncertainty.

DISCUSSION

Water impact assessment models for application in LCA were not developed until recently. On the one hand, data availability has been an issue to develop operational models, but recently this is improving with a growing interest in water. On the other hand, scientists face many challenges when modeling impacts from water use, e.g., the different impact pathways and the fact that impacts are dependent on local and temporal conditions. Regional specificity is perhaps the most important reason why it took so long before the first models to be developed. Traditionally, impact assessment models have struggled with site specific modeling (Pant et al. 2004; Gallego et al. 2010). The discussion of our analysis on the water impact assessment methods evaluated in this study will focus on these 2 key aspects: the importance of good data and spatial differentiation.

Importance of data availability

As explained in the introduction, data availability on water use is not as widespread as for other environmental indicators, such as energy. Only in recent years, with an increased interest in water, corporations start to comprehensively build water data into their environmental and sustainability reports. This may explain why the commercially available LCI databases also lack information on water use. In addition, to build such LCI databases, it is important to have a framework for how water use should be reported in the LCI phase, which in turn depends on what the impact assessment methods require from the inventory model. This type of water LCI framework was first proposed by Owens (2001) and further developed by the UNEP/SETAC Life Cycle Initiative working group on water (Bayart et al. 2010; Kounina et al. 2012). Building on this, a water database was developed by Quantis (2012). It has a global scope but is mostly populated with European and Swiss data and is built on the elementary water flow information from the Ecoinvent database.

When this study was initiated, we applied data from a beta version of the Quantis water database. For our case study, country specific data was used for tapwater supply in Germany and Spain (Table 3). It is a coincidence that the sourcing of tapwater in Spain and Germany has a similar distribution. Should we not have had data for one of these countries and used the European average, the share of groundwater would have doubled. This could have led to a significantly different result on renewable groundwater withdrawal (Figure 2) and the Van Zelm endpoint result (Figure 4). This tells us that water impact assessment studies require high data representativeness.

Table 3. Sources for the production of tap water used in the case study[a]

	Europe (%)	Spain (%)	Germany (%)
Groundwater	36	18	17
Surface water	64	82	83

[a]Quantis database (beta version).

At the same time, given the site specificity of water impacts, water assessment studies should rely on good data in the foreground system. This sets high challenges in terms of data collection of water use information for the type of product under study. The majority of water assessment case studies are for food products, which usually have a limited number of suppliers with a lower geographical spread. The assessment of all water impacts becomes much more complicated for products with several ingredients sourced from a global supply chain. The hand dishwashing product in this study including the perfume as a single ingredient (because perfumes are complex mixtures whose composition is highly confidential), has only 12 ingredients in the formula, but even in this case study a few assumptions were necessary. In fact, quite a number of hand dishwashing ingredients are not covered and were modeled with the generic ingredient "chemical organic," which is an important limitation. More complex formulations, such as laundry detergents, pose an even greater challenge to scientists when it comes to data collection.

Spatial differentiation at the inventory model

Within the goal and scope definition of a water impact assessment study, it is very important to pay sufficient attention to data representativeness, which may set challenges for screening studies. Building an accurate inventory model is equally important, as shown in the following 2 examples where the importance of data representativeness related to water turned out to be very complicated.

The baseline scenario assumed use of electrical boilers to heat the water used for the cleaning of the dishes in the use stage. As shown in the results analysis for the inventory methods (Figure 2), and even further in the midpoint (Figure 3) and endpoint results (Figure 5), indirect water use associated with electricity production is important. Therefore, the result could be very different for some water indicators if a gas boiler was used to heat the water. Figure 6 shows this effect for the blue water consumption midpoint method by Pfister et al. (2009). Similarly, consumers apply different practices to clean their dishes, as explained in the Materials and Methods section. Figure S1 shows the effect on the blue water consumption midpoint method by Pfister et al. (2009) for 10 plates cleaned by the DA or FS method. This shows that not only the use of representative electricity grid data is important, but also the need for a good understanding of the key influencing parameters in the inventory model. This is another element that sets high expectations for practitioners during the data collection phase in water assessment studies. The goal and scope definition of a water assessment study has to reflect the importance of these considerations.

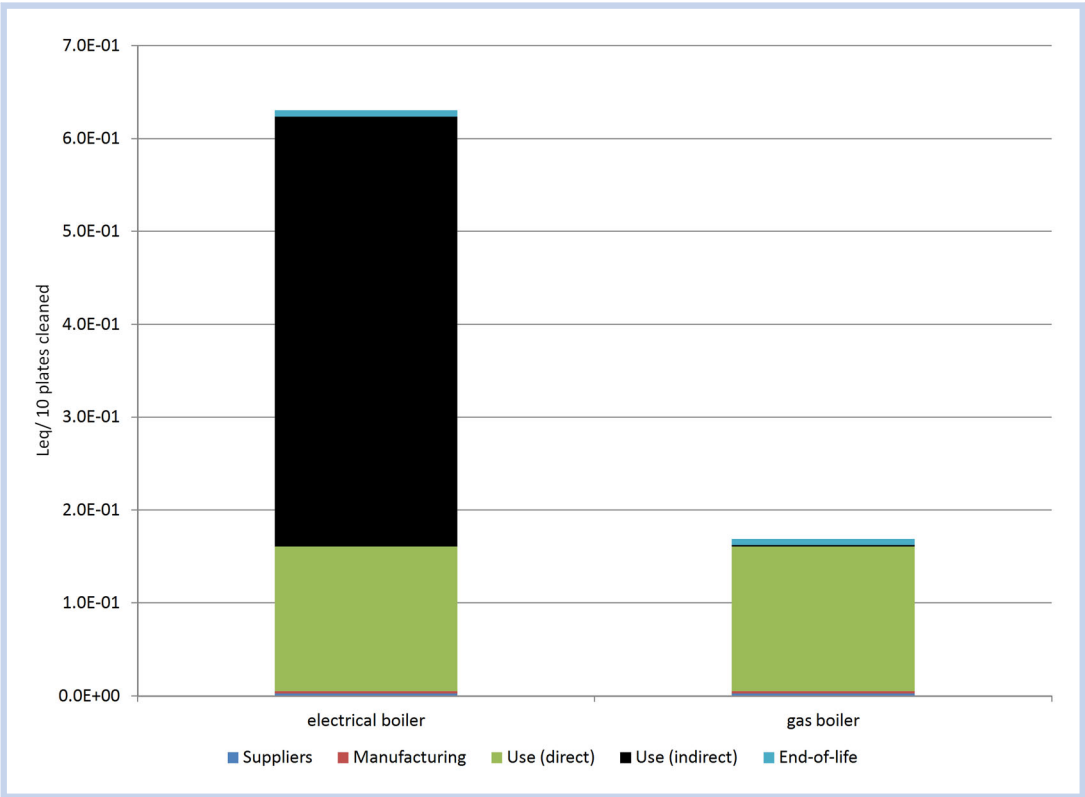

Figure 6. Effect of using an electrical or gas boiler to heat the water at the use stage. Results are split by their contribution across life cycle stages, using the Pfister midpoint method that assesses water stress from blue water consumption (Pfister et al. 2009).

Spatial differentiation at the impact assessment level

The previous section showed how results for some water indicators may change depending on the spatial differentiation in the inventory model. Another key aspect in a water impact assessment is whether or not spatial differentiation is applied.

To show the relative importance of spatial differentiation at the impact assessment level, a comparison is made between the Pfister and Van Zelm endpoint calculated for Germany and Spain (Table 4). The Van Zelm endpoint is not spatially differentiated (i.e., the characterization factor for Germany and Spain is the same), whereas the Pfister et al. (2009) method applies spatial differentiation (different WSI applied for both countries). The results are calculated for cleaning 10 plates in Spain and Germany, using the same product, and applying the same cleaning practice (DA) and heating appliance (electrical boiler). In the first step, the results for Spain are calculated. In the second step, inventory data applicable for Germany are included, whereas the characterization factor for Spain is applied. Compared to the results for Spain, this leads to a 24% lower result for Germany using the Pfister et al. (2009) method and a 9% lower result for Van Zelm et al. (2011). The difference in inventory is attributed to 1) a different electricity grid, 2) the sourcing of the tapwater, which in this case is about the same (Table 3), and 3) different sewage treatment infrastructure and municipal solid waste treatment. In the third step,

Table 4. Pfister and Van Zelm endpoint results for the cleaning of 10 plates using the same product, applying the same practice (direct application) and water heating method (electric boiler)[a]

	Pfister endpoint[b] (PDF × m^2 × y)	Van Zelm endpoint[c] (PDF × m^2 × y)
Spain (%)	6.32E-5 (100)	2.82E-4 (100)
Germany, spatial differentiation from inventory only (%)	4.77E-5 (76)	2.56E-4 (91)
Germany, spatial differentiation from inventory and impact assessment (%)	2.16E-5 (34)	2.56E-4 (91)

Results in parenthesis are relative to the respective endpoint results for Spain. The effect from spatial differentiation in Germany is shown by first applying the spatial differentiation at inventory level only (using the Spanish characterization factor) and second by applying the spatial differentiation at the impact assessment (using the German characterization factor).
[a]Tables for hand dishwashing water assessment study
[b]Pfister et al. 2009.
[c]Van Zelm et al. 2011.

the characterization factor for Germany is applied. Because the Van Zelm endpoint is not spatially differentiated, no further difference from the impact method is observed when compared to the results for Spain, whereas for the Pfister endpoint method a 42% incremental decrease is observed. Note that in this case the change from the spatial differentiation in the Pfister endpoint method is larger than the change from that of the spatial differentiation in the inventory. Also, both endpoint methods model the potential impact on the diversity of terrestrial plant species from different water uses (blue water consumption for Pfister, renewable groundwater consumption for Van Zelm). It therefore needs to be further evaluated as to whether or not the potential impact, as modeled by Van Zelm et al. (2011) requires further spatial differentiation at the impact assessment level.

Selecting the right spatial scale for the study

Interest in providing different stakeholders with life cycle thinking-based indicators is growing, as shown by initiatives such as the EU Product Environmental Footprint (Environmental Footprint of Products 2012), the French Grenelle legislation (Cros et al. 2010), and voluntary sustainability initiatives (The Sustainability Consortium). For LCA-based water indicators, a critical element in this context is the selection of the scale on which water indicators should be based. Providing consumer-relevant and actionable information on water is a challenge due to the high sensitivity of location specificity. As shown in the previous section, thermally polluted water use and blue water consumption are important when electrical boilers are used to heat the water used for cleaning dishes. Communicating the outcome of a Verones endpoint (Figure 5) or a Pfister midpoint result (Figure 6) to consumers would only be relevant for those who use electrical boilers.

Providing water-based information on a single product may strongly depend on where and how products are used (Figure S1 and Table 4). The fact that modeling LCA-based water impacts varies with the cause–effect chains makes the selection of a single water indicator that includes all relevant aspects unfeasible. At the moment, practitioners are restricted to evaluate all potential cause–effect chains to select indicators relevant for the system under study. Even for spatially differentiated methods, such as the Pfister midpoint method, there is a choice to report the result at country level or at grid cell level. For this hand dishwashing case study, where the product is manufactured in London, the absolute value for the manufacturing stage differs by a factor 3 for the Pfister midpoint result at country versus grid cell level (Figure S2). This is attributed to the high WSI for the grid cell where the plant is located (3 times higher than the country average). Therefore, when reporting results to the London plant manager, it makes more sense to report the indicator at grid cell level. However, this is not manageable when assessing water impacts from uses throughout Europe. In fact, an LCA-based water assessment can only be part of a comprehensive water assessment framework, including other tools such as environmental risk assessment and water risk assessment. This study shows an overall low contribution from manufacturing, even though the product is manufactured in a water scarce area. Water risk assessment-based tools, such as those developed by the World Wildlife Fund (Water Risk Filter 2011) or the World Business Council for Sustainable Development (Global Water Tool 2010) are very useful tools to assess water risks associated with product manufacturing. Environmental risk assessment (ERA) is a useful tool to manage quality aspects related to chemical emissions in water bodies. Chemical regulations in many regions rely on ERA to protect ecosystems, e.g., REACh (EC 2006).

Interpretation of results for hand dishwashing product

The various water assessment methods (inventory, midpoint, and endpoint) in this study clearly demonstrate the importance of the use stage contribution on the overall water impact of a hand dishwashing product. Depending on the water assessment method used, the improvement options suggested by these assessments may differ, because water use at the use stage may be driven by direct or indirect uses. Indirect water use is driven by electricity production (turbined water, thermal pollution, and blue water consumption). Therefore, improvements on these indicators can mainly be expected from using colder temperatures for cleaning dishes. This could be achieved by use of cold water-performing technologies and changing consumer habits, which in practice is very difficult and requires a strong case and intensive awareness raising campaigns. Note that cold water washing leads to energy savings with other LCA-associated benefits, such as decreasing impacts on global warming, fossil depletion, etc. It is important, however, to understand that the absolute scale of these effects depends on the type of boilers and may thus differ by country.

Reduction of direct water use at the product use stage has the greatest potential to improve water indicators as it is mainly driven by renewable groundwater use. The scale depends on the amount of tapwater being sourced from groundwater, which may vary by location. In practice, this means that consumers should be encouraged to minimize water consumption during the cleaning process. Saving water has an additional benefit as less water needs to be heated, thereby also reducing indirect water use and other LCA associated benefits from saving energy. This improvement option could be achieved by development of easy rinsing technologies and building consumer awareness.

CONCLUSIONS

A selection of water impact assessment methods developed for application in LCA studies was evaluated on a hand dishwashing product used in Spain or Germany. Selected water impact assessment methods included water inventory methods and water impact assessment methods at both the midpoint and endpoint level. "Traditional" LCA indicators with a scope on water, such as freshwater eutrophication, were not included in this study, as they start from emission data and are not spatially differentiated. The selected endpoint methods solely focused on the ecosystem quality area of protection.

Large differences (4 orders of magnitude) are observed between the water inventory category results with turbined water use (highest) and green water consumption (lowest). Similar differences are observed between midpoint results (4 orders for those with similar units) and the endpoint results (3 orders of magnitude). Without uncertainty information, these differences are associated with the different water use types (inventory categories) or cause–effect chains modeled (midpoints and endpoints). An LCA-based water assessment should therefore ideally consider all of them to understand the overall behavior of the system in terms of potential impacts associated with water use.

The use stage is the most important life cycle stage for the majority of the methods evaluated in this study. Interestingly, depending on the method, either the tapwater used in the cleaning process (direct use) or the water use in the background processes associated with electricity production (indirect use) were predominant. Because of different levels of spatial differentiation in the inventory and impact assessment model, the same product used under similar conditions can have a very different water indicator result. The sensitivity to spatial differentiation makes the goal and scope definition more important than ever in selecting representative water indicators.

Much development is still necessary to build reliable water databases for application in day-to-day LCA-based water assessment practice. Information for this study was derived from a beta version of the Quantis water database. This version consisted of specifically collected data for some unit processes, whereas for other unit processes data are from a literature review. The beta version did not include uncertainty from stochastic variability for the water elementary flows. This puts a limitation on the interpretation of results. Further development is necessary in terms of quantifying uncertainty in the inventory and impact assessment methods. Commercial LCA software needs to be further developed to allow practitioners to adopt new data and methods in their daily practice.

It is recommended that practitioners pay attention to the accuracy of their inventory models. Special attention should go to the dependence of key parameters in the inventory model on water use and the collection of representative foreground data on water. This places more weight on the data collection phase, because traditional approaches followed by LCA practitioners to overcome data gaps may be leading to incorrect results. This makes the iterative nature of LCA even more critical for LCA-based water assessments.

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SUPPLEMENTAL DATA

Figure S1. Effect of using different cleaning practices at the use stage (DA=Direct Application, FS=Full Sink). Results are split by their contribution across life cycle stages, using the Pfister midpoint method which assesses water stress from blue water consumption (Pfister et al, 2009).

Figure S2. Comparison of the Pfister midpoint result for the manufacturing stage at London plant using the method at two different spatial levels (country vs. grid cell).

Table S1: list of ingredients in Fairy hand dishwashing product.

Table S2: Ingoing parameters in inventory model for formulation, packaging, distribution and use of Fairy hand dishwashing product.

Table S3: End of life infrastructure data for Spain and Germany applied in the inventory model for Fairy hand dishwashing product.

Table S4: Inventory method results for the cleaning of 10 plates in Spain, using the Direct Application method and an electrical boiler.

Table S5: Midpoint method results for the cleaning of 10 plates in Spain, using the Direct Application method and an electrical boiler. All methods are in Leq except for the Swiss ecological scarcity method which is expressed in kUBP. The water scarcity indices are used for the Milà -i-Canals method: one based on a water use per capita ratio (WRPC) and one based on a water use per resource ratio (WUPR).

Table S6: Endpoint method results for the cleaning of 10 plates in Spain, using the Direct Application method and an electrical boiler.

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