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DÉVELOPPEMENT MÉTHODOLOGIQUE ET APPLICATION DU CONCEPT DE L'EMPREINTE EAU EN ACV

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DÉDICACE

« À ma mère, pour son support sans conditions,

Pour m'avoir montré que tout *problème a ses solutions,*

Et que les seules limites sont notre perception et notre imagination»

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RÉSUMÉ

Introduction

L'Analyse du Cycle de Vie (ACV) est une méthodologie qui quantifie les impacts environnementaux potentiels à des fins comparatives dans un contexte de prise de décision. Alors que les impacts environnementaux potentiels liés aux émissions de polluants à l'eau sont déjà caractérisés en ACV, les impacts potentiels d'une utilisation et d'une subséquente baisse de disponibilité d'eau ne sont pas encore complètement quantifiés. En effet, alors qu'une utilisation d'eau peut rendre la ressource non disponible par un déplacement (incluant évaporation) ou une baisse de la qualité, cette dernière n'est pas considérée dans les modèles existants. Une baisse de disponibilité d'eau pour les usagers humains peut potentiellement affecter la santé humaine si les usagers ne peuvent pas s'adapter pour subvenir à leurs besoins. Les impacts sur la santé humaine ont lieu selon deux chaînes cause-à-effet : les maladies liées à l'eau, lorsque les usagers domestiques subissent la baisse de disponibilité, et/ou la malnutrition, lorsque la baisse affecte les usagers qui produisent de la nourriture (manque d'eau pour l'irrigation ou les pêches/aquaculture). Cette thèse remplit donc les cinq objectifs principaux suivants: 1) fournir une méthode d'inventaire et 2) d'évaluation des impacts pour quantifier ces dommages sur la santé humaine dans un cadre ACV, 3) effectuer une comparaison du modèle avec les modèles existants, 4) fournir une application sur une étude de cas et 5) évaluer le modèle et quantifier l'incertitude.

Modèle d'inventaire

Pour quantifier la baisse de disponibilité de l'eau due à la dégradation, la qualité de l'eau entrante et sortante doit être captée par les flux d'inventaire. Dans le cadre de ce projet, une méthode d'inventaire est établie permettant de catégoriser la qualité de l'eau afin de pouvoir quantifier un changement de celle-ci et le changement de fonctionnalité correspondant. La fonctionnalité est définie par les différents usagers humains qui peuvent l'utiliser sans risques et sans traitements supplémentaires. Des catégories d'eau qui considèrent la qualité de l'eau sont d'abord définies par la source d'eau (surface, souterraine ou eau de pluie), des paramètres qualités et les utilisateurs pour qui chaque catégorie est fonctionnelle. Les besoins des utilisateurs sont identifiés par une liste de paramètres bio et physico-chimique et les seuils

maximaux possibles par contaminant pour chaque utilisateur. Ces seuils sont basés sur des normes internationales, recommandations et normes industrielles. Sur la base de la qualité et des sources d'eau, dix-sept catégories sont crées en regroupant les besoins des utilisateurs selon le niveau de contamination toxique ou microbienne que l'utilisateur peut tolérer (faible, moyen et élevé). Le processus résulte en huit catégories pour l'eau de surface, huit pour l'eau souterraine et une pour l'eau de pluie. Chaque catégorie est définie par jusqu'à 136 paramètres de qualité et permet d'établir les utilisateurs pour lesquels l'eau est fonctionnelle. Ces catégories d'eau permettent de qualifier les flux d'eau à l'étape d'inventaire, afin d'être utilisés avec un modèle d'évaluation des impacts potentiels associés à une baisse de fonctionnalité pour les utilisateurs humains, modèle qui fait l'objet de la prochaine étape du projet.

Modèle d'évaluation des impacts

Le modèle proposé prend en compte l'eau prélevée et rejetée, sa qualité et sa rareté afin d'évaluer la perte de fonctionnalité pour les autres usagers. Cette perte de fonctionnalité est ensuite multipliée avec deux paramètres : 1) une capacité d'adaptation, qui détermine dans quelle mesure l'eau non-disponible pourra être compensée par le biais de moyens financiers (ex : désalinisation), et 2) un facteur d'effet qui quantifie les impacts sur la santé humaine causés par la perte de fonctionnalité qui ne peut être compensée (i.e.: malnutrition ou maladies associées à un manque d'accès à l'eau).

Les impacts sur la santé humaine d'une utilisation d'eau, menant à une baisse de disponibilité d'eau pour les usages humains (domestiques, agricoles, ou pêches/aquaculture) sont présentés à l'échelle mondiale en résultats régionalisés et exprimés en années de vie perdue équivalentes. Un cadre pour l'évaluation des impacts causés par les moyens compensatoires dans les régions pouvant s'adapter est présenté en addendum.

Comparaison des modèles

L'évaluation du modèle développé dans ce projet a été effectuée à travers une comparaison systématique avec des modèles publiés dans la littérature et qui couvrent les mêmes chaînes cause-à-effet, notamment la rareté d'eau et les impacts d'un manque d'eau sur la santé humaine. Le but était de 1) identifier les choix de modélisation clés qui expliquent les différences principales entre les modèles, 2) quantifier l'importance des différences entre les modèles, incluant l'évaluation de l'incertitude associée et 3) discuter les choix méthodologiques principaux et fournir des recommandations pour orienter les développements méthodologiques futurs et les efforts d'harmonisation.

Les résultats ont permis d'identifier les choix de modélisation qui influencent significativement les indicateurs et qui doivent être analysés davantage et harmonisés, tels que l'échelle géographique à laquelle l'indicateur de rareté est calculé, la source de données d'entrées du modèle et la fonction qui décrit la rareté d'eau en fonction de la fraction d'eau disponible prélevée (WTA) ou consommée (CTA).

L'inclusion ou l'exclusion des impacts liés à la privation d'eau pour les usagers domestiques et l'inclusion ou l'exclusion du "trade effect" influencent les résultats d'impacts sur la santé humaine.

De plus, tant au niveau problèmes que dommages, la comparaison a démontré que de considérer une réduction de disponibilité due à une dégradation de l'eau affecte significativement les résultats. D'autres choix ont été analysés et sont moins significatifs pour la majorité des régions du monde. Des cartes sont fournies pour identifier les régions où ces choix sont pertinents.

Application sur une étude de cas

Le modèle développé est ensuite appliqué à une étude de cas sur l'empreinte eau d'un détergent à lessive, illustrant comment le modèle s'insère dans le concept de l'empreinte eau en complémentant les méthodes existantes adressant les différentes chaines cause-à-effet. En effet, l'intégration des différentes méthodes d'évaluation des impacts à l'intérieur d'une empreinte eau est toujours en cours et seulement quelques études de cas ont été publiées à ce jour illustrant le concept de façon exhaustive. Alors que les industries sont de plus en plus intéressées à évaluer leur empreinte eau au-delà d'un simple inventaire de volumes d'eau consommée, ils sont à la recherche de directives quand à l'application et l'interprétation des différentes méthodes disponibles.

Le modèle développé est également évalué et comparé à d'autres modèles adressant les mêmes chaines cause-à-effet. Une discussion sur l'applicabilité des différentes méthodes dans un contexte d'empreinte eau aborde les sujets tels que la définition des flux d'inventaire, la disponibilité des données, la régionalisation et l'inclusion des systèmes de traitements d'eau usée.

Le concept de l'empreinte eau tel que décrit dans la norme DIS ISO 14046 est illustré par l'étude de cas en incluant les catégories d'impacts liées à la disponibilité et à la dégradation. Au niveau problèmes, celles-ci incluent la rareté, le stress et les indicateurs de pollution tels que l'eutrophication, l'acidification et la toxicité. Au niveau dommages, les impacts sur la santé humaine et les écosystèmes sont évalué pour un manque et une dégradation de l'eau. Des analyses de sensibilité sont réalisées sur les choix de modélisation les plus sensibles, identifiés dans la comparaison mentionnée ci-haut.

Validation du modèle et incertitudes

Bien que les résultats du modèle ne puissent être validés directement avec des données réelles, une validation partielle de l'ordre de grandeur peut être effectuée en comparant les résultats que le modèle fournit si les impacts associés à toute l'eau consommée d'un pays sont évalués et comparées avec les données de l'Organisation Mondiale de la Santé décrivant les dommages sur la santé humaine liés à la malnutrition et au manque d'accès à l'eau. Ces données fournissent un seuil maximal, puisque ces impacts peuvent être causés par une utilisation d'eau ou d'autres causent, permettant d'identifier si les résultats du modèle se retrouvent dans un ordre de grandeur raisonnable. La comparaison montre que pour 75% et 71% des pays respectivement, les impacts évalués dus à la malnutrition et aux maladies liées à l'eau, sont en-dessous des données de l'OMS, tel que prédit. L'évaluation par le modèle à l'échelle mondiale donne une valeur du même ordre de grandeur que l'OMS pour les maladies liées à l'eau et un ordre de grandeur supérieur pour la malnutrition. Les incertitudes sont évaluées avec les données disponibles sur les paramètres d'entrée du modèle ou par des jugements d'experts, et elles sont comparées avec la variabilité spatiale du modèle à travers l'index UII (Uncertainty Increase Indicator), qui montre que l'incertitude intrinsèque du modèle est en général comparable ou supérieure à l'incertitude associée à variabilité spatiale à l'échelle du pays.

Conclusion

Cette thèse présente une méthode novatrice pour l'évaluation de l'inventaire et des impacts liés à une baisse de disponibilité causée par la consommation ou la dégradation de l'eau pour les usages humains en ACV. La méthode, qui représente une plus grande pertinence d'un point de vue logique en intégrant un plus grand nombre de paramètres et en offrant une plus grande complexité, a également démontré une différence dans les résultats obtenus. Le travail approfondi ensuite la compréhension du modèle et des autres modèles de rareté, stress et d'impacts sur la santé humaine en identifiant les choix de modélisations pertinents et les différences, permettant ainsi de quantifier l'incertitude du modèle et l'importance de ces choix dans un contexte régional spécifique, par l'utilisation de cartes mettant en évidence les régions où certaines analyses de sensibilité seraient pertinentes.

Décomposer les modèles existants et identifier les différences et similitudes, a permis d'identifier les principales composantes et ainsi supporter le développement éventuel d'une méthode consensuelle. Finalement, l'application à l'étude de cas a démontré que la méthode développée peut déjà être appliqué à un produit de détergent à lessive dans un contexte d'empreinte eau telle que présentée dans la norme ISO. La science et la disponibilité des données évoluent rapidement, mais les résultats obtenus permettent déjà aux entreprises d'identifier où dans le cycle de vie et dans le monde les impacts potentiels auront lieu.

En conclusion, malgré des incertitudes parfois élevées, un potentiel de surestimation des impacts dans certains pays, le besoin de données plus robustes et d'une meilleure opérationnalisation, ce travail contribue significativement à élargir les possibilités et l'exhaustivité de l'évaluation des impacts liés à l'utilisation de l'eau, et à la connaissance scientifique nécessaire pour appliquer, comprendre et développer davantage les modèles d'impacts.

ABSTRACT

Introduction

Life cycle assessment (LCA) is a methodology that quantifies potential environmental impacts for comparative purposes in a decision-making context. While potential environmental impacts from pollutant emissions into water are characterized in LCA, impacts from water unavailability are not yet fully quantified. While water use can make the resource unavailable to other users by displacement (including evaporation) or quality degradation, this latter is not yet considered in existing models. A reduction in water availability to human users can potentially affect human health if users cannot adapt to meet their needs. Health impacts may occur via two main impact pathways: water-related diseases, when domestic users are deprived of water, and malnutrition, when food-producing users are deprived of water (agriculture and aquaculture/fisheries). This thesis therefore meets these five main objectives: 1) an inventory and 2) impact model to quantify these potential damages to human health within an LCA framework, 3) a comparison of the model with other existing models, 4) an application on a case study and 5) an evaluation of the model and assessment of its uncertainty.

Inventory model

In order to assess a change in water quality and availability, the quality of the input and output inventory flows must be quantified. In the context of this project, an inventory method is established in order to categorize water quality and thus quantify a change, and the corresponding change in functionality. Functionality is defined by the different users by which the water can be used with no risks or additional treatments. Water categories that consider water quality are therefore defined by the source (surface, ground or rain), quality parameters and users for which the water is functional. A list of parameters was defined, and thresholds for these parameters were determined for each user. The thresholds were based on international standards, country regulations, recommendations and industry standards. Based on the quality and water sources, categories were created by grouping user requirements according to the level of microbial and toxic contamination that the user can tolerate (high, medium or low). Seventeen water categories were created: eight for surface water, eight for

groundwater and one for rainwater. Each category was defined according to 136 quality parameters and the users for which it can be of use. These categories allow qualifying the water flows at the inventory level in order to be used with a model assessing potential water use impacts caused by a loss of functionality for human users, which was the following step of this project.

Impact assessment model

The proposed model considers water that is withdrawn and released, its quality and scarcity in order to evaluate the loss of functionality for other users. This decrease in functionality is then multiplied by two parameters: 1) an adaptation capacity which determines how much of this decrease in water availability can be compensated through financial adaptation (ex: desalination), and 2) an effect factor to quantify the specific health impacts caused by the resulting loss that cannot be compensated for (i.e.: water-related diseases and/or malnutrition). World-wide regionalized results are presented for impacts on human health expressed in disability-adjusted life years (DALY). A framework for impact assessment caused by the use of backup technologies in regions able to adapt is presented in addendum.

Model comparison

The model comparison that followed was performed on methods that describe similar impact pathways, namely water scarcity and human health impacts from water deprivation. The aim was to (i) identify the key relevant modeling choices that explain the main differences between characterization models leading to the same impact indicators; (ii) quantify the significance of the differences between methods, including the assessment of model uncertainty and (iii) discuss the main methodological choices and provide recommendations to guide method development and harmonization efforts.

The results determined the modeling choices that significantly influence the indicators and should be further analyzed and harmonized, such as the regional scale at which the scarcity indicator is calculated, the sources of underlying input data and the function adopted to describe the relationship between scarcity and the withdrawal-to-availability (WTA) or consumption-to-availability (CTA) ratios. The inclusion or exclusion of impacts from domestic user deprivation and the inclusion or exclusion of trade effects boteh influence human health impacts. At both midpoint and endpoint, the comparison showed that considering reduced water availability due to degradation in water quality, in addition to a reduction in water quantity, greatly influences results. Other choices are less significant in most regions of the world. Maps are provided to identify the regions in which such choices are relevant.

Case study application

The model developed is then applied to a case study on the water footprint of a laundry detergent, illustrating how the model can be integrated in the water footprint concept while complementing existing methods addressing different impact pathways. Indeed, the integration of different water impact assessment methods within a water footprint concept is still ongoing and a limited number of case studies have been published presenting a comprehensive study of all water-related impacts. Although industries are increasingly interested in assessing their water footprint beyond a simple inventory assessment, they often lack guidance regarding the applicability and interpretation of the different methods available.

The model is also evaluated and compared to other models addressing impact pathways. A discussion on their applicability covers issues such as inventory flow definition, data availability, regionalization and inclusion of waste water treatment systems. Method-specific discussion covers the use of interim ecotoxicity factors, the interaction of scarcity and stress assessments and the limits of such methods and the geographic coverage and availability of impact assessment methods. Lastly, possible double counting, databases, software, data quality and integration of a water footprint within an LCA are discussed.

The concept of water footprinting as defined by the forthcoming ISO Draft Standard, is illustrated through the case study of a load of laundry using water availability and water degradation impact categories. At the midpoint it covers scarcity, stress and pollution indicators such as eutrophication, acidification, human and eco-toxicity. At the endpoint, impacts on human health and ecosystems are covered for water deprivation and degradation. Sensitivity analyses are performed on the most sensitive modeling choices identified in the aforementioned model comparison.

Model validation and uncertainty assessment

Although the model results cannot be directly validated with actual data, a partial validation of the order of magnitude can be performed by comparing the results obtained by characterizing the entire consumed water volume of a country with the model with the World Health Organization (WHO) data for water-related diseases and malnutrition. This data provide an upper threshold for the model results, since these health damages can be caused by water consumption or other factors, and hence allow a validation of the order of magnitude of the model results. The comparison showed that for 75% and 71% of the countries respectively, impacts obtained from the model for malnutrition and water-related diseases are below the WHO data threshold, as predicted. The world-wide assessment results in values in the same order of magnitude as WHO data for water-related diseases, and one order of magnitude higher than WHO for malnutrition.

Uncertainties are assessed based on available data for the input parameters of the model or based on expert judgments, and they are compared with spatial variability within the UII (Uncertainty Increase Indicator), which shows that the model uncertainty is generally comparable or higher than the uncertainty associated with spatial variability at the country scale.

Conclusion

This work presents a novel inventory and impact assessment approach for evaluating impacts from water consumption and water degradation on human health in LCA. The model, which integrates several new relevant parameters and presents a higher complexity level, also showed a difference in the results obtained. It then deepens the understanding of the model and other existing models on scarcity, stress and human health impact by identifying the key relevant modeling choices and differences, making it possible to quantify model uncertainty and the significance of these choices in a specific regional context. Maps of regions where these specific choices are of importance were generated to guide practitioners in identifying locations relevant for specific sensitivity analyses in water footprint studies. Deconstructing the existing models and highlighting the differences and similarities has helped to determine building blocks to support the development of an eventual consensual method.

Finally, the case study application shows that the model developed can already be applied to a laundry detergent product within a water footprint, as proposed in the ISO draft standard. The science and the data availability are rapidly evolving, but the results obtained with present methods already enable companies to map where in the life cycle and in the world impacts might occur.

In conclusion, despite sometimes high uncertainties, a potential overestimation of impacts in certain countries, the need for more robust data and better operationalisation, this work contributed significantly to the comprehensiveness and possibilities of water use impact assessment, and to the scientific knowledge necessary to apply, understand and further develop impact models.

TABLE DES MATIÈRES

LISTE DES TABLES

LISTE DES FIGURES

LISTE DES SIGLES ET ABRÉVIATIONS

AC	Adaptation Capacity
ACV	Analyse du Cycle de Vie
BWS	Blue Water Scarcity
CF/FC	Characterisation Factor / Facteur de Charactérisation
CTA	Consumption-To-Availability
CU	Consumptive Use
DALY	Disabled-Adjusted Life Years
DAU	Distribution of Affected Users
DBO/BOD	Demande Biologique en Oxygène / Biological Oxygen Demand
DCO/COD	Demande Chimique en Oxygène / Chemical Oxygen Demand
EF	Effect Factor
ES	Ecosystems
FU	Funtional unit
GNI	Gross National Income
GWR	Ground Water Renewable (resource)
HH	Human Health
ISO	International Standards Organisation
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
MCDA	Multi-Criteria Decision Analysis
MDC	Mean Difference Coefficient

MES	Matières en Suspension
PDF	Potentially Disappeared Fraction
RCC	Rank Correlation Coefficient
SEE	Socio-Economic and Effect
SEP	Socio-Economic Parameter
TDS	Total dissolved Solids
WFN	Water Footprint Network
WSI	Water Scarcity Index/Water Stress Index
WTA	Withdrawal-To-Availability
WULCA	Water Use in Life Cycle Assessment
WWTP	Waste Water Treatment Plant

LISTE DES ANNEXES

INTRODUCTION

Alors que l'importance de préserver la ressource eau prend de plus en plus d'importance au cœur des débats et initiatives environnementales, la méthode d'Analyse du Cycle de Vie (ACV), elle, se perfectionne en parallèle, fournissant ainsi un outil d'aide à la décision de plus en plus robuste en évaluant les impacts potentiels liés à la fabrication de produits ou services. La ressource eau n'a cependant commencé à être intégrée à la méthodologie d'ACV et à la notion d'empreinte environnementale que dans les dernières années et le processus n'est pas encore terminé (Koehler 2008; Kounina et al. 2013). Des groupes de travail internationaux tel que ISO et UNEP/SETAC (ISO 14046 2013; UNEP-SETAC Life Cycle Initiative 2013) sont encore en discussion afin d'évaluer et proposer des directives consensuelles sur l'intégration des impacts liés à l'eau en ACV.

Le projet de doctorat présenté ici contribue directement aux discussions et réflexions menées dans le cadre de ces deux groupes internationaux. Il s'est intégré dans les livrables scientifiques du groupe de travail WULCA (Water Use in LCA, de l'UNEP-SETAC Life Cycle Initiative) qui vise à promouvoir et encadrer le développement méthodologique et les applications industrielles pour l'évaluation des impacts liés à l'eau en ACV. En tant que déléguée représentante du Canada dans le groupe de travail de la prochaine norme ISO sur l'empreinte eau (DIS 14046) visant à fournir un cadre standard permettant une évaluation pertinente d'une empreinte eau, j'ai apporté un support scientifique au développement de la norme et au sein du groupe de travail, et mon exposition à ce groupe formé d'industriels et de groupes sectoriels et (para)gouvernementaux, m'a permis de mieux comprendre les besoins existants et ainsi orienter mon travail de recherche.

Ce projet de doctorat fut entamé à l'automne 2009 suite à un passage direct de la maîtrise, dans le cadre duquel ce projet a vu le jour. Il a pour but de proposer un modèle décrivant et quantifiant l'une des chaînes cause-à-effet du cadre méthodologique proposé par Bayart et al. (2010a) : celle liée à un manque d'eau pour les usages humains. Ce sont ces impacts sur la santé humaine qui sont quantifiés dans le modèle développé au cœur de ce doctorat, qui propose d'abord une méthode d'inventaire permettant de décrire les flux d'eau entrant et sortant d'un système par leur source et leur qualité en fonction des usages pour lesquels ils sont fonctionnels (chapitre 3), avant de quantifier les impacts sur la santé humaine d'une privation d'eau pour les usagers domestiques, agricoles et aquacoles (incl. pêches), liés soit à une consommation ou à une dégradation d'eau (chapitre 4). En supplément, un modèle conséquentiel permet d'évaluer les impacts indirects de l'utilisation de l'eau lorsqu'elle a lieu dans une région où les ressources économiques permettent aux usagers privés d'eau de s'adapter (addendum, chapitre 4). Le modèle développé est ensuite évalué et comparé aux autres modèles existants, afin d'en approfondir la compréhension des différents modèles, leurs choix de modélisation, les sources de données utilisées et l'impact de ces choix sur les résultats, ouvrant ainsi la porte à un travail d'harmonisation (chapitre 5). Les différents modèles sont ensuite appliqués à une étude de cas qui permet d'illustrer leurs interactions et leurs différences, en élargissant la portée de l'étude pour inclure les méthodes nécessaires à l'illustration d'une empreinte eau tel que décrit par le standard ISO DIS 14046 (chapitre 6). Finalement, le modèle est validé partiellement en comparant les résultats avec des données réelles, et les incertitudes son quantifiés et mises en perspectives avec la variabilité spatiale (chapitre 7). Une discussion suit ensuite sur les forces et faiblesses et les pistes à suivre pour de futurs travaux sont identifiées (chapitre 8).

CHAPITRE 1 REVUE CRITIQUE

1.1 L'eau

1.1.1 L'eau : une ressource unique

L'eau est la seule ressource qui est synonyme de vie, humaine ou non, et qui n'a aucun équivalent. L'eau relie l'atmosphère, les terres et les océans à travers son cycle global, en circulant à travers chacun de ces domaines, changeant de phase entre solide, liquide et gazeuse; supportant la biosphère et les humains, érodant les continents et nourrissant les zones côtières. L'eau sert également de système de transport pour les substances biochimiques, incluant les substances toxiques, qui éventuellement fraient leur chemin depuis leur source sur les continents jusqu'aux océans. Après avoir perdu leur contenu en eau à l'atmosphère par l'évaporation et l'évapotranspiration, les nappes et cours d'eau de surfaces sont rechargés par les précipitations.

Particulièrement pour les humains, l'eau est la première ressource source de vie. Sa disponibilité est une composante essentielle dans le développement socio-économique et la réduction de la pauvreté. Aujourd'hui, un nombre important de facteurs ont un impact à la fois sur cette ressource et sur sa gestion, incluant la pauvreté, la malnutrition, les impacts dramatiques des changements démographiques, de l'urbanisation croissante, les effets de la globalisation et les récentes manifestations des changements climatiques. Tous ces facteurs affectent le secteur de l'eau de façons de plus en plus complexes (World Water Assessment Program 2006).

La quantité d'eau douce dans le monde est pratiquement constante, environ 35 millions de km^3 desquels 24 millions sont contenus dans les glaciers, les sols gelés ou les neiges éternelles, 10,4 millions dans les eaux souterraines et les sols et 0,09 km^3 dans les lacs et marécages (Shady 2008). Cependant, alors qu'il est vrai que seulement une très petite fraction de l'eau douce sur terre est disponible pour les usages humains, elle est présente en quantité suffisante pour remplir les besoins humains domestiques, industriels et agricoles pour tous (Rijsberman and Mohammed 2003).

1.1.2 L'eau et les humains

L'humanité s'est embarquée dans un projet d'ingénierie global et écologique immense, avec aucune ou très peu de connaissance sur les conséquences. En l'espace de très peu de temps, d'un point de vue planétaire, nous avons cherché à reconcevoir et imposer un nouvel ordre sur les systèmes de la planète ayant évolués pendant des millions d'années (World Water Assessment Program 2006). D'un point de vue de l'eau, obtenir une source d'eau fiable et sécuritaire pour la santé, les besoins alimentaires, industriels, et pour la production énergétique a grandement changé l'ordre naturel de plusieurs rivières dans le monde. Maintenant, plusieurs régions du monde font face à des crises concernant l'eau douce. La distribution inégale des ressources hydriques dans le temps et l'espace et la façon dont les activités humaines affectent cette distribution aujourd'hui sont des sources importantes de crises liées à l'eau dans plusieurs régions du monde (World Water Assessment Program 2009). De plus, les changements climatiques sont superposés à un système hydrologique déjà complexe, rendant son influence difficile à isoler. Cependant, le *World Water Vision* a conclu que la crise liée à l'eau n'en était pas une causée par un manque, mais était plutôt une crise de gestion (Cosgrove and Rijsberman 2000) tout comme le *Global Water Partnership* qui appelle cette crise une de "gouvernance" (Rijsberman and Mohammed 2003).

Sandra Postel a tenté d'évaluer l'ampleur du déficit hydrique du monde, c'est-à-dire la quantité d'eau pompée en surplus par rapport à la quantité d'eau renouvelable disponible localement. Elle a conclut, en utilisant des données pour l'Inde, la Chine, le Moyen-Orient, l'Afrique du Nord et les États-Unis que nous prélevons chaque année 160 milliards de m^3 d'eau en trop dans ces régions. Puisque chaque tonne de grain nécessite mille mètres cubes d'eau, 160 millions de tonnes de grains sont produits chaque année avec de l'eau prélevée en trop (Kumar and Singh 2005).

1.1.3 Les problèmes liés à l'eau

Les conséquences de cette crise se font voir à plusieurs niveaux. Dans le monde, plus de 2 milliards de personnes sont infectées par des schistosomes et des helminthes et 300 millions de ceux-ci souffrent de sérieuses maladies suite à ces infections directement associées à la consommation d'eau non potable. Au Bangladesh seulement, près de 35 millions de personnes sont quotidiennement exposées à des niveaux d'arsenic présent naturellement dans leur eau potable qui menace leur santé et réduira leur espérance de vie. Tous les jours, les maladies entériques causent près de 5500 décès, principalement chez des enfants de moins de cinq ans (World Water Assessment Program 2003). La malnutrition, qui cause près d'un demi-million de décès chaque année, est souvent une conséquence directe soit de maladies entériques qui préviennent une absorption adéquate des nutriments, soit d'un manque de nourriture disponible à un prix abordable, souvent dû à des pénuries d'eau pour l'agriculture (FAO 2009a). Dans les pays plus développés qui doivent faire face à une pénurie d'eau, les besoins industriels et domestiques doivent souvent être comblés par le dessalement d'eau de mer, à fort coût environnementale, énergétique et économique.

La compétition pour l'eau existe déjà et va augmenter avec la demande dans pratiquement tous les pays. En 2030, 47% de la population vivra dans une région de stress hydrique élevé. La gestion de l'eau autour du monde est déficiente en performance, efficacité et équité. L'efficacité de l'utilisation de l'eau, l'abattement de la pollution et l'implantation de mesures environnementale sont insuffisantes dans la plupart des secteurs. L'accès à des services de base pour l'eau potable, l'hygiène et la production alimentaire demeure insuffisant dans les régions en développement et plus de 5 milliards de personnes, 67% de la population, n'auront toujours pas accès à des installations sanitaires adéquates en 2030 (World Water Assessment Program 2009).

Bien que les discussions sur les prochaines guerres concernant l'eau se multiplient, et que la compétition de la demande et les conflits augmentent, il y a cependant peu de preuves historiques que l'eau comme tel ait mené à un conflit international, ou qu'une guerre pour l'eau serait sensée d'un point de vue stratégique, hydrographique ou économique. Un conflit existant peut souvent être exacerbé si l'eau est un enjeu supplémentaire, mais au niveau international, l'eau semble davantage être une raison de collaboration transfrontalière plutôt qu'une raison de guerre, prévenant souvent plutôt que causant, des conflits. L'exemple de la résolution du conflit entre le Mexique et les États-Unis sur le partage des coûts et bénéfices liés aux mesures de conservation de l'eau des rivières Grande et Colorado en est un bon exemple (World Water Assessment Program 2009).

1.1.4 Les sources d'espoir

Considérer la ressource eau de façon différente est nécessaire si l'on veut atteindre notre triple objectif de sécurité alimentaire, réduction de la pauvreté et conservation des écosystèmes. Bien qu'il soit possible de produire la nourriture dont nous avons et aurons besoin avec l'eau disponible, il est probable que la production alimentaire et les tendances environnementales d'aujourd'hui, si elles se poursuivent, mènent à des crises supplémentaires dans plusieurs parties du monde. Alors que l'augmentation des prélèvements d'eau pour l'irrigation dans les pays en développement est favorable à la croissance économique et à la réduction de la pauvreté, elle est souvent mauvaise pour l'environnement. D'un point de vue global, le potentiel de l'agriculture à l'eau de pluie est assez grand pour subvenir aux besoins présents et futurs si la productivité est augmentée. Seulement en agissant pour augmenter l'efficacité de l'eau en agriculture arriverons nous a surmonter les défis importants que l'humanité affrontera dans les prochaines 50 années (Comprehensive Assessment of Water Management in Agriculture 2007). Le *Comprehensive Assessment of Water Management in Agriculture*, qui a rassemblé pendant cinq années de travail plus de 700 scientifiques à travers le monde, a émis un message fort et urgent : les problèmes vont s'intensifier s'ils ne sont pas adressés, et ce, dès maintenant (Comprehensive Assessment of Water Management in Agriculture 2007).

1.1.4.1 L'eau virtuelle

L'eau virtuelle définit toute l'eau qui est nécessaire à la fabrication d'un produit (Allan 1996). Ce concept est plus souvent employé dans le cadre du commerce de biens nécessitant une grande quantité d'eau : céréales, coton, etc. Le principe de l'eau virtuelle suggère qu'une augmentation stratégique du commerce alimentaire en fonction de la disponibilité de l'eau pourrait atténuer la rareté de l'eau et réduire la dégradation environnementale. Au lieu de miser sur une indépendance alimentaire, les pays présentant une pénurie d'eau devraient importer de la nourriture des pays qui sont plus riches en eau. De plus, la production alimentaire dans les régions riches en eau est ironiquement souvent plus efficace en termes de consommation d'eau par kg produit que celle dans les pays où l'eau est plus rare (Hoekstra and Chapagain 2007). Ainsi, en important des biens agricoles, une nation "économise" la quantité d'eau dont elle aurait eue besoin pour produire ce bien localement. En d'autres mots : elle exporte sa consommation d'eau. Par exemple, l'Égypte, un pays où la rareté de l'eau est importante, a importé 8 million de tonnes métriques de grains des États-Unis en l'an 2000. Pour produire cette quantité, elle aurait eu besoin de 8.5 km³ d'eau pour l'irrigation (à titre de référence, le lac Nasser fournit annuellement 55.6 km³)(European Environment Agency 2009). Mais les pays pauvres dépendent, en grande partie, de leur agriculture nationale, et le pouvoir d'achat requis pour couvrir les besoins alimentaires à partir du marché mondial est souvent bas. Alors que ces pays tentent d'atteindre une sécurité alimentaire, ils sont méfiants de dépendre des importations pour assurer leur besoins alimentaires de base et une certaine autonomie est souvent au cœur de leur politique. Malgré les problèmes liés l'eau, le développement de la ressource par de nouvelle sources semble une option plus sécuritaire pour atteindre la production nécessaire et stimuler l'économie.

De plus, d'autres facteurs interviennent, tel que décrit par Kumar et collaborateurs (Kumar and Singh 2005) qui affirment que la dynamique des flux d'eau virtuelle est davantage contrôlée par l'accès à des terres cultivables que par l'accès à l'eau. Ces propos sont renchéris par Chapagain et collaborateurs (Chapagain and Hoekstra 2008) qui expliquent que la raison pour laquelle les dynamiques sont plus diffuses qu'on ne s'y attendrait par rapport aux échanges d'eau virtuelle est que sous le régime des marchés actuels, l'eau est rarement le facteur dominant déterminant les importations et exportations de biens à haute consommation d'eau. D'autres facteurs tels que la disponibilité des terres et de la main d'œuvres jouent un rôle important, tout comme les politiques nationales, les subventions d'exportation et les barrières d'échanges internationaux. Leurs propos sont supportés par l'analyse de Wichelns (2010) et Yang et collaborateurs (Yang and Zehnder 2007) qui critiquent le fait que l'eau virtuelle soit considérée comme une solution aux problèmes de rareté d'eau.

1.1.4.2 L'utilisation agricole

Hamdy et collaborateurs (Hamdy et al. 2003) identifient que la solution afin de satisfaire les besoins grandissants des secteurs municipaux et industriels doit venir d'économies du secteur agricole. Non seulement parce que ce secteur représente la plus grande partie de l'utilisation totale, mais également parce qu'il comporte un important potentiel d'amélioration de l'efficacité. Dans un système d'irrigation traditionnel, aussi peu que 45% de l'eau utilisée est en fait absorbée par les plantes, avec plus de 50% de pertes. Ainsi, pour une situation typique où 80% de l'eau utilisée dans une région l'est pour fin d'irrigation, une augmentation de l'efficacité d'irrigation de 10% fournirait 50% d'eau additionnelle pour les secteurs industriel et municipal. Ceci démontre bien le potentiel d'économie d'eau en agriculture. Également plusieurs pays incluent la réutilisation d'eaux usées pour les usages agricoles dans leur planification hydrique, ce qui est déjà largement utilisé dans les régions arides de la méditerranée, libérant ainsi l'eau de meilleure qualité pour des usages potables, pratique qui devrait se répandre de plus en plus dans les régions arides et semi-arides.

1.1.4.3 La gestion de la demande

Le 2^{e} rapport des Nations Unies sur l'évaluation de la ressource eau dans le monde (World Water Assessment Program 2006) identifie le fait qu'une approche holistique et intégrée de la gestion de l'eau est nécessaire, en réponse à un système secteur-par-secteur (irrigation, municipal, énergie, etc.) souvent critiqué. Une telle approche intégrée favoriserait la coopération trans-sectorielle et également la gestion et le développement de l'utilisation des terres, de l'eau et des autres ressources, maximisant les bénéfices sociaux et économiques de façon équitable, sans compromettre la durabilité.

Alors que traditionnellement la réponse au stress hydrique est d'augmenter l'offre en développant des nouvelles sources et en augmentant les prélèvements aux sources existantes, les solutions portent maintenant davantage sur des approches efficaces et équitables, intégrant davantage la gestion de la demande, en intégrant une utilisation plus efficace de l'eau, en améliorant la balance entre la disponibilité actuelle et la demande, et en réduisant les usages excessifs. Des outils permettant d'évaluer la demande, les opportunités de réduction de consommation d'eau et les meilleures décisions à prendre dans un contexte intégrant la ressource en eau dans les diverses problématiques environnementales, sociales et économiques sont donc indispensables pour une meilleure gestion de la ressource. L'Analyse du Cycle de vie est un outil qui offre ce potentiel.

1.1.5 Rareté

L'un des plus grands défis concernant la quantification des impacts liés à l'eau est la définition de la rareté d'eau, ou plutôt l'absence de consensus à son sujet. Rijsberman (2006) défini une personne comme *water insecure* quand cette personne n'a pas accès à suffisamment d'eau pour subvenir à ses besoin, et une région présentant une rareté d'eau comme étant une région où plusieurs personnes sont *water insecure*. Cependant, l'accès à l'eau, les besoins d'une personne, et la région spatiotemporelle sont autant de facteurs qui influent sur cette approche et qui ne sont pas définis. Les indicateurs proposés varient du très simple au très complexe, selon le nombre de paramètres pris en compte. La totalité des indicateurs sont basés sur un ratio reliant l'eau disponible à la population ou aux prélèvements humains. Par exemple, le plus simple et répandu est le *Falkenmark indicator* (Falkenmark et al. 1989) qui représente l'eau disponible par capita, proposant 1700 m³ d'eau renouvelable par personne par an comme étant le seuil d'un stress hydrique, et 1000 m³ et 500 m³ pour une rareté et rareté absolue respectivement. Alors que cet indicateur a l'avantage d'être facile à comprendre et que les données sont facilement accessibles, les variations saisonnières, infrastructures et les besoins spécifiques d'une population ne sont pas pris en compte, et l'échelle du pays semble non pertinente pour de grands pays comme la Chine ou les États-Unis. Ohlsson l'a alors adapté en y intégrant le UNDP *Human Development Index* pour créer l'index de stress hydrique social (Ohlsson 2000).

Les indicateurs qui suivirent tentèrent d'évaluer de façon plus représentative le besoin de la population, d'abord par Shiklomanov (1997) qui évalua la demande de la population pour les secteurs agricole, industriel et domestique, puis amélioré par Raskin et collaborateurs (1997) qui remplacèrent la demande par les prélèvements actuels, dans l'intention de représenter une rareté plus objective qu'une notion théorique basée sur la demande. Ils ont aussi proposé des seuils de 20% et 40% pour définir la rareté et la rareté extrême respectivement. Cette définition fut également utilisée par Alcamo et collaborateurs (1997) dans leur *ratio de criticalité* évalué à l'aide de leur modèle WaterGap et par Vorosmarty et collaborateurs (2000a) qui utilisent un modèle climatique pour l'évaluer. Les limites de ce ratio de criticalité sont 1- ni les infrastructures ni les variabilités saisonnières ne sont prises en compte, 2- les prélèvements ne représentent pas la consommation d'eau et une fraction de l'eau prélevée peut être à nouveau disponible après usage, 3- la capacité d'adaptation d'une population à une rareté d'eau n'est pas prise en compte et 4- l'eau de surface et l'eau souterraine ne sont pas distinguées.

Le *International Water Management Institute* (IWMI) a tenté de résoudre une partie de ces limites (Seckler et al. 1998) en 1- prenant en compte l'infrastructure par l'entremise de la fraction de l'eau renouvelable disponible pour les usages humains, 2- ne considérant que l'eau évapotranspirée et 3- évaluant la capacité future du pays à s'adapter par le développement d'infrastructure et l'amélioration de l'efficacité d'irrigation par la mise en place de politiques pour la période 2000-2025. Le modèle résulte en la séparation des pays présentant une rareté d'eau pour des raisons physiques ou économiques, mais le résultat est non-numérique, catégorisant les pays qualitativement en pays présentant une « physical water stress » ou une « economical water stress ».

Le plus complexe des index est probablement le *Water Poverty Index* développé par Sullivan et collaborateurs (2002) qui tente de refléter à la fois la disponibilité physique de l'eau, le degré avec lequel les humains sont desservis par cette eau et le maintient des milieux écologiques. L'indicateur regroupe cinq aspects : l'accès à l'eau, sa qualité, sa quantité et sa variabilité, les utilisations de l'eau pour les utilisations agricoles, domestiques et industrielles, la capacité de gestion de l'eau et les aspects environnementaux. Alors que celui-ci adresse la majorité des limites présentées par les précédents, plusieurs choix de pondération doivent être faits et des données difficilement mesurables sont utilisées (i.e. temps passé à la collecte de l'eau domestique). La nature davantage « communautaire » de l'index justifie son objectif d'évaluation à une échelle plus locale que nationale.

Plus récemment, Pfister et collaborateurs (2009) ont proposé une méthode d'évaluation des impacts liés à l'utilisation de l'eau en ACV dans lequel ils présentent également un index de rareté qu'ils nomment le *Water Scarcity Index* (WSI). Celui-ci est basé sur le ratio de criticalité discuté plus haut, donc basé sur les prélèvements et non la consommation d'eau, mais il intègre un paramètre de variation saisonnière basé sur des données climatiques. Aussi, Döll et collaborateurs (2009) dans un contexte d'évaluation des impacts des changements climatiques sur la ressource souterraine ont proposé un paramètre de rareté d'eau qui prend en compte la consommation d'eau (CU), et non les prélèvements, ainsi que les variations saisonnières par un paramètre de faible débit statistique (Q90). Celui-ci représente le débit d'eau renouvelable le plus faible pour 9 mois sur 10, résultant en une valeur plus faible que la moyenne et permettant ainsi de mieux représenter la rareté d'eau dans les régions semi-arides. L'avantage du ratio de la consommation sur le Q90 (CU/Q90) est son interprétation physique plus représentative que le ratio de criticalité : une valeur de un ou plus pour le CU/Q90 implique que toute l'eau disponible dans un bassin versant est consommée 10% du temps. Le désavantage principal est que celui-ci dépend de données de consommation qui sont moins certaines et moins disponibles que celles des prélèvements pour les secteurs domestique et industriel (Alcamo et al. 2007).

Finalement, en 2012, le Water Footprint Network a publié un indice de *Blue Water Scarcity* (Hoekstra et al. 2012), qui évalue un ratio d'eau consommée – le Blue Water Footprint – sur l'eau disponible, en réservant 80% pour les besoins des écosystèmes. Les résultats sont fournis en indexes mensuels, et pour les grands bassins versants du monde seulement, excluant ainsi de nombreuses régions.

1.2 L'analyse du cycle de vie

1.2.1 Méthode

L'analyse du Cycle de Vie (ACV) est une méthode scientifique, obéissant à des normes ISO (ISO 14040 2006), permettant d'évaluer et de quantifier les impacts environnementaux potentiels générés par un produit ou un service, pendant tout son cycle de vie. Celui-ci inclut les étapes suivantes : l'extraction et la transformation des matières premières, la fabrication, l'emballage et la distribution, l'utilisation et la fin de vie du produit. La méthode est divisée en quatre étapes itératives, tel que présenté dans la figure 1-1. Le projet proposé ici concerne particulièrement les étapes d'inventaire et d'évaluation des impacts.

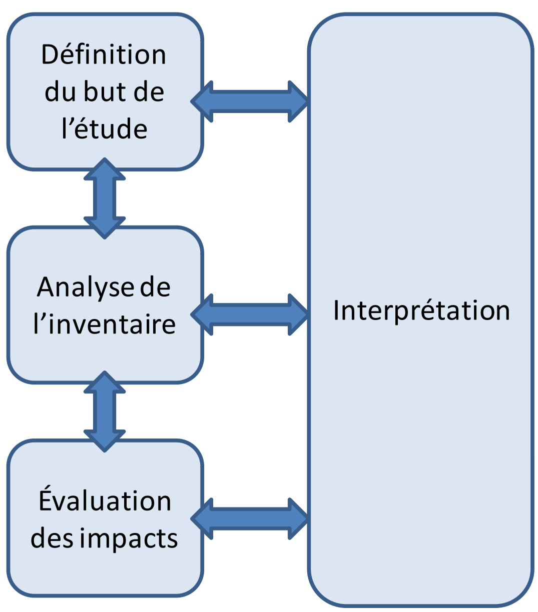

Figure 1-1 Étapes de l'Analyse du Cycle de Vie

1.2.1.1 But de l'étude

La définition du champ de l'étude est probablement l'étape la plus importante, puisque c'est ici que les frontières du système, l'objectif de l'étude, son application et l'audience visée sont définis. L'unité fonctionnelle, la mesure de la fonction du produit ou service sur laquelle seront basés tous les calculs, est également définie à cette étape.

1.2.1.2 Analyse de l'inventaire

L'analyse de l'inventaire consiste à recueillir les données sur tous les intrants et extrants du système, représentant les ressources extraites de l'écosphère vers la technosphère (faisant partie du système économique), et les émissions depuis la technosphère vers les différents milieux de l'écosphères (eau, air, sol). Cet inventaire est recueilli pour la valeur de l'unité fonctionnelle, pour chaque étape du Cycle de vie du produit ou service évalué. Des bases de données favorisent le travail de collecte de données en fournissant des données d'inventaire pour des milliers de processus industriels. La plus exhaustive est ecoinvent (Frischknecht and Jungbluth 2004).

1.2.1.3 Évaluation des impacts

Le but de l'analyse d'impacts est de comprendre et évaluer l'importance des impacts potentiellement générés sur l'environnement. L'analyse d'impacts caractérise l'inventaire recueilli en impacts en passant par différents modèles permettent de traduire et de regrouper ces données en différentes catégories d'impacts, d'abord à un niveau « Problèmes » ou « midpoints », situé au milieu de la chaîne cause-à-effet, puis à la toute fin de celle-ci, à un niveau « Dommages », ou « endpoint », tel que présenté par la figure 1-2.

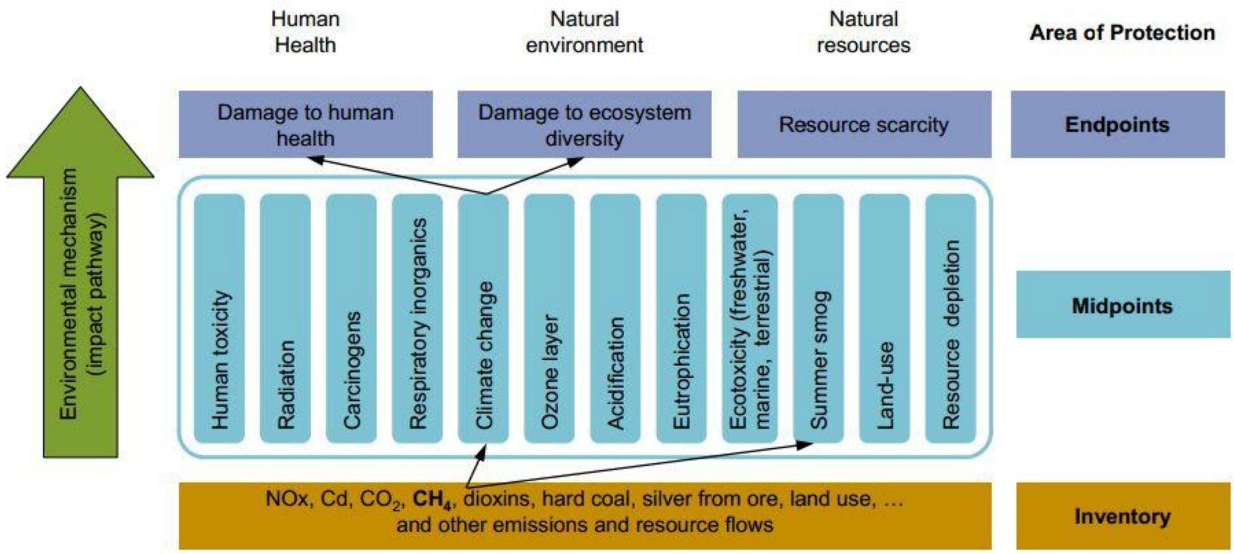

Figure 1-2 Caractérisation de l’inventaire en problèmes et dommages en ACV (ILCD handbook (European Commision Joint Research Center 2010))

L’évaluation des impacts en ACV se fait par la multiplication de la valeur d’inventaire par un facteur de caractérisation propre à ce flux élémentaire, résultant en un score d’impact pour une catégorie spécifique, tel que décrit par l’équation 1.1 (Udo de Haes et al. 1999).

$$I_j = \sum_{1\ldots x}(M_p \times FC_{j,p}) \qquad \text{Équation 1.1}$$

Où I_j est l’indicateur de la catégorie d’impact “j” (ex : écotoxicité), Mp est la masse de la substance émise ou extraite “p” (ex : kg plomb) et $FC_{j,p}$ représente le facteur de caractérisation de la substance “p” pour la catégorie d’impact “j” (ex : Impact écotoxicité/kg plomb). Un FC intermédiaire qui multiplie l’inventaire mène aux impacts au niveau problèmes, qui peuvent normalement être eux-mêmes multipliés par un FC dommages pour aboutir à des impacts à ce niveau.

1.2.1.4 Interprétation

Finalement, l’étape d’interprétation permet aux résultats des étapes précédentes d’être évalués en fonction de l’objectif de l’étude afin de fournir des conclusions et recommandations. De plus, tout au long de l’étude, les résultats sont interprétés pour identifier les contributeurs principaux, effectuer des études de sensibilité et d’incertitudes afin de revenir peaufiner les paramètres les plus critiques de l’étude.

1.2.2 Attributionnelle vs conséquentielle

Une étude ACV sera abordée soit avec une approche attributionnelle, de type comptabilisation, ou alors conséquentielle, qui étudie les conséquences possibles d'un changement entre deux systèmes de produits alternatifs. La distinction entre ces deux approches est importante dans le cadre de ce projet puisque les deux approches sont abordées, les scénarios de compensation présentés en addendum au chapitre 4 et en annexes (3) se référant davantage à l'approche conséquentielle. La différence conceptuelle entre ces deux approches ressemble à la différence entre la comptabilité et la planification financière : en comptabilité, le coût approprié de chaque item est attribué au compte correspondant alors qu'en planification financière un estimé est fait de comment les activités prévues affecteront les coûts futurs. En ACV, cela se traduit par une différence des frontières du système qui incluent uniquement les processus affectés par la décision dans le cas de la conséquentielle, versus tous les processus nécessaires à la réalisation de la fonction pour l'attributionnelle. Bien qu'une ACV attributionnelle n'est peut–être pas la meilleure option dans un contexte décisionnel, les méthodes attributionnelles sont plus souvent utilisées vu les problématiques liées à l'identification des conséquences, et le manque de consensus ou de méthodes permettant de le faire de façon systématique. Aussi, la disponibilité des données reflète davantage les opérations moyennes plutôt que les conséquences d'un changement d'opérations (Weidema 2003).

1.2.3 Sensibilité et incertitudes

Une analyse de sensibilité permet de tester la robustesse des résultats et leur sensibilité aux données, au modèle et aux choix effectués. Il est donc nécessaire d'identifier les paramètres qui ont le plus d'influence sur le résultat (Jolliet et al. 2005). Ceci peut se faire en faisant varier chaque paramètre d'un même pourcentage, et comparer l'incidence sur le résultat : les paramètres les plus sensibles feront davantage varier le résultat. Le paramètre peut également être varié sur la base du minimum et maximum de la donnée, lorsqu'un tel intervalle est disponible, résultant en une sensibilité plus représentative. Finalement, un ensemble entier de paramètres peuvent être variés afin d'évaluer la sensibilité d'un ensemble cohérent de ces choix de paramètres (analyse de scenario) sur les résultats. Ces analyses permettent d'identifier les paramètres et choix qui méritent une plus grande attention de ceux qui peuvent supporter une plus grande incertitude.

Comme tout outil d'aide à la décision, les incertitudes sont rarement considérées en ACV, bien qu'elles puissent être très élevées. Il est important d'évaluer les incertitudes puisqu'elles sont la réelle mesure du progrès scientifique, permettant d'évaluer le progrès d'un modèle à un autre, de quantifier la confiance que nous avons dans nos résultats d'ACV ainsi que de tester l'impact des choix qui sont faits, augmentant ainsi la crédibilité de l'ACV. Les incertitudes peuvent être définies comme les divergences entre une quantité mesurée ou calculée et la vraie valeur de celle-ci (Finnveden et al. 2009). Le problème en ACV c'est que cette « vraie valeur » est souvent impossible à identifier. Ainsi, tel que discuté par Heijungs (2004), la seule validation possible consiste en la validation de chaque étape individuelle, et, assumant que l'agencement de ces étapes suit des règles mathématiques et procédures strictes, on peut espérer que l'ACV en soit sera pertinente. En ACV, les incertitudes proviennent des données, des choix (scénarios) et des relations (modèles).

Une des sources d'incertitude des modèles en ACV est reliée à la variabilité spatiale des impacts. En effet, alors que l'ACV intègre les impacts dans le temps et dans l'espace et que cela peut sembler pertinent pour des catégories globales telles que le réchauffement climatique ou la destruction de la couche d'ozone, il n'en est pas de même pour les autres catégories dont les impacts dépendent de la localisation de l'émission : l'acidification (aquatique et terrestre), l'eutrophisation (aquatique et terrestre), le smog photochimique, la toxicité humaine, l'écotoxicité et l'utilisation des terres (Toffoletto et al. 2007). Les impacts liés à l'utilisation de l'eau sont également certainement très sensibles à une variabilité spatiale vu la grande disparité de la distribution et de l'utilisation de la ressource dans le monde.

1.2.4 Limites de l'ACV

Bien que l'ACV ait connu des développements importants et que plusieurs de ses limites aient été surmontées, quelques unes demeurent, bien que souvent l'objet d'études supplémentaires. Finneveden et collaborateurs (2009) résument bien les principales limites dans leur revue des nouveaux développements en ACV. Elles sont : 1- le manque de données, bien que les bases de données sont en constante croissance, 2- le manque de modèle d'évaluation d'impacts pour certains secteurs, tel que l'utilisation des terres et la ressource eau, 3- les choix possibles et manque de consensus sur certains aspects de la méthode tel que la délimitation des frontières du système et des choix méthodologiques pouvant mener à des conclusions différentes, 4- les incertitudes liées au manque de données et 5- le manque de consensus sur les méthodes de pondération et normalisation.

1.3 L'eau en analyse du cycle de vie

1.3.1 Ressources en ACV

Dans les 10 dernières années, des efforts ont été posés pour harmoniser les méthodes de prise en compte des ressources en ACV. Une ressource est définie comme étant une entité qui, une fois extraite, comporte une valeur pour les usages humains (Lindeijer et al. 2002). L'utilisation d'une ressource est définie comme étant l'allocation exclusive pour les usages humains, temporaire ou permanente, d'un objet de la nature présent en quantité limitée, ce qui peut mener à la destruction ou la dégradation de la ressource. On peut les diviser en trois principales catégories : les ressources biotiques, c'est-à-dire vivante lors de l'extraction (bois, poisson, etc.), les ressources abiotiques, donc non-vivantes (charbon, minerais, eau) et la surface terrestre. Seuls les avancements liés aux ressources abiotiques seront présentés ici. Les méthodes développées à ce jour pour la caractérisation des impacts liés à l'utilisation des ressources ont été classées en 4 catégories selon Lindjier et collaborateurs (2002).

La première catégorie se base sur la *somme de l'énergie ou des matériaux* liée à l'extraction. Elle inclut Baumann et collaborateurs (1992) qui agrègent la ressource sur une base massique et Lindfors et collaborateurs (1995) qui divisent cette agrégation selon différentes catégories : a) ressources renouvelables ou non, b) taux de renouvellement de la ressource et c) utilisation réversible ou non. D'autres auteurs agrègent plutôt les ressources sur une base énergétique (Baumann et al. 1992; Berg et al. 1995) en multipliant l'énergie spécifique d'une ressource avec la masse utilisée.

La deuxième catégorie inclut les méthodes liées *aux réserves disponibles et aux consommations actuelles*, par exemple en établissant le ratio de la ressource utilisée sur la réserve disponible (Heijungs et al. 1992). La réserve peut être basée sur plusieurs choix (Guinée and Heijungs 1995; Heijungs et al. 1992): la réserve physique, la réserve économique, la réserve ultime ou la réserve ultimement extractible. Ces méthodes peuvent être modifiées en ajoutant un paramètre représentant l'extraction annuelle de la ressource ou la proportion de la réserve que celle-ci représente (Fava et al. 1993; Guinée and Heijungs 1995).

La troisième catégorie se base sur l'agrégation des impacts liés à *l'énergie basée sur des scénarios futurs* (retour de la ressource à son état initial). Ces impacts peuvent être comptabilisés par exemple par les impacts environnementaux d'un procédé d'extraction durable de métaux, défini par une utilisation de ressource énergétique renouvelable pour concentrer un minerai 10 fois moins concentré qu'il ne l'est présentement (Steen and Ryding 1992). Une autre approche qui modélise les impacts futurs est celle de Blonk et collaborateurs (1996) qui inclut, en plus de l'exergie (voir ci-bas) la quantité d'énergie future nécessaire à l'extraction de la ressource sur une échelle de 50 ans, en plus de la surface des terres affectées par les opérations d'extraction. Finalement Mueller-Wenk (1999) propose de prendre en compte le taux d'épuisement actuel de la ressource et celui dans le futur, en intégrant un facteur d'amélioration de la productivité de la ressource, compte tenu de l'amélioration des technologies. L'épuisement ici inclut également les ressources présentes dans la technosphère.

La quatrième catégorie regroupe les propositions de méthodes qui utilisent *l'entropie ou l'exergie* d'une ressource comme mesure, corrélant une ressource disponible à une faible entropie, puisque l'entropie est une mesure thermodynamique qui évalue le désordre d'un système. L'exergie elle est une mesure de l'énergie disponible, ou utile, soit par combustion, échange de chaleur ou autre. Celle-ci dépend par contre de l'environnement, elle peut donc être perçue comme une correction de l'énergie par sa qualité (Heijungs et al. 1997) et être utilisée également pour exprimer la qualité d'une ressource non-énergétique. Alors que Finnveden et Ostlund (1997) considèrent la somme des exergies des ressources utilisées, Blonk et collaborateurs (1996) incluent l'énergie et les matériaux nécessaires à leur extraction et purification, et Ayres et Ayres (1996) appliquent le concept de l'exergie à tout le cycle de vie et l'utilisent comme un indicateur incluant donc l'émission de polluants à l'environnement. Plus récemment, Bosch et collaborateurs et Dewulf et collaborateurs (Bösch et al. 2007; Dewulf et al. 2007) ont combiné le concept de demande cumulée en exergie aux procédés d'écoinvent dans le but de fournir un indicateur de catégorie d'impact additionnel pour l'application de l'ACV, démontrant ainsi la faisabilité d'une telle approche. Leurs résultats ont démontré une dominance particulière des ressources fossiles et de l'utilisation des terres (Dewulf et al. 2007), et que la ressource eau contribuait en moyenne à 8% de la demande totale en exergie, mais jusqu'à 90% pour certains procédés.

Une autre méthode, non catégorisée plus haut, proposée par Heijungs (1997) propose d'évaluer l'impact sur la ressource basée sur la production annuelle per capita de la ressource. Ce type de normalisation ne recommande toutefois pas l'agrégation de plusieurs ressources.

Les deux premières catégories focalisent sur la consommation actuelle et les types 3 et 4 se concentrent sur les conséquences futures. Alors que les ressources ont une valeur pour la société humaine basée sur la fonctionnalité qu'elles apportent à la société humaine, les méthodes 1 et 2 sont défaillantes de par la possibilité d'évaluer la perte de fonctionnalité associée à leur utilisation. Les méthodes de types 4 présentent des problèmes conceptuels puisque l'énergie et l'exergie sont des indicateurs plutôt abstraits par rapport à une perte de fonctionnalité, ce qui les rend plus difficile à faire accepter comme étant représentative des situations propres à chaque type de ressource. Les méthodes de types 3 sont donc les plus adaptées pour l'évaluation d'une perte de fonctionnalité d'une ressource telle que l'eau.

Stewart et Weidema (2006) ont proposé un cadre pour l'évaluation des impacts liés à l'utilisation des ressources en ACV qui est cohérent avec la troisième catégorie décrite plus haut. Ils proposent trois types d'usage de la ressource : un usage où la ressource (d) est remise à l'environnement dans le même état ou meilleur que celui dans lequel elle a été prise, un usage où la ressource (c) est dégradée et n'est pas directement utilisable et un usage ou celle-ci (b) est rendue indisponible par son utilisation ou son élimination. D'un point de vue d'évaluation des impacts, il est alors intéressant d'agréger les impacts de transformation de la ressource prélevée (a), qu'elle soit transformée en b, c ou d.

Cependant, la ressource eau est différente des autres ressources de par le fait qu'elle est en grande partie renouvelable et essentielle tant aux humains qu'aux écosystèmes. Ainsi, son utilisation mène à une privation, souvent temporaire, de ses usagers, entraînant des impacts dans les catégories d'impacts santé humaine et écosystèmes. Cette ressource doit donc être traitée de façon particulière, tel que spécifié par le ILCD Handbook (European Commision Joint Research Center 2010) et seule la partie non-renouvelable de la ressource cadre bien avec la catégorie d'impacts « ressources ».

1.3.2 « Backup Technology »

Selon Stewart et Weidema (2006), une technologie utilisée pour transformer une ressource dégradée ou inutilisable en ressource utilisable est une *backup technology*. Différentes qualités de ressources peuvent être associées à différentes *backup technologies*, et être employées à différent moments. Pour l'eau, la *backup technology* est donc celle qui sera utilisée pour transformer une ressource qui a été consommée ou dégradée vers sa qualité originale lorsque celle-ci n'est plus disponible. Un débat entre Stewart et Weidema et Finneveden (Finnveden 2005) a ensuite émergé concernant la pertinence de traiter les impacts additionnels futurs de l'extraction de la ressource, soit de la *backup technology*, en tant qu'impacts ou comme faisant partie de l'inventaire. Un article commun (Weidema et al. 2005) a ensuite statué sur le fait que seuls les impacts qui ne sont pas prévus d'être compensés (remédiés par une *backup technology*) devraient être inclus dans l'évaluation des impacts.

1.3.3 L'empreinte eau

L'empreinte eau, ou *water footprint*[1], est un terme longtemps associé aux développements méthodologiques du *Water footprint Network (WFN)*, mais l'idée de base d'une empreinte eau est que de visualiser l'utilisation d'eau cachée derrière les produits peut aider à comprendre le caractère global de l'eau douce et à quantifier les effets de la consommation et des marchés sur la ressource eau. Cette meilleure compréhension peut ensuite servir de base à une meilleure gestion des ressources hydriques de la planète, tel que suggéré par le concept d'eau virtuelle présenté plus haut. Alors que celui-ci était davantage limité aux produits agricoles et même alimentaires, et principalement associé à la gestion des importations et exportations de denrées, l'idée de considérer l'utilisation d'eau tout au long de la chaine de production a gagné en intérêt depuis l'introduction du concept d'empreinte eau par Hoekstra en 2002 (Hoekstra and Hung 2002).

Le *Water Footprint* proposé par le WFN est un indicateur d'utilisation d'eau douce qui prend en compte le volume d'eau nécessaire à la fabrication d'un bien tout au long de la chaine de

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[1] Dans ce document, les termes "empreinte eau" et "water footprint" sont utilisés de façon interchangeable, et « Water Footprint » fait généralement référence à la méthodologie du WFN.

production, c.-à-d. dans une approche cycle de vie. Ce volume est en fait décrit par trois types d'eaux qui peuvent éventuellement être additionnés: la bleue, la verte et la grise. L'eau bleue réfère à la consommation d'eau de surface ou souterraine. Il est important de noter que l'on parle ici de consommation, et non de prélèvement, et que celle-ci réfère à la perte d'eau d'un bassin versant vers un autre, vers la mer, ou par l'évaporation ou l'intégration dans un produit. L'eau verte réfère à la consommation d'eau entreposée dans le sol en tant qu'humidité du sol et l'eau grise au volume d'eau requis pour assimiler la charge de pollution basée sur des standards de qualité d'eau ambiante. Bien que l'eau grise soit un concept intéressant de mesure de la dégradation de la qualité, il est rarement appliqué dû au manque de normes de référence en question et aux limites de la pertinence scientifique associée à un volume critique de dilution, négligeant ainsi les questions de persistance et de sort dans l'environnement (Katsoufis et al. 2010; Ridoutt et al. 2009). Le *Water Footprint* est donc une mesure de la consommation et de la pollution mais ce n'est pas une mesure de la sévérité de leurs impacts environnementaux locaux. Tel que critiqué par Ridoutt et Pfister (2010), les empreintes eau de différents produits ne sont pas comparables puisqu'ils proviennent de régions qui diffèrent de par la rareté locale de l'eau. L'impact environnemental local d'une quantité d'eau consommée et polluée dépend de la vulnérabilité du système hydrique local et du nombre d'usagers de ce système. Ceci dit, le *Water Footprint* fournit donc une information spatio-temporelle sur la consommation d'eau pour les usages humains, qui peut s'avérer utile pour alimenter les discussions sur les usages durables et équitables de la ressource, et servir de base à une évaluation locale des impacts environnementaux, sociaux et économiques.

Plus récemment, un standard ISO a été proposé afin de standardiser le cadre méthodologique, les exigences et les directives pour effectuer une empreinte eau dans un cadre cycle de vie. Ce standard, attendu en 2014, définit une empreinte eau comme étant l'ensemble des impacts sur la ressource eau causés par un procédé ou système de procédés. Selon le standard, les volumes d'eau ne sont donc pas suffisants pour effectuer une empreinte eau et l'évaluation des impacts générés par sa dégradation est nécessaire. Ces impacts sont associés à la consommation et à la dégradation d'eau, et doivent être régionalisés. Ils peuvent être quantifiés au niveau problèmes (i.e. baisse de disponibilité de l'eau, eutrophication, ecotoxicité, etc.) ou au niveau dommages (i.e. santé humaine et écosystèmes), en utilisant les méthodes d'impacts couvrant le maximum de chaines cause-à-effet pour répondre à l'objectif de l'étude. Les méthodes existantes en ACV permettent déjà de caractériser les impacts liés à la dégradation, principalement par l'eutrophication, (éco)toxicité et l'acidification. Les sections qui suivent décrivent le cadre qui permet de caractériser les impacts liés à la consommation d'eau. Les principes généraux du cadre méthodologique sont vulgarisés dans l'Annexe 1 qui présente un chapitre de livre sur la caractérisation des impacts de l'utilisation de l'eau écrit pour un manuel de cours d'ACV, en cours de publication.

1.3.4 Développement du cadre méthodologique pour la caractérisation des impacts de l'utilisation de l'eau en ACV

Suivant les concepts liés aux ressources en ACV présentés ci-haut, l'eau est définie comme une ressource abiotique qui peut être catégorisée en trois sortes différentes selon son taux de renouvellement : 1- Dépôt/stock (taux de renouvellement pratiquement nul, aquifères non renouvelable), 2- Fond/fund (taux de renouvellement bas, typiquement eaux souterraines) et 3- Écoulement/flow (taux de renouvellement élevé, typiquement eau de surface) (Finnveden 1996; Heijungs et al. 1997). L'utilisation de l'eau amène généralement deux types de problèmes distincts : la compétition pour la ressource dans les conditions actuelles, et l'épuisement de la ressource, causé par une extraction supérieure au taux de renouvellement, ce qui affecte les générations futures (Finnveden 1996; Lindeijer et al. 2002).

Les développements méthodologiques visant à intégrer la ressource eau en ACV ont commencé à prendre forme avec les travaux d'Owens (2002) qui a proposé une série d'indicateurs et de définitions visant à harmoniser les développements futurs. Il propose cinq indicateurs liés à la quantité d'eau : 1- utilisation d'eau *in-stream*, 2- consommation d'eau *in-stream*, 3- utilisation d'eau *off-stream*, 4- consommation d'eau *off-stream* et 5- appauvrissement de la ressource *off-stream*. *In-stream* et *off-stream* réferrant à un usage « dans le flot » (ex : eau turbinée) et « hors du flot » (ex : irrigation), respectivement. Il discute ensuite du potentiel de plusieurs paramètres de qualité de l'eau comme partie d'un indicateur de qualité : eutrophisation, demande biologique en oxygène (DBO), température, microorganismes pathogènes, couleur et turbidité, matières en suspensions (MES), toxicité, toxicité humaine et ecotoxicité.

Des efforts ont ensuite été faits pour intégrer la ressource eau, soit par Brent (2004a) qui propose une méthode d'évaluation pour comparer l'utilisation de certaines ressources d'un point de vue

« distance-à-la-cible » pour le contexte Sud Africain, ou par Bauer et Zapp (2005) qui ont mis en évidence la grande variabilité spatiale de la ressource parallèlement avec les points d'exploitation d'aluminium. Cependant, aucune ne permettait encore de modéliser les mécanismes environnementaux liés à l'utilisation de la ressource.

La *UNEP-SETAC Life Cycle Initiative* a été mise en charge d'examiner les problèmes liés à la consommation de ressources telles que l'utilisation des terres et de l'eau. Dans sa première phase, le projet a abouti à une série de recommandations concernant l'évaluation des impacts liés à l'eau, non publiées mais résumées dans Bayart et collaborateurs (2010b) : i- la méthode d'évaluation devrait être régionalisée en fonction du contexte hydrologique, ii- l'usage consommant de l'eau (différence entre l'eau prélevée et rejetée) génère des impacts en abaissant les niveaux d'eau et en privant les autres usagers de la technosphère et de l'écosphère de la ressource, iii- une série de types d'eau y est décrite, iv- l'épuisement de la ressource peut être considéré comme un problème alors que les impacts sur la santé humaine et les écosystèmes seraient plutôt des dommages, v- la catégorie de dommage sur les ressources peut ne pas être considérée si les impacts sont modélisés jusque sur la santé humaines et les écosystèmes, vi- une chaine cause-à-effet devrait décrire les impacts sur la santé humaine causés par l'utilisation d'une ressource de moins bonne qualité pour les usages domestiques et l'agriculture et vii- les impacts de la compensation de la production alimentaire et ceux sur la biodiversité causés par le desséchement et la perte d'habitat devraient aussi être inclus.

La deuxième phase du projet de la Life Cycle Initiative, appelé *Assessment of use and depletion of water ressources within the LCA Framework* (WULCA) a ensuite abouti à la publication d'un cadre d'étude pour l'évaluation de l'utilisation *off-stream* de l'eau douce en ACV (Bayart et al. 2010a). Les auteurs y proposent une terminologie, bâtie sur celle d'Owens (2002), définissant les usages *dégradatifs* - baisse de la qualité - et *consommants* - transfert d'eau à l'extérieur d'un bassin versant, par évaporation, intégration au produit ou transfert à la mer ou ailleurs en plus de la *compétition* - lorsque la disponibilité de l'eau est trop faible pour combler les besoins de tous les usagers – et de *l'épuisement de l'eau douce* – comme étant la réduction nette de la disponibilité d'eau douce dans un bassin versant pour une période donnée. Des recommandations y sont faites concernant l'inventaire lié à la ressource eau. Ils proposent que les flux élémentaires représentent différents types d'eau, chacun avec son propre facteur de caractérisation. Les types d'eau devraient être distingués par le type de ressource (surface, souterraine, etc.) et la qualité. Cette dernière pourrait être prise en compte soit par une approche « distance-à-la-cible», exprimé en volume d'eau équivalent pour une dilution ou en énergie nécessaire au traitement, ou par une approche basée sur les fonctionnalités, utilisant des standards de qualité établis par des organismes internationaux (Svobodová et al. 1993; WHO 2008; WHO and UNEP 2006). Ils recommandent que le volume d'eau remis à l'environnement soit considéré.

Bayart el al. (2010b) recommandent également trois chaines cause-à-effet décrivant les impacts liés à une utilisation d'eau douce. Ils proposent qu'une utilisation d'eau réduira la disponibilité de la ressource pour trois « sujets à protéger » (ou AoP pour *areas of protection*) : les usages humains, les écosystèmes et les générations futures. Ces chaines cause-à-effet sont cohérentes avec les recommandations de Stewart et Weidema (2006), qui proposent que le fait d'utiliser de l'eau et de priver d'autres usagers, pouvant causer des impacts par exemple sur la santé humaine, constitue une chaine cause-à-effet en soi, et devrait être adressé de façon additionnelle à l'impact généré sur la ressource même. Le cadre méthodologique incluant ces différentes chaines de cause-à-effet est présenté dans la figure 1-3 ici-bas.

Parmi les usagers humains identifiés (usagers domestiques, agriculture, industrie, transport, pêches, loisirs et hydroélectricité) seuls ceux affectés par un manque d'eau qui leur est fonctionnelle doit être considéré. Par exemple, une eau souterraine n'est pas fonctionnelle pour le transport par bateau, et la qualité est également un facteur qui influe sur la fonctionnalité. La baisse de disponibilité pour ces usagers peut alors entrainer deux scénarios, selon les conditions socio-économiques locales: le manque ou la compensation. Un manque d'eau pour les usagers domestique et agricole engendrera des maladies et de la malnutrition respectivement, et l'intensité de ce manque d'eau devrait être évaluée en fonction de la rareté d'eau locale et de sa qualité. La compensation fait référence à l'utilisation d'une *backup technology* telle que décrite plus haut et des scénarios génériques sont proposés pour chaque usager.

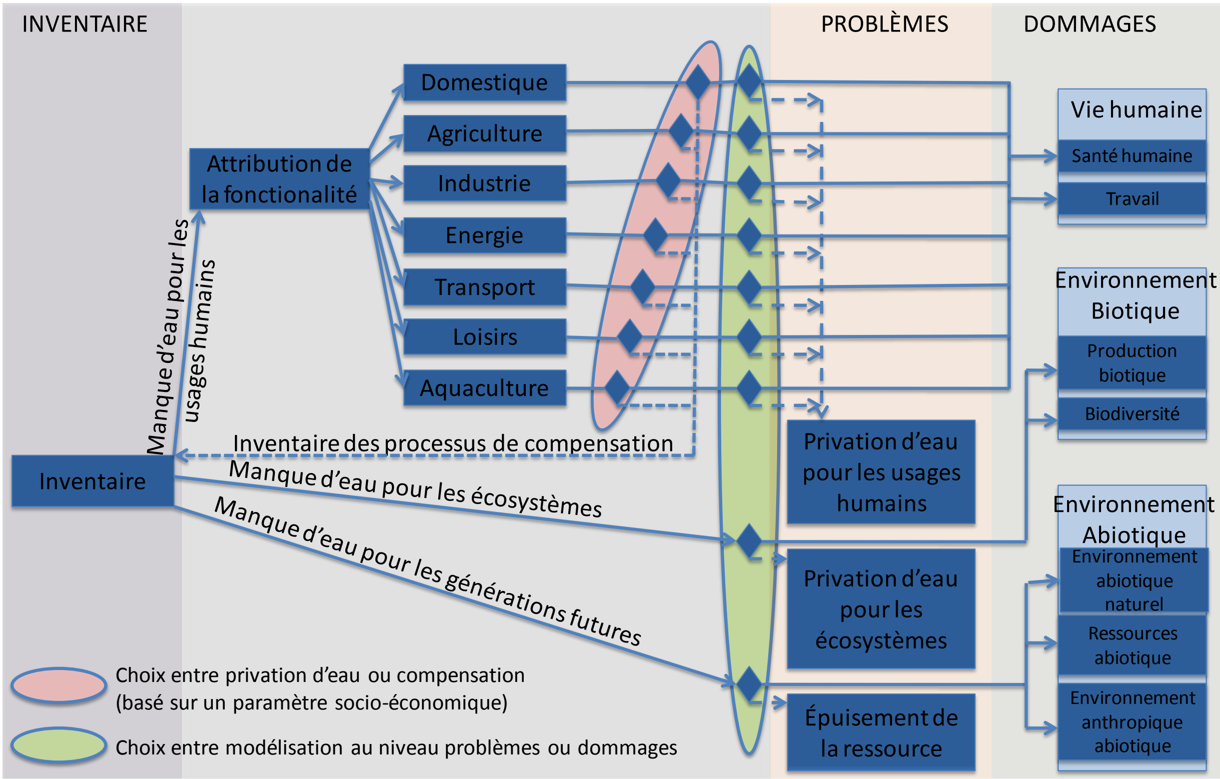

Figure 1-3 Cadre d'étude présentant les chaînes cause-à-effet proposé par Bayart et collaborateurs pour les impacts d'une utilisation d'eau en ACV (adapté de Bayart, Margni, et al., 2010)

Sans directement suggérer une voix d'impact pour la chaîne cause-à-effet sur les écosystèmes, ils proposent un indicateur au niveau problèmes exprimés en m³ d'eau non disponibles pour les écosystèmes et pointent vers diverses méthodes qui ont entrepris de modéliser certains aspects de cette chaine (Humbert and Maendly 2009; Milà i Canals et al. 2009; Pfister et al. 2009; Van Zelm et al. 2008). Finalement, ils discutent également d'une troisième catégorie d'impacts sur les générations futures causés par l'épuisement de la ressource par une consommation supérieure au taux de renouvellement de la ressource, menant à des impacts sur la catégorie de dommage « ressources naturelles » bien que l'existence de celle-ci soit discutable en ACV (Bayart et al. 2010b; Stewart and Weidema 2006). Ultimement c'est également par l'utilisation d'une *backup technology* que la quantité d'eau épuisée sera comptabilisée dans cette catégorie avec, par exemple, la quantité d'énergie supplémentaire nécessaire à l'extraction de la ressource, ultimement le dessalement d'eau de mer pour la ressource eau. Alors que ce cadre permet de canaliser les efforts de développement dans la même direction, il manque encore une méthode d'inventaire qui permettrait de supporter une telle méthodologie, et les chaines de cause-à-effet restent à être modélisées.

1.3.5 Inventaire et catégorisation d'eau

L'inventaire est la base de l'analyse du Cycle de Vie et aucune méthode consensuelle n'existe présentement pour la ressource eau. Les bases de données permettaient jusqu'à cette année de distinguer les différentes sources d'eau ou activités pour les prélèvements, mais aucun flux élémentaire n'était associé à l'eau remise à l'environnement. Par exemple, ecoinvent et Gabi (Frischknecht and Jungbluth 2004; PE International GmbH 2006) distinguent l'eau provenant de lacs ou rivières, souterraine, turbinée, salée ou de refroidissement. Au moment de la soumission de cette thèse, la nouvelle version d'ecoinvent venait d'être rendue publique avec des valeurs pour l'eau remise à l'environnement, mais non disponible encore dans les logiciels d'évaluation des impacts. De plus, une base de données privée développée par Quantis a récemment fait l'inventaire pour tous les processus ecoinvent des flux d'eau entrante, sortante, consommée et turbinée (Quantis 2012a).

Des outils tels que le *Global Water Tool* (World Business Council for Sustainable Development (WBCSD) 2007) ou l'outil du *Water Footprint Network* décrit plus haut permettent d'inventorier les volumes d'eau prélevés ou consommés par une industrie, en spécifiant la région géographique de cette utilisation d'eau. Bien que ce soient des méthodes d'inventaires valables en terme de quantité d'eau, elles n'offrent pas une information complète concernant la qualité de l'eau prélevée ou remise. L'eau grise du *Water footprint Network* est basée sur la qualité de l'eau rejetée, mais celle-ci est déjà caractérisée sommairement en utilisant les volumes de dilution et n'est donc pas une méthode d'inventaire. Similairement, Mila-i-Canals et collègues (Milà i Canals et al. 2009) proposent une méthode d'évaluation des impacts où l'inventaire se distingue par l'usage consommant de l'eau de surface et souterraine, l'usage (consommant ou non) de ressources non-renouvelable (eaux fossilisées) ou souterraine sur-utilisée (fond/fund) et le changement de disponibilité de l'eau de pluie dû à un changement des terres.

Finalement un cadre méthodologique proposé par Bayart (2010b), basé sur des travaux non publiés de Vince, propose de distinguer les flux élémentaires par leur source et leur qualité, et suggère huit flux décrits de façon qualitative : eau souterraine potable, de bonne et mauvaise qualité, eau de surface potable, de bonne et mauvaise qualité, eau usée et eau de mer. Bien qu'il s'agisse d'un pas dans la bonne direction, cette méthode ne quantifie pas la qualité des flux de « bonne » et « mauvaise » qualité.

Une méthode d'inventaire adaptée pour l'ACV et permettant de prendre en compte la qualité de façon quantitative devra probablement passer par un indicateur de qualité ou par des catégories d'eau. Stewart et Weidema (2006) notent que la qualité de l'eau ne peut être définie comme un seul indicateur puisque c'est un paramètre multidimensionnel. Elle doit donc être définie par un vecteur de qualité présentant plusieurs caractéristiques.

Des indexes de qualité de l'eau et des classifications ont été proposés dans plusieurs domaines autres que l'ACV, spécialement pour décrire et catégoriser les eaux de surface. Alors que ces méthodes remplissent l'objectif pour lequel elles ont été élaborées, elles sont principalement orientées vers les besoins des écosystèmes et non des usages humains. Tel que décrit par l'OMS en se référant aux classification existantes: "En règle générale, l'orientation des systèmes de classification vers la vie aquatique implique que les limites des catégories sont plus conservatrices que si elles étaient basées sur d'autres utilisations" (Enderlein et al. 1997). Les agences gouvernementales cependant ont un intérêt pour une classification orientée vers les fonctionnalités humaines et des directives partielles ont été proposées. La Communauté Économique Européenne a proposé des standards de qualité pour l'eau de surface en fonction des usages domestiques et du traitement à effectuer (EEC 1975). L'agence environnementale du Japon (Overseas Environmental Cooperation Center 1998) et l'EPA Taïwanaise (Taiwan EPA 1998) vont plus loin et identifient des paramètres et des seuils associés avec plusieurs utilisateurs incluant domestique, industriel, aquaculture, irrigation, récréatif et conservation environnementale. D'autres classifications incluent l'Inde, la Thaïlande et le Royaume-Uni (Enderlein et al. 1997). Alors que ces classifications pourraient servir, au moins partiellement, les besoins de la communauté ACV, seulement quelques paramètres sont définis et aucune information n'a pu être obtenue quant à la façon dont les seuils furent identifiés. De plus, les catégories sont créées d'une façon qui ne permet pas une grande distinction entre les différents besoins des utilisateurs, ce qui signifie qu'une perte de fonctionnalité par utilisateur individuel demeure impossible.

1.3.6 Méthodes existantes d'évaluation des impacts

Les travaux de Kounina et collaborateurs (2013) présentent et analysent les méthodologies existantes, tant au niveau des indicateurs de rareté, des méthodes d'inventaires, que des impacts au niveau problèmes et dommages. Celles-ci sont catégorisées selon les trois sujets à protéger tel que résumées dans la figure 1-4.

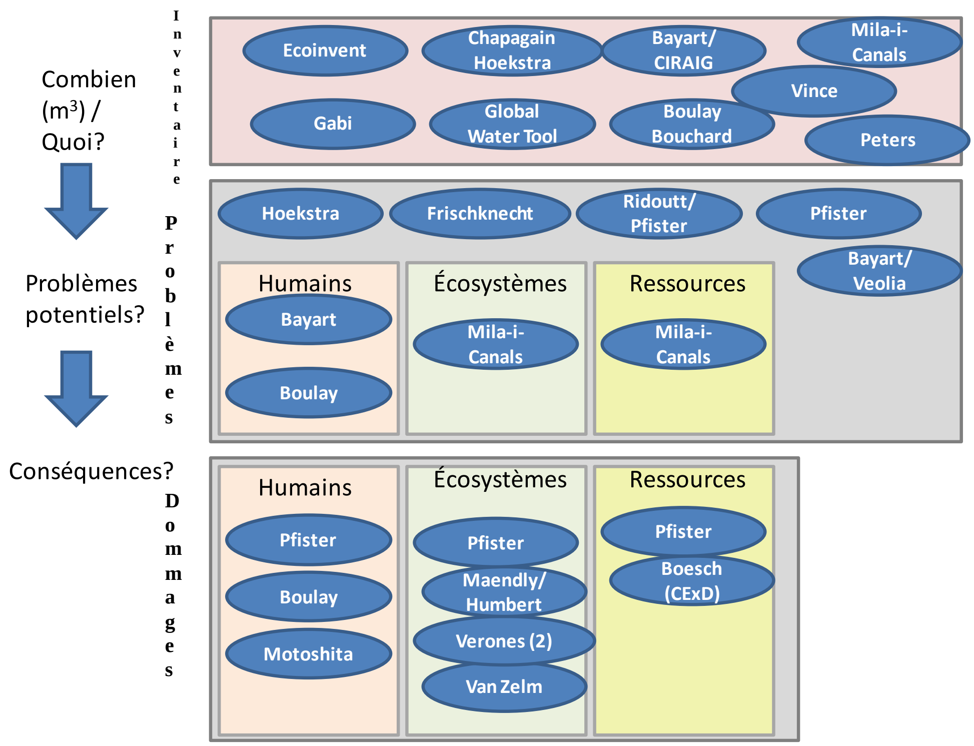

Figure 1-4 Portée et envergure des différentes méthode d'inventaire et d'évaluation d'impacts (identifiées par leurs auteurs, références dans le texte, adapté de Kounina et collaborateurs (2013)).

1.3.6.1 Méthodologies niveau problèmes

Au niveau problèmes, cinq méthodologies proposent un indicateur générique pour les trois sujets à protéger. En plus du *Water footprint Network* discuté plus haut, la *Swiss Ecoscarcity method* est une méthode « distance-à-la-cible » qui évalue le flux élémentaire en relation avec un flux critique, fixé par des objectifs politiques ou des recommandations législatives. Les résultats sont donnés en éco-points pour plusieurs impacts sur l'environnement. Pour l'utilisation de l'eau, le prélèvement d'eau douce est comparé à un taux de prélèvement par défaut de 20% des ressources disponibles, basé sur les recommandations de l'OCDE (Vörösmarty et al. 2000b) comme étant un stress moyen, et ce par bassin versant. Pfister et collaborateurs (Pfister et al. 2009) proposent un indicateur, le WSI (Water Scarcity Index). Celui-ci est une variation du ratio de criticalité décrit dans la section « rareté », mais incluant un paramètre de variabilité saisonnière basé sur des données climatiques et une distinction des débits fortement régulés ou non. L'indicateur est ajusté pour que le résultat du WSI soit de 0.5 lorsque le ratio de criticalité (appelé WTA) est de 0.4, valeur choisie comme étant le seuil entre un stress hydrique modéré et sévère. Cet index, ainsi que tous les index de rareté décrit jusqu'à maintenant mis à part le Blue Water Scarcity du Water Footprint Network, définissent cependant la rareté sur la base de l'eau prélevée dans une région, et non de l'eau consommée, suggérant ainsi qu'une eau prélevée et remise à l'environnement (i.e. utilisée pour le refroidissement) contribue à la rareté de la ressource, et aucun ne différencie l'eau de surface de l'eau souterraine ni ne prend en compte la qualité de l'eau.

Un indicateur proposé par Véolia (Veolia Water 2010), intègre cet index WSI de rareté locale et un index de qualité de l'eau prélevée et remise à l'environnement, évaluée par un index distance-à-la-cible basé sur des normes environnementales pour la qualité ambiante des eaux de surface. Ridoutt et Pfister (2010) utilisent également l'index WSI pour produire un « *stress-weighted water footprint* » de produits, en le multipliant avec l'inventaire, incluant un volume d'eau fictif quantifiant la pollution (le concept de l'eau grise du Water Footprint Network). Cette méthode a cependant mené à des résultats souvent aberrants et les auteurs eux-mêmes ne recommandent pas cette méthode (Ridoutt and Pfister 2013). Dans cette dernière publication, ils recommandent plutôt un système de pondération, tel qu'utilisé par Recipe (Goedkoop et al. 2012), afin d'agréger les impacts causés par la consommation et la dégradation de l'eau. Bien qu'intéressante, cette méthode comporte plusieurs choix de valeurs et pondération qui ne sont pas facilement identifiables avec les résultats, limitant ainsi le potentiel d'interprétation des résultats.

Finalement Mila-i-Canals (2009) propose deux catégories d'impacts spécifiques au niveau problèmes : impacts sur les écosystèmes et épuisement de la ressource, et ne sont donc pas orientées vers les usages humains. La méthode propose d'évaluer les impacts sur les écosystèmes par un indicateur qui inclut les usages consommants (évaporation par l'irrigation, le refroidissement, les réservoirs, etc.) de l'eau de surface et souterraine et par la transformation de l'utilisation des terres. L'indicateur pour l'épuisement de la ressource calcule le potentiel d'épuisement de ressource abiotique appliqué aux usages, consommants ou non, des eaux souterraines appartenant à des aquifères surexploités.

La difficulté liée au choix d'un indicateur au niveau problèmes provient du besoin d'un tel indicateur de pouvoir mener directement à des impacts au niveau dommages, et ce sans devoir revenir en arrière dans la chaine cause-à-effet parce que des informations auraient été perdues, tel que sur la qualité, ou des informations spécifiques à la région de prélèvement (Bayart et al. 2010b). De plus, alors qu'un indicateur au niveau problème représente un point de la chaine de cause-à-effets, l'indicateur de dommage devrait représenter le point au bout de cette même chaine, avec, idéalement, des résultats proportionnels. Mais de tels indicateurs n'existent pas toujours pour certaines catégories d'impacts (Bare et al. 2000). Également, la question peut se poser si un seul indicateur est désiré pour les trois chaines cause-à-effet, appelant ainsi un indicateur décrivant davantage la rareté de la ressource, ou si trois indicateurs distincts sont davantage souhaitables.

1.3.6.2 Méthodes niveau dommages

Bien que le projet concerne plutôt la catégorie d'impacts santé humaine, d'autres approches adressant l'une ou l'autre des catégories de dommage existent et sont présentés ci-dessous.

Santé humaine : Pour la santé humaine, l'indicateur de stress de Pfister et collaborateurs (WSI) multiplie une série de paramètres permettant de prendre en compte les impacts sur la santé humaine de la malnutrition causée par un manque d'eau pour l'agriculture : le pourcentage de l'eau utilisée pour l'agriculture, un facteur de développement humain, la quantité d'eau nécessaire pour soutenir l'alimentation et un facteur de dommage en DALY causé par la malnutrition. Ce dernier est dérivé à partir de différentes régressions ayant des valeurs R^2 de 0.71 et 0.26, qui laisse donc place à l'amélioration. De plus, l'hypothèse est faite qu'un cas de malnutrition survient lorsque toute l'eau nécessaire à la production de nourriture pour une personne pour une année est utilisée, alors qu'en réalité la malnutrition survient bien avant puisqu'une personne ne peut survivre normalement un an sans nourriture. Finalement, la méthode considère qu'une utilisation d'eau n'affectera pas les usagers domestiques et ainsi ces impacts sur la santé humaine ne sont pas considérés. Les facteurs de caractérisation résultants sont exprimés en DALY/m³ d'eau consommée. La seule autre méthode évaluant les impacts sur la santé humaine d'un manque d'eau est celle de Motoshita et collaborateurs (2010a) qui relient un manque d'eau pour les usages domestiques à des impacts en DALY sur la santé humaine, à travers une série de régression linéaires décrites par un module d'accès à l'eau potable et un d'évaluation d'impacts sur la santé humaine pour un manque d'eau, c.-à-d. une consommation d'eau non-potable, proposant ainsi des facteurs de caractérisation par pays. Les mêmes auteurs proposent également une méthode décrivant les impacts d'un manque d'eau pour l'agriculture, en caractérisant un facteur d'effet associé à un manque de calories. Celui-ci est causé par une baisse d'accessibilité à la nourriture dans les pays affectés, dû à une situation économique plus faible, par une baisse de production dans un autre pays. Ce facteur d'effet n'est pas applicable directement à une valeur d'inventaire et les travaux sont en cours pour perfectionner la méthode ((Motoshita et al. 2010b). Aucune des méthodes considèrent les impacts causés par une baisse de disponibilité de l'eau due à une dégradation, la rendant non-fonctionnelle pour les usagers, ni les impacts potentiels qu'un manque d'eau pour l'aquaculture ou les pêches peut avoir sur la malnutrition.

Ressources : Au niveau des ressources, la méthode de Pfister (2009) passe par le concept de la *backup technology* en évaluant les impacts sur la ressource par l'énergie nécessaire à dessaler l'eau surconsommée dans un bassin versant. Un facteur d'épuisement de la ressource, dérivé du ratio de criticalité, multiplie l'énergie nécessaire au dessalement et l'eau consommée pour obtenir des dommages en MJ d'énergie supplémentaire pour rendre la ressource disponible dans le futur. La seule autre méthode adressant cette chaine cause-à-effet est celle de Bosh (2007) qui passe par l'éxergie, chimique et potentielle, de l'eau douce en relation avec l'eau salée, qui aurait une éxergie de 0. Cependant, cette méthode ne prend pas en compte la rareté locale et tel que discuté plus haut, le concept d'éxergie demeure davantage théorique et peu accepté au niveau pratique vu le manque de pertinence en lien avec l'épuisement ou les fonctions de la ressource.

Écosystèmes : Finalement, au niveau des écosystèmes, Pfister et collaborateurs (2009) proposent un facteur de caractérisation basé sur la fraction de la Production Primaire Nette (NPP) qui est limitée par la disponibilité d'eau et les précipitations locales. La NPP est utilisée comme proxy à la vulnérabilité de la biodiversité des plantes vasculaires, normalement utilisée pour évaluer la fraction d'espèce potentiellement disparue (PDF). Les facteurs de caractérisations résultants, en m^2-an/m^3, sont multipliés par l'usage consommant, résultant en impacts sur la qualité des écosystèmes en m^2-an.

Trois autres méthodes évaluent les impacts sur les écosystèmes de façons plus spécifiques. Humbert et Maendly (article en révision) ont développé une méthode spécifiquement applicable pour évaluer les impacts générés par les barrages et réservoirs. Ils évaluent la fraction d'espèces potentiellement disparues d'un système aquatique sur une certaine superficie, soit par m^3 d'eau turbinée ou par kWh produit. Les résultats en $PDF \cdot m^2 \cdot an$ permettent une comparaison facile avec d'autres impacts sur les écosystèmes présentés dans les mêmes unités. Cette méthode étant spécifique aux barrages, elle n'est pas suffisante en soit pour évaluer les impacts de l'utilisation de l'eau sur les écosystèmes, mais offre un ajout important aux méthodes évaluant les impacts par d'autres chaines cause-à-effet. Tout aussi spécifique est le modèle développé par Verones (2011) qui évalue la perte de biodiversité due à la pollution thermique, par un facteur d'effet provenant d'observations empiriques (régression multiple). Van Zelm et collaborateurs (2008) évaluent la réduction de la biodiversité terrestre causée par un abaissement de la nappe d'eau causé par l'extraction d'eau souterraine. La chaîne cause-à-effet est basée sur le fait qu'une extraction d'eau souterraine abaisse la nappe ce qui provoque la disparition d'espèces de plantes terrestres, exprimées en fraction potentiellement non présentes (PNOF). Les facteurs de caractérisation expriment la réduction en biodiversité en $PDF\text{-}m^2\text{-}an$ par m^3 d'eau souterraine extraite. Deux méthodes sont ensuite apparues plus récemment. Hanafiah et collaborateurs (2011) ont calculé des facteurs de caractérisation qui quantifie la perte de richesse d'espèces de poissons causé par une consommation d'eau. Amores et collaborateurs (2013) ont calculé un facteur de caractérisation qui évalue les dommages écologiques associés, en fraction d'espèces potentiellement affectés, à une augmentation de la salinité causée par une consommation d'eau.

En bref pour le niveau dommage, plusieurs méthodes existent, décrivant plusieurs chaines cause-à-effet. Au niveau de la santé humaine, Pfister et Motoshita décrivent les impacts d'un manque d'eau pour l'agriculture ou les usages domestiques, mais les corrélations sont parfois faibles, et ni la qualité ni la compensation ne sont pris en compte. Plusieurs méthodes explorent différentes chaines cause-à-effet menant à des impacts sur les écosystèmes qui gagneraient à être combinées afin qu'un seul indicateur permette de convertir une utilisation d'eau en impacts sur les écosystèmes, incluant les barrages et réservoirs, la pollution thermique, l'abaissement de la nappe, la perte d'habitat pour les espèces aquatiques et la perte de ressource pour les espèces terrestres. L'épuisement de la ressource devrait être évalué par une méthode cohérente avec les deux autres chaines cause-à-effet.

Pour toutes ces méthodes proposées, la régionalisation des impacts liés à l'utilisation de l'eau diffère grandement, allant de l'échelle du modèle WaterGap de 0,5 x 0,5 degré, à l'échelle du pays en entier. Ultimement, indépendamment des méthodes choisies, l'échelle optimale se doit d'être identifiée. Celle-ci doit être représentative des impacts locaux générés par un manque d'eau : la limite entre deux régions permettant de différencier des zones où les impacts d'utiliser de l'eau seront en effet différents, tout en considérant la disponibilité des données à des échelles spécifiques.

CHAPITRE 2 PROBLEMATIQUE, HYPOTHÈSE DE RECHERCHE ET OBJECTIFS

2.1 Problématique traitée

Les impacts associés à l'utilisation de l'eau en ACV ne sont pas encore caractérisés de façon exhaustive, ni au niveau problème (rareté d'eau), ni au niveau de dommages sur la santé humaine. Alors que plusieurs indices de rareté d'eau existent, aucun ne différencient l'eau de surface et l'eau souterraine, et tous (sauf un développé en 2012 (Hoekstra et al. 2012)) se basent sur les prélèvements d'eau plutôt que la consommation pour évaluer la rareté. De plus, aucune méthode ne considère la baisse de disponibilité associée à une dégradation de la ressource, qui rendrait sa qualité impropre à une utilisation spécifique sans l'ajout d'un traitement supplémentaire. Ce manque se retrouve également au niveau des indicateurs de dommages sur la santé humaine. De plus, à ce jour, les impacts sur la santé humaine n'incluent que partiellement les impacts liés au manque d'eau pour les usagers domestiques et excluent les usagers « in-stream », i.e. les pêches et aquaculture.

Les aspects mentionnés ci-haut mettent en évidence le besoin de modèles plus exhaustifs et pertinents. Le concept de l'empreinte eau en ACV étant en pleine effervescence, cette thèse contribue au débat et à l'avancement des connaissances de l'évaluation des impacts potentiels dus à l'utilisation de l'eau en ACV. Ce besoin existe non seulement au sein des industries et praticiens, mais également au sein de la communauté scientifique qui s'intéresse de plus en plus à cette problématique.

2.2 Hypothèse de recherche

L'hypothèse de recherche de cette thèse est la suivante :

Une méthodologie d'évaluation des impacts liés à l'utilisation d'eau en analyse du cycle de vie intégrant la qualité de l'eau entrante et sortante, sa source et sa fonctionnalité pour des usagers humains spécifiques permet d'augmenter le pouvoir de discrimination de l'évaluation des impacts potentiels de l'utilisation de l'eau (consommation ou dégradation) sur la santé humaine.

2.3 Objectifs

L’objectif général de ce projet est de développer une méthode permettant d’évaluer les impacts associés à l’utilisation de la ressource eau cohérente avec le cadre méthodologique de l’ACV. La méthode focalisera sur les impacts potentiels générés par un manque d’eau pour les usages humains, en prenant en compte la qualité de l’eau prélevée et rejetée et sa fonctionnalité pour les usagers humains, la source et la capacité des utilisateurs à s’adapter à un manque d’eau.

L’objectif général est atteint par le biais des cinq objectifs spécifiques suivants :

1. Développer une méthode d’inventaire permettant de quantifier les flux entrants et sortants par des catégories d’eau définies par la qualité, la source d’eau (surface, souterraine, pluie, etc.) et les usagers pour lesquels elles sont fonctionnelles
2. Développer un modèle permettant de caractériser les impacts sur la santé humaine d’un manque d’eau pour les usages humains lorsque les conditions économiques ne permettent pas une adaptation et proposer un cadre méthodologique pour l’évaluation des impacts lorsque les usagers peuvent s’adapter
3. Évaluer la sensibilité et l’incertitude associée au modèle et à ces choix inhérents, ainsi que l’incertitude de modélisation à travers la comparaison de celui-ci avec d’autres modèles existants
4. Démontrer la pertinence et applicabilité du modèle par une étude de cas et comparer les résultats avec ceux d’autres modèles existants
5. Évaluer les résultats du modèle en les comparant avec les données disponibles sur la charge totale d’impacts sur la santé humaine liés aux problématiques modélisées (malnutrition et manque d’eau pour les usages domestiques) et en faisant une analyse d’incertitudes

2.4 Présentation du document

Les prochains chapitres correspondent aux quatre articles scientifiques publiés ou soumis, suivi d'un chapitre de résultats complémentaires, répondant dans l'ordre, aux cinq objectifs spécifiques ci-haut :

Chapitre 3: Categorizing water for LCA inventory

Chapitre 4: Regional characterization of freshwater use in LCA: Modeling direct impacts on human health

Note: Ce chapitre inclut en addendum un article en préparation qui explore le concept de la compensation en présentant un cadre méthodologique et une série d'algorithmes décisionnels permettant d'identifier la technologie marginale associée à une utilisation d'eau, spécifique à une région donnée.

Chapitre 5: Analysis of water use impact assessment methods (Part A): Evaluation of modeling choices based on a quantitative comparison of scarcity and human health indicators

Chapitre 6: Analysis of water use impact assessment methods (Part B): Applicability for water footprinting and decision making with a laundry case study

Chapitre 7: Résultats complémentaires : correction au modèle, évaluation des incertitudes et évaluation du modèle

Une discussion générale (chapitre 8) présente ensuite les forces et faiblesses de la recherche présentée, et met celle-ci en perspective dans le contexte actuel et futur de l'évaluation des impacts liés à l'utilisation de l'eau dans une approche cycle de vie.

Les trois annexes présentent les Informations Supplémentaires publiées avec chacun des articles présentés dans les chapitres 4, 5 et 6 respectivement, et référencées dans le texte des articles en question par *SI,* complémenté par un document Excel joint à cette thèse pour le premier article présenté au chapitre 4.

CHAPITRE 3 ARTICLE 1: CATEGORIZING WATER FOR LCA INVENTORY

3.1 Introduction

3.1.1 Background

Water use impacts assessment is currently undergoing significant changes. Until recently (Boulay et al. 2011b; Frischknecht et al. 2008; Milà i Canals et al. 2009; Pfister et al. 2009), there were no methods or guidelines to assess water use impacts in LCA, and only the volume of withdrawn water was listed in inventory databases. This impact assessment method development is therefore leading the evolution in water inventory requirements.

Inventory analysis involves collecting input and output data for all unit processes included in the scope of the assessment. From a water perspective, this translates into assessing the quantity, quality, type of resource (ground or surface water) and geographical location of the water that is withdrawn and released. These key characteristics will affect the functionality of the water – a loss of which would generate environmental impacts. Water functionality can be lost either through consumption (water is unavailable for use in the same watershed) or degradation (water is too contaminated to be used for a specific function) (Bayart et al. 2010b). Current databases such as ecoinvent (Frischknecht and Jungbluth 2004) and LCA Food (Nielsen et al. 2003) only distinguish the water source, at best differentiating between lake, river, ground, sea, sole and cooling/turbine water, without any quality differentiation. The purpose of this article is therefore to advance a functionality-based regionalized inventory method allowing impact assessment associated with quality degradation and consumption.

Bayart *et al.* (2010) propose “that the inventory flows represent a set of water types each representing an elementary flow with its own characterization factors”. They add that these water types should be differentiated based on their source and quality and should therefore be described by a set of quantitative values. However, the lack of quantitative methodology was highlighted in a recent assessment of water use in red meat production in Australia (Peters et al. 2010) where a qualitative classification of water of high, moderate, low or alienated quality is used. Bayart and colleagues mention two possible approaches to consider quality: either functionality-based or distance-to-target. The former “assesses to which users the water withdrawn and released is functional” and this should be based on international and accepted quality standards for each user. Water is considered functional if it can meet users’ needs without generating adverse effects or a change in activities. For example, the need for an extra influent treatment because of quality degradation caused by human intervention changes the activity. The impacts of this change should be accounted for in LCA through boundary extension. The distance-to-target approach can either be based on dilution or the energy required to treat the water to reach a reference water quality. Stewart and Weidema (2005) state that water quality is multidimensional and should not be defined in a single indicator but rather as a vector of water quality characteristics.

Water quality indexes and classifications have been advanced in many fields outside LCA, especially to describe and categorize surface water. While these methods serve their purposes, they are mainly geared towards ecosystem quality needs and not human uses. When describing the existing classification schemes, the WHO states: “As a general rule, the orientation of the classification system towards aquatic life implies that the category limits are more conservative than they would be if targeted at other water uses” (Enderlein, Enderlein et al. 1997). Government agencies, however, have shown interest in a function-based classification, and partial guidelines have been advanced. The European Economic Community has presented quality standards for surface water with respect to domestic uses and required treatment (EEC 1975). The Environmental Agency of Japan (Overseas Environmental Cooperation Center 1998) and Taiwan’s Environmental Protection Administration (Taiwan EPA 1998) have gone one step further and determined parameters and thresholds associated with several users, including domestic, industrial, aquaculture, irrigation, recreation and environmental conservation. Other classification schemes have been developed in India, Thailand and the UK (Enderlein, Enderlein et al. 1997). While these classifications can, at least partially, meet the needs of the LCA community, only a few parameters are defined and no information can be obtained on how the thresholds were determined. Moreover, the categories were created in a way that does not allow for much distinction between the users’ quality requirements. It is therefore impossible to assess functionality loss for individual users.

3.1.2 Objectives

This paper aims to create an appropriate inventory scheme/classification that allows quality to be considered and evaluated in a subsequent impact assessment in LCA through a functionality-based approach. The objective of the method is to create water categories defined by source and quality parameters. There should be as few categories as possible, yet sufficient enough to cover and differentiate the different user needs based on quality.

3.2 Methodology

The main steps in defining the water category are illustrated in Figure 3-1. They refer to the different parameters considered when defining water categories: the different users for which a category should be functional or not, the quality parameters that will define each category and their associated thresholds, and the sources of water to be considered. These steps are defined in detail in this section and led to the resulting water categories.

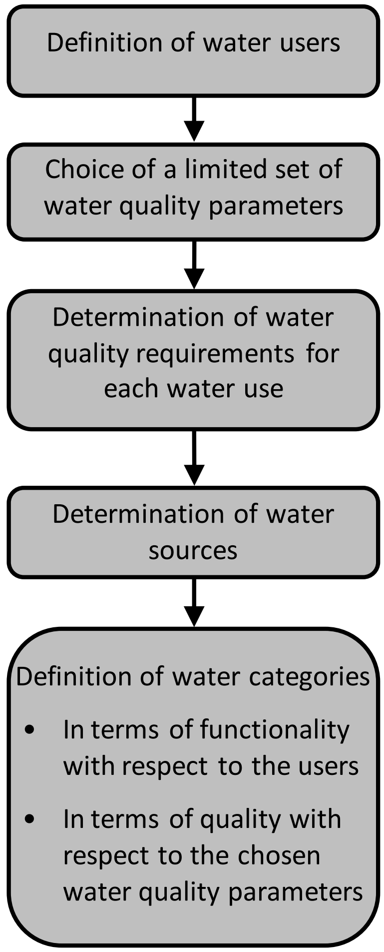

Figure 3-1 General methodology

3.2.1 Definition of water users

The first step consists in defining the water users. Bayart *et al.* (2010) identified seven main water users: agriculture, domestic users (drinking water), industry, transport, fisheries, hydropower and recreational users. However, for some of these activities, the quality of water that can be used varies greatly. This is especially the case for domestic, industry and agriculture. Sub-categories were therefore created to account for this diversity. In total, 11 distinct users were set out (see Table 3.1).

Domestic users differ regionally and across the world in their use of different water-treatment technologies based on available water quality. Available water quality therefore dictates the necessary treatment. However, while an increase in water contamination may not affect a user that already applies an advanced water treatment, this isn't the case for a user that relies on a simple disinfection method. Therefore, three domestic users were differentiated according to their drinking water production mode based on the three water-treatment levels.

Two types of water users were considered for agriculture: Agriculture 1 is the use of good quality irrigation water (needed to grow crops that are usually eaten raw) while Agriculture 2 is the use of relatively poorer quality irrigation water (needed to grow crops that are not eaten raw, such as cereals, and non-food agriculture). It should be noted that even though international standards for irrigation water exist (Ayers and Westcot 1985), national and/or regional standards or practices may vary greatly from one part of the world to the next. This is particularly true for microbiological standards, namely faecal coliforms. The standard ranges from the very strict Washington State standard for water reuse (Washington State 1997) to the use of polluted streams in Europe (UNEP Global Environment Monitoring System (GEMS) Water Programme 2009) or the use of untreated wastewater in many less-developed countries (Van der Hoek 2004; WHO and UNEP 2006). This translates into a wide range of infection risks for the populations that eat the crops and the agriculture workers who are more directly exposed. The descriptions of the users considered in this method are summarized in Table 3.1.

This definition of water users can be compared to the similar approaches of the Environment Agency of Japan (Overseas Environmental Cooperation Center 1998) and the Environmental Protection Agency of Taiwan (Taiwan EPA 1998) in their respective definitions of surface water quality standards. Japanese standards include three water supply categories, three fishery categories, three industrial water categories, one irrigation water category and one environmental conservation category. Taiwanese standards define three public water categories, one swimming category, two aquaculture categories, two industrial categories (manufacture and cooling) and one environmental conservation category. The European Communities (1975) also set out three domestic user types according to the treatment required to obtain drinking water. This distinction for users such as domestic and agriculture is important to avoid major misconceptions when later identifying the quality requirement for each user.

Table 3.1 Types of water users

Water user	*Definition*
Domestic 1	Domestic user performing no treatment or simple chemical disinfection to the water prior to use
Domestic 2	Domestic user performing a conventional chemical-physical treatment (coagulation or precipitation, solid removal process, disinfection) or equivalent treatment to the water prior to use
Domestic 3	Domestic user performing an advanced treatment (i.e. conventional treatment plus additional treatment (UV disinfection, adsorption, etc.)) or specific advanced treatment (reverse osmosis, nanofiltration, adsorption, ion exchange, desalination, etc.)or desalination to the water prior to use
Industrial	Industrial user (manufacturer) withdrawing available water and treating it to the required level
Cooling	Once-through cooling water energy production
Agriculture 1	Agriculture that requires good quality irrigation water
Agriculture 2	Agriculture that requires only poor quality irrigation water
Fisheries	Freshwater aquaculture and capture of fish
Hydropower	Hydroelectricity production
Transport	Transportation of goods through inland waters
Recreation	Recreational activities such as swimming and water sports

3.2.2 Choice of quality parameters

The second step consists in choosing the water quality parameters. This is not an easy task for two main reasons. The first is obviously the large number and diversity of parameters that can characterize the quality of a water stream, either natural or not. For example, the United Nations Global Environment Monitoring System (GEMStat) database (UNEP Global Environment Monitoring System (GEMS) Water Programme 2009), which aims at improving water quality data access and monitoring by providing quality data on surface and groundwater for 104 countries, includes 155 water quality parameters distributed as follows: physicochemical characteristics (22), microbiology (4), organic matter (8), nutrients (24), major ions (19), metals (56) and organic contaminants (22). Also, the Environmental Protection Agency of the United States (USEPA 1982) lists 126 priority organic and inorganic pollutants, and the United States Geological Survey (United States Geological Survey 2006) lists 95 emerging contaminants.

The second challenge lies in the fact that water quality characterization parameters may differ depending on the type of contamination, measurement methodologies or other field-specific issues (e.g. sodium adsorption ratio, SAR, in irrigation). For example, in the wastewater field, suspended particles are directly measured whereas, in the drinking water field, the relatively low concentrations of particles are indirectly measured through turbidity. Organic matter is usually measured in terms of biochemical oxygen demand (BOD) and chemical oxygen demand (COD) in the wastewater field, but these parameters are rarely used in the drinking water field because they would lead to values below the detection limit.

In order to define a workable list of parameters, choices had to be made according to the objectives of the water classification, as outlined by (Owens 2002), to define the functionalities of a water body. Except for transport and hydropower, for which quality is not an issue, the relevant parameters were based on international standards and guidelines. Table 3.2 is a sample of Table A2 provided in the Supplementary Information (SI) and lists the water quality parameters that define the functionality of water for irrigation, fisheries, drinking water production, recreation and cooling. Three parameter categories are set out: general parameters (which include microbial parameters), inorganic compounds and organic compounds. For each water use and quality parameter, the reference of the standard or guideline is indicated. In several cases, more than one reference is listed. The data were taken from different sources, namely WHO (WHO and UNEP 2006) for agriculture, FAO (Svobodová, Lloyd et al. 1993) for fisheries, WHO (WHO 2008) and the European Economic Community (EEC 1975) for drinking water, WHO (WHO 2003) and the government of Québec (MDDEP 2010) for recreation, EPRI (EPRI 2003) for cooling water and Taiwan EPA (Taiwan EPA 1998) for several users.

In the last column of Table 3.2 (and A2), the distinction is made between the parameters retained to define water categories and those that were not. The rationale behind each selected or discarded parameter is also briefly described. Toxicity (humans, plants or fish) is the main justification for parameter selection based on the guidelines for each user and which therefore define water functionality. Other selected parameters include indicators for scaling or clogging potential and aesthetic parameters for drinking water. The latter are indirectly related to human health issues since an aesthetically unpleasant source of water may favour risky behaviours such as the use of a contaminated water source or long and inadequate water storage.

Not all microbial parameters were considered in this study. Microbial indicators such as faecal coliforms were preferred to reflect the reality of microbial monitoring. Also, there do not appear to be any established parameters pertaining to organic compounds for irrigation. This is because the WHO and UNEP (2006) have defined thresholds for several inorganic compounds but none for organic compounds. This may be explained by the fact that, despite the concern over chemicals, most known illnesses relate to microbial contamination, and surveillance systems seem to solely focus on potential causes of human illness (Todd 2008). Phosphorus was not retained to evaluate drinking water functionality since it does not appear in drinking water standards. The microcystin-LR (cyanobacterial toxin) concentration was selected instead because it is part of the WHO guidelines for drinking water quality (WHO, 2008). However, it has been shown that relatively high concentrations of phosphorus in water (among other factors) favour the growth of cyanobacteria. Regional correlations between microcystin-LR and phosphorus concentrations may be established, as shown by Giani et al. (2005) for southern Québec. This type of relationship could be used to estimate the microcystin-LR concentration from the phosphorus concentration – the latter being more readily available. However the exclusion of dilution effects would here strongly affect the results.

Table 3.2 Sample of references and parameter selection rationale (complete in Table A2, Supplementary Information)

Parameter	Agriculture	Fisheries	Drinking water	Sources for drinking water	Recreation	Cooling	Parameter selection/Rationale
General parameters							
Faecal coliforms		TAI98	WHO08	EEC75, TAI98	QUE10, TAI98, WHO06		Retained because it is a good indicator for faecal pollution
Suspended solids	WHO06, TAI98	TAI98		EEC75, TAI98	TAI98	EPRI03	Retained (aesthetic parameter for drinking water, indicator for clogging potential)
Total dissolved solids	WHO06		WHO08	EEC75			Retained (aesthetic parameter for drinking water, indicator for scaling potential); correlation with electrical conductivity
…	…	…	…	…	…	…	…
Inorganics							
Arsenic	WHO06	FAO93	WHO08	EEC75, TAI98			Retained (toxicity)
Cadmium	WHO06	FAO93	WHO08	EEC75, TAI98			Retained (toxicity)
…	…	…	…	…	…	…	…
Organics							
Benzene			WHO08				Retained (human toxicity)
Atrazine			WHO08				Retained (human toxicity)
…	…	…	…	…	…	…	…

3.2.3 Determination of water quality thresholds per user

The selection of water quality thresholds was mostly based on those provided by the aforementioned references. WHO standards define most thresholds for Domestic 1 since this type of water must meet drinking water quality standards after disinfection. The faecal coliform threshold was set according to North American mandatory filtration regulations (MDDEP 2006; USEPA 2004), which are also in line with EEC guidelines (1975).

For Domestic 2, thresholds for specific inorganic and organic contaminants are equivalent to WHO drinking standards since conventional treatments do not remove these contaminants. There are two exceptions to this rule: Fe and Mn, which can easily be removed through conventional treatments (Crittenden 2005). The faecal coliform limit is consistent with that of the EEC guidelines (EEC 1975). From a drinking water perspective, the limit also corresponds to what is considered to be moderately- to highly-contaminated water, since this parameter is not normally the treatment limiting parameter but rather an indicator of microbial contamination (Payment et al. 2000).

For Domestic 3, thresholds for specific inorganic and organic contaminants are ten times higher than those of the WHO drinking standards. This is based on the assumption that an advanced water treatment system may remove 90% of inorganic and organic contaminants. There are three exceptions to this rule: Na, Cl and SO_4, for which the seawater concentrations are considered limiting values. The same approach was applied for TDS, alkalinity and hardness. This recognizes that desalination is part of the advanced treatments available for Domestic 3. As for Domestic 2, the faecal coliform limit is consistent with that of the EEC's guideline (1975) as well as what is considered to be highly-contaminated water from a drinking water perspective (USEPA, 2004; MDDEP, 2006). It also corresponds to well-treated wastewater (secondary treatment with disinfection), which is generally considered to be the highest contamination level water that can be used as a water source (MDDEP 2006).

For the three domestic users, all of the parameters that were not considered in the WHO standards were based on EEC guidelines (EEC 1975) and the Taiwanese classification (Taiwan EPA 1998). For the three domestic users, the threshold for oil and grease content was set as the detection limit of the partition-gravimetric method, which is 1.4mg/l (APHA AWWA and WEF 1998), even if lower values than this detection limit are mentioned in EEC guidelines (EEC 1975).

For agricultural uses, thresholds for specific inorganic contaminants and for most conventional parameters were set according to WHO and UNEP (2006) criteria. Plant toxicity, which occurs under very specific conditions or for very specific plant species, was not considered since it has very limited impacts in terms of water functionality for agriculture (Ayers and Westcot 1985). Regarding microbiological contamination, thresholds for Agriculture 1 and 2 categories were set from the risk reduction goals proposed in the WHO guidelines for wastewater use in agriculture (WHO and UNEP 2006), for unrestricted irrigation and restricted irrigation, respectively. These guidelines define the acceptable risks for crop consumers and agricultural workers and the minimal treatment that should be done on wastewater in order to reuse it for irrigation purposes.

It therefore indirectly sets the quality requirements for irrigation water. This indirect approach was chosen because, to our knowledge, there is no international consensus on microbial contamination thresholds for irrigation water, as confirmed by the experts consulted during this study.

The water requirements for industry are assumed to be equal to the thresholds for the Domestic 2 water category. While water quality needs for industry vary greatly, one can suppose that industries have adapted to the available water quality. This therefore implies two assumptions: 1) surface water used by industries around the world is of functional quality for Domestic 2 users or better; and 2) industries have adapted to a water quality that meets their needs. This can be justified by considering that a water-consuming industry would not likely settle in an area where desalination is required. Cooling processes are excluded from this hypothesis and 1998 industrial standards from the Electric Power Research Institute (EPRI) were used to identify the few thresholds that apply to cooling water (EPRI 2003). However, some were adjusted based on typical seawater composition, which can also be used for cooling purposes.

Thresholds for water use in fisheries were mostly defined by reviewing the factors affecting fish health, as published by the FAO (Svobodová, Lloyd et al. 1993). The Taiwanese surface water quality standards (Taiwan EPA 1998) were also used to set these thresholds when not specified, such as the faecal coliform threshold for Aquaculture . Whenever a maximum concentration range is recommended by the FAO (1993) and the lower limit for this range is extremely low ($2x10^{-4}$mg/l for cadmium, for example), the functionality criterion becomes the absence of this contaminant. This is the case for cadmium, chromium, lead and zinc, which can be highly toxic to fish. The same approach was used for pesticides, which can be extremely toxic to fish. The threshold for oil and grease content was set as the detection limit of the partition-gravimetric method, which is 1.4mg/l (APHA AWWA and WEF 1998), even if lower values are stated in FAO guidelines (Svobodová, Lloyd et al. 1993).

WHO guidelines for safe recreational water environments (WHO 2003) were used to set thresholds for recreational use. As per the WHO’s risk analysis-based suggestion, thresholds for specific inorganic pollutants were set at ten times the thresholds for drinking water. These guidelines specify that the faecal coliform count is a very good indicator of fresh water microbial contamination. Because of its common use in many fields, this indicator was therefore preferred over intestinal enterococci, which is a very good indicator of fresh and marine water contamination but is less common in other applications such as drinking water or irrigation.

3.2.4 Determination of water sources

Bayart *et al.* (2010) and Owens (2002) suggest including surface and groundwater as distinct sources. This distinction is important since the two types of water do not necessarily serve the same users and are not available in the same amounts throughout the world. They therefore represent different scarcities – an important factor in water use impact assessment (Bayart, Margni et al. 2010). In addition, we propose the inclusion of rain as an extra source of water to enable the life cycle inventory accounting of rainwater harvesting. This would prevent the water from reaching ground and surface water bodies as well as its potential subsequent extraction. Sea water was not categorized since it can be classified as poor-quality surface water, as described below.

3.2.5 Definition of water categories

Eight water quality categories were created from all the quality thresholds obtained for each user. User functionalities were identified as either sensitive to microbial contamination (represented by faecal coliforms) and/or toxic contamination (most other parameters). Users were then grouped based on the level of contamination they could handle (low, medium, high). While Agriculture 1 and Recreation are more sensitive to microbial contamination, toxic contamination is more crucial for Fisheries. Domestic 1 is very sensitive to both types of contamination, while Agriculture 2 and Domestic 2 (and consequently Industrial as well) are moderately sensitive to both. Finally, Domestic 3 and Cooling are the least sensitive, except for Transport and Hydropower, which do not present any quality restrictions. For each group, the more critical value was chosen for each parameter, ensuring that all user thresholds are respected within a group. This is important to ensure that all users can safely use a water category that is functional for them. However, it may also be restrictive for some users who could actually use lower-quality water for certain parameters. The consequences of this are discussed below. The quality categories were then associated with water sources to create 17 categories (see Table 3.4). Quality parameters have not been set for rainwater since rain is considered to serve all users.

Categories were created in such a way that all the quality parameters of a given water stream must be below the thresholds for a given water category in order to be put in a category. When one parameter of the water to be categorized does not meet the specified limit for a water category, the category is no longer relevant, since only one excess contaminant can severely restrict a user functionality (high faecal coliforms, high lead content, etc.). However, while many parameters are defined for each category, it is not necessary to know the values for all parameters before categorizing a water stream. This is discussed in section 3.3.

3.2.6 Application

To show applicability, the water category was evaluated for the world's main surface waters using available data. Specific values are presented for the Amazon basin, but all watersheds with available data were characterized. Data from the GEMStat database (UNEP Global Environment Monitoring System (GEMS) Water Programme 2009) was used. This database is currently the only available international water quality database. Although data frequency in both time and space is not fully consistent (countries report quality parameters of their choice), data are available for several countries or at a watershed scale. Only parameters with a minimum of ten samples were considered in the characterization, and the median of all samples was used for each parameter. This value was then compared to the resulting category thresholds (tables 3.3 and A3), and possible water categories were identified for each parameter. The overall category for a water type corresponds to the best quality common to all parameters.

3.3 Results and discussion

3.3.1 Determination of water quality thresholds for each water use

The chosen thresholds are reported in Table A3 in the SI (a sample of which is presented in Table 3.3). Each threshold is referenced, and a short rationale or comment was added whenever necessary. When no clear standard, guideline or recommendation was provided, no threshold was retained.

The procedure described above, based on the available references, was used to select the thresholds for each value, with four exceptions described herein. First, the limiting value for BOD_5 was set at 5 mg/L even though Taiwan EPA (Taiwan EPA 1998) adopted a lower value for its Domestic 1 and Swimming categories. This is based on the fact that BOD_5 less than 5 mg/L is very difficult to measure because of the uncertainty associated with the BOD analysis. Second, one exception was made to the Cooling thresholds of EPRI guidelines for the $CaxSO_4$ threshold, which is instead based on typical seawater composition. The value of this parameter is indeed equal to $10^6 (mg/L)^2$ for seawater, whereas the value recommended by EPRI is $5x10^5$ $(mg/L)^2$. Third, whereas iron could be regarded as an aesthetic parameter for Domestic 1 due to the taste and color it can give to drinking water, no limit was assigned in the WHO guidelines. It was therefore omitted from drinking water requirements. Lastly, the WHO's (2006) proposed agriculture pH requirements are between 6.5 and 8.4 , but, the lower value was extended to 4.5 to include naturally occurring rainwater pH range (Charlson and Rodhe 1982), since it would be incoherent to characterize water as being too acidic for irrigation if it has the same pH as natural rainwater.

The selected approach favoured the use of an exhaustive list of parameters, since no rationale would support limiting the proposed thresholds to fewer parameters. While it is obvious that not all of the parameters are known for any one water use, it is best to provide a threshold for when they become available to ensure that the characterisation is as exhaustive and robust as possible (i.e., based on as many available quality parameter as possible to describe water quality). While a characterisation is possible with only a few parameters, the more parameters are used, the more relevant the water category becomes.

Table 3.3 Sample of quality thresholds for each user: Values and detailed reference (complete in Table A3 in the SI)

Parameter	Units	Agriculture 1	Agriculture 2	Fisheries	Domestic 1	Domestic 2	Domestic 3	Recreation	Cooling
General parameters									
Faecal ccoliforms	UFC/100 ml	100 (based on risk analysis in WHO6)	10000 (based on risk analysis in WHO6)	10000 (TAI98)	20 (EEC75; it also corresponds to a threshold for mandatory filtration in the Province of Québec, Canada)	2000 (EEC75)	20000 (EEC75)	200 (QUE10 & WHO06; TAI98's threshold for swimming is 50)	
Suspended solids	mg/l	100 (WHO06, TAI98)	100 (WHO06, TAI98)	40 (TAI98)	25 (TAI98 & EEC75)			25 (TAI98)	300 (EPRI03)
Total dissolved solids	mg/l	2000 (WHO06; threshold for severe effect)	2000 (WHO06)		500 (EEC75)	500 (EEC75)	40000 (seawater is considered as a limiting case)	40000 (seawater is considered as a limiting case)	
…	…								
Inorganics									
Arsenic	mg/l	0.1 (WHO06; wide toxicity range from 0.05 to 12)	0.1 (WHO06; wide toxicity range from 0.05 to 12)	3 (FAO93; lower limit of the toxicity range)	0.01 (Drinking water standard of WHO08)	0.01 (Drinking water standard of WHO08)	0.1 (ten times the WHO08 standard; EEC75)	0.1 (ten times the WHO08 standard)	
Cadmium	mg/l	0.03 (WHO06: conservative limit due to its potential for accumulation)	0.03 (WHO06: conservative limit due to its potential for accumulation)	Absence (FAO93 toxicity range from 0.0002 to 0.001)	0.003 (Drinking water standard of WHO08)	0.003 (Drinking water standard of WHO08)	0.03 (ten times the WHO08 standard)	0.03 (ten times the WHO08 standard)	
…									
Organics									
Benzene	mg/l			Absence (FAO93 consider the lighter oil fractions as more toxic than heavier fractions)	0.01	0.01	0.1	0.1	
Atrazine	mg/l				0.002	0.002	0.02	0.02	
…									

3.3.2 Determination of water categories

The seventeen resulting water categories (eight surface water, eight groundwater and one rainwater) are presented in tables 3.4 and 3.5. The different categories based on quality, associated sources and users for which each category is functional are identified in Table 3.4. Table 3.5 presents a sample of Table A5 in the SI, providing an exhaustive list of thresholds for each water category.

Table 3.4 Water category functionalities per user (S = Surface water, G = Groundwater)

Quality	***1***	***2a***	***2b***	***2c***	***2d***	***3***	***4***	***5***	***Rain***
Sources	*S or G*	*S or G*	*S or G*	*S or G*	*S or G*	*S or G*	*S or G*	*S or G*	*Rain*
Quality level	*Excellent*	*Good*	*Average*	*Avg-Tox*	*Avg-Bio*	*Poor*	*Very Poor*	*Un-usable*	
Contamina-tion	Low microbial low toxic	low microbial medium toxic	Medium microbial medium toxic	Low microbial high toxic	High microbial low toxic	High microbial medium toxic	High microbial high toxic	Other	N/A
Dom 1	√	**X**	**X**	**X**	**X**	**X**	**X**	**X**	√
Dom 2	√	√	√	**X**	**X**	**X**	**X**	**X**	√
Dom 3	√	√	√	√	√	√	√	**X**	√
Agri 1	√	√	**X**	√	**X**	**X**	**X**	**X**	√
Agri 2	√	√	√	√	√	√	**X**	**X**	√
Fisheries	√	**X**	**X**	**X**	√	**X**	**X**	**X**	√
Industry	√	√	√	**X**	**X**	**X**	**X**	**X**	√
Cooling	√	√	√	√	√	√	√	**X**	√
Recreation	√	√	**X**	√	**X**	**X**	**X**	**X**	√
Transport	√	√	√	√	√	√	√	√	√
Hydro	√	√	√	√	√	√	√	√	√

√: Functional **X: Non-functional**

Table 3.5 Sample of water category threshold values (Complete in Table A5 of the Supplementary information)

Parameter	Units	*1*	*2a*	*2b*	*2c*	*2d*	*3*	*4*	*5*
General parameters									
Faecal coliforms	UFC/100 ml	20	200	2000	200	10000	10000	20000	
Suspended solids	mg/l	25	25	100	25	40	100	300	
Total dissolved solids	mg/l	500	500	500	2000	2000	2000	40000	
…	…								
Inorganics									
Arsenic	mg/l	0.01	0.01	0.01	0.1	0.1	0.1	0.1	
Cadmium	mg/l	0	0.003	0.003	0.03	0	0.03	0.03	
…									
Organics									
Benzene	mg/l	0	0.01	0.01	0.1	0	0.1	0.1	
Atrazine	mg/l	0.002	0.002	0.002	0.02	0.02	0.02	0.02	
…									

While these categories were created by grouping user's quality thresholds and choosing the lowest value, as explained above, certain parameters were adjusted to avoid incoherence. These adjustments are described below.

The coliform parameter was adjusted to 200 CFU/100 mL for Agriculture 1, instead of the 100 mg/L value previously identified, to harmonize it with recreational uses. This is justifiable as there is no solid or uniform international guidelines for good quality irrigation water and the limit of 200UFC/100 mL faecal coliforms for Agriculture 1 comes from British Columbian standards (2003). The WHO presents its information based on a debatable risk assessment based on the acceptable risk of illness – a definition that may differ from one country to the next.

The sodium adsorption ratio is an important parameter for irrigation, but its guideline is a function of the conductivity or total dissolved solids (TDS) parameter. At this point, it is recommended to refer to the WHO and UNEP (2006) for the threshold assessment of the sodium adsorption ratio. The relation between these parameters and the acceptable zone for agriculture functionality is presented in the SI.

Cooling water guidelines are proposed for aluminium, copper and iron but are more severe than for most other users and were therefore omitted. This is based on the fact that Cooling is for once-through cooling water with a large range of water qualities. The specialists consulted as part of this study agree that a higher than recommended metal concentration is likely to, at worse, shorten the service life of the cooling equipment or require more frequent cleaning. This was considered to be within the limit of functionality for cooling purposes (Klvana 2010). This hypothesis seems acceptable if one agrees that drinking or irrigation water can be used for once-through cooling in power plants.

Lastly, dyes were excluded since little information was found on thresholds requirements, aside from the qualitative toxicity information associated with malachite green (Svobodová, Lloyd et al. 1993).

3.3.3 Application

From the example described above, the water category for the Amazon watershed is classified as S3. As shown in Table 3.6, the combination of high fecal coliforms and suspended solids drive this classification. Results for all available watersheds worldwide are presented in Figure 3-2 and the associated data and sources supporting this classification are presented in the SI. While data for all watersheds was not available, hypotheses were set out to use the closest available data, whenever possible. This classification can be used to determine the water quality entering a process (withdrawn). Similarly, the water released from a given process into the environment (e.g. an industrial effluents), can be classified combining the amount of chemicals "released to water" as reported in existing LCI databases with the volume of water being released. This latter information is traditionally not given by LCI databases. If no primary data on the released volume are available, a hypothesis could be made on the fraction of withdrawn water that is evaporated based on industrial standards. For example, Shiklomanov et al. (2003) proposes a range of 5-20% evaporation in industry.

Table 3.6 Classification of available water in the Amazon basin

Parameter	Median Value	Number of samples	Units	Lowest accepted threshold (identified from table A5)	Accepted water category
Nitrites	0	434	mg/l	3	All
Nitrates	0.073	734	Mg N/l	50	All
Ammonia	0.2	37	mg/l	0.3	2b, 2d, 3,4,5
pH	7.9	37		4.5 – 8.4	All
Suspended solids	64	45	mg/l	100	2b,3
Phosphorus (total)	0.030947	491	mg/l	0.1	All
Sulfate	5.243	259	mg/l	500	All
Zinc	0.0042	290	mg/l	2	2a,2b,2c,3,4,5
Arsenic	0.0007	332	mg/l	0.01	All
Fecal coliforms	3500	33	UFC/100ml	10000	2d, 3, 4, 5
	Resulting water category				3

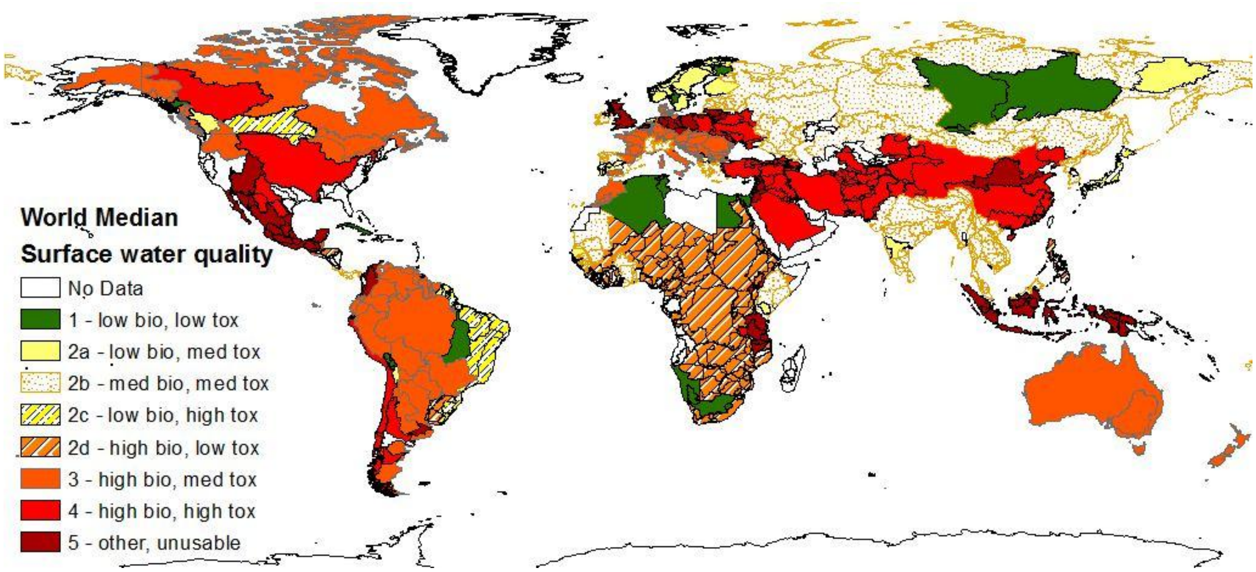

Figure 3-2 Classification of the world's surface water

Not all parameters may be available when categorizing an industrial effluent and, it is therefore important to make an expert decision to determine the parameters that are most likely to be affected by the industry. US EPA guidelines on effluent limitations may be used to identify the sensitive parameters per industrial sector (2010). For example, when evaluating a pulp and paper effluent, parameters such as BOD and suspended solids will be more important than faecal coliforms. The effluent quality is highly dependent on the industry and national legislation, and, while industrial effluents may often correspond to category 5 as sampling has demonstrated in Pakistan and Malawi (Phiri et al. 2005; Sial et al. 2006), it may be otherwise in other regions or for other industries. For example, the pulp and paper industry in Québec, Canada, reports BOD and MES averages that meet water category 3 criteria (18,4 mg/L and 30 mg/L, respectively) making the water functional for Domestic 3, Agriculture 2 and Cooling if the other contaminants also conform. A simple Excel tool that evaluates the resulting water category based on input parameters is provided in the SI.

Water quality regulations and guidelines vary from one country to the next, and while this study attempts to advance widely accepted functionality-based categories, it is clear that certain parameters may be subject to discussion. Moreover, as with any threshold-based methodology, one should exercise good judgment when faced with a value that is close to the threshold. In cases of doubt, referring to the original source used to determine the threshold may help.

When creating a limited number of categories that group different combinations of eight users with specific requirements, gaps that will sometimes misrepresent the actual users for which a category is functional are unavoidable. For example, one parameter may cause a much lower quality category to be chosen and lead to an overly restrictive list of users for which the water is functional and eventually to the overestimation of the impacts for the process effluent or an underestimation for an influent. Such cases are especially prone to occurring when guidance is not provided for some parameters for a specific user when grouped in the same category as one for which guidance is provided. However, a lack of guidance does not necessarily mean a high tolerance to a parameter but rather a lack of consensus or information on the toxic effect. Other cases are generated from threshold differences between users that are considered to have the same sensitivity to toxicity (e.g. Domestic 1 and Fisheries). These gaps are unavoidable with a water category strategy, unless a number of categories as large as 256 (2^8, for eight users with different quality requirements) is applied.

In order to implement this approach in LCI databases, quality and quantity information on water entering and leaving each process is necessary. Elementary flows are defined as each corresponding to a water class. While this may seem insurmountable, much of this information is actually already available. In most disaggregated ecoinvent processes (Frischknecht and Jungbluth 2007), the inventory already indicates the volume and source of water entering the process, and the amount of chemical emitted into water. The quality of water entering the process can be taken from the classification proposed in this paper through Figure 3-2 and based on GEMStat data (2009). Therefore, only the released quantity of water must be collected to make the approach operational. When not available, hypotheses can be formulated on the percentage of water evaporated from a process and then deducted from the water withdrawn. Alternatively, we suggest collecting generic industry data on effluent quality and assessing a default water category for each industry type. However, as previously discussed, even within an industry, the effluent quality can vary depending on geographical location and the level of regulation. Effluent regionalization could be carried out by allowing the user to choose between good, average or bad quality effluents, each representing the extremes and the average of worldwide practices for a specific industry.

These water categories can serve as elementary flows in a database such as ecoinvent (Frischknecht and Jungbluth 2007) and enable the quantification of the functionality loss associated with withdrawn and consumed water or the water released at a lesser quality. This functionality loss should also consider scarcity and the distribution of the different users sharing the same resource in order to assess the actual m^3 of water whose function has been lost, as carried out in Boulay et al. (2011b), whose methodology assesses impacts on human health or the amount of water needed to be compensated based on a loss of functionality. One interim use for this approach would be to apply the categories as a quantitative enhancement in LCI reporting instead of just total water use or the relatively qualitative categories in the Australian example cited above. Such results of water withdrawn and discharged by category can be used in a MCDA decision making framework until such time as the mechanistic linkages and data necessary for full integration in proposed endpoint models are established.

3.4 Conclusions

This method first determined eleven different human users based on the difference in water quality each of them require. In total, 137 quality parameters and their associated thresholds were then used to guide the creation of 17 distinct water categories based on the source, quality and potential users. These categories were created in an attempt to operationalize the functionality-based water categories proposed by Bayart et al. (2010). The resulting inventory method fills the existing gap in LCA associated with the assessment of the potential impacts of the degradative use of water by providing the elementary flows necessary to evaluate a loss of functionality for human users. The result constitutes a step forward in extending these classifications for worldwide acceptance as compared to existing classifications (EEC 1975; Overseas Environmental Cooperation Center 1998; Taiwan EPA 1998). Moreover, it was found that only one additional data are required to operationalize this methodology in existing databases: the volume of water that is released. This latter parameter could, however, be approximated based on industrial evaporation hypothesis.

While this article has explored water category development, limitations regarding the feasibility of grouping the quality requirements of different users into a manageable amount of categories were determined. Along with the simplicity of water categories, choices and simplifications have to be made and may lead to the overestimation or underestimation of impacts when used with an impact assessment method. These limitations could be overcome if functionality-based inventory flows were to be considered directly, avoiding the water category simplifications. Inventory information would then provide a volume of water, the source, and the different users it can be functional for. This information would be obtained from a quality parameter comparison assessment with the user thresholds proposed here, resulting in a functionality vector in which each element represents the functionality or not (1 or 0) of the water type for a particular user. The functionality comparison between influents and effluents could then enable impact assessment from functionality loss, as already advanced by Boulay *et al.* (2011b), but avoiding the error associated with modeling gaps from water categories.

Another limitation associated with assessing user functionality is the lack of internationally-recognized thresholds for many users and parameters. While there is generally good guidance for domestic users and fisheries, agriculture quality requirements remain few and inconsistent throughout the world. Moreover, the threshold approach can always be criticized at values near the threshold, since reality is rarely black or white when it comes to water functionality for a specific use. However, at this point, method developments do not capture the subtleties of the impacts associated with quality degradation, apart from functionality loss.

CHAPITRE 4 ARTICLE 2: REGIONAL CHARACTERIZATION OF FRESHWATER USE IN LCA: MODELING DIRECT IMPACTS ON HUMAN HEALTH

4.1 Introduction

Vital to life, water is a unique natural resource. While it cannot disappear, it can be made unavailable to specific users (ecosystems, human users and future generations (Mila I Canals et al. 2008)) either by displacement or quality degradation. This change in availability can lead to environmental impacts. Life cycle assessment (LCA) is a methodology that quantifies potential environmental impacts for comparative purposes in a decision-making context (Hauschild 2005). It is used by governments and industries alike to support their impact reduction strategies. In LCA, the resources consumed and emissions generated by a product or service over its entire life cycle are compiled, characterized and grouped into different impact categories using formal models. While potential environmental impacts from pollutant emissions are characterized in LCA, impacts from water unavailability are not yet fully quantified.

Based on a review of existing methods to characterize water use impacts in LCA, Bayart *et al.*(2010b) suggested a general framework that considers three main impact pathways leading to water deficits for human uses, ecosystems and future generations (freshwater depletion). This paper focuses solely on human uses and proposes a method that assesses the consequences of insufficient access to water for human needs. Bayart and colleagues (2010b) distinguish the following human users: domestic, agriculture, industry, fisheries, hydropower, transport and recreation. A decrease in water availability for human uses can lead to impacts on human health. If there is sufficient economic wealth in the area, users will adapt to the lack of water by compensating with a backup technology (e.g. desalination, water or goods import, etc.). The impacts of these processes can be assessed in existing impact categories through traditional LCA and included in the results by expanding the product system under assessment (i.e. the system for which water use is being studied).

In Bayart *et al.*'s (2010b) review, only one method addresses the impact pathway leading to impacts on human health. Pfister *et al.* (2009) proposed a breakthrough by quantifying in DALY (disability-adjusted life years) the impacts of malnutrition stemming from a lack of water for agriculture and addressing spatial and temporal variations for over 10 000 watersheds. These regionalized impacts include a scarcity parameter that accounts for seasonal variations. More recently, Motoshita et al. (2010a) also proposed a methodology to assess the human health impacts of water scarcity on domestic users. Outside the LCA field, Fry et al. (2010) assessed the avoided health impacts from increased water availability for domestic uses. While these methods can assess potential impacts for consumptive water use, they do not consider the fact that the users' adaptation capacity may lead to compensation instead of direct human health impacts. In addition, they do not address the consequences of change in water availability due to a degradative use limiting the potential uses (i.e. functionality) of the resource. Water is considered degraded when it is returned to the body of water with a lower quality, while a consumptive use refers to evaporation, integration within a product or the return of the water to a different watershed or the sea (Falkenmark 2000). As most industrial and domestic water uses can be considered degradative, it is important to account for the fact that returned polluted water will not provide the same function as clean water, but may still provide some in comparison to consumed water (Boulay et al. 2011a). Also, none of the existing methodologies consider in-stream users, such as fisheries, nor do they distinguish between surface water and groundwater use.

The objective of this work is to develop a characterization model that assesses the potential impacts generated by a loss of water availability or functionality for human uses caused by consumptive or/and degradative use. These potential environmental impacts are modeled using two distinct and complementary impact pathways: one leading to direct human health impacts (in DALY) caused by malnutrition and disease and the other modeling compensation scenarios to overcome water shortage. This paper focuses on modeling the first impact pathways and will only discuss the second aspect within a comprehensive LCA perspective.

4.2 Method

GENERAL FRAMEWORK DESCRIPTION: Figure 4-1 illustrates the impact pathway from a water use inventory to direct and indirect impacts generated by a deficit for human uses. Boulay *et al.* (2011a) determined two important concepts needed to identify water deficits for human uses: human users and water categories, the latter defining water of sufficient quality and adequate source to be functional for the former. Human users are identified according to domestic use, agricultural use, fisheries (here referring to catch and aquaculture), industry, cooling, transport, hydropower and recreational use. Three domestic user categories were created in an effort to account for the different qualities of water used for domestic purposes based on local availability, each quality requiring different levels of treatment. Each user is further categorized as in-stream and off-stream according to Bayart *et al.* (2010b) and as shown in Figure 4-1. In this paper, off-stream hydropower was not considered. The water categories represent the 17 possible elementary flows described by source (surface, ground or rain) and water quality (see Supplementary Information, referred to as SI in the text). Elementary flows describe the exchanges between the assessed product system and the environment. Water quality is evaluated based on a series of parameters and their thresholds, which determine the users for which a specific water category is functional (sufficient quality and adequate source). For example, groundwater is generally not functional for transport, hydropower or recreational use, and poor quality water is not functional for clean domestic water users (domestic 1) who do not treat their water before consumption (Boulay et al. 2011a). Water quality ranges from 1- excellent (functional for all users) to 5 - unusable (only functional for transport or hydropower). When facing water scarcity, users can adapt and compensate the loss of functionality previously provided by the water resource (water treatment, import of water or goods). Alternatively, if the socio-economic situation is not favourable enough, users will directly suffer from a reduction in available water. This could lead to disease caused by limited access to domestic water (lack of hygiene and access to safe water) or agricultural productivity losses leading to malnutrition (World Water Assessment Program 2006).

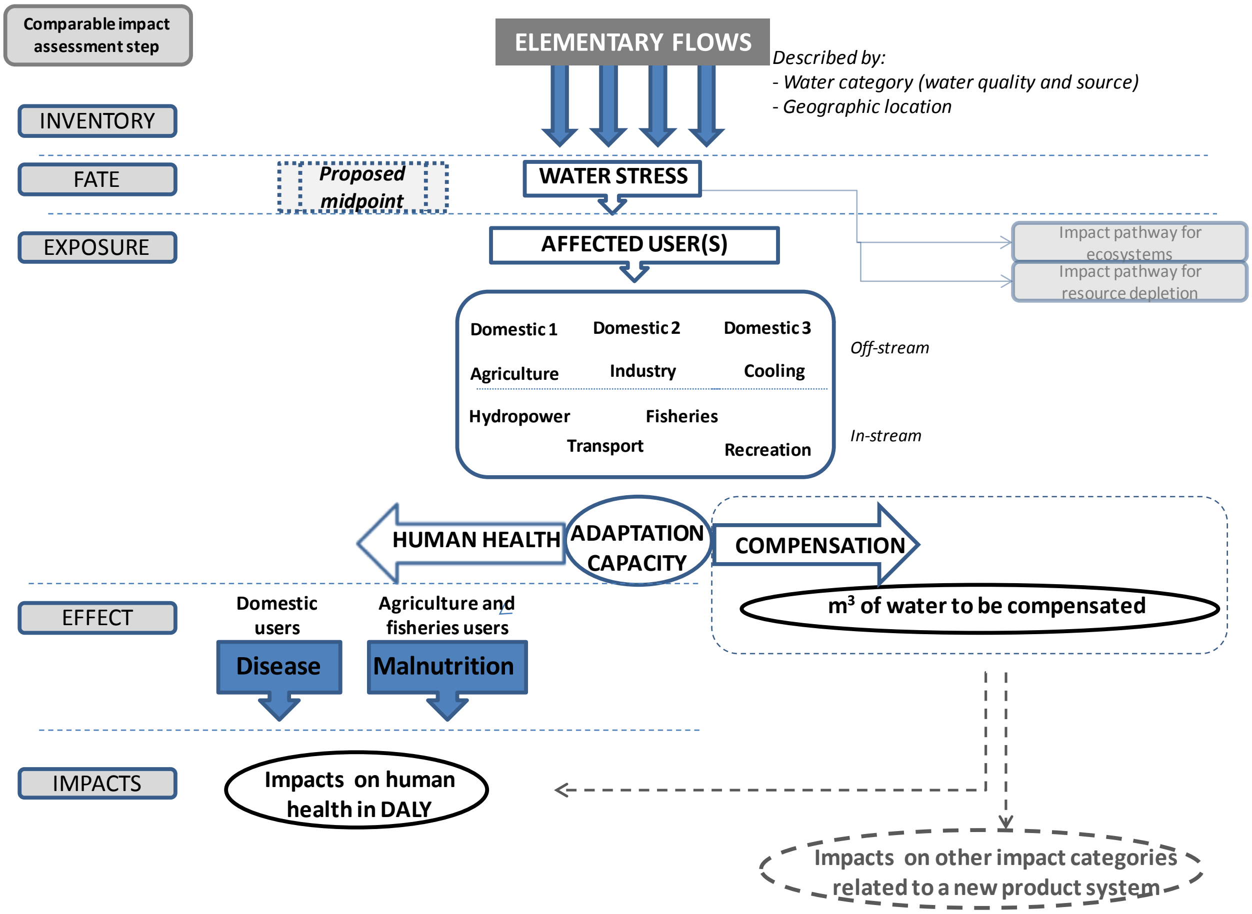

Figure 4-1 Water use impact pathways for human users leading to compensation or human health impacts

MODEL DESCRIPTION **-** The model characterizes the impacts associated with the amount of water entering and leaving the product system for a given category. Potential impacts on human health are calculated based on the difference between resource extraction and emission into the environment, as per Equation 4.1.

$$HH_{impact} = \sum_{i=1}^{17}(CF_i \times V_{i,in}) - \sum_{i=1}^{17}(CF_i \times V_{i,out}) \qquad \text{Equation 4.1}$$

Where, HH_{impact} expresses the human health impacts in DALY, CF_i is the characterization factor of water category i for the human health impact category (in DALY/m^3 of water category i) and V_i (in and out) is the volume of water category i entering and leaving the process or product system, i.e. the elementary flows (in m^3).

Characterization factor CF_i includes three main components that can be compared to the three factors traditionally used to define emissions-related impact categories: 1) fate, 2) exposure and 3) effect. As described in Equation 4.2, they respectively represent: 1) local water stress, 2) the extent to which user(s) will be affected by a change in water availability and their ability to adapt to this change, and 3) the human health impacts of a water deficit for user j.

$$CF_i = \sum_{j=1}^{10} (\underbrace{\alpha_i}_{\text{FATE}} \times \underbrace{U_{i,j}(1 - AC)}_{\text{EXPOSURE}} \times \underbrace{E_j)}_{\text{EFFECT}} \qquad \text{Equation 4.2}$$

Where α_i expresses the water stress index of category i (dimensionless), $U_{i,j}$ the user(s) j that will be affected by the change in water category i availability (dimensionless), AC the adaptation capacity (dimensionless) and E_j the effect factor for user j (DALY/m³). The following section describes these three components.

FATE: WATER STRESS (α_i) – In Equation 4.2, the stress index represents the level of competition among users due to the physical stress of the resource, addressing quality and seasonal variations and distinguishing surface and ground water since these two types of resources often do not present the same scarcity in a same region and may not serve the same users. In this paper, the scarcity parameter α^*_i for surface water was first calculated based on the CU/Q90 ratio proposed by Döll (2009). A discussion on the underlying choice of this scarcity parameter is reported in the SI. The consumed water (CU) in the numerator was calculated using data from the WaterGap model (obtained from the developers (Alcamo et al. 2003a)). While no seasonal effects were taken into account for renewable groundwater resource availability (GWR), they were considered in the denominator for surface water by the Q90 parameter. The latter, the *statistical low flow*, represents the flow that is exceeded 9 months out of 10. It is therefore a lower value than the average or median flow and allows for the exclusion of the effect of very high flows (e.g. during monsoon season), since this water is rarely fully available unless extensive storage facilities are also available (Alcamo et al. 2000). However, the monthly discharge used for the Q90 assessment accounts for the presence of reservoirs.

The scarcity parameter α^*_i for surface and ground water is described in Equations 4.3 and 4.4.

$$\alpha *_{surface,i} = \frac{CU \times (1-f_g)}{Q90} \times \frac{1}{P_i} \quad \text{Equation 4.3}$$

$$\alpha *_{GW,i} = \frac{CU \times f_g}{GWR} \times \frac{1}{P_i} \quad \text{Equation 4.4}$$

Where CU represents the consumptive use in km³/yr, Q90 the statistical low flow in km³/yr, f_g the fraction of usage dependent on groundwater (obtained from WaterGap), GWR the renewable groundwater resource available in km^3/yr and Pi the proportion of available water that is of category i.

It should be noted that the less functional a water category is, the more abundant it will be, since all higher quality categories will also meet the category's functionality requirements. This is a consequence of water categories being defined by upper thresholds and not by ranges. They are functionality-based, and category 3 would therefore include 2 and 1, and so on. The availability of water for a given category is considered through parameter P_i in equations 4.3 and 4.4, which is the fraction of freshwater of category i or better available in a region. The proportion of each water category per watershed is evaluated based on data describing surface and groundwater quality from GEMStat (UNEP Global Environment Monitoring System (GEMS) Water Programme 2009). Since seawater may be considered very poor or unusable (as per water categories 4 or 5, see the SI for a detailed description), scarcity for these categories was considered to be null in regions with access to seawater, consequently considering infinite availability. When no water of a certain quality was available, no scarcity was calculated. The consumptive use was not available for each water category, which would ideally be needed to calculate the $\alpha *_i$ specific to each water type. Only total water consumption CU was available. Therefore it was assumed that the best quality water is consumed first, before lower quality water, thus resulting in higher scarcity for better quality categories. Alternatively, one could have assumed that water of different quality was consumed in proportion to its availability, which would lead to discarding the parameter Pi. The first proxy was chosen and this, in some cases, may lead to an overestimation of the physical scarcity of good quality water if lower quality water is consumed first. It would be best to use the consumptive use per water category, if it ever becomes available, instead of either of the two proxies. For the assessment of rain water use from harvesting or as green water, please refer to the SI.

The stress index (α_i) is then modeled in order to obtain an indicator ranging from 0 to 1 based on accepted water stress thresholds. Assessments of low, moderate, high and very high water stress are associated with water withdrawals of 10, 20, 40 and 80% of available water, respectively (Alcamo et al. 2000; Pfister et al. 2009; Vörösmarty et al. 2000c). Correlations between these withdrawals-to-availability ratios and consumption-to-availability ratios were generated (see SI) and used to establish corresponding thresholds of 10, 12, 18 and 40%, respectively. Because the stress index (α_i) is meant to reflect the competition between users, the α_i is set to result in 0 for low water stress (meaning that the consumption of 1 m^3 of water will not affect other users when water is abundant) and up to 1 for very high stress (meaning that each consumed 1 m^3 will deprive other competing users of 1 m^3). The in-between data were fit in an S-curve passing through 50% scarcity when the high-stress threshold is reached, as also proposed by Pfister et al. (2009) and shown in SI in equation S1.

MIDPOINT – WATER STRESS INDICATOR (WSI) - The proposed midpoint uses the availability of water and differentiates source and quality by weighing the stress of each water type. The results of each flow are aggregated in Equation 4.5 in the same way as in Equation 4.1 for the endpoint, here using the water stress α_i as a midpoint CF.

$$WSI = \sum_i(\alpha_i \times V_{i,in}) - \sum_i(\alpha_i \times V_{i,out}) \qquad \text{Equation 4.5}$$

Where WSI expresses the midpoint result in m^3 equivalent of water, α_i the stress index of water category i (in m^3 of water equivalent per m^3 of water of category i withdrawn/released) and V_i (in and out) the volumes of water category i entering and leaving the process or product system (i.e. elementary flows (in m^3)). It represents the equivalent amount of water of which other competing users are deprived as a consequence of water use.

AFFECTED USER(S) ($U_{i,j}$) - A change in water i availability will not affect all users to the same extent. The impacts depend on 1) the water's functionality for a specific user j, $F_{i,j}$ (based on its quality and type of water resource), and 2) the identification of the user(s) most likely to be affected by a change in water availability in the area of interest (U_j). Together, these parameters make it possible to assess the extent to which user j will be affected by a change in availability of water category i, as shown in equations 4.6 and 4.7.

$$U_{i,j} = \frac{U_j \times F_{i,j}}{\sum_{j,offstream}(U_j \times F_{i,j})} \qquad \text{where j is an off-stream user} \qquad \text{Equation 4.6}$$

$$U_{i,j} = U_j \times F_{i,j} \qquad \text{where j is an in-stream user} \qquad \text{Equation 4.7}$$

Where $U_{i,j}$ represents the proportion in which user j is affected by a change in water availability for category i (dimensionless), U_j the proportion in which user j is affected by a change in water availability (dimensionless) and $F_{i,j}$ the functionality of water category i for user j (dimensionless).

Functionality – $F_{i,j}$ - Water categories presented in Boulay *et al.*(2011a) are related to users through a binary functionality parameter (1 or 0) that reflects whether or not the water category is functional for a given user. The functionality $F_{i,j}$, is based on the potential use of water i by user j without any additional treatment.

User's identification U_j – For off-stream users, this parameter represents the user(s) that will be affected by a change in water availability (i.e. the one from which this additional water will be taken). Such users are referred to as marginal users. It is still debated which is the marginal user(s); different approaches have been suggested (Motoshita et al. 2010a; Pfister et al. 2009) and mainly differ on the inclusion or exclusion of domestic users as users potentially affected by a decrease in water availability. A UNESCO report states that "unbridled competition from richer farmers and industrial concerns for water, productive land and fisheries, often put the poor at a serious disadvantage. It is also often very difficult for the poor to assert their rights and needs so as to receive a fair entitlement to public goods and services" (World Water Assessment Program 2006). However, there is a debate (Pfister et al. 2009) as to whether an additional water use as typically assessed in LCA would indeed deprive domestic users or whether the change in water availability would be absorbed by the agricultural sector, which would be considered the sole marginal user due to its lower willingness to pay. In reality, there is not enough information to provide robust evidence of such a choice. To satisfy both ends of the debate, two approaches were chosen. The first, the distribution hypothesis, implies that all users are likely to be affected proportionally to their use. A regional distribution of water withdrawals per user was used to represent the probability of being the affected user. In the second approach, the marginal user hypothesis, agriculture is considered to be the only off-stream user that is affected. Therefore 100% of the water use will affect agriculture and no human health impacts will be generated from water deprivation for domestic uses. Results for both scenarios are given, leaving the final choice up to the LCA practitioner depending on the information available at the local level.

For in-stream users, a change in availability of 1 m^3 of water in a country will deprive each of the in-stream users proportionally to the intensity at which they use the surface water bodies. For human health impacts, this only concerns fisheries. The details of the calculation and hypotheses for both off-stream and in-stream users can be found in SI.

ADAPTATION CAPACITY (AC) - The adaptation capacity defines whether the change in water availability will create deficit or compensation scenarios. The World Bank gross national income (GNI) classification (UNEP 2009) was chosen as the socioeconomic parameter to indicate a country's adaptation capacity (AC) (see Figure 4-2). Its correlation with access to an improved water source or improved sanitation was reported by the United Nations and reproduced in the SI (World Water Assessment Program 2009).

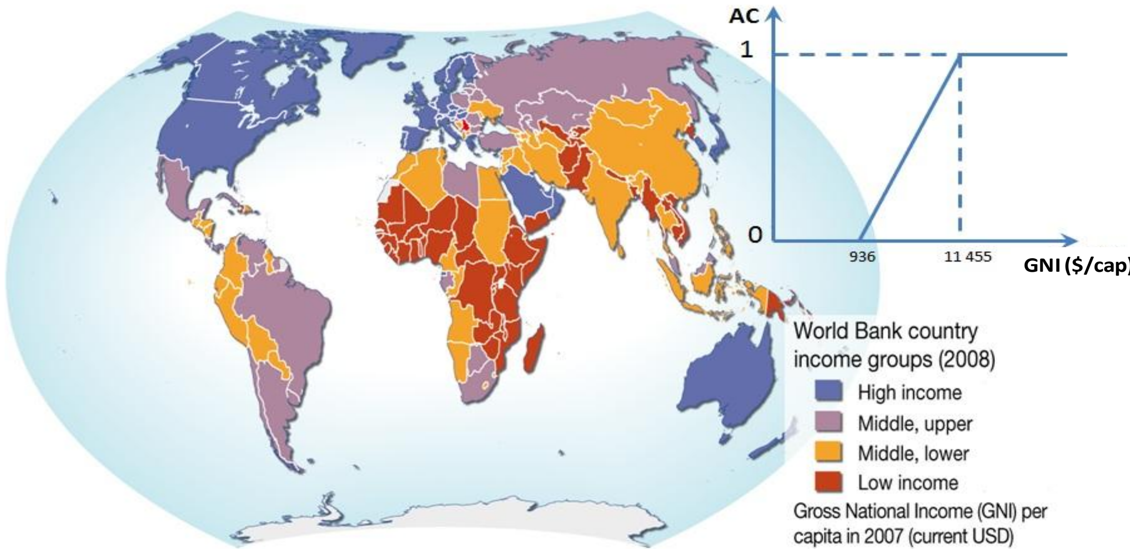

Figure 4-2 Adaptation capacity based on World Bank country classification: No adaptation for low-income countries, complete (100%) adaptation for high-income countries and partial adaptation for middle-income countries

It is proposed that low-income countries (GNI< \$936/cap) will not be able to adapt to a change in water availability and will therefore suffer water deficits, whereas high-income countries (GNI> \$11 455/cap) will have the means to fully compensate for this type of change. Middle-income countries (\$936/cap < GNI <\$11 455/cap) are attributed an adaptation capacity proportional to their incomes, meaning that, in these countries, both compensation and deficit occur. This relation is shown in Figure 4-2 and described in Equation 4.8.

$$AC = (9.5\ 10^{-5} \times GNI) - 8.9\ 10^{-2} \qquad for\ 936\ \frac{\$}{\text{cap}} < GNI < 11{,}455\ \frac{\$}{\text{cap}} \qquad \text{Equation 4.8}$$

EFFECT FACTOR (E_J) - Effect factor E_j assesses the importance of human health impacts caused by a water deficit for user j for five of the ten users: domestic (1, 2, and 3), agriculture and fisheries. If a water deficit occurs for the remaining users (transport, hydro, industry, cooling and recreation), impacts will only be generated through a compensation process when occurring in countries able to compensate. This is reflected by the E_j zero value for these users.

For agriculture and fisheries, it is generally accepted that a lack of water would result in malnutrition health impacts due to lower food availability (Pfister et al. 2009; UNESCO 2003).

The effect factors (DALY/m^3) were determined by first assessing the health impacts generated by malnutrition in DALY/kcal and dividing this value by the amount of water needed to produce one kcal, either from agriculture or fisheries. For domestic use, the effect factor (DALY/m^3) relates the human health impacts associated with a lack of hygiene and sanitation when water is scarce to the water deficit for domestic use. It is calculated by dividing the ratio of health burdens from water-related hygiene and sanitation issues by the actual volume of water in deficit for domestic uses (based on a value of 50 l/cap/day to ensure low health concerns and cover most basic needs) (Howard and Bartram 2003). The resulting effect factors are 6.53×10^{-5}, 2.02×10^{-5} and 3.11×10^{-3} DALY/m^3 for agriculture, fisheries and domestic, respectively. A domestic use deficit is therefore critical, since it shows health impacts that are two orders of magnitude greater than those for agriculture or fisheries. The details on how these parameters were obtained are presented in the SI.

4.3 Results

WATER STRESS - Along with the elementary flows, the water stress indexes (α_i) are proposed to calculate the water stress indicator (WSI) at the midpoint level. Figure 4-3 shows the water stress index for good quality surface water (α_{2a}). Whereas high scarcity is found where expected, in addition, some major watersheds in North America do not have good quality water (UNEP Global Environment Monitoring System (GEMS) Water Programme 2009) and these quality data are also used in as default for the northern watersheds when primary data are lacking. Assuming water quality data are representative of the entire region, no water of this quality would be present in that particular region, hence no assessment of its use would be needed. See the SI for all water category indexes.

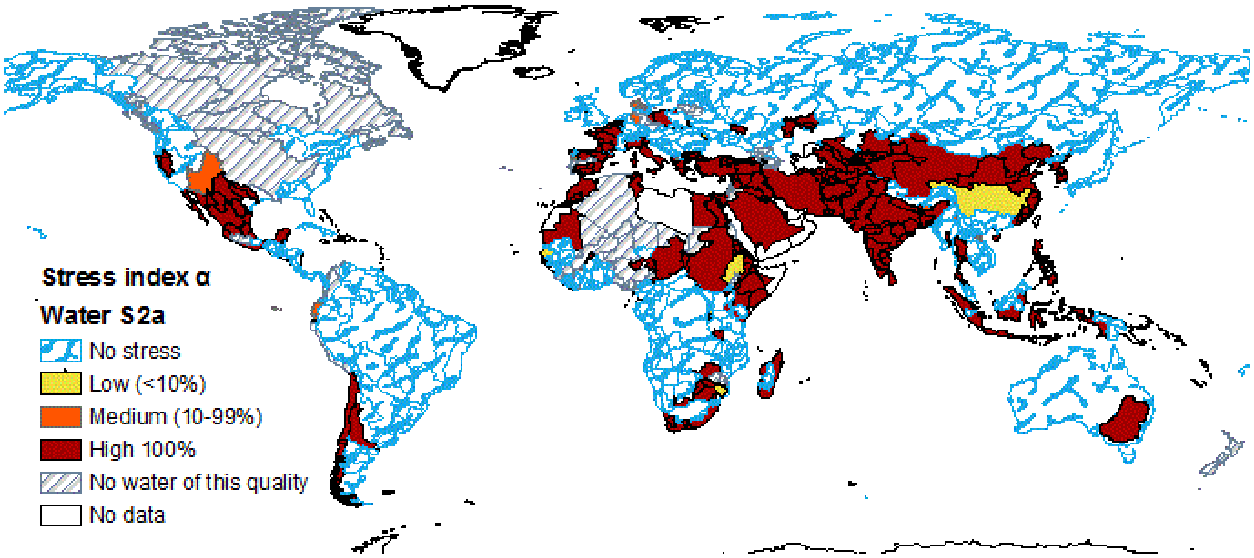

Figure 4-3 Regional water stress index α_{S2a}, based on the ratio of consumptive use over renewable available resource, including local water quality data and modeled based on accepted stress thresholds

HUMAN HEALTH IMPACTS – Figure 4-3 shows the results of the direct potential impacts on human health (in DALY/m^3) from the use of 1 m^3 of good-quality surface water (S2a). The CFs in Figure 4-4b are based on the hypothesis that several users are affected proportionally to their use and therefore include impacts brought about by a lack of hygiene and sanitation related to a water deficit for domestic use and impacts generated by malnutrition from both agriculture and fisheries. The CFs in Figure 4-4a refer to health impacts generated only by malnutrition from both agriculture and fisheries. As expected, high-income areas such as North America, Europe and Australia show no direct impacts on human health because they have maximum adaptation capacities. They would, however, generate potential impacts from compensation. CFs for other water types are detailed in the SI.

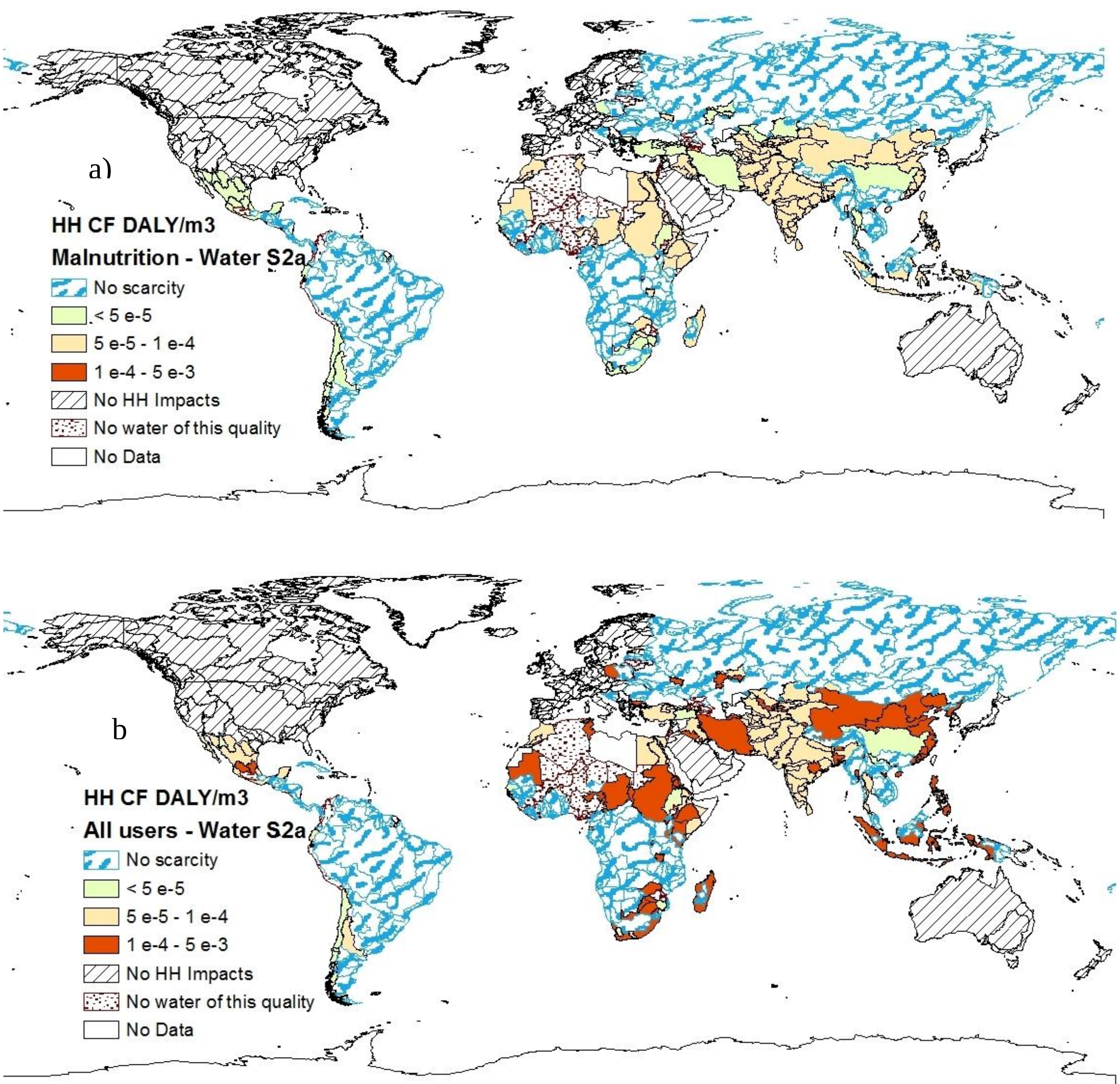

Figure 4-4 Human health characterization factors for good quality surface water (category S2a) in DALY/m³ a) considering that agriculture is the only marginal user affected (along with fisheries), leading therefore to malnutrition and b) considering all users are affected proportionally to their use (i.e. also including domestic users and therefore considering health burdens due to lack of hygiene and sanitation in addition to malnutrition).

ILLUSTRATIVE EXAMPLE - A paper mill producing cardboard from recycled fibres was studied as an illustrative example using the generic ecoinvent data set *Corrugated board, recycling fibre, single wall, at plant* characterized with the Impact 2002+ methodology (Jolliet et al. 2003). The generic dataset was adapted with available primary industry data from the paper producing company Cascades. For each ton of corrugated board produced, 17.4 m^3 of water are withdrawn from a nearby river and 16.4 m^3 are released into the same river (average pulp and paper plant data). The effluent is categorized as average quality water (category 2b, see Boulay et al. (2011a) and SI), while the influent is considered to be good quality surface water (S2a). This scenario B (average effluent S2b) is shown along with two other hypothetical variations: A. the effluent is better treated, up to the good quality surface water of the influent (well-treated effluent, S2a) and C. the effluent is considered null (i.e. 100% of the water is consumed). The purpose of this sensitivity analysis is to assess the variability of the impacts associated with the water that is consumed or used but released at different quality levels. The magnitude of the difference between both hypotheses on the marginal user affected is also illustrated by this example, as discussed below.

Impacts on human health are shown for the hypothetical production of cardboard in the Ganges Basin in India. This region was chosen because it shows a low adaptation capacity and high water scarcity. In this respect, adaptation will still occur ($AC \neq 0$), implying that both the direct impacts and impacts generated by compensation scenarios should be addressed. Only the direct impacts are shown in Figure 4-5. The 10 main contributors (elementary flows) to human health impacts are identified, including water use, which is split into impacts generated by malnutrition and by diseases related to water deficit for domestic use. All three scenarios (A, B and C) are presented for both hypotheses: (1) Distribution: all users are affected proportionally to their use (i.e. considering health burdens due to the lack of hygiene and sanitation and malnutrition) or (2) Marginal: reporting that only malnutrition impacts occurs. The reduced quality of the effluent in B may significantly contribute to the overall human health damages. The additional impacts from a degradative use (in 1B) are, in this case, even higher than all other human health contributors. However, considering that all water must be consumed (1C) instead of considering the polluted released flow (1B) would more than double the impacts on human health from water use. In addition to these results, compensation scenarios for each user should be modeled with a traditional LCA method to yield impacts scores that would then allow for the integration of indirect impacts into other impact categories.

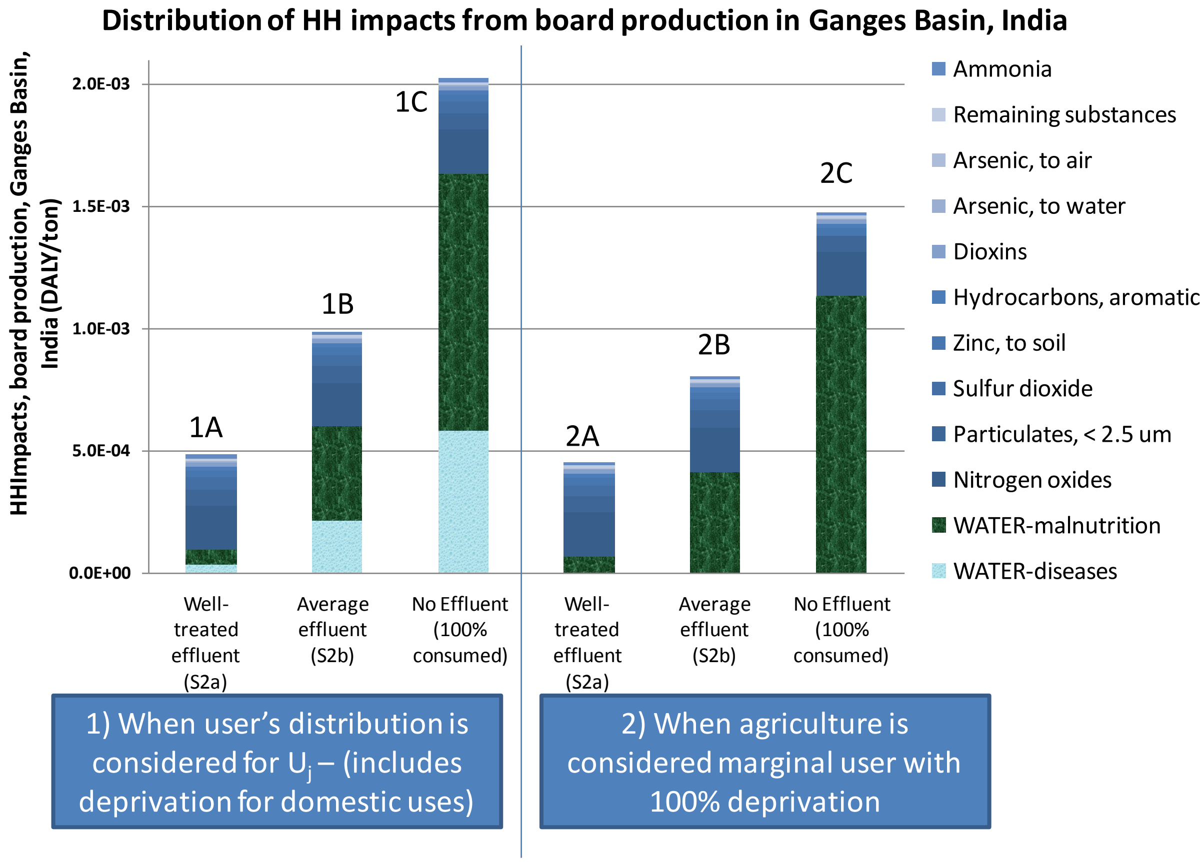

Figure 4-5 Human health impacts of water use for board production in comparison with other human health contributors from the production process for 3 different effluent scenarios and for both hypotheses on marginal use: 1) considering all users to be affected proportionally to their use (i.e. including domestic users) and 2) considering that agriculture is the only marginal user affected (along with fisheries), leading therefore to malnutrition (not including domestic use).

4.4 Discussion

This methodology modeled the impacts generated by a change in water availability for human uses in an LCA perspective. The water stress parameter α was proposed to calculate a midpoint indicator (WSI) that could be used to characterize physical inventory flows into a common metric, as suggested by Frischknecht et al., Pfister et al. and Mila-i-Canals et al. (Frischknecht et al. 2008; Mila I Canals et al. 2008; Pfister et al. 2009). In this respect, α can be considered representative for three areas of protection, including human health, ecosystem quality and resource depletion. However, it is important to note that no ideal midpoint has been found for the impact pathway to allow for a partial characterization that could then be used directly as an input for endpoint modeling in all categories, as presented in Bayart *et al.* (2010b). This is due to the fact that each impact pathway involves different mechanisms. Here, the water stress parameter α corresponds to a fate factor (see Equation 4-2). The human health impact pathway was further modeled up to the endpoint level by developing CFs expressed in DALY/m^3. Additionally, indirect impacts must be considered through the compensation scenario by identifying the marginal technology that is appropriate for each water type, as proposed by Weidema et al. (1999). These indirect impacts could be consistently compared with the direct impacts by further modeling a new LCI (by system expansion) and assessing the related impacts on human health and other categories generated by the compensation scenario, allowing for a coherent comparison with other sources of impacts. The application of this method to a straightforward example illustrates the potential significance of considering water use impacts in an LCA carried out for a region with low adaptation capacity. However, further case studies, including higher income countries, are needed to further evaluate this approach. Moreover, regionalization was shown to be a critical issue since impacts can be dominant or non-existent from one region to another.

While other methods have been advanced to assess water use impacts in LCA, the methodology herein differs in the following aspects:

Quality – The model does not only consider the consequences of consumptive use leading to reduced access to water. It also assesses the consequences of a loss of functionality for downstream users due to the degradative use of water. Seeing as the inventory procedure integrates the functionality of withdrawn and released water based on quality, the released water and its corresponding functionalities are considered to be returned to the environment, avoiding an overestimation of the potential impacts by considering that the water was consumed. This is especially important for heavy water users (e.g. thermal plants for cooling). Assessing the quality of the returned water would also serve as an incentive for effluent treatment. In the aforementioned example, if the plant were to treat its effluents to attain the same water quality as the influent (2a), water use impacts would be reduced by a factor of 6 as compared to the water released as 2b. Finally, including quality in the methodology also curbs the impacts of using low-quality water vs. high-quality water since not all users will be affected the same way.

Stress – Unlike other widely-used scarcity indicators, the one proposed here considers the ratio of consumptive use over a statistical low-flow parameter. The numerator allows for a better representation of the physical scarcity of water without considering the withdrawn water that is released in the same watershed (e.g. for cooling). The denominator takes seasonal variations into account, which is very important in countries that face monsoons and droughts. Moreover, the indicator distinguishes surface water, groundwater, rain water and quality, which is significant since the water categories are not available in the same quantities and not functional for all users. A more specific scarcity parameter per water type could be obtained by adjusting the CU term to reflect the actual consumption of each water quality category. However, the regional breakdown of the volume of water consumed for each specific water category is not available. The parameter therefore currently assesses the scarcity of a water category based on its availability and assumes that water of good quality will be consumed before a lower water quality.

Users – All users are considered, and some are even differentiated based on the quality of water they require. While not all suffer human health impacts, it would be especially important to consider them in a compensation perspective, since industry and the energy business in developed countries are unlikely to stop their activities due to water shortage without first considering compensation strategies.

Human health – Although a first attempt to model human health impacts for malnutrition has already been proposed by Pfister and colleagues (2009) with a statistical correlation of $R^2 = 0.26$, in this paper, a probabilistic approach was proposed instead to evaluate the health burdens associated with malnutrition and with water shortages for domestic use. Both show acceptability in terms of the distribution – log-normal, which allowed for the use of geometric averages. The p-values describing the distribution of the correlation are of 0.0839 and 0.15, respectively. A p-value higher than 0.05 reflects an acceptable distribution. Malnutrition was modeled for a water shortage affecting agriculture, fisheries and livestock (through agricultural feed). With regards to domestic use, while infrastructure plays an important role in the health burden caused by water-related diseases, this role did not need to be considered since the impacts of a change in water availability in current socioeconomic conditions were modeled. The health burden for 1 m^3 of water that is unavailable for domestic use (whether from lack of access or scarcity) was assessed from the correlation between the actual water used for domestic purposes, the water required to avoid health issues and the actual health burden of water-related issues. This rationale is based on strong epidemiological evidence that access to water is correlated with diarrhoeal diseases, worm infections and malnutrition (World Water Assessment Program 2009). However, whether or not access to water for domestic users will be reduced from typical inventory data in LCA, and, consequently, the inclusion or exclusion of the associated impacts, are still open for debate. Pfister and colleagues (2009) chose to exclude them, but Motoshita and colleagues (2010a) developed a model that suggests they should be included. When addressing the impact pathway, it is important to understand whether a water use in a low-income, water-stressed region would indeed lower water availability for domestic users or rather only affect other users (e.g. agricultural or industries). At this stage, the information to support the choice and determine the marginal affected user is insufficient. For this reason, we chose to present CF for two main scenarios: 1) all users are affected proportionally to their water use in a given region, implying that the burden caused by a lack of hygiene and sanitation is considered in addition to malnutrition, and 2) only agriculture is affected among the off-stream users as it would present the lowest willingness to pay. As demonstrated in the example, human health impacts from water use may be of the same order of magnitude as the human health impacts generated by toxic emissions, thus supporting the conclusion that including or excluding this impact pathway has a relevant influence in a LCA.

Compensation – Compensation accounts for the capacity of human users to adapt to a freshwater deficit through the use of technology. Model results indicate that direct human health impacts are expected to be null in developed countries, but indirect impacts may be generated by compensation scenarios. In this respect, the use of backup technologies is not only meant to alleviate depletion, as proposed by Pfister (2009), but also to compensate when local resources are not necessarily being depleted but are still used to the extent that they reduce availability for users due to competition. Marginal technology use should therefore be included as having potential impacts within LCA and not solely as a depletion metrics. The volume of water needed would be obtained through the marginal technology in place to compensate for a specific water category. Impacts from this technology should be included in the water use impact assessment by system boundary expansion. However, the assessment should be done consistently with the method proposed here. This implies that only a fraction of the volume, based on the adaptation capacity, is produced from the marginal technology for the withdrawn water category. The identification of the regional marginal technology for each water type should be performed and its impacts included in the scope of the assessment.

Limitations – Though it demonstrates spatially-differentiated capabilities, the model does not predict absolute or real impacts. It addresses the potential environmental impacts used to characterize environmental interventions within an LCA with an underlying hypothesis of linearity between a change in water availability and the resulting impacts. The default characterization factors in this model may be well-suited to exploring the potential impacts of water use but the robustness of the results and required level of detail can only be evaluated once the model is used and the results are compared with those from other models. Obtaining better data on water quality per watershed or region is necessary to increase the accuracy of the methodology. Current data are not adequately distributed in time or space, and the parameters and uncertainty of the model have yet to be evaluated. Several specific limitations apply to this model. Interactions between surface and groundwater or between watersheds are not considered and the consecutive use of the water resource is not taken into account – a fact that could lead to an underestimation of potential impacts. These limitations also apply to (downstream) transboundary watersheds. However, the results could be corrected on a case-by-case basis by identifying the country in which the impacts are most likely to occur for a specific water use or by adding the potential impacts generated by water use in downstream watersheds. Also, user and water availability distribution within a watershed is considered to be homogeneous, implying that there is no difference if the water is withdrawn upstream or downstream of other users. However, it is unlikely that sufficient data would be available on the exact location of the water use to allow for such distinctions or modeling. Moreover, as a modeling choice, the quality of the effluent was taken *as is* and therefore does not account for dilution or degradation time. While this option is believed to be the best choice since dilution would require the availability of more water, thus creating a loop effect, it may become a limitation in cases in which the natural buffering capacity of the environment would assimilate some of the pollutants. Lastly, limitations associated with water categories, as described in Boulay *et al.* (2011a) may lead to either an overestimation or underestimation of the impacts by grouping different user functionality requirements into a limited set of water categories. The same reference proposes a more detailed functionality-based approach to overcome this.

To ensure the integration of the method within daily LCA practices, life cycle inventory databases must be expanded to account for released water volume and therefore support the calculation of water categories. The inclusion of impacts from compensation scenarios may be facilitated by adapting a life cycle inventory database to a consequential approach by including the use of the local marginal source of water, which may come from desalination, reuse, etc.

As for other regionalized impact categories, datasets should allow for region selection and appropriate CFs. To facilitate the use of these CFs, a generic dataset of effluent water quality by industry type could be generated. Moreover, a water mix similar to a grid mix could be set out based on the local surface/ground water consumption data and water quality data that could be used when actual inventory input data are not known.

4.5 ADDENDUM: Assessing indirect impacts of water use through marginal technology

The work prensented in this section is the object of a publication in preparation, in co-authorship with Eléonore Loiseau[2], Christian Bouchard[3], Cecile Bulle[1], and Manuele Margni[1].

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4.5.1 Introduction

The impact assessment framework of water use in LCA links inventory flow to three Area of Protection: human life, biotic environment or abiotic environment (Bayart et al. 2010a) . When it comes to the effect on human life, using water can incur impacts on human health when domestic, agriculture or aquaculture users are deprived (Bayart et al. 2010b; Boulay et al. 2011b). These potential impacts occur either from a decrease of water availability for hygiene purpose and leading to diseases, or from a decreased productivity of agricultural production or aquaculture leading to malnutrition (Boulay et al. 2011b). However, these impacts do not occur in regions where economic resources are sufficient to allow the deprived users to turn towards technology to meet their needs, for example by desalinating water. On the other hand, this technology may generate environmental burdens that should be somehow addressed when assessing impacts from water use in LCA. A consequential approach can help in assessing these additional impacts.

A conscequential approach in LCA is one that will look at the changes in the present situation and the impacts caused by the related changes in the system. This is different from an attributional approach where impacts are assessed by attributing a fraction of the present situation proportionnally to the change being considered. The conscequential approach often translates into identifying one affected *marginal* technology rather then using a mix of available technologies, based on the fact that the other technologies are either constrained, i.e. they have reached their maximal capacity, or they are more expensive to provide the same output (Weidema et al. 1999).

This paper proposes a consequential framework and operational model using the marginal technology concept in order to assess LCA impacts from water consumption and degradation. This is done by identifying additional processes which can be added to the inventory of the study performed and thus included in the impact assessment. A methodology to identify water treatment steps based on a set input and output is therefore also proposed. The entire paper builds on existing methods for inventory (Boulay et al. 2011a) and impact assessment (Boulay et al. 2011b), further discussed below, which have already addressed issues related to inventory modeling of water quality and impact assessment of water use on human health, respectively.

4.5.2 Methodology

Based on the framework presented by Bayart el al. (adapted in Fig.4-6), the socio-economic context determines whether impacts from water deprivation of human users will occur from direct impacts on human health or rather through adaptation, which may generate indirect impacts in all impacts categories. In order to model these later, the adaptation capacity first needs to be assessed. This is done in section 4.5.2.1 below. However, adaptation will only need to occur when there is competition between users for the availability of the resource, and this is assessed by quantifying the state of constrain of water in section 4.5.2.2. Using a consequential approach, a model is proposed in section 4.5.2.3 and 4.5.2.4 respectively to identify the marginal source of water that will be used to meet the additional water demand on constrained resources and the associated technologies needed. Finally, how in-stream users are affected is addressed in section 4.5.2.4.5.

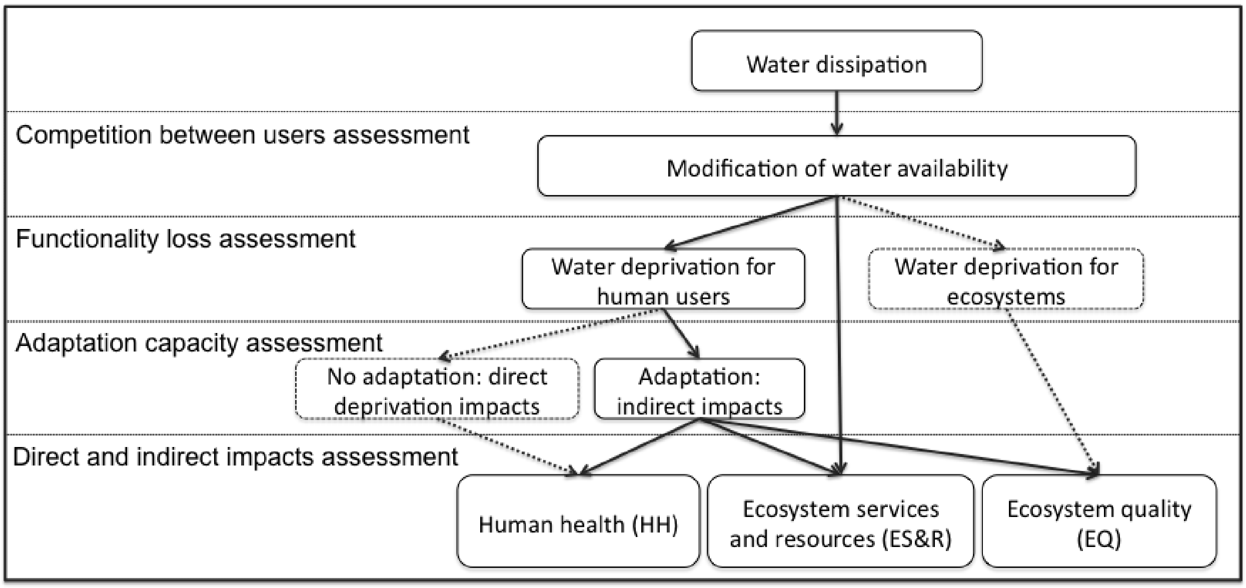

Figure 4-6 Impact pathway for water use impact assessment in LCA (adapted from Bayart el al.(2010b))

4.5.2.1 Identify regions capable to adapt to water deprivation

While water deprivation for human users can result in direct impacts on human health, deprived users in more developed regions will adapt and use technology to meet their water needs. Methods developped to assess impacts from lower water availability for human users all consider some socio-economic indicator in their modeling (Boulay et al. 2011b; Motoshita et al. 2010a; Pfister et al. 2009), resulting in no, or low, direct impacts on human health when the socio-economic context allows for adaptation. According to Boulay et al. (2011b), the Adaptation Capacity (AC), i.e. the capacity of a region to adapt to water deprivation, is related to the Gross National Income (GNI). The World Bank classifies countries (UNEP 2009) into low, middle and high income country which was used to set out that low income region will not be able to adapt (AC = 0) and will suffer human health impacts from a change in water availability, high income countries will fully adapt (AC = 1) and not suffer any human health impacts, and middle income countries will partially adapt and partially suffer from human health impacts, proportionally to their income level (0<AC<1) (Boulay et al. 2011b).

The same AC parameter is used here to assess the extend to which an additional demand for water can be met by increasing the availability through adaptation technology, and hence without using resources already allocated to other users. The AC, ranging between 0 and 1, therefore serves to assess which fraction of the additional volume of water used will ultimately be provided by the adaptation technology which will increase availability, and which (the balance) will rather deprive other users. The present paper addresses the adaptation pathway and is consistent and complementary with Boulay et al. (2011b) which addresses the deprivation pathway.

4.5.2.2 Identify the constrain on the resource

Adaptation occurs when the water withdrawn is constrained. The water is described by its quality and source (surface or ground), as these characteristics define whether or not it can be used by a specific user without further adaptation. For this purpose, water categories from Boulay et al (2011a) are used to describe water flows, and indexed as category i. Tab.4.1 summarizes the eight water quality categories with their qualitative description. Threshlods for 137 parameters can also be found for each category in the orignal paper. Whether users will compete or not for the resource, here a specific water quality, depends on whether the resource is constrained or not. The assessment of a water category being constrained is done through the scarcity parameter α_i as defined in Boulay et al (2011b). This parameter relates the local consumption of water with the local availability of a specific water category, and is defined in such a way that scarcity starts as soon as competition is present between users, which is when more than 10% of the water is withdrawn, or 3% consumed. Hence, for our purpose, an unconstrained resource is one below these thresholds, and its scarcity is defined as $\alpha_i = 0$. For $\alpha_i > 0$, users will start to compete and the use of a backup technology will be needed for any additional water use, even if it is not directly the “last” user who will pay the adaptation price.

Table 4.1 Water quality categories description adapted from Boulay et al (2011a)

Water quality category	Excellent	Good	Average	Average - Tox	Average-Bio	Poor	Very Poor	Unusable
Category number	1	2a	2b	2c	2d	3	4	5
Microbial contamination	low	low	medium	low	high	high	high	unusable
Toxic contamination	low	medium	medium	high	low	medium	high	unusable

4.5.2.3 Identify the marginal water source

In order to identify the marginal source of water for each water category a decision algorithm is used. This algorithm is a “market activity” type model as introduced in ecoinvent 3(Weidema B P, Bauer C, Hischier R, Mutel C, Nemecek T, Vadenbo C O 2011). In such a model, all possible sources supplying a given commodity for a specific geographical location are identified as inputs to produce a consumption mix as an output, when following an attributional approach. This concept is usually widely understood with electricity production and the use of an electricity mix. However, a consequential approach can also use the concept of *market activity* by identifying the one single marginal source for the commodity, as described in Weidema et al. (1999). When referring to electricity, this corresponds to identifying the one source of electricity (nuclear, coal, hydropower, etc.) that will expand if the demand increases. The same can be done with water.

A *market activity* model (referred to as *market* from now) for each of the 8 different water qualities i was set out. For each water *market*, all possible sources to supply water of this quality (i) were considered. These include locally available water of type i (surface and ground), treated water from a lower quality source of water (Q<i), desalinated water, and transported water (short or long distance). The released water (j) also contributes as an input to a water *market*: the same *market i* if the quality has not been degraded (or improved) or the specific water *market* of this water category in other cases. As illustrated in Fig. 4-7 below, the water market activities are indirectly linked with the markets of the goods that water serves to produce (product X). An additionnal demand on water i market can be met by an increase in supply, or a decrease in demand from another market, whose own supply could be met by an increase of import. In the present paper, we choose to consider a model for a marginal use of water only, as usually assessed in LCA. This implies that the volume of water is too small to cause major changes in the system, and in this case it was assumed that ripple effects would be too small to affect the product X market and its imports.

Within each *market*, instead of considering averages, each process delivering to the *market* is investigated as to whether or not any of them are constrained or if they have the capacity to expand as a result of a change in demand coming from an additional process. The most economical of the unconstrained options will typically be the one affected by a market expansion, i.e. an additional demand, and will be identified as the marginal source (Weidema et al. 1999). Translating this into a water *market*, when using water of a certain quality i, this additional water demand is equivalent to an expansion of the water i *market*. It will be met by the marginal technology producing this water in this region. Conversely, the release flow will displace the marginal source of water for this *market j*.

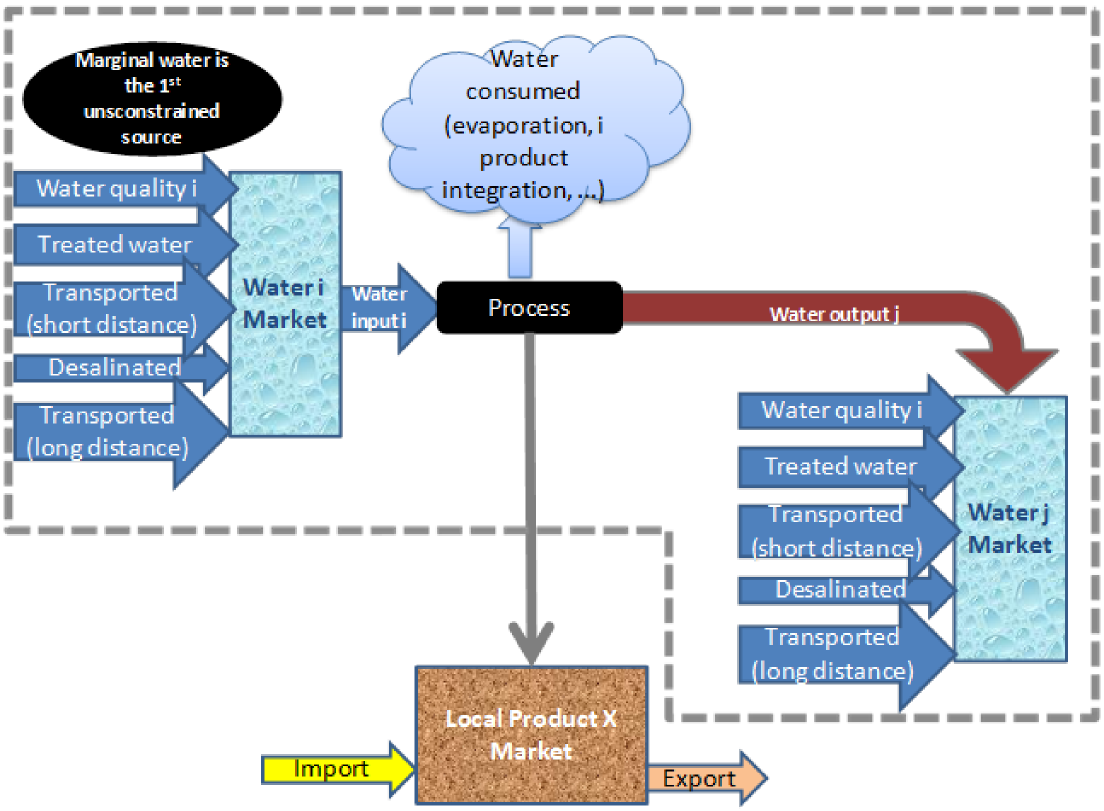

Figure 4-7 Representation of a water market activity used to identify marginal water source

In general, it can be assessed that the marginal technology is the unconstrained technology with the lowest long-term cost (Weidema et al. 1999). This will most likely be the treatment of the next best quality water available (or pumping of groundwater) and is identified as water m. When no water is available in a region, transporting it from the next watershed is more economical then desalination (see cost evaluation below). Only when no water including sea water, is available in a region, would long-distance water transport (including from other countries) be considered an option. That is, when sea water is available, desalination is chosen over import as political and economical reasons have demonstrated in the past. The case of Singapore is an example of short distance international water transfer, and in 2010 the agreement was not renewed as the price fixed by Malaysia was no longer interesting and Singapore chose a more protective strategy towards becoming self sufficient in terms of water supply, with desalination. Another example was studied by Dawoud et al. (2006) concerning the Gulf Cooperation Council (GCC) countries, which represent well the countries for which this model applies, as water is scarce yet financial resources are not. The authors concluded that sustainable development of GCC countries will depend in the future on large scale desalination, as mass water import/transfer had lost their economic advantages compared with the rapid development of new and cheaper desalination technologies. Moreover, import would need to occur from a region with unconstrained water source otherwise a loop effect is created.

Both surface and ground water are considered for each quality. In theory, unconstrained water of the higher quality is used as a marginal source. The exception to this is when the cost of treating a lower quality surface water is lower than pumping (and maybe also treating) the unconstrained groundwater source. When both surface and ground water of the same quality are available, surface was chosen as being more convenient and cheaper to use. The following algorithm (Fig.4-8) was used to identify the marginal source (M0 to M6) of water for a specific water *market* i in a region.

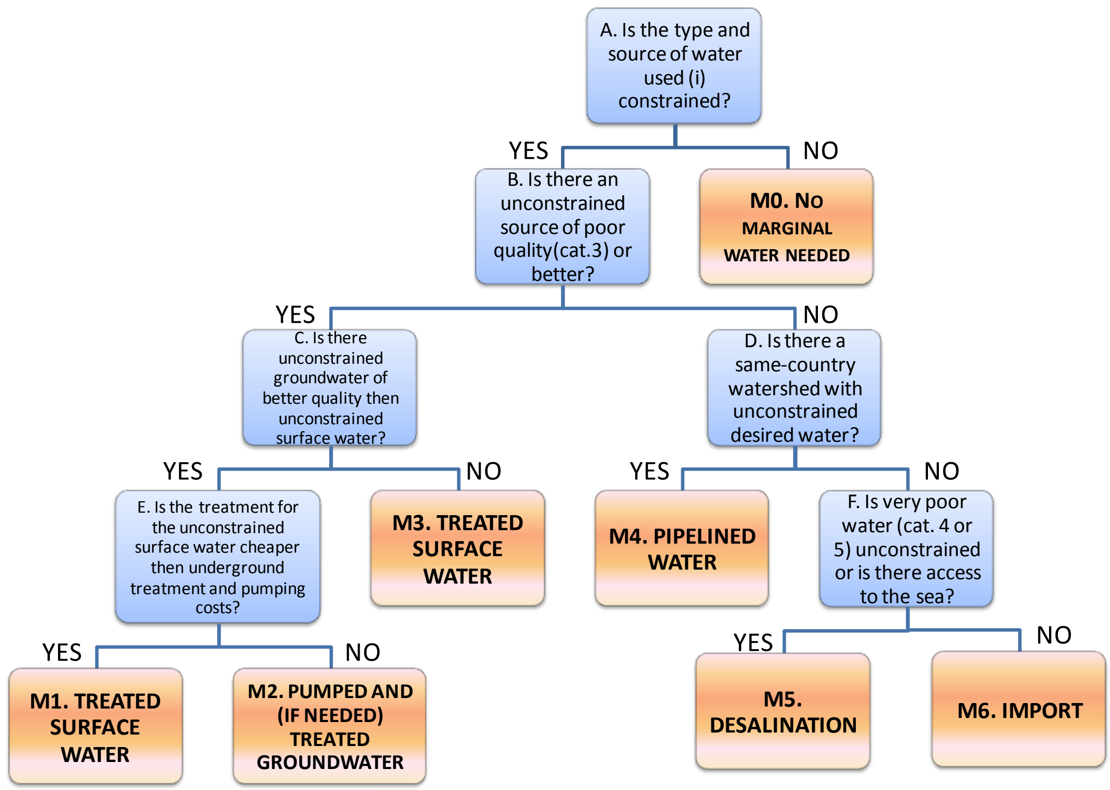

Figure 4-8 Decision algorithm for the choice of the marginal water. Boxes A to F assess the local situation leading to one of the 6 marginal sources, M1 to M6, or to no marginal water needed, M0.

In figure 4-8, boxes A to F assess the local situation leading to one of the 6 marginal sources (M1 to M6) or to no marginal water needed (M0). All questions relating to the availability of an unconstrained source of water of a certain quality (Boxes A, B, C and D) were addressed based on the scarcity parameter alpha, as described in Boulay et al (2011b) and available on a Google Earth Layer (CIRAIG 2012a). Questions of boxes E and F are addressed as per the following.

Box E assesses whether the most economical option would be to use lower quality surface water or to pump groundwater of higher quality. The pumped groundwater may still have to be treated, but with a more economical treatment then the one needed by the available surface water. In order to answer this question, the costs of treatment needed to obtain one category from another, as described in the following section, were evaluated and compared with the costs of pumping groundwater. The later was assessed based on the depth of the groundwater table specific to a region. While this information is not readily available, data on the number of deep wells in a country (none, few or many) obtained directly from IGRAC (International Groundwater Resources Assessment Centre) was used to assess whether the groundwater level was high (5 m), medium (20 m) or deep (200 m) respectively. When no data were available, the medium value was used by default. Most of the cost assessments were performed using the calculator WaTER from the US Department of Interior(US Department of the Interior). All cost assessment were performed including fixed and variable costs, an electricity rate of 0.07$/kWh, and an actualisation over 20 years with a 6% interest rate. All other hypothesis and calculations can be found in the Excel Supplementary Information. For the groundwater pumping, additional costs for drilling were added to the pumping costs based on Macdonald et al(2009).

Box F assesses if there is water available nearby that could be pipelined instead of using desalinated water. For this, watersheds sharing a common boundary with the watershed in which the withdrawal is occurring were selected. If the desired water, or one for which the cost of treatment and pumping is still lower then desalination costs, was found to be unconstrained in one of the possible watershed, this option was then chosen. When more than one watershed was possible, then the one with the lower cost was selected. The cost was evaluated based on the treatment (if required) and the height differential between the average altitudes of the two watersheds in order to reduce the pumping energy required (National Oceanic and Atmospheric and Administration; 2011).

This decision algorithm resulted in a local marginal water source being identified for each water category for a specific region, at the scale of the country intersected with the main watersheds. The next step consists in identifying the processes required to supply each *market* with the appropriate water based on these marginal sources of water.

4.5.2.4 Identify the marginal technology to supply the marginal source of water

The marginal technology is the technology that is necessary to supply the market with the marginal source identified in the previous section. Apart from pipelined and imported water, all other marginal technologies consist of one or many water treatments. According to quality input and output, different water treatment chains will be needed to meet a marginal demand for a given water category. A methodology is proposed to determine the appropriate treatments for passing from one water category to another. The choice of a marginal treatment chain first depends on technical constraints. An economic criterion is also considered in the selection of the marginal treatment chains as recommended by Ekvall et al (2004). Therefore, decision trees for water treatment selection based on technical and economic rules were developed and are detailed in each section below.

4.5.2.4.1 Classification of pollutant substances

In the contaminant list describing the water categories (Boulay et al. 2011a) some have similar chemical or physical properties and can hence be removed through the same treatment processes. In order to simplify the determination of treatment chains, the quality parameters were grouped into 9 families: Pathogens, Suspended Solids, Organic pollutants, Nitrogen Compounds, Gas, Inorganic Salts/anions/cations, Heavy Metals, Refractory COD, and oil/grease/hydrocarbons(IPPC) (see supplementary information).

4.5.2.4.2 Definition of technical rules

To determine water treatment sequences, it is important to consider technical constraints which will differ according to the quality of water inputs and the desired final quality. Consequently, 7 technical rules were defined to take into account technical criteria in the selection of water treatment chains (see Fig.4-9). These rules are detailed in supplementary information.

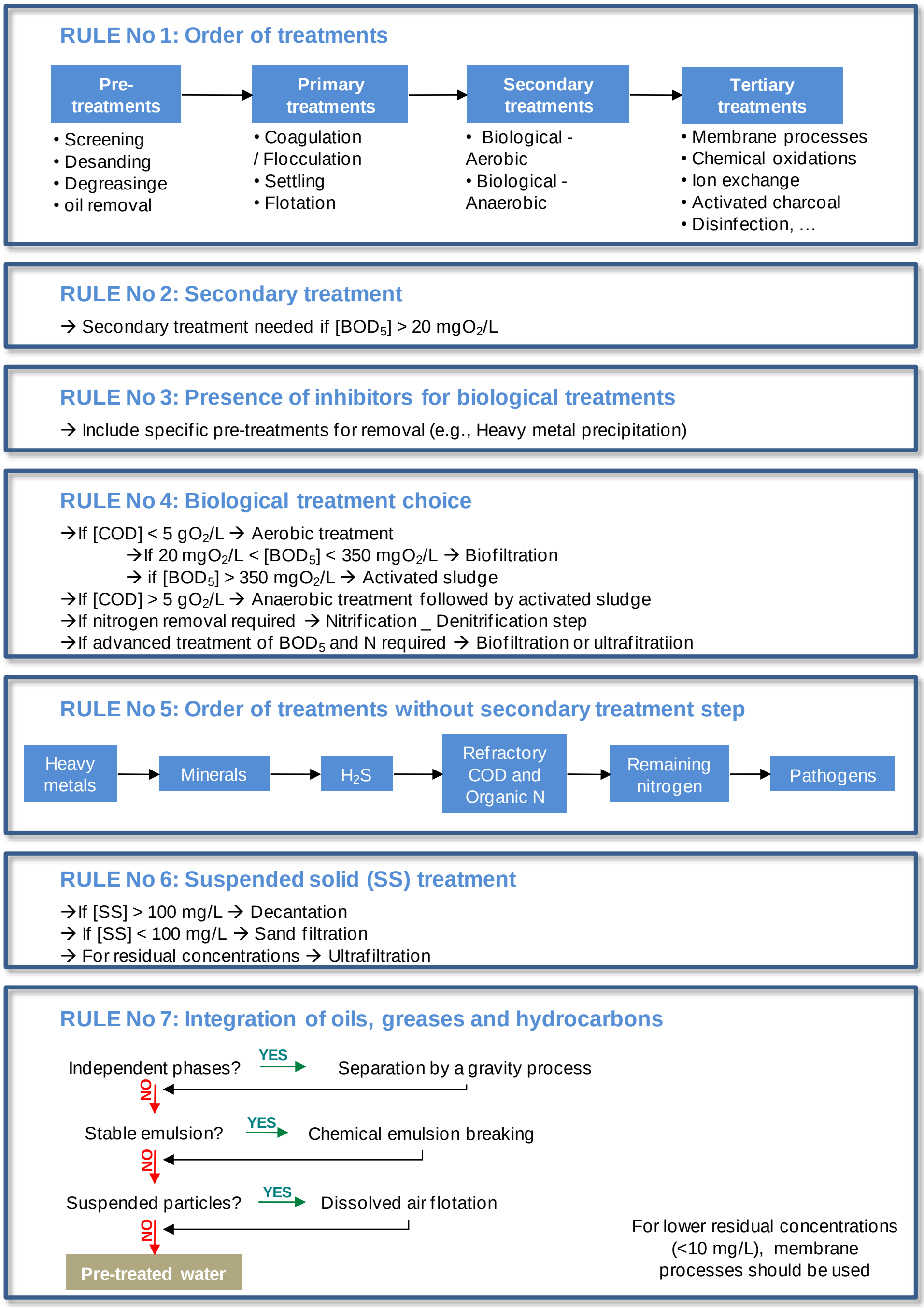

Figure 4-9 Seven technical rules required to determine appropriate water treatment chains between a fixed input and output quality

4.5.2.4.3 Inclusion of an economic criterion

It appears that, for tertiary treatments, several technologies can be used to remove a pollutant substance. The final choice is then based on an economic criterion, meaning that the technology with the lowest treatment cost (\$ per m^3 treated) is selected (8^{th} rule). The treatment cost includes capital costs as well as maintenance and operation costs for a given technology. In order to rank the technologies according to this criterion, the WaTER software (Water Treatment Estimation Routine) was used (US Department of the Interior). This tool is based on spreadsheets developed by the United States Bureau of Reclamation and The National Institute of Standards and Technology in order to facilitate the estimation of drinking water treatment cost (El-Fadel and Maroun 2002). Though drinking water and waste water treatments operate quite differently, this was deemed sufficient as an approximation of the costs to support a simple economic ranking of the technologies.

As the capacity or scale of the water treatment system can significantly influence the treatment cost (USEPA), a reference capacity was set. It corresponds to the supply of a population of 20000 inhabitants (i.e. a daily flow around 9000 m^3 / day) (USEPA). This choice is supported by the fact that, as shown by (WHO and UNICEF), half of the world population is living in cities (defined by the presence of a population of more than 20000 inhabitants). This hypothesis was tested for sensitivity with plant size as low as 1000 m^3/day.

The WaTER tool does not include the stripping technology but it seems that this technology's cost is comprised between the costs of exchange ion (IPPC) and ozonation (Spencer and Witco). Using these references, advanced technologies were ranked from lower to higher costs as follow: chlorination, resin ion exchange, stripping, ozonation, microfiltration (MF), granular activated carbon, nanofiltration/reverse osmosis.

4.5.2.4.4 Decision tree making

The 8 rules introduced above were used to build 5 decision trees (see supplementary information). Depending on water quality input and output, the choice of a decision tree will be on one or the other as described in Fig.4-10. Their use will lead to the determination of appropriate water treatment chains corresponding to the marginal technology.

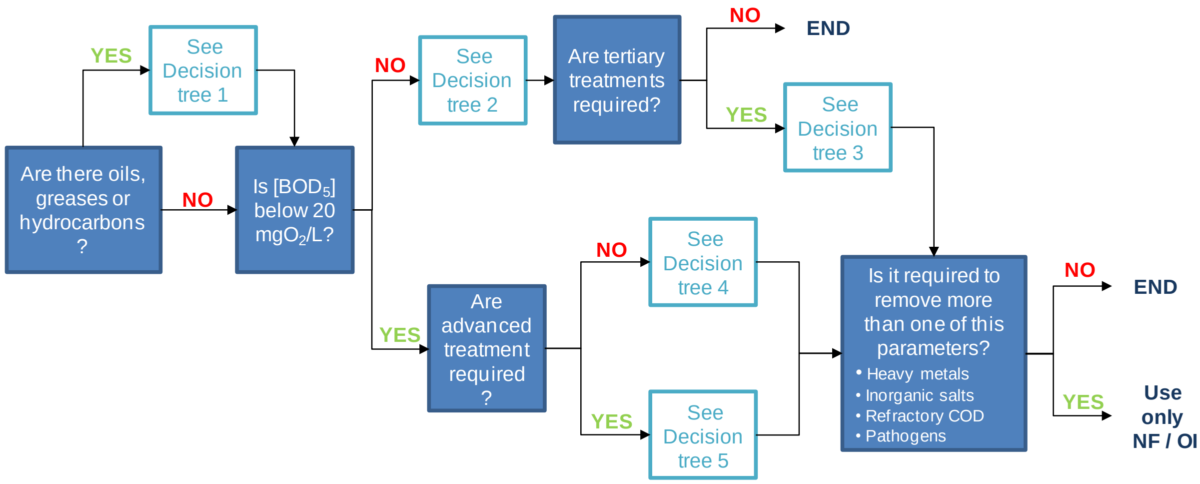

Figure 4-10 Procedure for determining appropriate treatment chains depending on water quality input and output

4.5.2.4.5 Instream Users

The market activity model and marginal source approach works for water when it behaves similarly to other commodities: it is removed from the source in order to be used. However, there are two types of water users (Bayart et al. 2010b; Boulay et al. 2011a; Boulay et al. 2011b), off-streams (agriculture, domestic, industry) and in-streams (transport, hydropower, fisheries and recreation), the latter using the resource without removing it from the source. Hence, a cubic meter of water consumed in a region will deprive several users simultaneously: one off-stream user and all in-stream users. In order to address if and how these in-stream users would adapt from an additional demand of water, we have to look at them individually.

Transport

In order for freight transport to be affected by an additional water use, this demand would have to be sufficiently large to lower a stream's level enough to prevent a ship to pass. As state in the introduction of this paper, the goal of the present method is to assess indirect impacts associated with additional marginal water use, hence this is outside the scope of this paper.

Hydropower

While hydropower harnesses the energy carried by the water flowing, it is far from being exploited at its full potential. Globally around 19% of the potential has been developed, raising to 60% in countries having actively developed their hydropower potential (International Energy Agency 2010). This means that the limiting factor to the hydropower production is not the amount of water available but rather the facilities to exploit it. Moreover, generally, water withdrawals occur downstream from a dam. Therefore impacts from adaptation due to a lower hydropower production caused by a marginal water use should not be readily considered unless such reduction in capacity of production is known to occur. If this is the case, the local marginal electricity technology should be used as a backup technology for the electricity production loss. The conversion factor of 8 m^3/kWh as used in ecoinvent for non-alpine conditions (Frischknecht and Jungbluth 2007) can be used to obtain the amount of electricity to be compensated for.

Fisheries

Not yet determined

Recreation

Whereas water bodies are often used for recreational activities, it is unlikely that an additional use of water would change the behaviour of the recreational users unless the water level or quality was to be sufficiently affected, which is outside the scope of this paper. Moreover, uncertainties associated with such modeling would be quite high.

4.5.3 Results

4.5.3.1 Adaptation

Results from this parameter are taken directly from the one proposed in Boulay et al. (2011b), and re-calculated with updated 2010 values for the GNI (UNEP 2009). Fig.4-11 shows the extent to which countries will adapt and market will be able to expand.

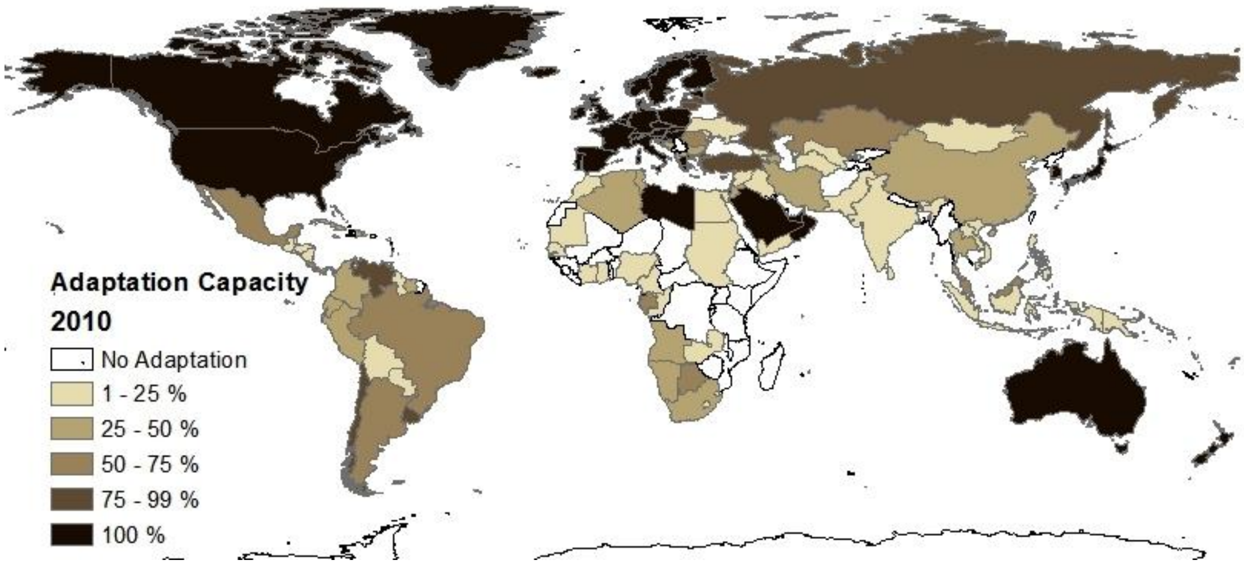

Figure 4-11 Adaptation capacity of countries (AC) ranging between 0 and 100%

4.5.3.2 Marginal water source

The decision algorithm in Fig.4-8 was used to identify the marginal water source for each water quality category for all regions where some adaptation occurs (AC>0). Results for all 808 regions of the world created by the intersection of the country and watershed scale, are available in the Excel Supplementary Information. The map of these regions is available on the online Google Earth layer (CIRAIG 2012a). Fig. 4-12 presents the results for the marginal water source for good quality water (2a) worldwide.

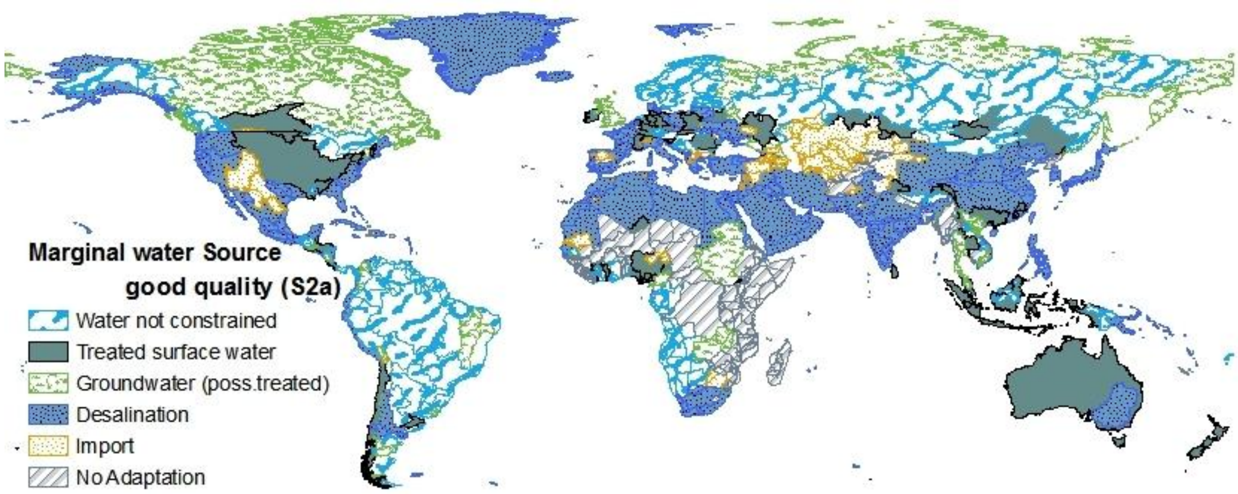

Figure 4-12 Marginal water source for good quality water (2a)

For all 808 regions, 620 (77%) present some form of adaptation (AC > 0), the balance corresponding to regions in low income countries (AC = 0). Tab. 4.2 represents the distribution of marginal technologies for these cases.

Table 4.2 Distribution of marginal water sources per water quality category for 620 regions of the world with AC>0

Desired water	No marginal water needed (M0)	Treated Surface water (M1 and M3)	Pumped and (if needed) treated groundwater (M2)	Pipelined water (M4)	Desalination (M5)	Import (M6)
1	70	141	131	0	180	98
2a	184	80	77	0	180	98
2b	232	39	71	0	180	98
2c	198	76	68	58	122	98
2d	102	121	119	0	8	98
3	282	0	60	0	180	98
4	480	0	32	0	10	98
5	492	0	22	0	8	98

4.5.3.3 Marginal technology

For the marginal water sources including a water treatment (M1, M2 and M3), the selection of marginal technologies for water treatment relies on the decision trees defined in section 4.5.2.4.4. Using this procedure, treatment chains have been determined for each pair of water categories defined by Boulay et al. (2011a) and results are presented in Tab.4.3. The starting water quality refers to the marginal water source in the region where the water is being used and the desired water quality refers to the actual quality of water used in the process being assessed.

Table 4.3 Determination of water treatment chains for each couple of categories defined by Boulay et al. (2011)

	Desired water quality							
Starting water quality	1	2a	2b	2c	2d	3	4	5
1								
2a	A				A			
2b	B	B		D	B			
2c	A	A	A		A			NA
2d	B	B	B	D				
3	B	B	B	E	B			
4	C	C	C	C	C	C		
5	NA							

A:	Ultrafiltration + Nanofiltration/Reverse Osmosis
B:	Sand Filtration + A
C:	Coagulation/Flocculation/Decantation + A
D:	Sand Filtration + Nitrification + Chlorination
E:	D + Ultrafiltration

4.5.4 Discussion

Results of this paper can be simplified in five distinct cases covering most water types and most of the world: 1) Socio-economic context is limited and no impacts will occur from adaptation, but rather all on human health which can be assessed with other existing models, 2) Water (of this quality) is not constrained and no adaptation is needed, 3) Groundwater is available and unconstrained and can simply be pumped to meet the additional water demand, 4) Water treatment will be used as a backup technology, and is very likely to include a membrane treatment or 5) Special cases for some limited regions which have to be assessed for if and how water transport may occur (mainly represented by the yellow regions in Fig.4-12). These simplifications greatly facilitate the inclusion of indirect impacts from water use in LCA and no further data are required unless results show to be significant and prove to require additional attention.

While this paper presents an innovative approach to further improve the inclusion of indirect impacts from water use in LCA, data uncertainty and modeling choices are sometimes arguable and a discussion is provided here on the main points.

First, the choice of the resource being constrained or not is binary, as often the case in consequential LCA, and is based solely on the presence of stress or not. A binary choice here actually seems consistent with the highly binary results obtained for the stress indexes by Boulay et al.(2011b) or Pfister et al. (2009), implying that for most cases, either water stress is a problem, or it is not. This can easily be explained by the high contribution of irrigation to water use (around 70% worldwide), and its effect particularly present in already dry regions. A water abundant region will practice most of its agricultural activities with rain water whereas a water scarce region which practices agriculture will need to irrigate, hence increasing the gap of resource availability between these two types of regions. However, as discussed in Boulay et al. (2011b) for the modeling of the stress parameter α_i, the actual volume of water use for each water category would be necessary to make this method more robust. Currently, some stress indexes are overestimated, and may result in some resources being defined as constrained when they are not.

When applying the decision tree procedure to determine marginal technology, a conservative approach has been conducted. Thus, for each category of water, it was considered that all pollutant substances were at their maximum concentration. This assumption leads to the definition of treatment chains mobilizing processes covering a wide range of pollutant substances, i.e., membrane technologies. Besides, the use of membrane technologies is also justified by the relatively low threshold concentrations defined for certain parameters in water categories (i.e., concentration in BOD_5). To reach these values, it is necessary to perform advanced treatments. This conservative approach may explain that water treatment chains are very similar as there are only 5 types of combinations for a priori 25 different. A more specific approach can be conducted with case study data. By using decision trees, more appropriate treatment chains would be generated. However, this second approach will be more time and data consuming.

Data regarding groundwater depth is not readily available, and somewhat arbitrary values were used based on limited data. However, it was found that the conclusions are not affected by a pumping depth between 3 and 20 m. Deep wells will however affect the decisions as the costs of pumping from 200m deep are relatively important.

This model does not consider the fact that an increase in water use in a water-stressed region may lead to a re-allocation of resources, rather than an expansion of the water supply by technology, concentrating the water uses on the highest value usages. Since this is a possible response to water stress, if an important water use proves to have significant impacts, a deeper study should be done to better understand the dynamics between the users in a specific region. Moreover, one of the main limits of this model is the choice to apply it solely to marginal uses of water and assuming that import and export from the market of the produced goods will not be affected. While these may be reasonable for relatively small water uses, when assessing larger volumes of water, the influence on the local import and export should be analyzed.

This paper presents a first and innovative approach at assessing impacts from water use, which is especially relevant for developed countries presenting water scarcity and/or poor water quality. Until now, impacts from water use were solely considering deprivation impacts and related impacts on human health, but human activities induced by additional water use should be included. This is done here with a consequential approach, identifying the unconstrained marginal water source and the treatment processes required to produce the water used. By supplying a new inventory, this method offers the possibility to include and assess these impacts with any impact assessment method.

CHAPITRE 5 ARTICLE 3 : ANALYSIS OF WATER USE IMPACT ASSESSMENT METHODS (PART A): EVALUATION OF MODELING CHOICES BASED ON A QUANTITATIVE COMPARISON OF SCARCITY AND HUMAN HEALTH INDICATORS

5.1 Introduction

5.1.1 Background and goal

In LCA, potential impacts from water pollution were traditionally captured by impact categories such as (eco)toxicity, acidification and eutrophication. The impacts of using the resource itself (impacts of water use) and reducing the availability of water for other users—humans and ecosystems—were not yet captured until recently. Since the preliminary discussion on the topic began in the early 2000s (Bauer and Zapp 2005; Brent 2004b; Owens 2002), several methods have emerged and entirely or partially address the different impact pathways outlined in the general framework proposed by Bayart et al.(2010b). Kounina et al.(2013) (see SI, Fig.S1) reviewed and analyzed the developed methods and their scopes, strengths and weaknesses. At the midpoint level, most existing methods quantify water scarcity based on a use-to-availability ratio, referred to as scarcity or stress index. At the damage level, impacts are generally modeled up to specific endpoints within a given area of protection: human health, ecosystem quality or resources.

The review showed that existing methods sometimes model complementary impact pathways or the exact same ones based on different modeling approaches and assumptions. Building on Kounina et al.'s (2013) review, this paper aims to: (i) identify the key relevant modeling choices that explain the main differences between characterization models leading to the same impact indicators for human health impacts; (ii) quantify the significance of the differences between methods, including an assessment of model uncertainty and (iii) discuss the main methodological choices in order to guide method development and harmonization efforts. This paper constitutes the 3rd deliverable of the UNEP/SETAC Life Cycle Initiative working group on water use in LCA (WULCA), and represents a stepping stone towards its goal to develop a harmonized method through scientific consensus on existing methods (UNEP-SETAC Life Cycle Initiative 2013).

5.1.2 Presentation of methods analyzed

The methods chosen for this comparison focus on human health, with scarcity as an intermediate indicator along the impact pathway. Fig.5-1 provides a detailed description of the selected methods along the impact pathways leading to human health. A summary table of the methods and their associated names is presented in SI, Tab.S1. Damage-oriented methods assessing impacts on ecosystems address impact pathways that are considered complementary (Kounina et al. 2013) and are therefore excluded from the scope of the comparison. The resource depletion damage category is still under debate and not yet mature enough to be included in the scope of this paper.

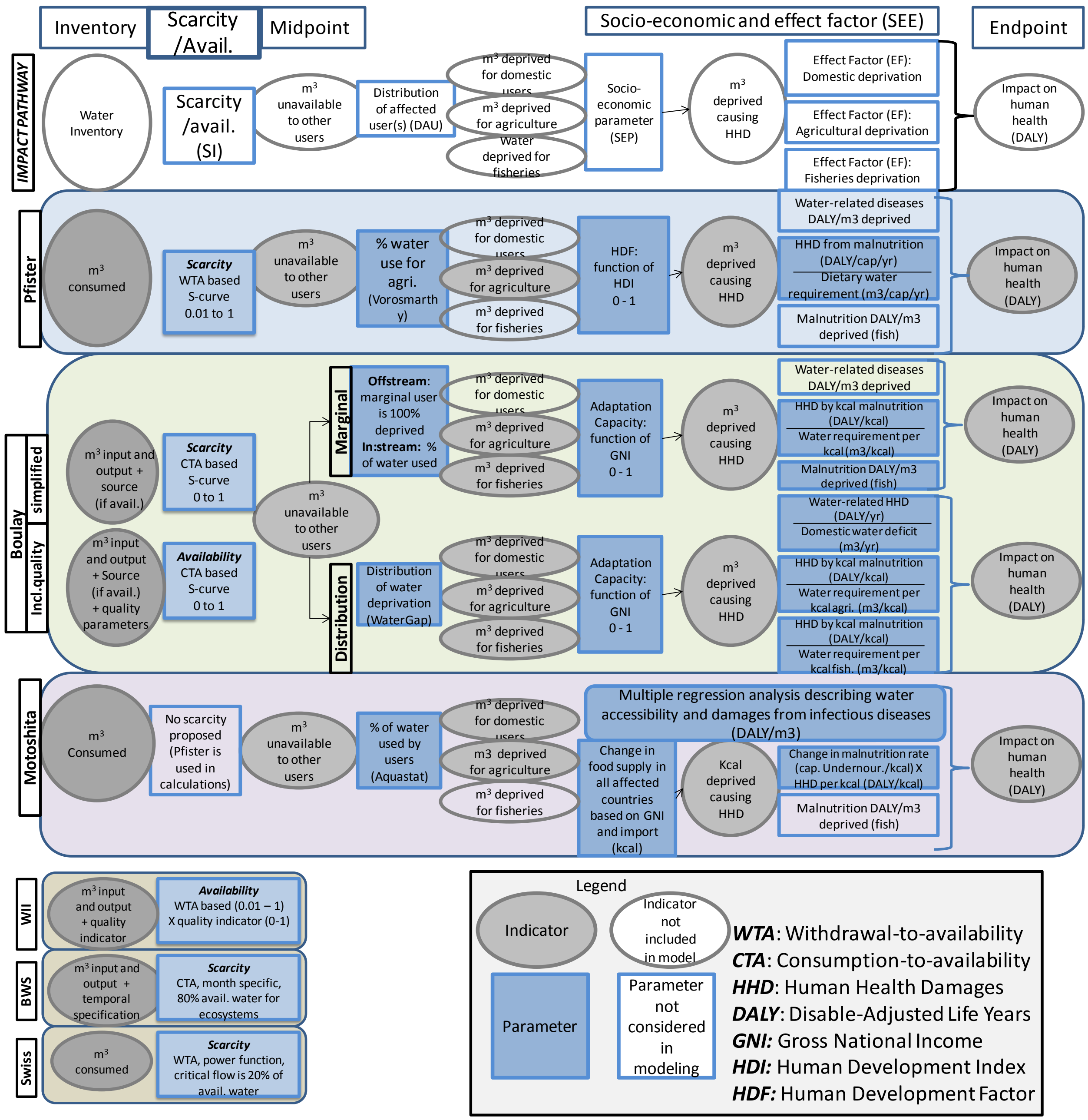

Figure 5-1 Description of method specific impact pathways leading to potential impact to human health. Choices made related to the inventory and in the modeling of the parameters (blue boxes) are analyzed.

5.1.2.1 Midpoint: scarcity and availability

The scarcity and stress methods reviewed by Kounina et al. (2013) are selected for the model comparison, except for Ridoutt and Pfister (2010). The latter was excluded because the authors suggest using a different approach (Ridoutt and Pfister 2013). In this paper, and as a proposal for future consistency, scarcity refers to the pressure on the resource from a quantity perspective only, and availability refers to an assessment of lower water availability due to water quality degradation and quantity depletion. This is in line with the terminology used in the ISO standard available as a Draft International Standard at the time of submission (ISO 14046 2013).

Scarcity methods

- Swiss Ecoscarcity (named M-SwissSc)(Frischknecht et al. 2008)

The Swiss ecological scarcity method is based on the distance-to-target principle, which is similar to using a withdrawal-to-availability (WTA) ratio based scarcity indicator. All withdrawn volumes in a region are considered and divided by the critical water use volume for this region with data from WaterGap (Alcamo et al. 2003b) (in the updated version). The critical volume is defined as the rate of water use at which scarcity begins to occur, set by default to 20% of the water renewal rate. This fraction is then squared and normalized using a reference region (the default is Switzerland). Results are given in eco-points (scaled by a constant to obtain readily presentable numerical quantities) at the country and grid-cell levels (0.5° x 0.5°). The indicator is applied to the volume of water that is consumed or withdrawn and therefore assesses consumptive water use or all water use reported in ecoinvent 2, except for hydropower production.

- Pfister WSI (named M-PfisterSc) (Pfister et al. 2009)

This scarcity indicator is based on a WTA ratio, modified to account for seasonal variations and modeled using a logistic function (S-curve) in order to obtain resulting indicator values between 0.01 and 1 $m^3_{deprived}/m^3_{consumed}$. The curve is tuned using OECD water stress thresholds, which define moderate and severe water stress as 20% and 40% of withdrawals, respectively (Alcamo et al. 2000). The model is available at the grid-cell level (0.5° x 0.5°), and data for water withdrawals and availability were obtained from the WaterGap model (Alcamo et al. 2003b). The indicator is applied to the consumed water volume and assesses consumptive water use only.

- Blue water scarcity (named M-BWSc)(Hoekstra et al. 2012)

This scarcity indicator is based on a consumption-to-availability ratio (CTA) calculated as the fraction between consumed (referred to as blue water footprint) and available water. The latter considers all runoff water, of which 80% is subtracted to account for environmental water needs. The data are from Fekete et al.(2002) for water runoff and Mekonnen et al. (Mekonnen and Hoekstra 2011) for water consumption. Results are available for the main watersheds worldwide but many outlying regions are not covered. The indicator is applied to the consumed water volume and only assesses consumptive water use.

- Boulay – Simplified methodology considering consumptive use only (named M-BoulaySc) (Boulay et al. 2011b)

This scarcity indicator is based on a CTA ratio (using statistical low-flow to account for seasonal variations) and modeled using a logistic function (S-curve) in order to obtain resulting indicator values between 0 and 1 $m^3_{deprived}/m^3_{consumed}$. The curve is tuned using the same water stress thresholds as the OECD water stress thresholds in M-PfisterSc (Alcamo et al. 2000) but converted with an empirical correlation between WTA and CTA. More specific scarcity indicators are also available for surface and groundwater based on the same approach as for water from unspecified origin. Water consumption and availability data for surface and ground water are taken from the WaterGap model (grid cell level). Results are available at a scale that originates from the intersection of the watershed and country scales, resulting in 808 cells worldwide. The simplified method does not consider changes in water quality, unlike the original one (presented in the next paragraph). The indicator is applied to the consumed water volume and assesses consumptive water use only.

Stress methods

- Boulay – Original method, including quality aspects (named M-BoulayAv) (Boulay et al. 2011b)

This stress indicator assesses degradative and consumptive water use. The same characterization model as M-BoulaySc is used, though it is differentiated for eight water categories that each correspond to an inventory flow that describes a type of water (surface or groundwater) of a given quality that is acceptable for specific human uses (domestic, etc.). This indicator assesses degradative and consumptive water use by characterizing input and output flows of water from a process and their difference in quantity and quality. Default values on local availability and water quality are taken from the GEMStat database (UNEP Global Environment Monitoring System (GEMS) Water Programme 2009).

- Veolia Water Impact Index (named M-WIIXAv) (Veolia Water 2010)

The impact index is calculated as the product of (1) a water scarcity indicator (M-PfisterSc) and (2) a quality indicator. The latter is calculated as a ratio between a reference concentration—based on environmental quality standards (EQS) targeted to protect the receiving water bodies—and the actual concentration of the inventory flows. Since EQS are pollutant specific, the quality index is driven by the most penalizing ratio. It is set to a maximum of 1 when the concentration of the inventory flow is below the reference concentration for all pollutants, meaning that the impact of consuming EQS-compliant water yields maximum impacts whereas consuming non EQS-compliant water for at least one contaminant has fewer impacts (the higher the index, the greater the impacts). This indicator assesses degradative and consumptive water uses by characterizing the input and output flows of water in a process.

5.1.2.2 Endpoint impacts on human health

So far, the human health impacts of water deprivation have been modeled using four parameters: 1-scarcity (how much of the water used will deprive other users?); 2-distribution of affected users (which users will be deprived by which fraction of unavailable water?); 3- socio-economic parameter (to what extent will the deprived users suffer health impacts and remain unable to adapt through economic resources?) and 4- effect factor (what is the effect on human health of a specific user being deprived of a certain amount of water?). Equation 5.1 represents how these parameters interact in a generic model. Pfister (2009) and Boulay (2011b) model each parameter explicitly, whereas Motoshita_dom (2010a) uses a statistical regression that merges steps 3 and 4 into a single modeling step (see Fig.5-1). The intermediary parameters express different modeling components along the cause effect chain (see Fig.5-1) and are kept distinct to gain insight into their individual contributions to the total results. Characterization models and factors assessing the impacts of depriving domestic users, agricultural users and/or fisheries of sufficient water are analyzed individually, since the pathways can lead to different direct human health endpoints expressed in DALY (disability-adjusted life years).

$$CF_i\left(\frac{DALY}{m^3}\right) = SI \times DAU_i \times \underbrace{SEP_i \times EF_i}_{SEE_i \text{ factor}} \qquad \text{Equation 5.1}$$

Where:

CF_i: Characterization factor describing the potential human health impacts of water deprivation of user i (agriculture, domestic user or fisheries)

SI: Scarcity or stress index, depending on the inclusion (stress) or exclusion (scarcity) of quality in the index

DAU_i= Distribution of affected users i (i.e. fraction of water use that affects user i)

SEP= Socio-economic parameter

EF_i= Effect factor for water deprivation of user i

SEE_i factor= Socio-economic and effect factor

- Pfister (named E-Pfister)(Pfister et al. 2009)

This endpoint indicator expressed in DALY is obtained by modeling the cause-effect chain of water deprivation for agricultural users (lack of irrigation water) leading to malnutrition. It assumes that there is no general causality from water consumption to lack of water for domestic use, arguing that water access for domestic use is mainly dependent on infrastructure (and not on water) availability. It builds on the scarcity indicator (M-PfisterSc) and models the cause-effect chain by multiplying it by 1) the agricultural users' share of water use (as DAU) from Vörösmarty (2000c)), 2) a socio-economic parameter defined as a human development factor for malnutrition, which relates the Human Development Index (a composite index representing human development by considering life expectancy, education and income published by the UNDP) to malnutrition vulnerability and 3) two values independent of location combined in an effect factor that describes the DALY/m^3 of water deprived for agriculture: the per-capita water requirements to prevent malnutrition (in m^3/(yr•capita)) and the damage factor denoting the damage caused by malnutrition (DALY/(yr•capita)). The effect factor therefore carries two underlying assumptions: 1) global malnutrition health impacts are exclusively caused by a lack of water for irrigation and 2) a case of malnutrition occurs only once all the water required for one person is no longer available. The first assumption may lead to an overestimation of impacts while the second may lead to an underestimation. The results are derived on a 0.5° x 0.5° grid cell scale and aggregated at the watershed level (>10 000 watersheds, as in (Alcamo et al. 2003a)).

- Boulay (named E-Boulay with different variants: _agri, _dom, _marg, _distri, _Q)(Boulay et al. 2011b)

This endpoint indicator expressed in DALY is obtained by modeling each water user's loss of functionality. It addresses three different impact pathways: malnutrition from water deprivation for agricultural users, malnutrition from water deprivation for fisheries and water-related diseases associated with a lack of water for domestic use. Four model scenarios are considered by E-Boulay as a cross-combination of both original versions (addressing consumption and degradation with suffix _Q) and the simplified version (which only addresses consumption) of the model and two key modeling hypotheses: distribution and marginal. Distribution (Boulay_distri or Boulay_distri_Q) refers to the impact assessment in which all users are competing and proportionally affected according to their distributional share of water use for off-stream users (here, agriculture and domestic). Marginal (E-Boulay_marg or E-Boulay_marg_Q) refers to a modeling choice in which an additional water use will deprive only one off-stream user (in addition to in–stream users here, fisheries). The one for which water has less value was set as agriculture by default. This hypothesis therefore excludes potential impacts to domestic users. The distribution among users from WaterGap is used to determine the distribution of affected users for Boulay_distri (factor giving the m^3 deprived distribution between affected users). The socio-economic parameter used in all E_Boulay methods is one minus the adaptation capacity (AC). High-income countries are considered to fully adapt (AC = 1) whereas low-income countries are considered not to adapt at all (AC=0) to water deprivation. The adaptation capacity of medium-income countries is considered to be linearly correlated with GNI per capita. The effect factor uses country-specific statistical data to obtain the relationship between health impacts from malnutrition (DALY/kcal of malnutrition x kcal produced/m^3 for agricultural use or kcal produced/m^3 for aquaculture use), associating 50% of malnutrition health impacts to a lack of calorie intake and the remaining 50% to water-related diseases, since they often lead to malnutrition. These 50% are added to the health impacts in DALY from water-related diseases and divided by the amount of water lacking for domestic use based on the minimum requirement of 50L/cap/day and the actual regional water use by domestic users, resulting in DALY/m^3 deprived for domestic use. For both effect factors, a linear relationship is therefore assumed between the health impact and the deprived water. The results are presented according to the M-Boulay spatial scale, resulting from the overlap of the country and main watershed scales for the simplified alternative versions (E-Boulay_distri and E-Boulay_marg) and original methods (E-Boulay_distri_Q and E-Boulay_marg_Q). In this paper, for several analyses, the aggregated CFs are separated into domestic and agricultural deprivation parts and referred to as E-Boulay_dom and E-Boulay_agri, respectively.

- Motoshita (named E-Motoshita with different variants:_dom, _agri, _agri (no TE))

This damage assessment model is based on the sum of two distinct models: one for infectious disease damage caused by domestic water scarcity (Motoshita et al. 2010a) (E-Motoshita_dom) and one for malnutrition damage caused by agricultural water scarcity (Motoshita et al. 2010b) (E-Motoshita_agri).

For domestic water scarcity, the method assumes that water resource scarcity caused by water consumption will lead to a loss of access to safe water. Subsequently, based on location, drinking unsafe water will result in the use of infectious sources and health impairment by disease. The method provides country-based CFs expressed in DALY/m^3 of water consumed obtained with M-PfisterSc as a scarcity assessment, multiplied by the share of water used by domestic users (from Aquastat) and a combined socio-economic and effect factor obtained by applying non-linear multiple regression analysis considering related socio-economic factors such as GDP, expenditure for capital formation, average temperature, sanitary facilities, nutritional conditions and health expenditure based on statistical data. The factor represents the inaccessibility to safe water due to domestic water scarcity and a subsequent increase in infectious diseases (intestinal nematode infection and diarrhea).

The impacts of malnutrition caused by agricultural water deficit are modeled using the same data source for scarcity and distribution as above, multiplied by a socio-economic parameter describing the trade effect. This illustrates how food supply shortage in a country will spread to other countries through international food trade. It applies a food shortage sharing model based on the proportion of world net import amount (in kcal) for net food importer countries that are not able to adapt (or only partially able to adapt) using the adaptation capacity defined in Boulay et al.(Boulay et al. 2011b) based on gross national income (GNI). For example, if 1 000 kcal of food are not produced in Spain due to water shortage and local crop productivity, the amount will be distributed among all the world's net importer countries proportionally to the amount they import (in kcal). Countries with low and middle incomes will be affected by the food shortage. This effect is quantified in DALY by using malnutrition-related DALYs in the importing countries (Dalys/kcal malnutrition). The method provides country-based characterization factors in the context of both domestic and agricultural water scarcity, expressed in DALY per m^3 of water consumed. The method can also be used without the trade model (E-Motoshita_agri (noTE)) to compare local effects.

5.2 Method

The following section describes how the analysis was performed. It is divided into three parts: comparison, analysis of modeling choice and uncertainty assessment. The comparison first assesses how the model results compare at the characterization factor level and the respective intermediary parameters (identified by the blue squares in Fig.5-1). A set of modeling choices were identified, and their sensitivity in terms of the final results was analyzed using two versions of the same model, differing only by the option being analyzed (e.g. the use of one or another source of data for modeling). For each model, the uncertainty assessment quantifies the uncertainty associated with the choice of model only.

Two statistical indicators were used to compare the models. The difference between the model responses was assessed through the mean difference coefficient (MDC), and the consistency of model response through the rank correlation coefficient (RCC), which are defined below. The correlation coefficient (Pearson's) was not considered an appropriate indicator because the data revealed heteroscedasticity (i.e. the difference between the values given by two methods is not independent of the value itself). When the homoscedasticity assumption is violated, Pearson's coefficient of correlation may overestimate the goodness of fit.

The comparison sought to analyze the degree of model response agreement and consistency from one model to the next, rather than their correlation. Two models can have 100% correlation but may still disagree. A mean relative coefficient (MDC), as described in Equation 5.2, was used to represent the difference between two models. It illustrates a mean relative difference, which is the mean of the absolute differences between each data pair divided by their average. It measures dispersion, just like the standard deviation would, but it is not defined in terms of a specific measure of central tendency: it represents the difference between two measurements, not their deviation from an arithmetic mean. Also, the standard deviation squares its differences, giving more weight to greater differences and less weight to smaller differences compared to the mean difference. It can be interpreted similarly to a coefficient of variation, with a higher value representing a greater difference between models. It should be noted that the maximum value for the MDC is equal to the number of datasets compared, such that when two datasets are compared, the maximum value of MDC is 2, since a large difference will result in one value being negligible as compared to the other, making the largest value divided by half of its value (i.e. equaling 2).

$$MDC = mean(\frac{Difference\ between\ data\ set}{Mean\ of\ data\ set}) \qquad \text{Equation 5.2}$$

The rank correlation coefficient (RCC) is also referred to as the Spearman coefficient and is used to represent the consistency between two models based on the respective ranks that each regional parameter (at country or region level) would occupy. The RCC ranges between 0 and 1: the higher the value, the more consistent the models. This method was successfully used by Fenner et al.(2005), who aimed to compare models by ranking model outcomes. This is especially relevant for comparative LCAs.

5.2.1 Model comparisons

5.2.1.1 Scarcity indicators

The first comparison is a generic comparison of all four scarcity assessment methods (midpoint), as identified in Fig.5-1: M-SwissSc, M-PfisterSc, M-BWSc and M-BoulaySc. The comparison was carried out at the watershed level with the 250 watersheds from the World Resource Institute as the finest common resolution (Aguilar-Manjarrez 2006). Since all four methods yield results in different units (m^3 equivalent referring to different equivalencies or ecopoints), they are normalized using their respective world weighted averages using withdrawal volumes as weighting factors. Normalized results therefore correspond to equivalent units of "world- m^3 equivalent" for all methods.

The RCC and MDC between each pair of methods were calculated (M-BoulaySc vs. M-PfisterSc, M-BoulaySc vs. M-BWSSc, etc.).

5.2.1.2 Availability indicators

The M-BoulayAv and M-WIIXAv availability indicators both consider water scarcity and change in quality, thus making them principally comparable. However, the fundamental basis upon which these methods assess the change in water quality is different. While M-BoulayAv assesses a change in quality based on the functionality of water for human users, M-WIIXAv quantifies the change in quality based on environmental standards for ambient water quality, mainly oriented ecosystems. The methods do not actually aim to model the same impact pathway, and the comparison is therefore irrelevant. This is further addressed in the discussion section.

5.2.1.3 Human health impacts: Overall CF

The characterization factors (CFs), as presented in each of the four main models, are directly compared in pairs. However, to enable an adequate comparison, only the simplified versions of the Boulay methods—those that disregard water quality—are used. The effect of this modeling choice is further analyzed in section 5.2.2. Since the Motoshita model results are only available at the country level, this scale was used for all endpoint analyses.

5.2.1.4 Human health impacts: Domestic user deprivation

The impacts of depriving domestic users are assessed in E-Motoshita_D and E-Boulay_distri. Only the domestic component of Boulay_distri is used in this comparison and is referred to as E-Boulay_dom. First, the entire CFs are compared. Then, the scarcity and distribution of affected users (DAU) parameters are removed from both methods and the socio-economic and effect factors (SEE) are compared. The removed components (scarcity and DAU) were compared in other parts of this paper (sections 5.2.1.1 and 5.2.2.3). The SEE factors are regionalized parameters in both methods and describe the human health impacts of domestic user deprivation in DALY/m^3 of lacking water. In E-Boulay_dom, the adaptation capacity provides a regionalized resolution, since the value of DALY caused per m^3 of water lacking for domestic users is the same worldwide. In E-Motoshita_dom, this value is regionalized by modeling the loss of accessibility to safe water and the subsequent increase of infectious disease damage is regionalized by applying statistical regression analysis based on country-specific data.

5.2.1.5 Human health impacts: Agricultural user deprivation

The impacts of depriving agricultural users are assessed in all four methods: E-Motoshita_agri, E-Pfister, E-Boulay_distri and E-Boulay_marginal. The agriculture component of E-Boulay_distri is considered here and referred to as E-Boulay_agri. The models are compared on three levels: (i) the CFs; (ii) the product of the socio-economic and effect factors (SEEs) (isolated and compared by removing the scarcity factors and distribution of affected users in both methods); (iii) the effect factors alone. This last comparison can only be done for E-Pfister and E-Boulay, which both assess a single worldwide value that describes the impacts in Daly per m^3 of water lacking for agricultural users.

5.2.2 Analysis of specific modeling choices

Several modeling choices may affect the inventory requirements and the four modeling parameters identified in Fig.5-1 and Equation 5.1: scarcity/availability, distribution of affected users, socio-economic parameter and effect factor. A specific number of key choices that differ from one method to the next are identified below, and, for each model, the importance of the choice is quantified by assessing the consistency (RCC value) and difference (MDC value) between the two versions of the same model in which different choice options are applied. No choice on effect factor is analyzed here as they are directly compared and analyzed in the previous section.

5.2.2.1 Inventory-related choices

Four model specifications that affect the level of detail required for the inventory flows are identified: temporal resolution scale, water source, regional resolution scale and quality aspect. We evaluated the extent to which the **models with** a ***higher level of detail*** leading to higher spatially- or temporally-resolved inventory flows and/or more detailed specifications on water source and water quality increase the discriminating power of model outcomes.

In daily practice, inventory data at lower (or unknown) spatial and temporal resolution, water withdrawals or releases without quality or water source specification are common situations. CFs for the corresponding inventory flows are mainly generated by two different approaches: i) by adopting a lower level of detail e.g. calculating national CF using national averaged model input parameters (such as water consumption and availability) or using total available and consumed water instead of differentiating surface versus ground water or ii) by keeping the highest level of detail to calculate specific regional CFs and aggregating them using weighted averages to calculate e.g. a national CFs using water withdrawals in each sub-watershed as weighting factor, or by calculating an "unspecified origin" CF based on surface and ground water CFs using ground and surface water withdrawals as weighting factors. We evaluated the influence of these choices, which resulted in **models with a *lower level of detail*** versus the aforementioned higher-resolution models.

Temporal resolution scale

Higher level of detail: Monthly assessment

Water scarcity is known to be a seasonal problem in many regions of the world. While most indices are annual, two methods provide monthly indicators: M-BWSSc and M-PfisterSc (Pfister and Baumann 2012). M-PfisterSc is used to compare the original annual values with individual monthly values. The largest absolute difference between a monthly value and the annual value is calculated for each region and geo-referenced on a map to identify the regions in which collecting inventory data with higher temporal resolution is worthwhile.

Lower level of detail: Annual assessment

To characterize the inventory data without any temporal specification, the generic CFs must be recalculated: i) using annual averaged input values or ii) using a weighted average of monthly CFs based on total monthly water withdrawals. The absolute difference between the two options is illustrated on a map, and the MDC was calculated.

Water source

Higher level of detail: Specifying surface and ground water sources

It is relevant to differentiate the water sources used in the inventory since the decreased availability of surface or ground water will not affect the same users. Even though surface and ground water are often interconnected, transport, hydropower and fisheries cannot use groundwater. The M-BoulaySc method is used to evaluate the importance of specifying the water source (surface or ground).

Lower level of detail: Unspecified source

If the source is not specified, two approaches may be used to characterize the inventory flow: i) assess all available and consumed water as a single resource or ii) use a weighted average of surface and ground water CFs using the fraction of total regional surface and ground water withdrawals, respectively, as weighting factors.

Regional resolution scale

Higher level of detail: Watershed and sub-watershed scale

The difference in geographical resolution between sub-watershed, watershed and country is assessed using M-BoulaySc. The last two resolutions are obtained from the withdrawal-based weighted averages of sub-watershed results. The MDC was calculated in comparison with the country scale.

Lower level of detail: Country scale

For a country-level assessment, the following scarcity indexes were compared using M-BoulaySc: i) country-level data based scarcity indexes ii) weighted average of watershed or iii) sub-watershed scarcity indexes, using water withdrawals as weighting factors.

Quality aspect

Higher level of detail: Water quality specification

Water that is released at a lower quality than withdrawn may become unusable by some users, thus reducing water availability for downstream users. The original M-BoulayAv method assessing both degradative and consumptive water use is compared with the simplified M-BoulaySc, which addresses only consumptive water use. The results were compared according to three hypothetical scenarios: i) 100% consumption of good quality surface water (S2a), ii) 100% consumption of poor quality surface water (S3) or iii) 100% degradation of good quality water (S2a) into very poor quality water (S4).

The same three hypothetical cases were analyzed at the endpoint level using the E-Boulay_distri and E-Boulay_distri_Q methods to assess the difference in human health impacts when considering the impacts of water consumption alone and those generated by water consumption and degradation.

Lower level of detail: Unspecified quality

Comparing i) M-BoulaySc (no quality specified) with ii) a weighted average of M-BoulayAv CFs using amounts of water of different quality withdrawn from different watersheds would be of interest. However, such quality-specific withdrawal data are not available, meaning that CFs of "unspecified quality" can only be calculated using the "lower level of detail" approach (i.e. using total water without any quality specification). Therefore, no comparisons were possible for this parameter.

5.2.2.2 Scarcity modeling choices

Water scarcity indexes were developed using withdrawal-to-availability ratios (WTAs) (i.e. M-PfisterSc, M-SwissSc) or consumption-to-availability ratios (CTAs) (i.e. M-BoulaySc, M-BWS-Sc). Moreover, hydrological data sources and scarcity model algorithm change from one method to the other. While M-SwissSc squares the WTA, M-BWSSc subtracts 80% of available water for ecosystems. M-PfisterSc and M-BoulaySc both use S-curve modeling to fit the ratio (WTA and CTA, respectively) to values between 0 (For M-BoulaySc) or 0.01 (for M-PfisterSc) and 1. The curve is tuned using withdrawal-based water scarcity thresholds (in M-PfisterSc), which describes it as moderate or severe when respectively 20 or 40% of the resource is withdrawn ((Alcamo et al. 2000; Vörösmarty et al. 2000c)). Alternatively, the curve is tuned using consumption-based equivalent thresholds (in M-BoulaySc) extrapolated from the withdrawal-based ones, as being 6 and 12% of the consumed resource (values updated from (Boulay et al. 2011b) with more recent data). The following analyses were performed.

Consumption-based vs. withdrawal-based scarcity (CTA vs. WTA)

Water withdrawals partly return to the catchment where they were extracted (Perry 2007), and it has therefore been argued that a consumption-based indicator (CTA) is more relevant than a withdrawal-based indicator (WTA) (Boulay et al. 2011b)(Berger and Finkbeiner 2012). Two analyses were carried out to evaluate the model choice. First, CTAs and WTAs were directly compared using the underlying data from WaterGap through the rank correlation coefficient for the 808 cells covering the globe, as used in M-BoulaySc. Second, using the same model, WTA-based scarcity (based on the original OECD thresholds) was compared with CTA-based scarcity (based on the aforementioned extrapolated scarcity thresholds). While the original M-BoulaySc model uses an S-curve to describe the relationship between CTA and scarcity between the two thresholds that define low and high scarcity, we linearized the curve in order to exclude the differences related to the algorithms used to fit the curves.

Scarcity model algorithm

The four modeling choices used to translate CTAs and WTAs ratios into scarcity indicators were evaluated: i) S-curve modeling between the thresholds for low and high scarcity, set at 0 and 1 respectively, as in M_BoulaySc and M_PfisterSc; ii) linear function between the thresholds for low and high scarcity, set at 0 and 1 respectively; iii) power function applied to the ratio of water consumed to a critical flow, as described by M-SwissSc and adapted to consumptive use; and iv) direct use of the ratio considering that 80% of available water is reserved for ecosystems, as modeled in M-BWSSc. These modeling choices were applied using CTAs calculated with the WaterGap data. Values were normalized for comparison purposes and plotted on an x-y graph.

Data sources for water availability and water use

In order to assess the importance of the hydrological data source (water availability and water use), the original version of M-BoulaySc with data from WaterGap was compared to the same model using water consumption and availability data from Fekete et al. (Fekete et al. 2002), as in M-BWSSc. This comparison was performed on the main watersheds for which data were available.

5.2.2.3 Affected users

While all current methods suggest that water use can lead to water deprivation for agriculture, the same is not true for domestic users or aquaculture/fisheries. Based on the existing models, the impact of the choice is analyzed along with the data source used to assess the extent to which a specific user is deprived (DAU).

Aquaculture/fisheries

Only the E-Boulay methods include the impacts of water deprivation on aquaculture/fisheries. The contribution of the impact pathway to the total human health impacts was analyzed by comparing E-Boulay_marginal method with and without the aquaculture deprivation impacts.

Domestic

While Pfister et al. stipulate that increased water use will not generally affect domestic users, Motoshita et al. set out a model that quantifies human health impacts from water deprivation for domestic users. In the Boulay et al. model, both options are offered, and the choice is left to the practitioner to include (distribution) or exclude (marginal) the effect on domestic users. The alternatives are compared, and MDC and RCC are calculated.

Data source for the distribution of affected users

National values for user distribution vary depending on the data source: WaterGap (used in E-Boulay), Aquastat (FAO 2009b) (used in E-Motoshita) or Vorosmarty et al. (Vörösmarty et al. 2000c) (used in E-Pfister). To assess the importance of these sources, E-Boulay_distribution was run using all three data sources. The results were compared.

5.2.2.4 Socio-economic

One of the main diverging choices that describes the influence of the economic context on malnutrition resulting from water use is the consideration of a trade effect in E-Motoshita_agri, which illustrates how a food supply shortage in a country will spread to other countries through international food trade. The extent to which the inclusion of this effect impacts the results is analyzed by comparing E-Motoshita_agri (no TE) to E-Motoshita_agri, E-Boulay_agri and E-Pfister.

5.2.3 Uncertainty assessment of model choice

There are several types of uncertainty, including parameter uncertainty, model uncertainty, decision rule uncertainty, natural variability, etc. Here, the uncertainty associated with the choice of model is assessed at midpoint and endpoint. At midpoint, the assessment is carried out for all major watersheds compared in 2.1.1 for scarcity assessment methods only (availability methods are not comparable). The uncertainty was determined by using each set of normalized data to identify the minimum and maximum values between the models for the same watershed. These values were then re-converted to the scale of each model (i.e. “de-normalized”) in order to provide a method-specific min-max range per watershed. Using the normalized results obtained through the different methods, an average value was also provided for each watershed in m^3 world-normalized equivalent per m^3 water consumed, including a 95% confidence interval.

The uncertainty of the choice of model was assessed for the different human health endpoints (from water deprivation for domestic and agricultural users). No normalization step was necessary since all models represent the same damage unit (DALY) and the minimum and maximum are identified across models assessing impacts on the same user. An average between the different method results is calculated for impacts on domestic and agricultural users, with a 95% confidence interval bracket.

5.3 Results and discussion

Table 5.1 Rank correlation coefficients (RCC) and mean differences (MDC) for model comparisons and choice analysis. The smaller the colored bars, the more consistent /in agreement the models or the choices. Sections in green refer to midpoint result analysis.

5.2.2 CHOICE ANALYSIS	Rank correlation coefficient (RCC)	Mean difference coefficient (MDC)
5.2.2.1 Inventory		
a) Temporal resolution scale		
Higher complexity: monthly assessment	96%	0.23
Annual modeling choice	98%	0.13
b) Water Source		
Higher complexity: Surface water specification	93%	0.18
Higher complexity: Groundwater specification	87%	0.27
Unspecified source modeling choice	95%	0.23
c) Regional resolution scale		
Higher complexity: Watershed vs country	81%	0.53
Higher complexity: Sub-Watershed vs country	48%	1.46
Country-scale modeling choices:Country vs watershed aggregation	90%	0.48
Country-scale modeling choices:Country vs sub-watershed aggregation	82%	0.88
d) Quality		
Higher complexity: water quality specification at midpoint		
100% consumption good quality water	64%	1.26
100% consumption poor quality water	91%	0.74
100% degradation good quality water into very poor quality	31%	1.56
Highercomplexity: water quality specification at endpoint		
100% consumption good quality water	46%	0.79
100% consumption poor quality water	59%	1.24
100% degradation good quality water into very poor quality	43%	0.83
5.2.2.2 Scarcity		
a) CTA vs WTA	96%	n/a
in M-BoulaySc method	94%	0.24
b) Scarcity Model algorithm		
Linear function vs S-curve	100%	0.19
Direct vs linear function	74%	1.70
Power vs linear function	74%	1.92
Power function vs DirectBWS	100%	1.91
Power function vs S-curve	74%	1.90
Direct vs S-curve	74%	1.73
c) Data Source (WaterGap vs WFN data)		
CTA WaterGap vs CTA WFN	91%	0.90
in M-BoulaySc method	75%	0.37
5.2.2.3 Affected users		
a) Aquaculture	100%	0.0004
b) Domestic	83%	0.75
c) Source of data		
Watergap - Aquastat	93%	0.30
Watergap - Vorosmarthy	87%	0.59
Vorosmarthy - Aquastat	89%	0.60
5.2.2.4 Socio-economic		
Trade effect		
E-Motoshita_agri / E-Motoshita_agri (no TE)	76%	1.31
E-Motoshita_agri(no TE) / E-Boulay_agri	65%	1.83
E-Motoshita_agri (no TE) / E-Pfister	75%	1.15
* the longer the red bar the lower the correlation		
*the longer the blue bar, the higher the difference		

Table 5.1 (Continued)

5.2.1 COMPARISON	Rank correlation coefficient (RCC)	Mean difference coefficient (MDC)
5.2.1.1 Scarcity		
M-PfisterSc/M-BoulaySc	71%	1.61
M-SwissSc/M-BWSSc	41%	1.58
M-PfisterSc/M-BWSSc	50%	1.19
M-PfisterSc/M-SwissSc	64%	1.62
M-BoulaySc/M-SwissSc	47%	1.58
M-BoulaySc/M-BWSSc	32%	1.56
5.2.1.3 Human Health CF		
E-Boulay_distri / E-Boulay_marg	83%	0.75
E-Boulay_distri / E-Motoshita	35%	1.81
E-Boulay_distri / E_Pfister	71%	1.70
E-Pfister / E-Motoshita	59%	1.41
E-Pfister / E-Boulay_marg	80%	1.60
E-Motoshita / E-Boulay_marg	45%	1.75
5.2.1.4 Domestic water deprivation		
Total domestic CF		
E-Motoshita_dom/E-Boulay_dom	45%	1.82
SEE Factor		
E-Motoshita_dom/E-Boulay_dom	78%	1.72
5.2.1.5 Agriculture water deprivation		
Total agriculture CF		
E-Boulay_agri/ E-Motoshita_agri	56%	1.70
E-Boulay_agri / E_Pfister	81%	1.59
E-Pfister / E-Motoshita_agri	49%	1.31
SEE Factor		
E-Boulay_agri/ E-Motoshita_agri	-25%	1.70
E-Boulay_agri / E_Pfister	88%	0.76
E-Pfister / E-Motoshita_agri	-40%	1.77
Effect Factor		
E-Boulay_agri / E_Pfister	20%	1.16
* the longer the red bar the lower the correlation		
*the longer the blue bar, the higher the difference		

5.3.1 Model comparison

5.3.1.1 Scarcity indicators

The highest consistency was observed between M-BoulaySc and M-PfisterSc (RCC = 71%), which is explained by the choice of similar low and upper scarcity thresholds and logistic function (S-curve). A comparative graph is included in SI (Fig.S2).

5.3.1.2 Availability indicators

The two availability assessment methodologies were not compared quantitatively since they target two distinct areas of protection. M-BoulayAv is a availability indicator at an intermediate modeling step to assess water deprivation for human uses and the resulting impacts on human health. M-WIIXAv addresses the potential impacts of a loss of quality based on ecosystem quality standards. This could be considered as a potential midpoint indicator for the impact pathway leading to the ecosystems quality area of protection. However, it is not clear which additional impacts are not already captured in specific pollution indicators. This indicator should be used with caution in an LCA context to avoid double counting with impact categories such as ecotoxicity or eutrophication. M-WIIXAv could be used in parallel to evaluate contaminants that are not addressed by other methods (e.g. fecal coliforms, COD, etc.).

For both methods, water quality data remains a weak point since global datasets providing environmental concentrations have limited measurement points for several regions of the world.

5.3.1.3 Human health CF

Fig.5-2 shows the comparison between endpoint CFs. Both E-Boulay methods (distribution and marginal) yield generally higher results than E-Motoshita, with the latter showing higher results than E-Pfister. E-Boulay_distri results are higher than E-Boulay_marg, since the impacts of domestic user deprivation are greater than those of agricultural user deprivation and only included in E-Boulay_marg. Since the graph is on a log scale, zero values are not plotted, despite their relevance (60 of the 175 countries).

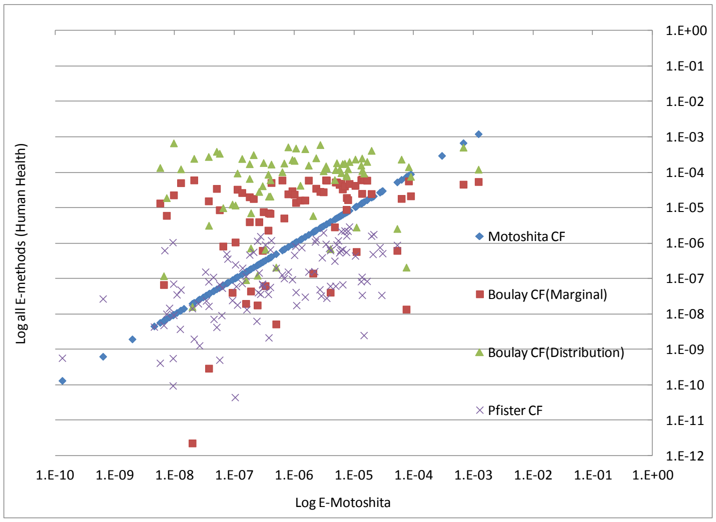

Figure 5-2 Comparisons of human health CFs from water use provided by E-Boulay_marg, E-Boulay_distri, E-Motoshita (domestic+agriculture) and E-Pfister

5.3.1.4 Human health: Domestic user deprivation

Fig.5-3 compares the E-Boulay and E-Motoshita model results for the pathways linking water deprivation for domestic users to human health impacts. The CFs of E-Boulay_dom are generally higher than E-Motoshita_dom. The rank correlation between the two models is low (45%), and they differ significantly (MDC is relatively high, 1.82). A higher correlation is observed for the intermediary parameter SEE (RCC of 78%) (i.e. excluding both the scarcity and distribution of the affected users intermediary parameters (see Equation 1). The MDC, however, remains relatively high at 1.72.

For the 127 CFs analyzed, in E-Boulay_dom, there are 60 values for which the result is 0 and only 6 in E-Motoshita_dom (Fig.5-3). The largest differences are in poor countries with no scarcity problem according to E-Boulay_dom but which show non-zero scarcity values in E-Motoshita_dom (coming from M-PfisterSc with the lowest scarcity equal to 0.01). These countries include Angola, Central Africa, Benin, Burundi, Congo and Ghana.

When focusing on the SEE factor (see SI, Fig.S3), E-Boulay_dom had non-zero values for 107 of 139 countries analyzed versus 128 non-zero values in E-Motoshita_dom.

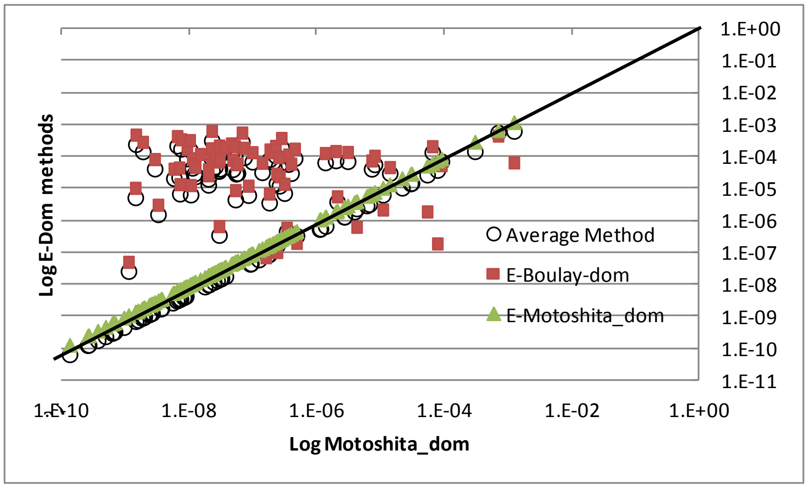

Figure 5-3 Comparison of human health model outcomes from domestic water deprivation impact pathways using Boulay and Motoshita models.

5.3.1.5 Human health: Agricultural user deprivation

The E-Pfister and E-Boulay_agri CFs show the highest consistency (RCC=81%), while both methods demonstrate low consistency with E-Motoshita_agri (56-49%). A comparison of regional human health CFs from water deprivation for agriculture is shown in Fig.5-4. Of the 124 countries analyzed, E-Boulay_agri generated zero-value for 57 versus 17 and 3 for E-Pfister and E-Motoshita_dom, respectively. The zero-values in E-Boulay_agri come from the scarcity and socio-economic parameters, which were both set at zero when below the threshold set to define each issue. In general, E-Boulay_agri yielded greater impacts than E-Pfister, and, in most cases, E-Boulay_agri also led to more significant impacts than E-Motoshita_agri.

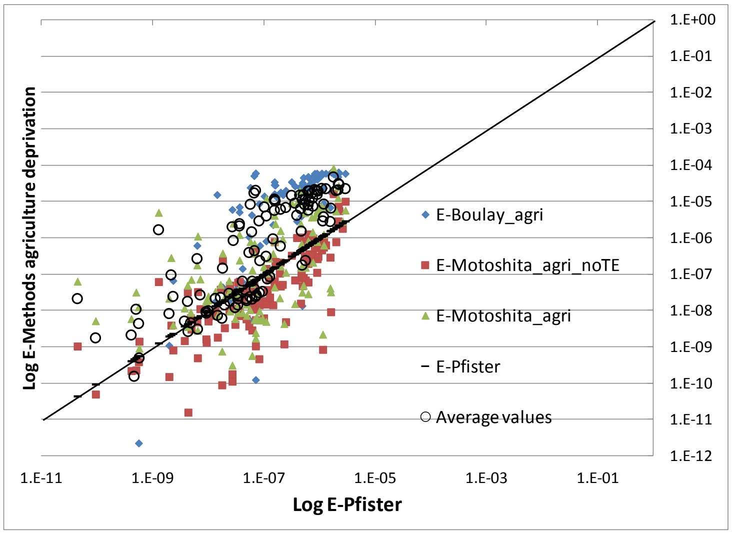

Figure 5-4 Comparison of agriculture water deprivation impacts on human health.

When comparing the SEE factors alone (see Equation 5.1), the correlation between E-Motoshita_agri and_Boulay_agri and E-Pfister E_drops to a negative value since the correlation was driven by scarcity and the distribution of affected users. The results of E_Boulay_agri and E-Pfister are very consistent (88%), and the MDC (0.76) is relatively low.

Focusing on the effect factor only (i.e. disregarding the distribution parameter (DAU)), E-Pfister and E-Boulay_agri show constants values (i.e. independent of location): 1.363 x 10^{-5} DALY/m^3 and 6.53 x 10^{-5} Daly/m^3, respectively. E-Pfister considers a minimum volume of water needed to meet direct human dietary requirements (1 350 m^3/(yr•capita)) and a damage factor from malnutrition. The latter is derived from a linear regression between country-specific malnutrition rates and human burdens related to malnutrition (DALY), resulting in a per-capita malnutrition damage factor of $1.84 \cdot 10^{-2}$ DALY/(yr•capita). The effect factor is obtained by the ratio between the two values. The effect factor of E-Boulay_agri directly relates the average health burdens caused by calorie malnutrition (DALY/kcal) to the total calorie deficit of a given population. The geometric mean across all low- and middle-income countries facing malnutrition was calculated (1.27 x 10^{-7} DALY/kcal). A similar value of 1 278 m^3/(yr•capita) is considered to meet a direct human dietary requirement of 2 800 kcal/(day•capita), resulting in an average agricultural productivity of 800 kcal/m^3, which is then corrected to account for the share of agricultural produce used to feed livestock. The effect factor is obtained by multiplying the malnutrition burden by the corrected agricultural productivity. The connection between malnutrition and water deprivation for agriculture in E-Pfister assumes that a case of malnutrition occurs only when the entire water requirement for one person to eat for one year is consumed. But the consequences of malnutrition will occur long before this amount of water is consumed, thus explaining the lower value than E-Boulay_agri, which assumes the linear effect of malnutrition per kcal deprived. SEE and EF factor graphics are included in SI.

Overall, with respect to E-Boulay, E-Pfister yielded a lower effect factor, higher SEE and lower CF. One can deduct that the socio-economic parameter is responsible for the higher SEE, and the distribution of affected users is responsible for the lower CF—a parameter analyzed in section 5.2.2.3.

5.3.2 Analysis of specific modeling choices

5.3.2.1 Inventory-related choices

Temporal resolution scale

Higher level of detail: Monthly assessment

Fig.5-5 shows the maximum absolute difference between the monthly water scarcity indicators versus the annual value. It is to be compared with the original range of 0.01 to 1 of the M-PfisterSc scarcity indexes. The difference remains below 0.1 for large areas of the world and is significant (0.1 – 0.5) and very large (> 0.5) in most of the US, Europe and India. This difference would lead to higher results for month-to-month comparisons. The high consistency between monthly values and annual values (96%) and relatively low MDC (0.23) suggest that, with the exception of certain specific locations identified in Fig.5-5, the water scarcity indicators for different regions that assess the water withdrawn at the same time of the year are not likely to be affected by a higher temporal resolution.

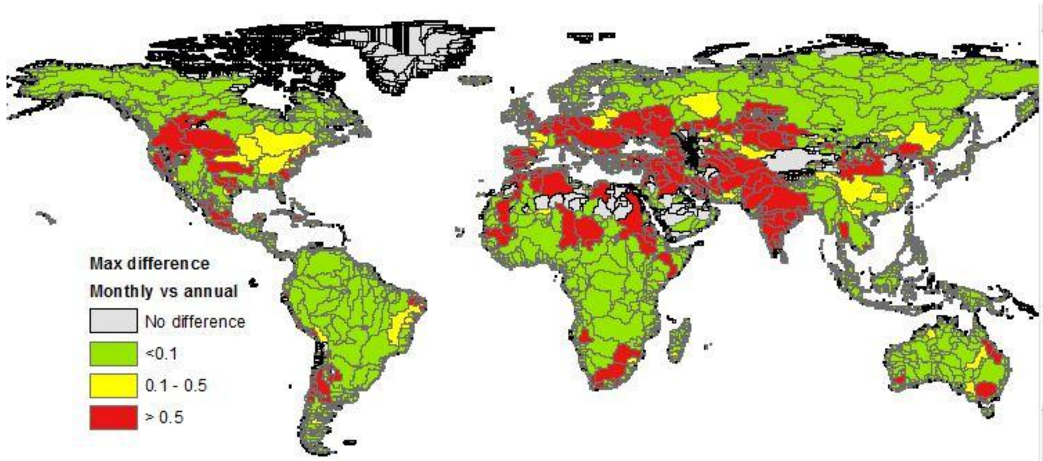

Figure 5-5 Maximal absolute difference between monthly water scarcity indicators of wettest/driest month and the annual value. Results are obtained with M-PfisterSc, which scarcity indexes range from 0.01 to 1.

Lower level of detail: Annual assessment (annual data or monthly weighted average based on withdrawals)

Scarcity indicators based on annual model input data versus indicators aggregated from monthly scarcity indicators based on a weighted withdrawal average are highly correlated (RCC = 98%) and show low MDC (0.13). Exceptions are in regions mainly located in the US and Europe (see map in SI). In these regions, which face peaks of higher scarcity during specific periods in the year, a weighted average of monthly scarcity is more representative to assess the impacts associated with constant year-round withdrawals than an annually calculated value.

Water source

Higher level of detail: Surface and ground water sources

Specifying surface and ground water sources in the assessment scarcity indicators leads to MDCs of 0.18 and 0.27 and RCCs of 93% and 87% for surface and ground water, respectively, when compared with a general scarcity indicator based on overall water use and availability. In over 80 % of cases, the resulting scarcity values are unchanged (see Fig.5-6). In approximately 11% of cases, scarcity indicators specific to surface water are higher; in 7% of cases, scarcity specific to ground water is higher.

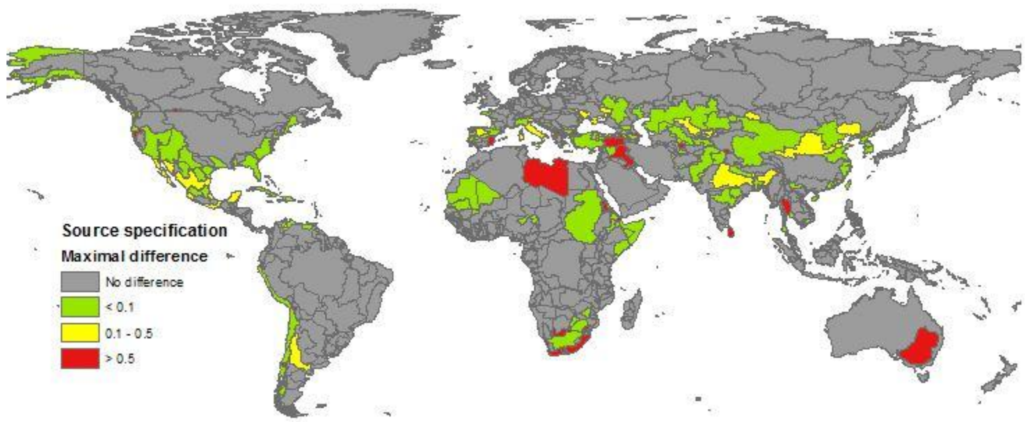

Figure 5-6 Maximal absolute difference between resulting scarcity indicators specifying surface and ground water vs. a generic scarcity indicator considering overall water use and availability. Results are obtained using M-Boulay-Sc, which values range between 0 – 1.

Lower level of detail: Unspecified source

Scarcity indicators based on overall aggregated water use and availability versus indicators aggregated from surface and ground water scarcity results based on the intensity of water withdrawal are generally highly correlated (RCC = 95%) with a relatively low MDC (0.23). Exceptions are mainly located in the US, Central Asia, southeast Australia and certain coastal regions (see map in SI).

Regional resolution scale

Higher level of detail: Country, watershed and sub-watershed scales

A higher spatial resolution than the country scale results in an MDC of 0.53 and an RCC of 81% when compared to the watershed scale. The difference increases when the values are compared to the sub-watershed scale: MDC of 1.46 and RCC of 48%. Fig.5-7 shows where the most significant differences lie.

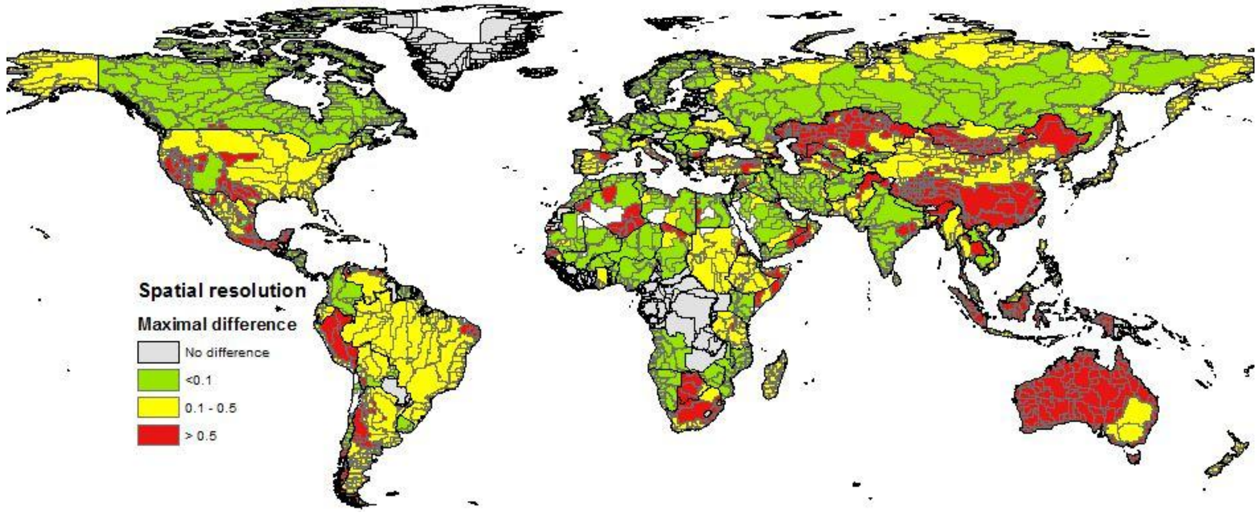

Figure 5-7 Maximal absolute difference between different spatial resolution choices: country scale (aggregated from sub-watershed), watershed scale (aggregated from sub-watershed) or sub-watershed scale scarcity. Results are obtained using M-Boulay-Sc, which values range between 0 – 1.

Lower level of detail: Country scale

Different aggregating choices to obtain country-scale scarcity values result in a moderate difference (MDC and RCC of 0.48 and 90%, respectively) when comparing countrywide values for water use and availability data versus a watershed-based scarcity aggregation. The difference increases (0.88 and 82% for MDC and RCC, respectively) when the countrywide model is compared to a sub-watershed scarcity aggregation. Fig.5-8 illustrates the greatest variation incurred from such modeling choices on the resulting country level scarcity indicator. The values are available in SI.

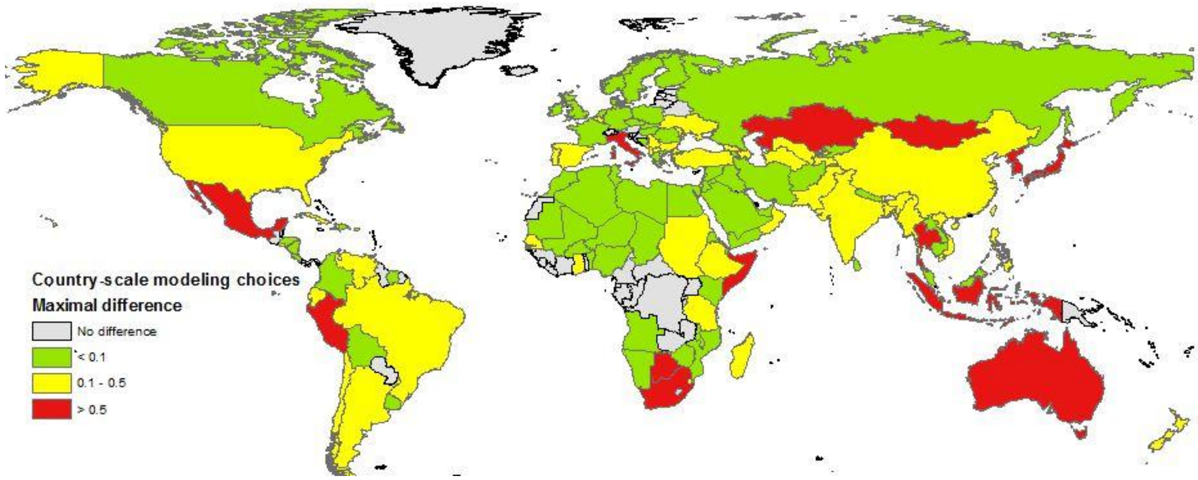

Figure 5-8 Maximal difference for different choices for country-scale scarcity modeling: using direct country data, aggregating scarcity from watershed or aggregating scarcity from sub-watershed, using M-Boulay-Sc (result from 0 to 1).

Quality aspect

Higher level of detail: Water quality specification

Model results accounting for water quality (M-BoulayAv) are not correlated with results that exclusively address water quantity (simplified M-BoulaySc). At midpoint, the MDC ranges between 0.74 and 1.56, and the RCC ranges between 31 and 91%. At the endpoint, the MDC ranges between 0.79 and 1.24 and the RCC between 43 and 59%. A detailed description of the differences is presented in SI. The results reveal significant country-specific variations. The variations in results between countries, and the map published in Boulay et al. (Boulay et al. 2011a) can help in identifying specific case for each region.

At midpoint, representing results based on scarcity or availability can greatly influence the conclusions of a study. The choice should therefore be made based on the question to be answered. If only physical scarcity is to be addressed or if no pollution occurs, then a scarcity indicator is appropriate. To assess the availability of the water resource for other users—ecosystems or human users (as described above)— availability is a more appropriate indicator. It has been argued that including quality could lead to double counting when used in parallel with specific water pollution indicators (Berger and Finkbeiner 2012), but, in reality, this is rarely the case. The contribution to the potential impacts of a specific contaminant must be considered in both: the loss of water functionality (Boulay et al. 2011a) and in human toxicity models (Rosenbaum et al. 2008). Moreover, the threshold for functionality must be exceeded for drinking water, in which case one could argue that the ingestion route of exposure may not occur, and the human toxicity impacts of drinking may lead to double counting. However, the pathway leading to the human health impacts from water deprivation is associated with hygiene and biological contamination and less so with toxicity, though some cases may fall in an ambiguous zone. Using the marginal version of the model helps to avoid potential double counting.

5.3.2.2 Scarcity

CTA vs. WTA

WTA and CTA results are generally consistent (RCC = 96%). Correlating the data from WaterGap shows that, on average, 30% of the water withdrawn in the world is consumed. Fig.5-9 shows the difference in results using M-BoulaySc. The most important variations are observed in agricultural-intensive regions, where a large fraction of water withdrawn is consumed, and in regions with significant water-cooling needs, where most withdrawn water is not consumed. Worldwide, the difference in scarcity results in MDC and RCC values of 0.24 and 94%, respectively.

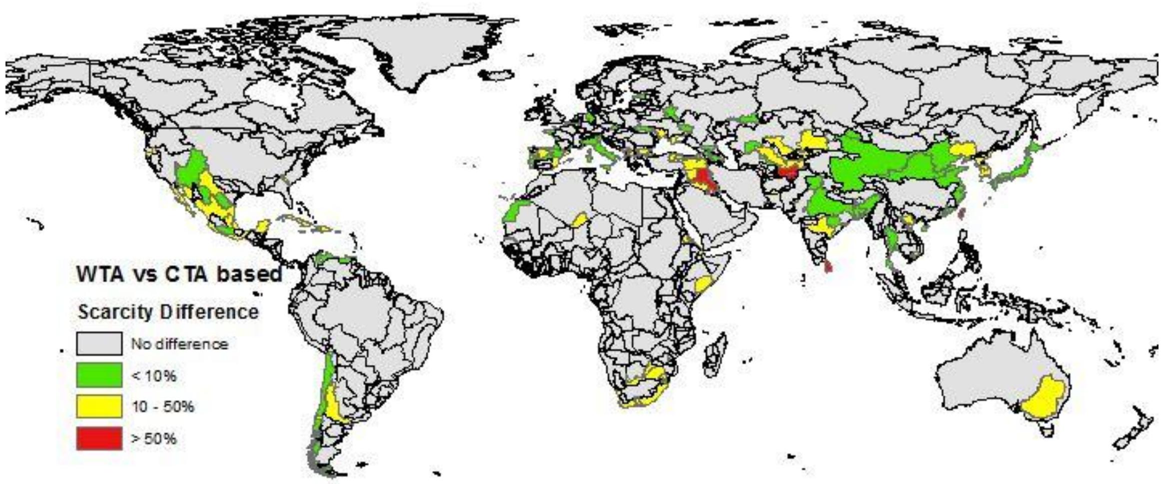

Figure 5-9 Comparison of CTA vs WTA- based scarcity (using M-BoulaySc, values ranging from 0 to 1)

Scarcity model algorithm

Modeling the scarcity index with an S-curve or a straight line yields a relatively small difference in scarcity results: MDC = 0.19 and 100% consistency, as illustrated in SI. The difference increases when an upper threshold of scarcity equal to 1 is excluded: MDC ranging between 1.70 and 1.92 with the higher value corresponding to the use of a power function. The consistency (RCC) is strictly related to the inclusion or exclusion of a threshold, which will make the rankings of low-scarcity regions (and high-scarcity regions) equal and less correlated with direct CTA. Adopting an S-curve or a straight line is therefore less important than defining scarcity with (or without) thresholds.

Data source

The underlying data used to calculate CTA (WaterGap or as used by the Water Footprint network ((Fekete et al. 2002; Mekonnen and Hoekstra 2011))) is first compared with CTA results (RCC 91% and MDC 0.90). Calculating the scarcity indicators with M-BoulaySc using consumption and availability data from one or the other source results in an MDC of 0.37 and decreases consistency to RCC = 75%. Fig.5-10 shows the regions for which the data source may lead to significant changes in water scarcity results. No difference is observed in most of the world, especially in regions where M-BoulaySc gives a value of 0 (no scarcity). The effect on consistency is again associated with the threshold effect of modeled scarcity versus the original CTA comparison.

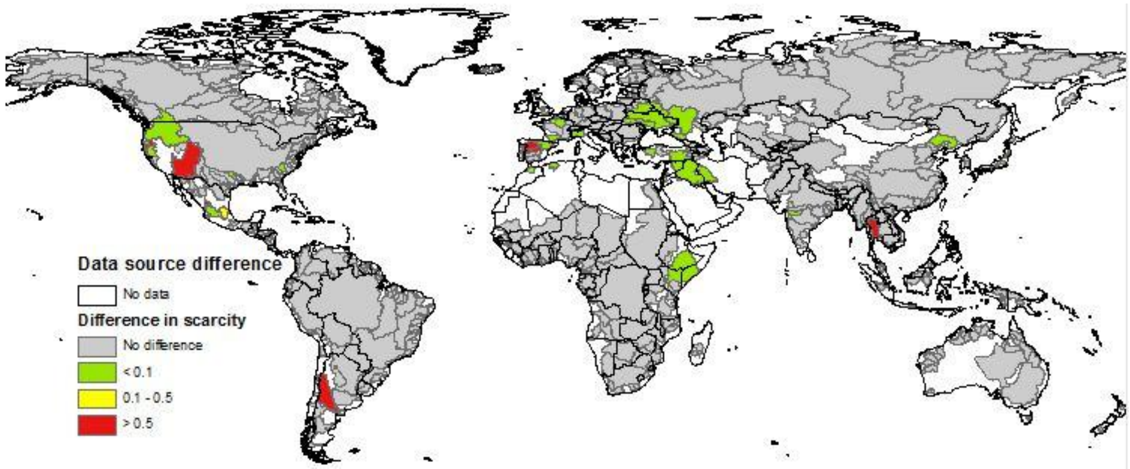

Figure 5-10 Absolute difference in scarcity indicators using model input data on water use and availability from WaterGap or as used by the WFN. Results are obtained using M-BoulaySc, which values range from 0 to 1.

Scarcity overview

The choice of model can have a significant impact on the scarcity results, since they differ in terms of consistency of response and absolute value. Among the most influential modeling choices, the scale at which the modeling data are used to calculate the index leads to important differences between sub-watershed and country scales. Maps of Fig.5-7 identify regions in which collecting regionalized data at the sub-watershed level, rather than the country level, is relevant. While spatial resolution is an influential aspect, the question of the optimal scale remains. Variations in terms of water use and availability may be observed at a very small scale—perhaps a neighbor has a pond and not the other—but scarcity does not need to be defined at such a local level. Different scales may be relevant depending on the type of impact and region. Since scarcity is only associated with the modeling of human health impacts (Fig.5-1), the scale at which human society can still use water with no further adaptation to water scarcity is the most relevant and may range from a few kilometers that populations must walk in developing countries to larger areas that already get water from a mountain hundreds of kilometers away through pipelines, for example. Determining a scarcity index with no socio-economic context, although practical, may therefore have little relevance as a midpoint for assessment on human health. A region-specific optimal scale must still be determined, and inventory efforts must then be adapted to the scale, since variations are important even when using withdrawal-weighted averages.

In addition, the relationship that describes scarcity as a function of CTA (or WTA), whether it is a curve with thresholds, a direct function, an exponential one, etc., was also shown to influence the results. The key issue is therefore how to scarcity and this is reflected in two choices: the choice of curve (direct, exponential or logistic) and the use of thresholds. On this later, both withdrawal-based methods and indexes (M-PfisterSc and M-SwissSc) use the OECD thresholds at which a region faces moderate or severe water stress when respectively 20 or 40% of the resource is withdrawn. While these thresholds are not defined based on scientific data, they at least provided a commonly agreed upon reference, which does not exist for consumption-based scarcity. This issue must be addressed in future research work since scarcity is caused by water consumption and not simply withdrawal. Regarding the choice of curve, logistic and exponential curves correspond to opposite views in the assessment of regions with a high fraction of water use (a logistic curve results in smaller differences, whereas an exponential curve increases the difference). At this point, no robust data exists upon which to base this choice; hence the direct curve represents the intermediate choice with the least potential of error.

The source of the data are not important for most of the world when using M-BoulaySc, although data from Water Gap or the figures used by the WFN will lead to different results for certain parts of the world (northwest America, Spain, Eastern Europe, Middle East, and other isolated watersheds). The type of model and data reference year may be possible sources of discrepancy. The WaterGap water use data are for year 2000 and the water availability data are for 1961-1990. The WFN data averages the 1996-2005 time period. Since the hydrological models were developed over a decade ago, there may be a need for improvements and updates.

Finally, the monthly resolution scale, the differentiation between withdrawn surface water versus withdrawn ground water and the use of a WTA- or CTA-based indicator made less of a difference at a global level, with, however, a few important exceptions in specific regions. Moreover, it is uncertain whether surface and ground water scarcity are meaningful midpoints. While they lead to different potential human health impacts, greater ground water scarcity does not necessarily lead to more significant impacts and perhaps this distinction is only necessary when modeling endpoint damages, where impacts associated with a specific type of water can be assessed. Hence, this type of differentiation may be useful depending on the objective of the study and only for the regions highlighted in Fig.5-6. Groundwater data of a satisfying quality is still not available and must be further developed from hydrological models.

5.3.2.3 Affected users

Aquaculture

Though fisheries are important water users in certain parts of the world, the proportion of water used for this purpose in comparison to agriculture or domestic use is generally small (Boulay et al. 2011b). Consequently, including or excluding the impact pathway does not affect ranking and leads to an MDC of 0.0004, with the largest difference (absolute) seen for Egypt and China at 1.3 $x10^{-6}$ DALY/m^3.

Domestic

Comparing both hypotheses proposed in E-Boulay (marginal and distribution approaches) leads to an RCC of 83% and an MDC of 0.75. The difference stems from attributing 100% of the water deprivation to agriculture or using the fraction of water used by each user (i.e. including domestic users). The greater impacts of depriving domestic users result in a significant difference for all low- and middle-income countries with water scarcity (see map in SI). This is because even though domestic users represent a generally smaller fraction of users than agricultural (10-20% of total use), the effect factors for domestic deprivation is higher than agricultural deprivation (Boulay et al. 2011b).

Data source

The world average fraction of water used for agriculture across watersheds differs according to the data source: 46% with WaterGap (used in E-Boulay), 61% with Aquastat (used in E-Motoshita), and 65% with Vörösmarthy et al.(2000a) (used in E-Pfister). Weighted averages using water withdrawal from WaterGap as a weighting factor yields 74%, 72% and 77%, respectively. Calculating the same results with E-Boulay_agri in DALY from agricultural water deprivation with these three different data sources for distribution of affected users shows a change in RCC from 87% to 93% and in MDC from 0.30 to 0.60. Aquastat and WaterGap are the best correlated with the smallest difference. Although the resulting difference is not as significant as the choice of model, for example, it is a source of discrepancy between models that may be easily harmonized by selecting the most robust data source. The difference may be observed in the relative difference between the SEE and CF of E-Pfister as compared to E-Motoshita_agri. Since only the distribution of affected users and scarcity differ between the SEE and CF and since they both use the same scarcity indicator, the difference in relative magnitude may be attributed to the user's fraction of water use (see SI).

5.3.2.4 Socio-economic

The Motoshita_agri model differs significantly when considering (or not) the trade effect (RCC of 76% and MDC of 1.31). When comparing Motoshita_noTE (instead of the original model) with E-Boulay_agri and E-Pfister, the correlation increases from 65 to 75% (as compared to 49 to 56% with the original model in section 3.1.4), thus demonstrating the significance of the trade effect.

5.3.3 Overview of human health impacts

When modeling the human health impacts of water use, all three models agree that scarcity should be considered, followed by a parameter that describes the extent to which each user is affected (DAU) and an assessment of the socio-economic situation and, finally, an effect factor that quantifies the health impacts in DALY for each m^3 for which a specific user is deprived. Differences arise out of the choice of scarcity indicator but also out of the choice of users affected by water deprivation. Considering the effect on domestic users impacts the results and, although there is no consensus on whether they actually are affected or not, efforts towards a consensual model should consider this as a sensitive choice. Aquaculture/fisheries are only considered in E-Boulay and, although it is conceptually relevant to include it, it was shown to be insignificant for most of the world. Also, the data source used to assess the fraction of water attributed to a specific user led to relatively small variations, but a consistant, updated and reliable source should be used.

The trade effect factor introduced in E-Motoshita_agri had an important effect on the results, and, although still under development, the results indicate that further research into trade effects modeling is appropriate since it constitutes an additional modeling step that is not yet included in other models. Excluding such effect could significantly change the conclusions of an assessment and underestimate water use impacts in richer countries. This is in agreement with the discussion in Boulay et al. (Boulay et al. 2011b) on the indirect impacts, and, ultimately, the two concepts should be combined: agricultural water deprivation in a rich country either leads to an increase in imports and associated indirect impacts or to a reduction in exports with malnutrition-related human health consequences in developing importing countries. Whether this should be included in the characterization factor or modeled separately as a model boundary extension should be agreed upon.

Although the effect factor from E-Motoshita_agri could not be directly compared, the value in DALY per m^3 deprived for agriculture obtained by E-Pfister and E-Boulay were compared, and the value of Boulay is more relevant to describe the physical connection between water, food deficit and health impacts.

Lastly, it is important to keep in mind that even though modeling choices are compared and general trends are uncovered, it does not certify that the damages that are modeled actually occur in the predicted way. Health damages are extremely hard to predict and the relation between water consumption, scarcity, and impacts is still at this point based on logical argumentation, and not a verified mechanism.

5.3.4 Uncertainty

The uncertainty associated with the choice of model is shown in Fig.5-11 as the maximum difference between model results (max-min) for scarcity and human health deprivation for domestic and agricultural users. The numerical values of the confidence intervals for each model are provided in SI. While uncertainty may be high in certain regions, this is not the case everywhere.

The average values were shown along with the original models in figures S2, 3 and 4, respectively, and the uncertainty data for each method are provided in SI. Although average values do not have any specific physical meaning, they are useful to carry out a sensitivity analysis on model choice. Uncertainty related to input data for WTA and socio-economic data has not been specifically addressed since it was quantified for the case of M-Pfister and E-Pfister

(Pfister and Hellweg 2011). However the uncertainty may be combined for a complete uncertainty assessment.

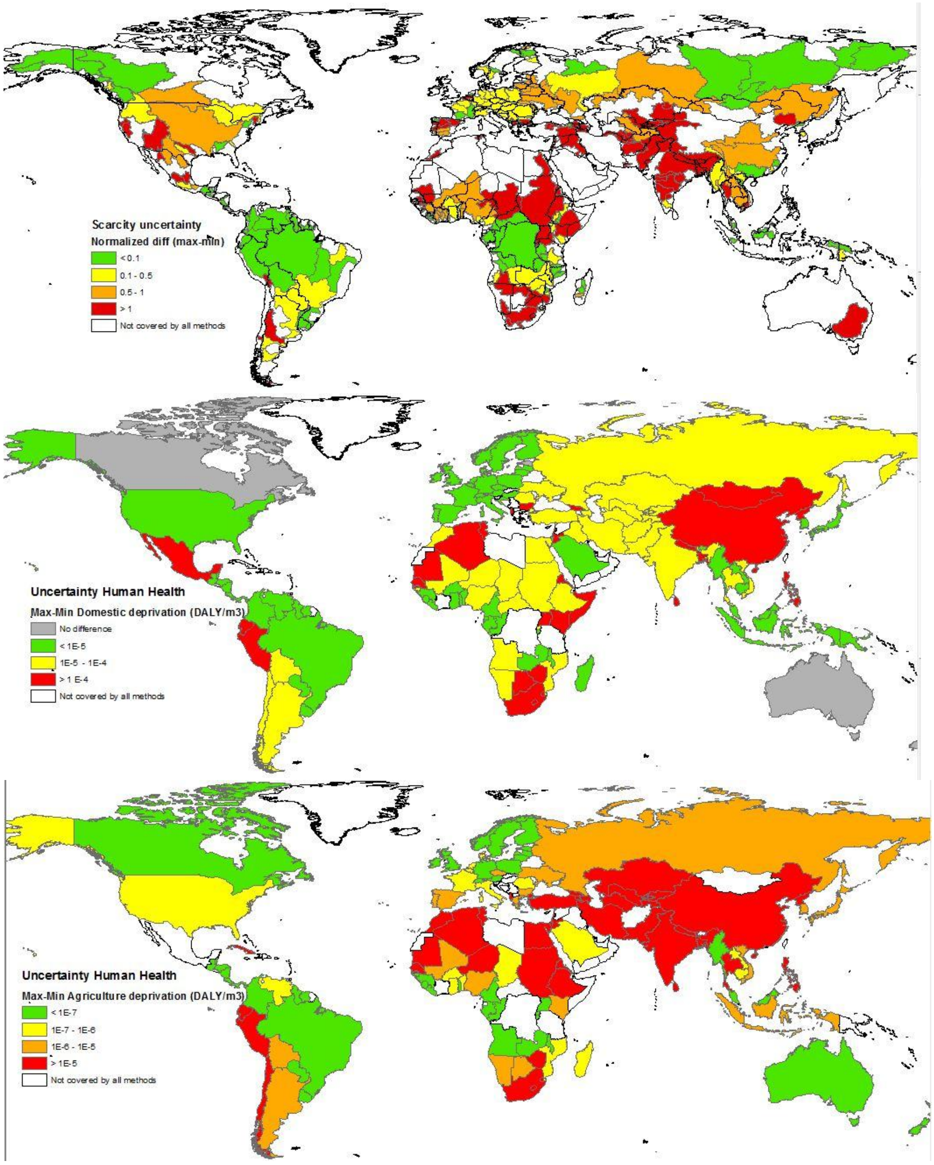

Figure 5-11 Uncertainty associated with the choice of model for a) scarcity, b) domestic water deprivation and c) agricultural water deprivation.

5.4 Conclusion

Since several methods characterize the same impact pathways, it is not clear which method to use or the consequences of the choice of method. This paper provides such insight and sufficient practical geo-referenced information to guide the identification of regions in which different models and underlying modeling choices yield diverging results. Moreover, deconstructing the existing models and highlighting their differences and similarities has helped to determine building blocks to support the development of a consensual method. Until such a method is developed, the uncertainty related to model choice in each method as well as the average values at midpoint and endpoint can help enrich the results of one of the methods compared in this paper. In a related paper (*Water impact assessment methods analysis (Part B): Applicability for water footprinting and decision making*, by the same authors), the insights outlined in this paper were applied to a case study on laundry detergent. An assessment of the applicability of the different models and the related uncertainty was also carried out.

CHAPITRE 6 ARTICLE 4 : ANALYSIS OF WATER USE IMPACT ASSESSMENT METHODS (PART B): APPLICABILITY FOR WATER FOOTPRINTING AND DECISION MAKING WITH A LAUNDRY CASE STUDY

6.1 Introduction

6.1.1 Background

When global warming started growing as a concern among different groups of society, the term carbon footprint became internationally known, and supported by a methodology endorsed by the scientifically recognized group IPCC, to represent this category of impacts that the world should focus on to reduce, and that diligent companies were willing to tackle, assess, communicate and lower. As water issues are now also gaining attention, expectations are that an equivalent concept, water footprint, should be developed grouping all water-related issues into one single and relevant indicator, and avoid any value judgment. The carbon footprint indicator managed to handle such a challenge relatively well, mainly because the selected impact indicator along the causality chain, (i.e. the increase of radiative forcing), covers the same impact pathway for all concerned emissions. This indicator describing the global warming potential is expressed as the relationship to a reference substance, i.e. in equivalent mass emission of CO_2, with a given time horizon. Moreover this impact category is considered global, i.e. the magnitude of the potential impact is independent from the emission location. Water impacts, however, consider accounting of several impact pathways (ISO 14046 2013; Kounina et al. 2013), many of which are highly sensitive to spatial (i.e. location where the environmental intervention occurs) and temporal factors.

Environmental impacts related to water have been historically addressed in LCA through a set of water pollution impact categories including aquatic acidification, aquatic eutrophication or aquatic ecotoxicity. Although research is still on-going to improve characterization models used for calculating these impact category indicators (2012a), recent developments in LCA have focused on water quantity aspects (Koehler and Aoustin 2008). Methodologies have been developed for assessing the effect of human activities on water availability and deriving impacts on human health and ecosystems (Boulay et al. 2011b; Hanafiah et al. 2011; Pfister et al. 2009; Van Zelm et al. 2008). In addition, The Water Footprint Network (Hoekstra et al. 2012) has developed a water footprint methodology to quantify the total volume of freshwater that is consumed and polluted directly and indirectly by a product or an organization. Finally, requirements and guidelines on how to assess impacts related to water – or "water footprint" will be consolidated into the forthcoming ISO 14046 standard (ISO 14046 2013).

Some of these developments are still recent and a limited amount of case studies have been published so far (Berger et al. 2012, Jeswani and Azapagic 2011). Undertaking the application of these water footprint methodologies is nevertheless essential for companies to assess and understand water related impacts of their products. Kounina et al. (2013) reviewed the scope, strengths and weaknesses of all methods related to water availability and Boulay et al. in the part A of this paper (Boulay et al. 2013) analyzed quantitatively the main differences among those assessing water scarcity, availability and human health impacts, including their hypothesis, behaviors, results and uncertainties. The outstanding questions include: How can these methods now be used to determine the water footprint of products, how sensitive are the results to the main modeling choices and what are the main challenges for applying these methodologies to products?

6.1.2 Objective

The objective of this paper is twofold. First, it aims to illustrate how to apply existing water-related methods within the concept of water footprint, as defined by ISO Draft Standard (ISO 14046 2013), through a case study of a laundry detergent (one wash). The sensitivity of the results are evaluated using the conclusions of part A of this paper (Boulay et al. 2013) as well as through a regional sensitivity analysis. Second, this paper discusses the applicability of the different methodologies and their interpretation within a practical water footprint case study.

6.2 Methodology

6.2.1 Case study: goal & scope

The goal of this study is to provide a comprehensive overview of potential impacts relating to freshwater, associated with one wash using a laundry product. The functional unit (FU) is expressed as “one wash at 40°C at average French conditions using 37g of concentrated laundry liquid detergent”. The study was intended to give the manufacturer insights into the feasibility of existing and developing water footprinting methods to assess home care products with a cradle-to-grave perspective.

The system boundaries include all life cycle stages from cradle-to-grave (Fig.6-1). Ingredients of the laundry are transported from various countries to Spain, where the detergent is manufactured and packed. The laundry product, is then transported to France where it is assumed to be used by the consumer under average conditions (i.e. average French washing machine/load/temperature) as defined by the producer. The washing machine is excluded from the system based on a 2% cut-off criteria (less than 2% of total water consumption volume) and considering the high uncertainty associated with this modeling (see SI). The wash waters from the machine are disposed to the sewerage system, treated and then ultimately discharged into the aquatic environment (e.g. river). Since households are not all connected to the sewerage system, a percentage of the wash waters are assumed to be discharged directly into the environment (see Tab.1).

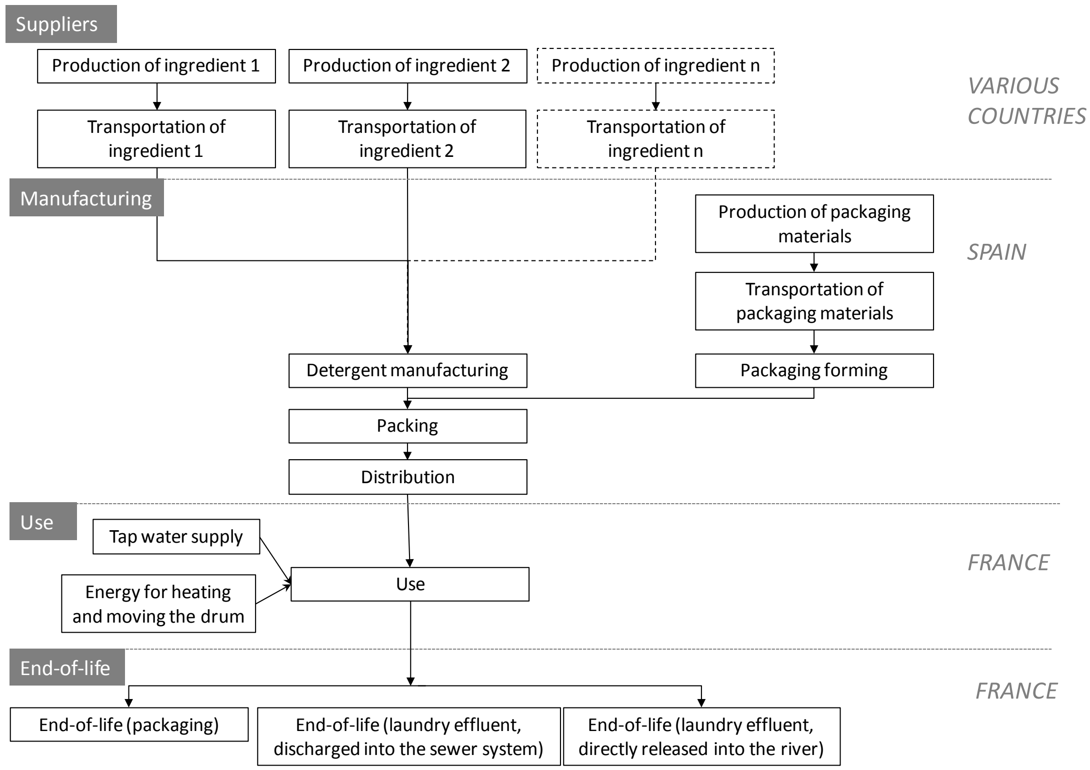

Figure 6-1 Product system studied to provide one washing operation, including spatial information

6.2.2 Water Inventory

Foreground data were provided by the detergent producer. Secondary data were obtained from the Water Database (Quantis 2012a) and the scientific literature. The Water Database builds on existing water data from ecoinvent 2.2 (Frischknecht and Jungbluth 2004) providing a comprehensive water balance for over 4'000 unit processes. Water inputs and outputs are classified by source (e.g. surface water, shallow groundwater, etc.) and use (e.g. agricultural, cooling, etc.) and are regionalized at a country scale. The Water Database fulfills requirements of most available life cycle impact category indicators related to water (Quantis 2012a).

For this project, the water balance (by volume) is calculated for each unit process. The difference between water inputs and water outputs is calculated as the consumptive water use. The difference in water quality between the input water and output water is considered as degradative water use. The consumption of soil moisture, called green water, is reported separately in the inventory, and excluded from the impact assessment, as the inclusion of this inventory flow in a water footprint is still debatable (Kounina et al. 2013; Berger and Finkbeiner 2012). The additional water evaporated from a specific crop could be assessed if the specific water consumed is compared with the water consumption at the natural state in the same region. The main modeling choices, assumptions, and data used are described in Table 6.1.

Table 6.1 Hypothesis and data source for modeling life stages

Life cycle stage	**Parameter(s)**	**Assumption / data**	**Sources**
Ingredients production	Chemicals used for modeling ingredients	No chemical specific data were available for some ingredients (representing 53% of the mass of the finished product) and in these cases proxies have been used for modeling these ingredients. When no relevant proxies were available (3.78% mass), the generic process “chemicals, organic, at plant/GLO” has been used	Wernet et al. (Wernet et al. 2012)
Laundry manufacturing	Treatment of process wastewater	100% treated	Unilever information, the plant has a WWTP
	Evaporation rate during wastewater treatment	10%	Quantis Water Database Technical Report (Quantis 2012b)
	Energy consumption	0.3 GJ/ton Electricity 0.89 GJ/ton Gas	Unilever data
Use	Volume of tap water used in the washing machine	62.4 liters	Unilever internal data
	Electricity consumption by the washing machine (thermal and	2.74 MJ	Unilever internal data

	mechanical)		
	Clothes drying	Clothes are air dried and not ironed	
	Evaporation during clothes drying	0.67kg of water / kg of dry clothes	Milà i Canals 2009(Milà i Canals et al. 2009)
End-of-life	Wastewater treatment plant connectivity	80%	Eurostat (Eurostat 2013)
	Evaporation rate during wastewater treatment	10%	Quantis Water Database Technical Report (Quantis 2012b)
	Loading factor (LF) used to calculate remaining pollutants in the effluent of the wastewater treatment plant – COD, nitrogen and phosphorous	Loading factor provided by ecoinvent have been used: COD: 18% Nitrogen: 33% Phosphorous: 29%	Ecoinvent calculation sheet (Doka 2009)
	Loading factor (LF) used to calculate remaining pollutants in the effluent of the waste water treatment plant – laundry ingredients	From 2 to 40%, depending on substances – as per (Hoof et al. 2011)	European Commission (1999)

6.2.3 Impact assessment

The impact assessment methods for the water footprint were selected to cover impacts associated with changes in water availability and quality, as required by ISO 14046 (ISO 14046 2013). Methods associated with water availability were chosen from Kounina et al. (2013) to cover the midpoint impacts through the scarcity and/or availability indicators and the endpoint impacts through the published methodologies assessing damages on human health and ecosystems. The endpoint category resource depletion was not included as it is not considered mature enough (Kounina et al. 2013). Methods associated with water degradation were chosen in order to cover the most common impact pathways: freshwater acidification, freshwater eutrophication, aquatic ecotoxicity and human toxicity through water exposure, as well as thermal pollution. All methods refer to freshwater, as specified in the goal of the study, and are summarized in Table 6.2. A detailed description of these methods can be found in the literature (Boulay et al. 2013; Kounina et al. 2013; Rosenbaum et al. 2008; Jolliet et al. 2003; Goedkoop et al. 2012).

Table 6.2 Summary of methods included in this water footprint (Methods with an * and corresponding names refer to part A (Boulay et al. 2013), HH: Human health, ES: Ecosystems)

		Indicator	Units	Reference	Details
		Water Availability			
Midpoint	1	Scarcity: M-PfisterSc*	m^3 equivalent	Pfister et al.(2009)	
	1	Scarcity: M-BoulaySc*	m^3 equivalent	Boulay et al.(2011b)	Method adapted from the original publication (M-BoulayAv)
	1	Scarcity: M-SwissSc*	ecopoints	Swiss Eco-Scarcity (Frischknecht et al. 2008)	
	1	Scarcity: M-BWSSc*	m^3 equivalent	Water Footprint Network, Hoekstra et al. (2012)	
	1a	Availability: M-BoulayAv*	m^3 equivalent	Boulay et al.(2011b)	Assessment performed assuming two input water qualities: (1) very good quality water or (2) ambient quality based on available data on world water quality from the GEMStat database(UNEP Global Environment Monitoring System (GEMS) Water

	1a	Availability: M-WIIXAv*	m^3 equivalent	Veolia Impact Index, Bayart el al.(Veolia Water 2010)	Programme 2009)
		Water Degradation			
	2	Eutrophication	Kg P equiv.	ReCiPe (Goedkoop et al. 2012)	
	3	Acidification	Kg equiv. SO2	Impact 2002+ (Jolliet et al. 2003)	
	4	Ecotoxicity	CTUe equivalent	Usetox (Rosenbaum et al. 2008)	Recommended and interim have been considered throughout all processes, and Unilever recalculated specific CFs for ingredients released in water
	5	Human Toxicity	CTUh equivalent	Usetox (Rosenbaum et al. 2008)	USEtox, emissions with fate in water, recommended and interim have been used for ingredients released to water
Endpoint		Water Availability			
	6	HH : E-Pfister*	DALY/m^3	Pfister et al. (Pfister et al. 2009)	Impacts from water deprivation for agricultural users
	6	HH: E-Motoshita*	DALY/m^3	Motoshita et al. (2010a; 2010b)	Impacts from water deprivation for agricultural and domestic users
	6	HH : E_boulay_marg*	DALY/m^3	Boulay et al (2011b)	Impacts from water deprivation for agricultural users and fisheries
	6	HH :E-Boulay_distri*	DALY/m^3	Boulay et al. (2011b)	Impacts from water deprivation for agricultural and domestic users, and fisheries

	7	ES : Terrestrial species deprivation	PDF*m^2*yr	Pfister et al. (2009)	Terrestrial species loss from water use
	8	ES : Aquatic species deprivation	PDF*m^3*yr	Hannafiah et al. (2011)	Aquatic species loss from water use, PDF*m3*yr converted in PDF*m2*yr using a depth of 3 m (Quantis 2012b)
	9	ES : Ground-water table lowering	PDF*m^2*yr	Van Zelm et al. (2008)	Terrestrial species loss from groundwater table lowering, due to water use
		Water Degradation			
	10	ES:Thermal pollution	PDF*m^2*yr	Verones et al. (2011)	Impacts on species from an increased in effluent temperature in PDF·day·m^3/(°C) converted to PDF*m2*yr using a 3 °C temperature raise and an depth of 4.87 m from the publication measurements.
	11	ES: Eutrophication	PDF*m^2*yr	Goedkoop et al. (2008)	Following Recipe conversion from species to PDF (7.89 x10^{-10} Species/m^3 and a depth of 3m)
	12	ES: Acidification	PDF*m^2*yr	Impact 2002+ (Jolliet et al. 2003)	
	13	ES: Ecotoxicity	PDF*m^2*yr	Usetox (Rosenbaum et al. 2008)	PDF*m3*yr converted in PDF*m2*yr using a depth of 3 m (Quantis 2012b)
	14	HH: Human Toxicity	DALY/m^3	Usetox (Rosenbaum et al. 2008)	

Some of these methods cover identical impact pathways, and thus should be interpreted as double counting. However, all methods are presented here for comparison and analysis. At the midpoint (M-), methods are divided in three categories: scarcity (Sc), availability (Av), and water degradation indicators. In this paper, following part A and as a proposal for consensual terminology, scarcity refers to a water pressure based on quantity, and availability refers to an assessment of lower water availability based on water quality degradation and quantity depletion. Scarcity indicators are based on a withdrawal-to-availability ratio (WTA) (Frischknecht et al. 2008; Pfister et al. 2009) or consumption-to-availability ratio (CTA), and are then modeled, following different functions, to result in a scarcity index expressed in m^3 equivalent (deprived) within each method, or ecopoints for Swiss Ecoscarcity. Availability indices (Boulay et al. 2011b; Veolia Water 2010), are based on scarcity and they add a parameter to assess the extent to which degradation contributes to lower availability, resulting also in m^3 equivalent units, though equivalence is not the same between methods (Boulay et al. 2013).

Midpoint indicators related to water pollution are traditionally emission-based impact categories. Freshwater ecotoxicity and human toxicity are addressed using the consensual multimedia and multi-pathways exposure model USEtox (Rosenbaum et al. 2008). Missing CFs were specifically developed for substances released into water at the end-of-life and the results were expressed for only the fraction of the emission in the aquatic compartment. For ecotoxicity, these were previously published (Hoof et al. 2011), whereas they were calculated specifically for this paper for human toxicity when sufficient data were available (73% of the mass or 19 out of 25 substances are characterized). Freshwater acidification is characterized with the IMPACT 2002+ methodology (Jolliet et al. 2003) and expressed as kg SO_2 Eq. Freshwater eutrophication is modeled based on the ReCiPe methodology (Goedkoop et al. 2012) and expressed as kg P Eq.

At the endpoint (E-), water availability impact assessment methods model the impact pathways from user deprivation (agriculture, domestic, and/or fisheries) to human health in DALY from (Boulay et al. 2011b; Motoshita et al. 2010a; Motoshita et al. 2010b; Pfister et al. 2009) and to ecosystem impacts in PDF-m^2-yr to aquatic (Hanafiah et al. 2011) and terrestrial species (Pfister et al. 2009; Van Zelm et al. 2008). The indicators of water degradation contribute to the same categories of impacts as the midpoint, modeled up to the damage level in DALY and PDF-m^2-yr.

An additional pathway assessing impacts from thermal pollution on ecosystems is modeled (Verones et al. 2011).

Results of this water footprint profile therefore consist of 14 indicators, 5 at midpoint (out of the 10 applied) and 9 at endpoint level (out of the 13 applied) as indicated by the numbers in the first column of Table 6.2. Indicators with the same number denote the same impact pathway. For the availability indicators, care should be taken to avoid double counting with scarcity and/or degradation indicators, and this is further discussed in section 6.4.1.3. At the midpoint, the profile is presented in three parts: water availability footprint using scarcity and availability - and water degradation footprint. At the endpoint, results are presented as both a human health (HH) water footprint and an ecosystem (ES) water footprint. This choice reflects the different ways that water footprint results can be presented.

6.2.4 Sensitivity analysis

Part A of this paper (Boulay et al. 2013) identified relevant methodological choices in modeling different indicators addressing selected impact pathways. The sensitivity of most influential choices is analyzed in this case study on the life cycle stages that contribute most to the overall water footprint, i.e. the use phase and end-of-life (section 6.3.2 and 6.3.3). Both are occurring in France. Results from Part A indicate that for France, the following choices are relevant for a sensitivity analysis: inclusion of quality in availability assessment, monthly temporal resolution at the midpoint and inclusion/exclusion of trade effect and of quality aspect in water deprivation for human health at the endpoint. Including or excluding domestic users, is a sensitive choice for all regions which may suffer health impacts from water deprivation and it is analyzed by comparing both versions of the E-Boulay method. All the underlying data needed for these analyses are available in Part A of this paper.

A sensitivity analysis on the regional effect is performed by virtually moving the use and end-of-life stages from France to Spain and India, two countries that present different hydrological and socio-economic conditions. Results for Spain are presented at the midpoint (scarcity) and for India at the endpoint, for water availability impacts only. This selection is made in order to limit redundant results and test the most likely different possibilities, since water degradation is generally assessed independently of location.

Results of the sensitivity analysis are shown alongside the main results. Geographical sensitivity analysis is shown separately after the main results, and additional results are shown in SI for a sensitivity analysis on the consideration of quality and domestic users at the endpoint for the use and end-of-life case in India, using E-Boulay_marg and E-Boulay_distri with and without quality.

6.3 Results

6.3.1 Water Inventory

Figure 6-2 shows the mass balance of water for the assessed product system. Inputs are dominated by the use phase linked to (1) tap water use (53% of the overall water withdrawal) and (2) indirect water withdrawal for power production (41% of the total) needed to heat the water and mechanically operate the washing machine. Evaporative losses from the cooling waters associated with electricity production represent about 60% of the amount directly evaporated from the consumer use phase. Water is mainly released to surface water (99.9%) at the use stage, i.e. cooling water for power production, and at the end-of-life stage as direct water discharge via the sewerage system.

Water used in hydropower production (not represented on the diagram), represents a substantial volume, i.e., 3.03 m^3 /FU but was excluded from the impact assessment with the exception of evaporative losses.

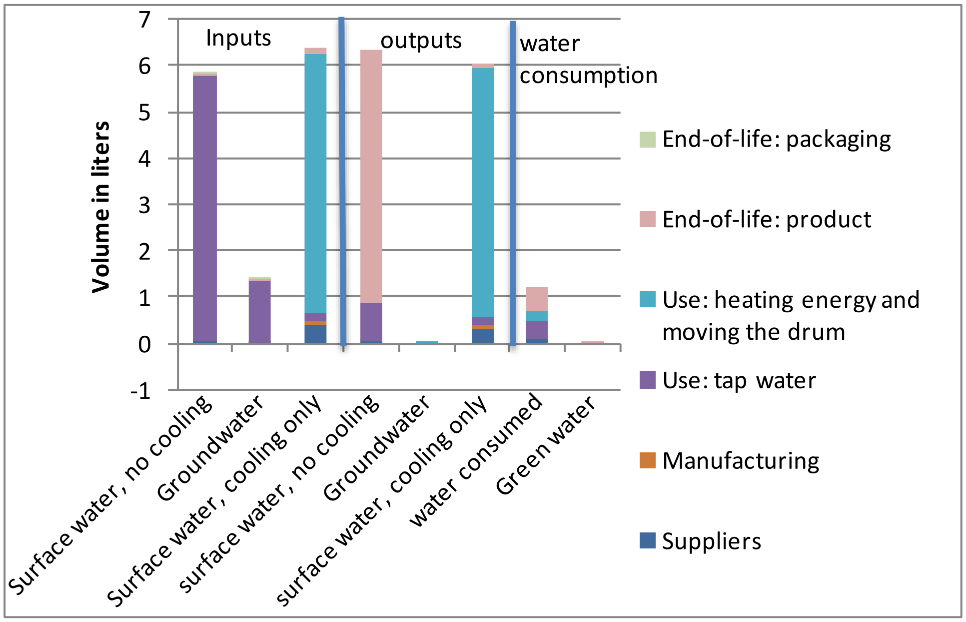

Figure 6-2 Water flow inventory results: input, output and consumed volumes of water for a load of laundry washed in France using a detergent produced in Spain.

6.3.2 Impact assessment results at midpoint

6.3.2.1 Scarcity results

The scarcity indicator results, calculated by the four methods, reveal very similar profiles across the different life cycle stages. Figure 6-3 shows the normalized results in order to bring all units to a common unit m^3 world-equivalent, using each method specific world weighted-average annual scarcity, with withdrawal volumes as a weighting factor. Water scarcity is mainly caused by water consumption during the use phase for cooling purposes in energy production, tap water evaporated (i.e. consumed) when drying the clothes, and water evaporated during the wastewater treatment at the end-of-life. Ingredients and packaging contribute between 10-20% of water scarcity impacts and manufacturing is below 5%. The variability associated with the choice of model can be seen with the difference between the lower result (M-BoulaySc) and the highest (M-PfisterSc), differing by a factor of almost 4.

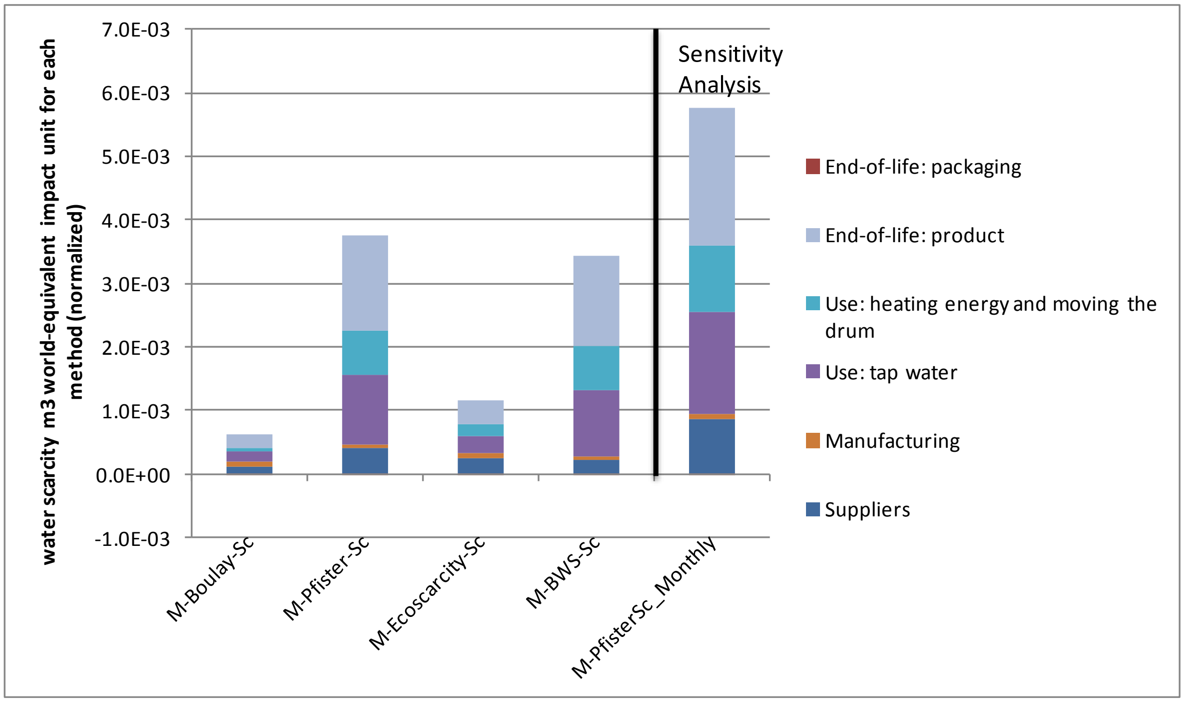

Figure 6-3 Midpoint scarcity indicators (normalized) results for a load of laundry washed in France using a detergent produced in Spain (Table with numerical results in SI).

Sensitivity – temporal variation. Results obtained using a scarcity index resulting from a monthly weighted average approach using M-PfisterSc in part A (Boulay et al. 2013) and based on intensity of withdrawals, are shown on the right hand side of the histogram of Fig.6-3. Since countries like Spain and France have an increased water demand in summer when water is less available, the aggregated scarcity index is higher than when considering all water resources availability and use year-round. This result represents the higher scarcity contribution of washing a load of laundry at times corresponding proportionally to the water use intensity (withdrawals) in the country. While this may be relevant for agricultural water use, it is less representative for domestic activities such as laundry which occur regularly throughout the year.

6.3.2.2 Availability results

Both availability indicators (M-BoulayAv and M-WIIXAv) are dependent on the input water quality. Two scenarios were tested on both methods for use and end-of-life phases. The first one assumes input water of best quality (Figure 6-4, middle section). It is represented by a quality index of 1 and a water input category of 1 by M-WIIXAv and M-BoulayAv, respectively. The second scenario assumes local ambient water quality using default data from the GEMStat database. The quality index of M-WIIXAv gets a score of 0.11 based on the limiting pollutant (Phosphorus). The input water category for M-BoulayAv, assessed based on all available pollutant data, results in category 3 (poor quality), phosphorus and faecal coliforms being the most limiting parameters (Figure 6-4, right section).

Results of M-BoulayAv, obtained applying an online tool to determine water output quality(CIRAIG 2012b), present a negative impact, or credit, associated with tap water discharged at end-of-life stage, corresponding to a reduction of 193% and 87 % of the total impact when assuming either ambient poor water quality or very good quality water input, respectively. The higher benefit in the first case arises from the higher difference between the qualities of discharged water (category 2d according to (Boulay et al. 2011a)) vs. withdrawn water (poor quality, category 3). This change of quality is caused by potabilisation and wastewater treatments, associated with the removal of phosphorus and fecal coliforms. These are present in larger amounts in ambient water in France (Boulay et al. 2011a; UNEP Global Environment Monitoring System (GEMS) Water Programme 2009) than in the effluent from a load of laundry (no mixing with black water was considered). Despite that from a water impacts perspective this might be seen as a benefit, a full LCA might highlight potential burden shifting, e.g. from emissions due to energy requirement due to water treatment processes.

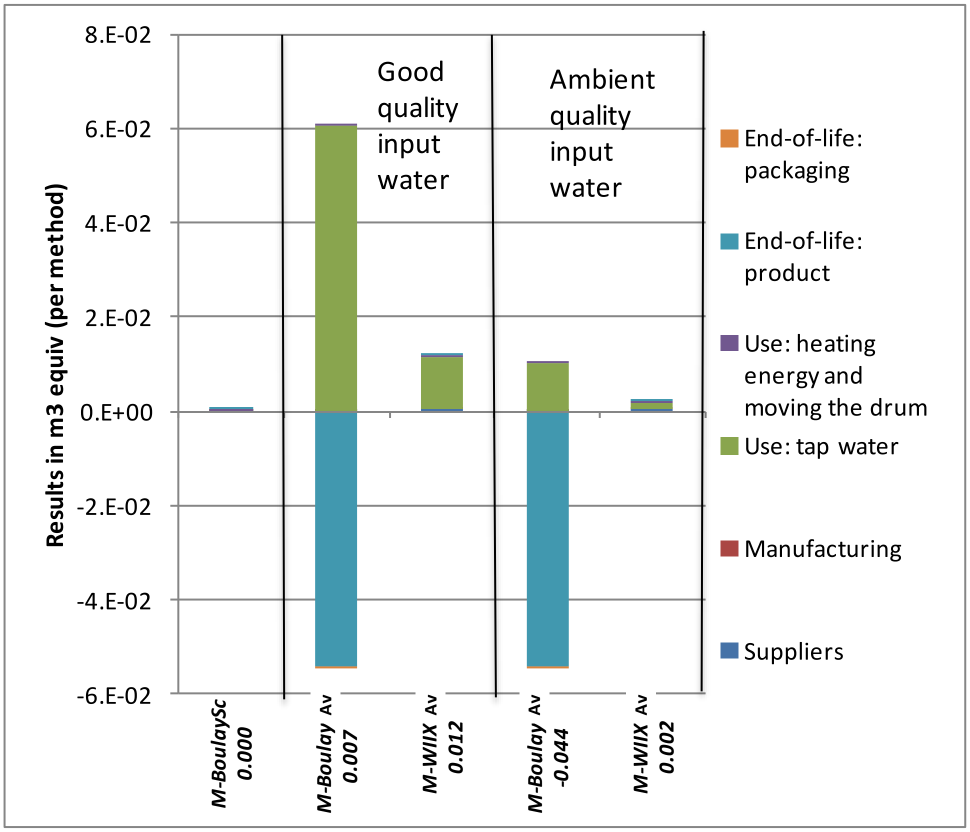

Figure 6-4 Scarcity indicator (left hand-side histogram that doesn't consider water quality) compared with availability indicators of M-BoulayAv and M-WIIXAv both calculated based on two input water quality scenarios: a) good quality water or b) ambient quality as defined by the mean data from GEMStat (UNEP Global Environment Monitoring System (GEMS) Water Programme 2009)

M-WIIXAv follows the same approach as M-BoulayAv method, i.e., characterizing withdrawn and released water volumes accounting for both water quality and quantity. However, it uses different standards to evaluate water quality and does not lead to any credit. The water released, limited by COD contaminant, is of lower quality than the water withdrawn. The magnitude of the results of both methods is heavily influenced by the choice of quality for input water, with a total result going from 0.012 to 0.0022 m^3 equivalent when the quality of input water is reduced from very good quality to available quality. The limiting contaminants used to assess the quality of the input and output in both methods also plays a crucial role. The fundamental difference in method definition is that M-WIIXAv is based on ecosystems water standards while M-BoulayAv is based on human user functionalities. This makes the comparison questionable as explained in Part A (Boulay et al. 2013) and this is further discussed below (section 6.4.1.9), together with the potential for double-counting of effects.

Sensitivity to water quality (scarcity vs availability). Figure 6-4 compares results of the scarcity indicator M-BoulaySc with the availability indicators of M-BoulayAv and M-WIIXAv. Taking into account the change in water quality (availability index) led to differences up to a factor of 45 compared to the scarcity index that only focuses on water quantity. Withdrawing water of poor ambient quality and releasing it at a higher quality generates net environmental benefits. This indicator represents the change in water availability for human users in France. Lowering water quality, or improving it, affects the availability of the resource for specific users and they may need to adapt to a change of water quality available, by additional water treatment or by changing water source for example.

6.3.2.3 Quality indicator results

The water degradation footprint, i.e. the results from the quality indicators, at the midpoint level is shown in Figure 6-5. Emissions into water from the end-of-life of detergent ingredients contribute the most in relative terms to the ecotoxicity impact category. However, results should be interpreted with care as characterization factors were mostly interim, due to the lack of high-quality data (Hoof et al. 2011). Human toxicity, eutrophication and acidification are mainly driven by emissions from electricity production required at the use phase (electricity mix in France) and in the case of air emissions the assessment includes only the fraction of emissions transferred into the freshwater compartment. Supplier activities contribute to acidification (31%) and eutrophication (20%) due to the manufacturing of detergent ingredients and packaging.

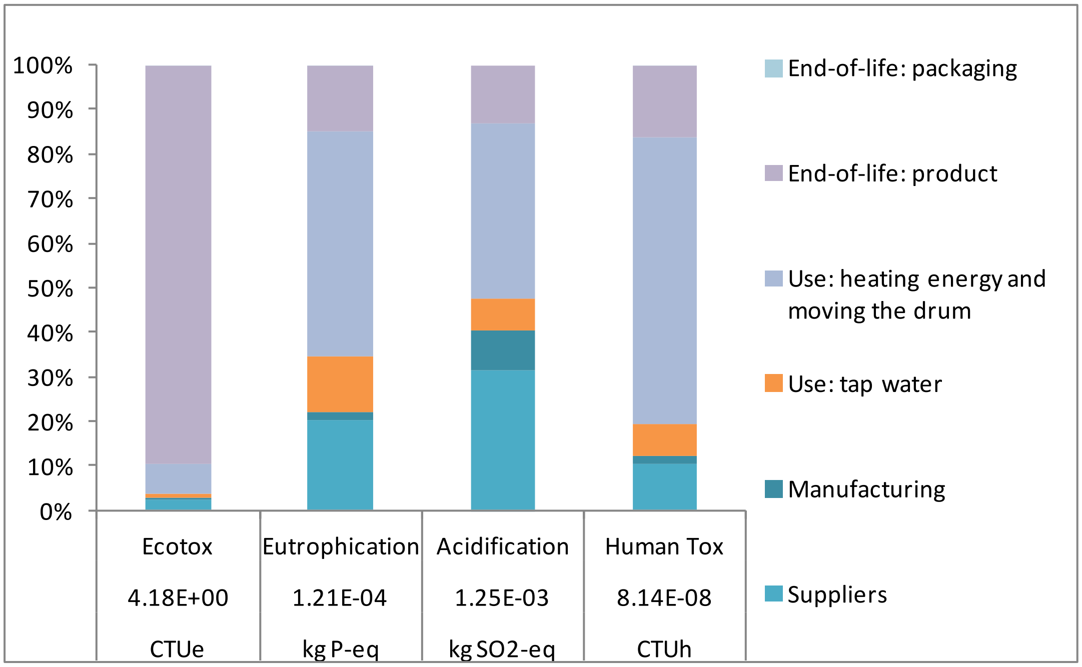

Figure 6-5 Water degradation footprint presented at the midpoint, including ecotoxicity, eutrophication, acidification and human toxicity impact categories.

6.3.3 Impacts assessment results at endpoint

6.3.3.1 Human health water footprint

Human Health water footprint is shown in Figure 6-6 including impacts from water degradation and the change in water availability on human health (numerical results are given in SI). Human toxicity results are 2 to 4 orders of magnitudes higher compared to impacts from water deprivation since they occur in all geographical contexts, whereas those from water deprivation are only generated in regions with a low socio-economic context. Most of the life cycle stages in this case study occur in Europe where water deprivation will typically not cause malnutrition or water-related diseases. Both Boulay methods (E-Boulay_distri and E-Boulay_marg) and E-Pfister method account for impacts from suppliers based in India only because no direct consequences on human health (impacts equal zero) are attributed to suppliers based in developed countries that can adapt to water scarcity. The contribution of the different life-cycle stages to human toxicity is dominated at 53% by the electricity consumption during the use phase and for the treatment process at the end-of-life (23%).

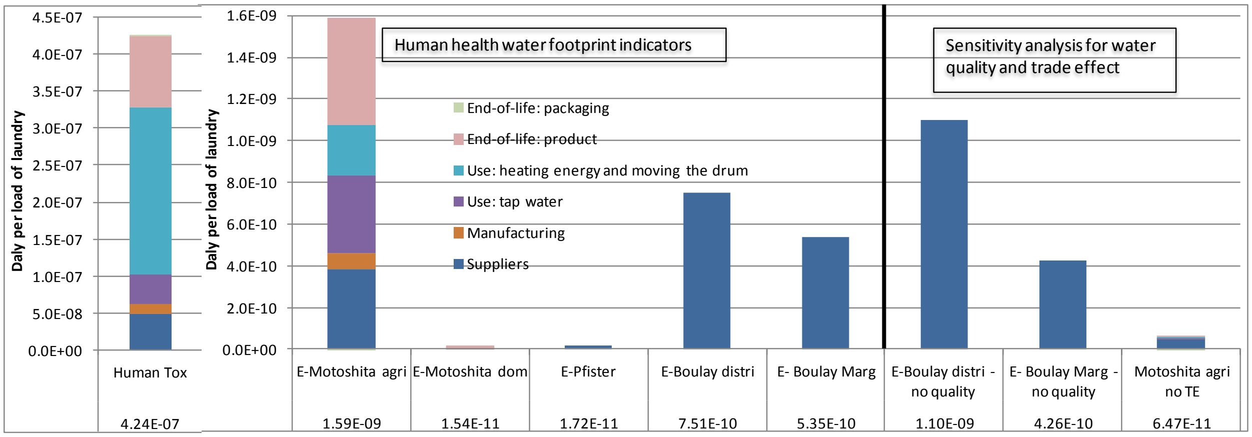

Figure 6-6 Human health water footprint presented at the endpoint (in Dalys) according to different methods and including sensitivity analysis on water quality and trade effect. A different scale is used for Human Tox to represent the higher order of magnitude.

Sensitivity of the trade effect. The E-Motoshita indicator shows human health impacts being generated also outside of India, because it includes a trade effect assessing impacts from a change of trade volumes caused by a decrease in agricultural export due to reduced water availability. In this case study, The Netherlands, France, Spain and Germany have enough economic power to avoid health burden from the decrease of food production and the loss is therefore shared by more economically vulnerable countries importing agricultural goods, namely Bangladesh, Mexico, China, Iran, etc. This effect was discussed in part A as bringing a substantially different contribution to the health impacts characterization factors.

Sensitivity of considering Quality. As for the midpoint assessment the difference in water quality between input and output is considered also at the endpoint (c.f. Figure 6-6). A benefit is gained in India from releasing water at a higher quality than the one withdrawn. However, it does not offset impacts from the consumptive use coming from evaporation during drying of clothes, tap water production and electricity production. In this case study, water of average quality (category S2b) is taken as default ambient water quality based on Gemstat (UNEP Global Environment Monitoring System (GEMS) Water Programme 2009), functional for agriculture and most domestic users. The quality of output water is altered to "average bio" (S2d) due to phosphorus content. It is therefore not anymore functional for most domestic users (Boulay et al. 2011a). In this case study, considering quality leads to a difference of impacts of a factor 1.5 for the distribution model (E-Boulay_distri), and a negligible difference for the marginal one (E-Boulay_marg), since domestic users are not included.

6.3.3.2 Ecosystem water footprint

The Ecosystem water footprint is shown in Figure 6-7 including impact categories for water availability and water degradation. Impacts from water degradation (from ecotoxicity and eutrophication) are dominating for the same reasons as the midpoint (see 3.2.3). They are followed by impacts on terrestrial species from lower water availability (using Pfister et al.(2009)), caused by water consumption from drying, energy production and wastewater treatment. However, uncertainty on this method is not assessed and one may question the higher impacts of water consumption (2 orders of magnitude) on terrestrial species than on aquatic species, hence results on these indicators should be interpreted with care.

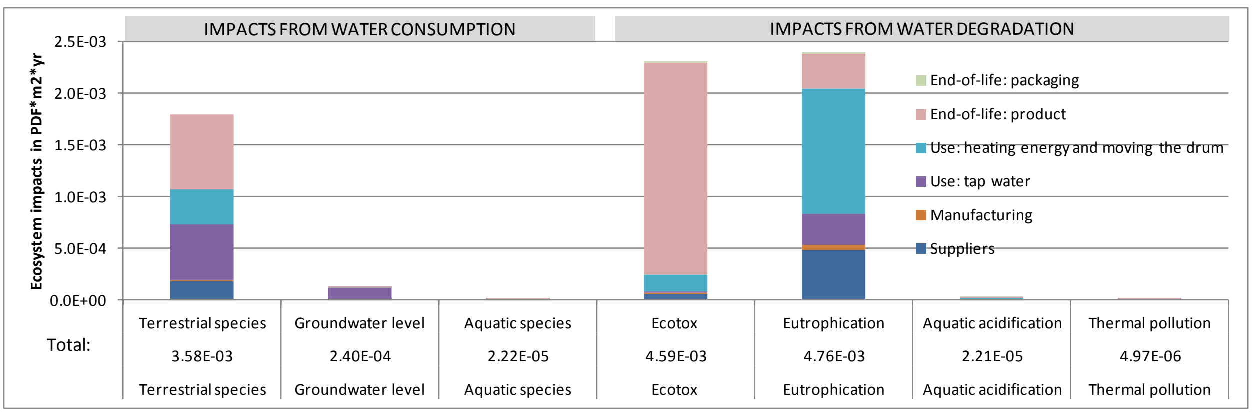

Figure 6-7 Ecosystem water footprint profile at the endpoint including impact categories for water availability and water degradation

6.3.4 Regional Sensitivity analysis

6.3.4.1 Midpoint: Spain

Results for the normalized scarcity obtained when considering the use and end-of-life phases in Spain (Fig.A, shown in SI), are increased by a factor 10 in comparison with the original case study. The uncertainty associated with the choice of model is lower, as methods agree more in that Spain is a water scarce country, whereas France is a country where scarcity is lower and variations among models are larger. However, as shown in Part A, regional variation within

Spain is important and a smaller resolution for the use and end-of-life phase would greatly influence the results.

6.3.4.2 Endpoint Human Health: India

Impacts on human health from water deprivation for an equivalent load of laundry done in India (with French use conditions) are shown in SI (Fig.B). Results from E-Motoshita_dom, E-Pfister and E-Boulay increase by 2, 3 and 4 orders of magnitude respectively, whereas impacts from the E-Motoshita_agri model (with the trade effect) remained in the same order of magnitude, since malnutrition impacts in low income countries from water deprivation for agriculture were already accounted for through the trade effect when the use and end-of-life phases were set in France.

6.4 Discussion and recommandations

This study has applied 10 midpoint and 13 endpoint water footprint methods to evaluate a household product, namely a laundry detergent. In terms of methods applicability, challenges for the practitioner are identified and discussed below.

6.4.1 Scope, inventory and impact method challenges

6.4.1.1 Collecting water inventory data.

To assess a given product life cycle, practitioners need to collect primary data related to foreground processes and identify appropriate background processes to model upstream burdens of the respective supply chains. Background processes are given by life cycle inventory (LCI) databases. Historically, the latter reported volumes of water abstracted from the environment by unit processes providing in some cases information about the type of water resources (e.g. river, lake, etc.), and the type of water use (e.g. cooling) (Frischknecht and Jungbluth 2007). No information about the quantity of water discharged into the environment was given, preventing a consistent calculation of water balance. The quantity of water evaporated, incorporated into products, or transferred to other watersheds or into the sea, also called consumptive water use (Bayart et al. 2010b; Kounina et al. 2013) could not be assessed. Several efforts have been recently invested to overcome these limitations with the publication of LCI databases such as

Water Footprint Network (Water Footprint Network 2011) and Pfister et al (Pfister et al. 2011). The Quantis Water Database (Quantis 2012a) used in this project provides a water balance for each unit process, allowing to determine the quantity and the quality of water withdrawn, discharged and consumed. Although not quantified, uncertainty associated with these inventory data are expected to be large in some cases, as generic hypotheses were often applied for several categories of processes (e.g. evaporation rate in industrial processes, or in hydropower production). Practitioners need to identify and collect primary data for foreground processes and whenever possible also for sensitive generic background processes for a more accurate assessment. This is not an easy task since industrial water flows are not necessarily collected or reported in a usable format. The same volume of water could be used for different purposes, and reused, leading to allocation. For this project, company primary data were used for estimating water consumptive use on manufacturing sites; generic data from the literature were used to estimate input/output flows at the use and end-of-life stage.

Water quality data are also required for some methodologies. M-BoulayAv and M-WIIXAv assess water degradation as the decrease (or increase) of water quality between input and output water flows. To do so, it is required to know the quality of the water abstracted from the environment. This parameter has been historically disregarded in LCA and hence it is not yet conventionally reported in LCA databases, although the Quantis Water Database has recently integrated this information using default quality data as provided by GEMSTat database as proposed by Boulay et al.(2011a). These can be used as default data, keeping in mind that specific data need to be further searched when there is doubt on the quality of input water or replaced when primary data are available. Especially regarding groundwater, qualities can largely vary even within small areas and data gaps are important, therefore default data are highly uncertain.

6.4.1.2 ***Availability of relevant process data.***

As per any LCA, the representativeness of the selected unit process to model the background system is key for the reliability of the assessment. Practitioners often use proxies to fill in data gaps. In this study, we used proxies for modeling some of the detergent ingredients.

6.4.1.3 *Water treatment systems.*

Water is not always directly abstracted from – and released back into - the environment by water users. Industrial effluents are generally discharged to sewer systems, from where they are treated in wastewater treatment plants (WWTP) improving water quality prior to discharge into the environment. WWTP have a direct effect on water pollution indicators and therefore on water footprint results. Moreover, since water is often treated prior to use, an increase in water quality will precede a water use, which influences the assessment of degradative water use. Generic processes from LCI databases can be used to tackle this issue. This certainly provides more accurate results for tap water treatment plant than for WWTP. Indeed, in sewer systems all effluents are mixed prior to treatment. Using generic processes for modeling WWTP such as those proposed in the ecoinvent database is not always relevant since these processes consider an average quality of effluents for modeling water input. This water quality does not necessarily reflect the system under study. The parameterized wastewater treatment tool provided by ecoinvent (Doka 2009) allows the user to define a specific wastewater input at least for a few conventional pollutants like suspended solids, phosphorus, biological oxygen demand, etc., whereas individual chemicals – organic or inorganic - cannot be modeled with the level of detail required in this study. We used this tool to define loading factors, i.e. the fraction of pollutant that is finally discharged into the environment after treatment, for COD, N and P. For micro-pollutants, loading factors have been obtained from the Detergent Ingredient Database from the European Commission (DID List (Detergent Ingredient Database (DID list) 2007)). However, loading factors vary among WWTP technologies. Local data on WWTP efficiency are difficult to access for water footprint practitioners. These parameters have nevertheless a significant impact on water footprint indicator results related to water pollution, as for the assessment of toxicity and ecotoxicity impact categories in a LCA. In the context of this case study, not all water users are connected to sewer systems with a WWTP. 80% of WWTP connectivity has been assumed. For other cases, since WWTP have significant impacts on water pollution indicators, it is essential for the practitioner to know whether the water used is sent to the sewer system or not.

6.4.1.4 *Regionalization*

Potential impacts from water use are highly dependent on the location. Thus, knowing where water has been used and released increases the representativeness of the water footprint results.

Although this information could be easily accessible to the practitioners for foreground processes, it is often not the case for background processes all along the life-cycle. The origin of materials is not always well identified by companies and the most important associated water flows do not necessary take place at the first-tier supplier location. For instance, a company will probably know the location of its chemical supplier, but although the chemical may have been blended and packed on-site, it may have been formulated somewhere else in a region with different water scarcity. Both the scarcity index and the quantity of water used by a given unit process might significantly vary depending on the location. For industrial activities, water efficiency is generally higher in areas facing water stress, since companies have to adapt to local constraints. For agricultural production, the quantity of water required for irrigation varies among locations for the same crop (Pfister et al. 2011). As shown in this paper uncertainty information from spatial variability and model uncertainty gives an indication on the confidence one can have on the results and where to target data collection efforts.

6.4.1.5 ***(Eco)toxicity.***

The USEtox model needs to be used to characterize substances for which no (eco)toxicity characterization factors (CF) exist. This is particularly the case for products like detergents, specialty chemicals and similar (Hoof et al. 2011). Despite the USEtox model being implemented into a freely accessible excel sheet available on-line (Www.usetox.org), a significant effort needs to be invested in collecting substance property data to calculate a new CF. This additional effort and the availability of underlying physico-chemical and (eco)toxicity effect data might be the limiting factor for obtaining robust results, especially when the number of substances to be characterized is large.

6.4.1.6 ***M-WIIXAv, Water Impact Index.***

The M-WIIXAv indicator is based on ambient water quality standards aiming to protect the environment. Different sets of environmental standards are proposed by different public organizations (Canadian Council of Ministers of the Environment 2007; Department of Water Affairs Forestry 2011; European Parliament 2000; Ministry of environmental protection the people’s republic of China 2002; 2000). These differences could have several reasons including differences of water quality requirements due to local conditions or types of resources, but also differences in the process of standard definition, subject to political compromises and priorities. For instance, environmental standards proposed by US EPA do not have the same values as water quality standards set by the European commission. This leads to consistency problems when combining these two sets of standards into the same water footprint study, e.g. for a product system encompassing unit processes located in the US and in Europe. For this study, we used standards recommended by Veolia in the on-line WIIX calculation tool (Growing Blue 2012), corresponding to a combination of European and French ambient water quality standards.

6.4.1.7 *M-BoulayAv categories.*

To apply M-BoulayAv, water flows need to be classified into water categories as proposed by the author (Boulay et al. 2011a). An on-line tool (CIRAIG 2012b) is available to identify the corresponding category of any water flow, and associated characterization factor based on the quality parameters available. Although this tool helps in implementing the methodology, presently it still represents an additional step required to perform a conventional LCA, until it is fully integrated and operational in databases and software. Boulay et al., however, propose a simplified method relying on default water quality input that can be used in combination with a qualitative assessment on the output flows as a preliminary assessment, requiring further data collection or calculation only if needed.

6.4.1.8 *Method availability and coverage.*

M-Boulay-Sc, M-BoulayAv and E-Boulay(all) factors are all available on a Google Earth layer online or for download free of charge (CIRAIG 2012a), similar to M-Pfister and E-Pfister (2009). M-SwissES-Sc are available by contacting the author. All three cover the entire globe and are available at different resolution scales. M-BWS-Sc is available online free of charge for the main watersheds of the world, excluding, however, large regions around the coastal areas. M-WIIXAv can be calculated from M-Pfister-Sc and recommended regulatory references, as discussed above. Lastly, E-Motoshita (all) methods are not directly usable in the publicly available form yet but factors can be obtained from the author and should be published soon. CF for groundwater extraction impacts (Van Zelm et al. 2008) and effects of thermal emissions to water (Verones et al. 2011) are so far only available for specific cases in The Netherlands and Switzerland, respectively, and are not available with global coverage. Applying these CFs to the case study processes (located in different areas) induces additional uncertainty, since they are not meant to be used generally in LCA and no testing of sensitivities to other areas has been done. While also ecotoxicity and eutrophication impacts are derived for models of specific regions, these CF are explicitly extrapolated to serve generic LCA assessments.

6.4.1.9 Availability indicators and double counting within a water footprint.

Midpoint indicators presented in this paper were categorized in three different types: quality indicators, scarcity indicators (water consumption) and availability indicators (water consumption and degradation). Quality indicators can, and should, be used alongside one scarcity indicator for a comprehensive assessment of impacts related to water. Availability indicators, however, also address quality aspect and care should be used when interpreting results. M-WIIXAv index applies environmental ambient water quality standards. Although it provides relevant information when applied as a standalone index, it remains debatable to integrate it into a water footprint profile as there is a clear overlap with other quality indicators such as ecotoxicity, eutrophication, etc. The M-BoulayAv method assesses water scarcity through specific water categories, with the underlying hypothesis that water availability for a given category is reduced for specific users when quality is degraded beyond its threshold level. Since water categories are based on human use standards, one can argue that the indicator captures water scarcity and related impacts from degradation on water availability for human users, while direct impacts from pollution are captured by indicators addressing ecosystem quality, hence avoiding double counting and allowing these indicators to be used in parallel. The exception may be human toxicity in some cases, see discussion in part A. On the other hand, if one agrees that the more polluted water is, the more damaging it is for ecosystems, and that this is correlated with direct impacts from pollution, then this indicator can be used on its own at the midpoint level to represent impacts related to both water consumption and degradation.

6.4.2 Outlook and future developments

6.4.2.1 *Databases and softwares.*

In order to make these methodologies more operational, it is essential to integrate LCIA and water footprint method with LCI databases within a common framework in commercial software.

The Quantis Water Database has been the first effort ensuring this integration beyond a simple multiplication of a physical elementary flow by a characterization factor. It implements methods such as M-BoulayAv and M-WIIXAv. Ecoinvent 3 will partially implement this Database in its framework ensuring the mass balance between water elementary flows and the calculation of water consumption and related impacts indicators. A full integration of the availability assessment methodologies in the commercial LCA softwares still requires the capacity to perform a regionalized assessment of input and output water quality (i.e. the calculation of pollutant concentrations). Veolia has created a footprint tool to facilitate application of the M-WIIX-Sc methodology. However its application is restricted to a water management scope as limited background data are available. For M-BoulayAv, the CIRAIG Water Tool allows calculating the water category classes and characterization factors for foreground water flows. This can be used to overwrite default data provided for each water category classes for the different countries meant to assess background processes. M-BoulayAv methodology has been included into the Quantis water database and integrated in IMPACT World+ (2012b), but it is not yet included in ecoinvent or in LCA software at the time of publication.

6.4.2.2 ***Water Quality data.***

Assessment of water quality of elementary flows is still limited. At this time water quality input can only be assessed using GEMStat database as processed in Boulay et al (2011a) or with a qualitative assessment. While this database is to our knowledge the most complete collection of water quality data worldwide to date, it is still far from providing a comprehensive and detailed data coverage. Moreover several inconsistencies between data provided by the member countries are observed, namely regarding the type of contaminant reported and the frequency of sampling. Improved data on water quality is necessary to properly assess and monitor human influence on water resources. Other sources of data like NEWS database (“Oceanographic Commission UNESCO’s Intergovernmental (IOC)” 2008) on N and P should be investigated, compared and if needed integrated with GEMStat data.

6.4.2.3 ***Water footprint as part of a complete LCA.***

Impacts related to water can be assessed at three different levels. The first one focuses solely on water availability impacts using scarcity indicators or availability indicators. It assesses diminished water availability from quantity and respectively quality aspects of water use. The second level represents a comprehensive assessment of impacts related to water that combines a scarcity indicator assessing water availability with existing water pollution indicators assessing water quality (this can be performed at both midpoint and endpoint). The third level expands the impact profile within the LCA framework. This latter level integrates the water footprint alongside other impacts associated with the product system under the evaluation. Such a fully integrated framework makes it possible to express an LCA profile as the sum of different footprints, making sure to avoid any double counting, as proposed by IMPACT World+ methodology (2012b). This latter was not presented in this case study, but would be necessary if an assessment of environmental performance was desired to ensure the risk of burden shifting was minimized. These different levels can each fulfill different purposes Guidance on each of them is provided in the ISO Draft Standard (ISO 14046 2013).

6.5 Conclusion

This study has shown that water footprinting as proposed in the ISO draft standard can already be applied to industrial products. The inventory data provided by databases such as the Quantis Water Database, along with the emerging impact assessment methods assessed in parts A and B of this article show that it is feasible to get an overall view of the impacts of products on water along the life cycle. The method developments and the data availability are rapidly evolving, but the results obtained with present methods already allow companies to map relevant hotspots along the product value chain around the world. However, the study has shown that at both the inventory and impact assessment levels further work is still required to improve the robustness and the confidence in the results. At the inventory level data gaps on production of chemicals are still common, and at the impact assessment level this article shows how different methods (and different assumptions within the same methods) lead to different results, especially at the endpoint level. This calls for further harmonization, which is desired in a decision-making context.

CHAPITRE 7 RÉSULTATS COMPLÉMENTAIRES

7.1 Correction au modèle

Lors de la publication du modèle en 2011, les données de prélèvements et de consommation ne provenaient pas de la même source. Ainsi, lorsque la corrélation entre les ratios d'eau disponible prélevée (WTA) et consommée (CTA) fut calculée à la section 4.2, dans le but de convertir les seuils définissant la rareté d'eau basés sur le WTA en seuils applicable au CTA, une inconsistance de source de données a pu influencer les résultats. Lorsque les données de prélèvements furent obtenues par WaterGap, les corrélations furent recalculées. Le coefficient de corrélation entre WTA et CTA, montré à la figure S1 de l'Annexe 4, a ainsi passé de 0.63 à 0.93 et les seuils équivalents trouvés en section 4.2, ainsi que l'équation des SI (Annexe 4, équation S1) ont été mis à jours. La table 7.1 et l'équation 7.1 résument les changements. Ces changements ont amené une différence dans les facteurs de rareté (et conséquemment dans les FC sur la santé humaine), en caractérisant davantage de régions sous « stress très élevé » (i.e. rareté $\alpha = 1$), et légèrement moins de régions à faible stress (i.e. rareté $\alpha = 0$).

Table 7.1 Mise à jour des seuils de rareté

Seuils WTA	Qualificatif du stress	Seuils CTA (publiés originalement)	Seuils CTA (mis à jours)
10%	Faible	10%	3%
40%	Élevé	18%	12%
80%	Très élevé	40%	24%

$$\alpha = \frac{1}{\left(1+0.3065e^{-0.568(\alpha*-2.2)}\right)^{1/0.0055}} \qquad \text{Équation 7.1}$$

7.2 Incertitudes et variabilité

Trois types d'incertitudes ont été analysés : l'incertitude associée 1-au choix du modèle, 2- au choix de l'échelle géographique (variabilité spatiale), et 3- aux paramètres du modèle (incertitude intrinsèque). Ces trois aspects sont décris ci-bas.

Les incertitudes associées au choix du modèle sont présentées à la section 5.3.4 et représentent l'incertitude des résultats causée par l'utilisation d'un modèle plutôt qu'un autre. Cette incertitude, parfois large vu les différences entre modèles, pourra éventuellement être réduite par l'harmonisation des hypothèses sous-jacentes des différents modèles à travers le processus de création d'un modèle consensuel. Dans l'intérim, les différents modèles peuvent être utilisés en analyse de sensibilité.

L'incertitude associée au choix de l'échelle géographique et à l'utilisation de facteurs de caractérisation agrégés sur la base d'une moyenne pondérée par les prélèvements est directement liée à la variabilité spatiale (section 5.3.2.1). Bien que l'incertitude associée au choix d'une échelle en particulier ne puisse être réduite, i.e. choisir un facteur pour toute la Chine amènera toujours une grande marge d'incertitude par rapport à une région chinoise donnée, des cartes ont été générées pour présenter visuellement la variation causée par le choix d'une échelle fine (sous-bassin versant) versus l'échelle du pays. Cette information permet d'identifier les régions présentant une grande variabilité et pour lesquelles une échelle plus fine peut faire une grande différence (ex : Australie, Chine, Mexique, Afrique du Sud) et cibler l'effort pour réduire l'incertitude comparativement aux régions pour lesquelles la variabilité spatiale est faible et la différence entre l'échelle des sous-bassins versant ou l'échelle du pays est faible (ex : Canada, France, pays scandinaves, Égypte).

L'incertitude associée aux choix de modélisation des paramètres, i.e. l'incertitude intrinsèque au modèle, a été quantifiée dans le cadre du projet Impact World + (Impact World +, 2012), une méthodologie complète et régionalisée d'évaluation des impacts pour l'ACV développée en partie par le CIRAIG. L'incertitude des paramètres d'entrée du modèle a été quantifiée, soit par les données disponibles décrivant les paramètres entrants du modèle ou, dans l'absence de celles-ci, par un jugement d'expert. Cette incertitude a ensuite été propagée avec une analyse Monte-Carlo. Le tableau 7.2 ci-bas décrit l'incertitude associée à chaque paramètre du modèle (équation 4.2).

Table 7.2 Évaluation de l'incertitude des paramètres du modèle

Paramètre	Distribution	Valeur	Source
Rareté (sort)			
Consommation d'eau (CU)	Log normale	$GSD^2 = 2$	Jugement expert
Eau disponible (Q90 et GWR)	Log normale	$GSD^2 = 1.1$	Jugement expert
Dépendance à l'eau souterraine (Fg) (%)	Uniforme	+/- 10%	Jugement expert
Indice de qualité de l'eau disponible (Pi) (%)	Uniforme	+/- 25%	Jugement expert
Exposition			
Capacité d'adaptation (AC) basée sur le GNI	Pas d'incertitude quantifiée		
Distribution des usagers (U_{ij}) (%)	Uniforme	+/- 10%	Jugement expert
Effet			
Effet de la malnutrition sur la santé	Log normale	5.39	Analyse des données
Effet d'un manque d'eau pour les usages domestiques sur la santé	Log normale	12.9	Analyse des données
Eau requise pour production en aquaculture/pêches (m^3/kcal)	Uniforme	Min : 5.71E-4 Moyenne : 6.17E-3 Max 1.18E-2	Données
Eau requise pour production agricole (m^3/kcal)	Uniforme	Min : 7.14E-04 Moyenne :1.25E-3 Max : 1.79E-3	Données
Kcal agricoles requise par kcal animale produite	Uniforme	min 4 max 11 Moyenne 7.22	Données
Fraction de l'agriculture destinée à la production animale	Uniforme	+/- 10%	Données

Finalement, une collaboration avec un collègue de doctorat, Guillaume Bourgault, a permis de comparer l'incertitude liée aux paramètres du modèle avec la variabilité spatiale, en combinant les deux dans un même indicateur, UII (Uncertainty Increase Indicator)(Bourgault et al. 2013). Lors d'une agrégation à une échelle de résolution moins fine par moyenne pondérée (basée sur les prélèvements d'eau) - notamment l'agrégation de facteurs de caractérisation à l'échelle de bassins versants en un facteur de caractérisation à l'échelle du pays - cet indicateur permet d'évaluer si l'incertitude relative agrégée est plus élevée (UII > 1) ou plus faible (UII < 1) que l'incertitude relative pré-agrégation en comparant le coefficient de variation (CV) du FC agrégé avec le coefficient de variation des FC natifs (équation 7.2).

$$UII = \frac{CV_{agrégé} - CV_{natif\ min}}{CV_{natif\ max} - CV_{natif\ min}} \qquad \text{Equation 7.2}$$

Si le CV du FC agrégé est plus élevé que celui du FC natif le plus incertain, le UII sera supérieur à un, ce qui fournit l'indication que l'échelle d'agrégation (pays) contient une grande variation spatiale comparée à l'incertitude des FC, tel qu'observé sur la Figure 7-1 pour l'eau de catégorie S2a au Guatemala, en Lituanie et en Guinée. Une collecte de données plus approfondie permettrait de réduire l'incertitude. À l'opposé, un CV agrégé qui se trouve entre le CV du FC natif le plus élevé et le plus faible résultera en un UII inférieur à un. C'est le signe que la variabilité spatiale contribue à l'incertitude de façon comparable, ou moindre, à l'incertitude intrinsèque au modèle, tel que pour le Népal par exemple (voir Figure 7-1).

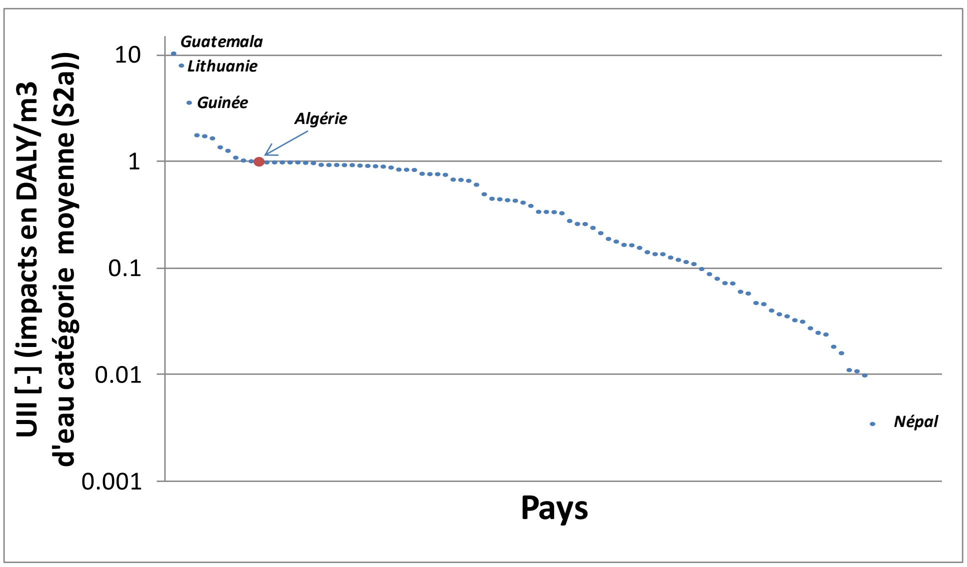

Figure 7-1 Présentation de l'index UII (tiré de Bourgault et al., 2013)

7.3 Évaluation de la charge globale des impacts sur la santé humaine due à l'utilisation d'eau

Le modèle présenté au chapitre 4 caractérise les impacts sur la santé humaine causés par un manque d'eau menant à la malnutrition et/ou à des maladies liées à l'eau. Ce modèle est partiellement évalué d'un point de vue global en comparant :

- ➔ les résultats obtenus avec le modèle pour l'évaluation des *impacts potentiels associés à toute la consommation d'eau annuelle* dans un pays

avec :

- ➔ les *charges globales* par pays exprimés en DALY (années équivalentes de vie perdues) reportées par Organisation Mondiale de la Santé pour chaque chemin d'impact, i.e. *malnutrition et maladies associées à un manque d'eau domestique*.

L'évaluation pour chacun de ces chemins d'impact est expliquée ci-dessous. Le modèle utilisé est une version « simplifiée », qui ne considère pas la dégradation de l'eau mais seulement la consommation, utilisant donc un FC qui caractérise toute l'eau disponible indépendamment de la qualité (i.e. une valeur de 1 est utilisée pour P_i dans les équations 4.3 et 4.4).

7.3.1 Malnutrition

Le modèle présenté au chapitre 4 évalue les impacts potentiels de la malnutrition causée par une consommation/dégradation d'eau et de la réduction de la production agricole qui découle de la baisse de disponibilité d'eau pour l'irrigation. En appliquant le modèle à toute l'eau consommée d'un pays, ou du monde, on obtient l'impact potentiel total de la malnutrition causé par la consommation d'eau tel que prédit par le modèle. L'Organisation Mondiale de la Santé (OMS) reporte des valeurs de dommages sur la santé humaine causée par un déficit calorique lié à un manque de nourriture. Une comparaison directe de cette valeur impliquerait l'hypothèse sous-jacente que tout le manque de nourriture est causé par un manque d'eau pour l'irrigation. Si on assume que d'autres facteurs entrent en jeux (disponibilité des terres, productivité, variation climatiques, etc.), cette validation permet de comparer l'ordre de grandeur des résultats obtenus, et de fixer un seuil de dommages potentiels maximal servant de point de référence. Un résultat inférieur à la donnée de l'OMS est obtenu pour 75% des pays. La charge mondiale calculée par le modèle est cependant supérieure : 4 x 10^7 DALY à la valeur reportée par l'OMS : 7.7 x 10^6 DALY. Près de la moitié de la différence provient de l'Inde. Ce pays représente 27% (valeurs OMS) contre 42% (Prédiction du modèle) des DALY liés à la malnutrition au monde et influence ainsi largement les valeurs mondiales. La figure 7-2 ci-dessous montre la relation entre les valeurs évaluées par le modèle et les valeurs reportées par l'OMS.

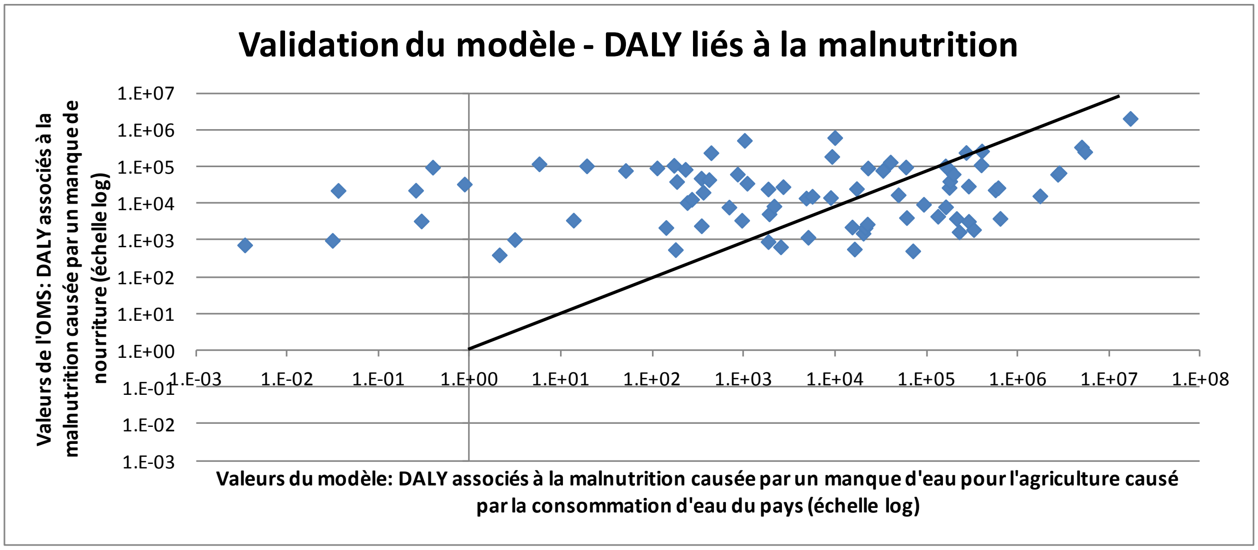

Figure 7-2 DALY associés à la malnutrition causée par un manque d'eau pour l'agriculture causé par la consommation d'eau du pays comparés aux valeurs de l'OMS en DALY associés à la malnutrition causée par un manque de nourriture

7.3.2 Maladies liées à l'eau

La section du modèle qui évalue les impacts d'un manque d'eau pour les usages domestiques quantifie les dommages potentiels causés par une consommation d'eau en privant les usagers domestiques. En conséquence, utiliser le modèle pour caractériser *toute* la consommation d'eau d'un pays évalue les dommages potentiels de cette consommation sur la baisse de disponibilité d'eau pour les usages domestiques et les maladies associées à ce manque d'eau. Si on considère que le manque d'infrastructure et les pratiques d'hygiène contribuent également aux dommages causés par les maladies associées au manque d'eau, les valeurs de l'OMS qui décrivent la charge globale exprimé en DALY associée au manque d'eau devraient être supérieures à celle évaluée par le modèle pour la consommation d'eau. C'est le cas pour 71% des pays. La charge mondiale calculée par le modèle est pratiquement égale à celle rapportée par l'OMS : 7.8 x 10^7 DALY (OMS) versus 9.4 x 10^7 DALY (modèle). La figure 7-3 montre la comparaison par pays.

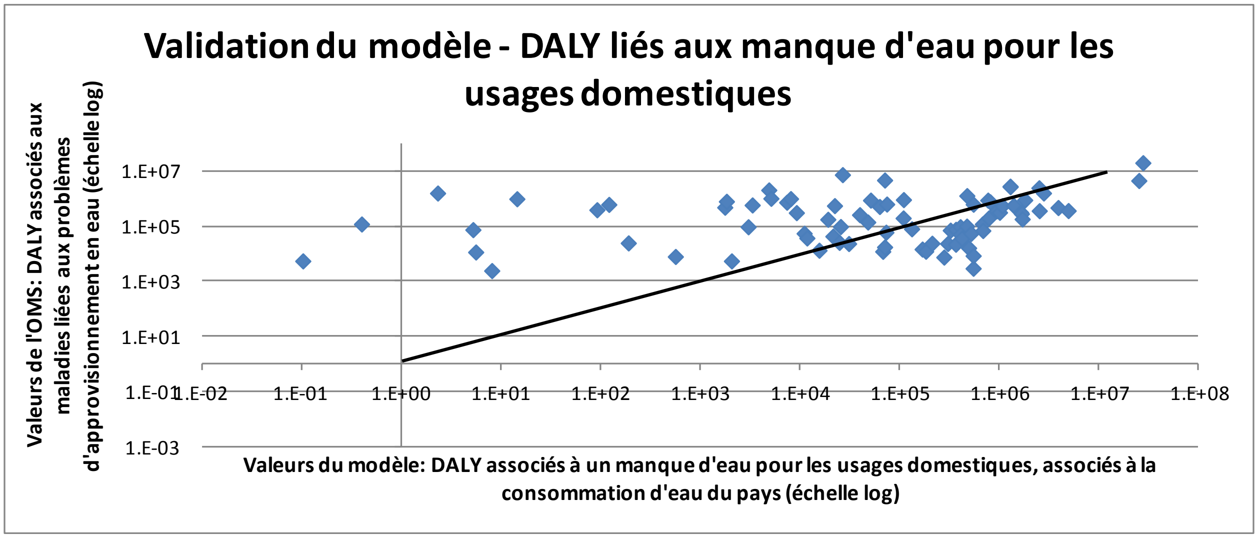

Figure 7-3 DALY associés à un manque d'eau pour les usages domestiques associés à la consommation d'eau d'un pays comparé aux valeurs de l'OMS en DALY associés aux maladies liées aux problèmes d'approvisionnement en eau

Pour les deux figures (7-2 et 7-3) les pays qui se trouvent le plus loin au dessus de la ligne d'équité (le plus à gauche sur le graphique) représentent les pays où d'autres facteurs que la consommation d'eau contribuent significativement aux dommages sur la santé humaine causés par le manque de nourriture ou les problèmes liés à l'eau. Pour la malnutrition, ces pays sont Uruguay, Laos, Honduras, Namibie, Niger, Bénin, Suriname, Costa Rica, Mali, Pologne, Malawi et Népal. Pour les maladies liées à l'eau, ces pays sont : Uruguay, Laos, Niger, Honduras, Namibie, Suriname, Mali, Bénin, Népal, Djibouti et Costa Rica. Ces pays, très similaires pour les deux types d'impacts, sont des pays qui ne présentent pas une rareté d'eau élevée, ce qui explique de faibles impacts prédits par le modèle et confirme l'hypothèse que d'autres facteurs qu'une baisse de disponibilité de l'eau sont à l'origine des dommages sur la santé humaine rapportés par l'OMS. Il faut noter que les pays pour lesquels le modèle prédit des impacts nuls, i.e. les pays qui présentent une capacité d'adaptation maximale, ou qui ne présentent pas de rareté d'eau, n'apparaissent pas sur le graphique qui est en échelle logarithmique. Dans ces pays, le modèle stipule donc que les dommages sur la santé humaine ne sont pas causés par un manque de disponibilité de l'eau pour l'irrigation ou pour la santé humaine.

Les pays qui se trouvent en dessous des axes d'égalité sont les pays pour lesquels le modèle prédit des valeurs supérieures aux valeurs reportées par l'OMS. Pour ces pays, le modèle surestime les impacts de 1-2 ordres de grandeurs. Parmi ceux-ci on retrouve pour les usages domestiques : Bulgarie, Arménie, Albanie, Ukraine, Iran, Tunisie, Chili, Cuba et Liban, et pour la malnutrition : Bulgarie, Liban, Arménie, Kazakstan, Kyrgystan, Azerbajan, Turkmenistan et Uzbekistan. Pour ces pays, on peut émettre l'hypothèse que malgré le faible GNI, considéré dans la capacité d'adaptation, la capacité des usagers domestiques et agricoles à pallier à un manque d'eau par le biais d'adaptation est plus élevée que celle évaluée par le paramètre AC du modèle.

7.4 Outils développés

Dans le cadre de ce projet de thèse, l'opérationnalisation de la méthode s'est effectuée à travers le développement de deux outils : une couche Google Earth et le Water Tool.

7.4.1 Couche Google Earth

L'accessibilité des facteurs de caractérisation est cruciale à l'utilisation de la méthode par le public. Ainsi, deux couches Google Earth ont été créées avec l'aide d'un stagiaire, Assane Gueye et mises en ligne à l'adresse suivante : http://www.ciraig.org/fr/wateruseimpacts.php. Ces couches, une à l'échelle du pays et une à l'échelle plus fine résultant de l'intersection des pays et des bassins versants, présentent les facteurs de caractérisation, tant au *midpoint* qu'au *endpoint*, et pour la version originale du modèle et pour une version simplifiée qui ne considère pas la qualité de l'eau (Pi égal à un dans les équations 4.3 et 4.4). Les couches peuvent être téléchargées ou consultées directement en ligne. La figure 7-4 montre un aperçu de l'outil.

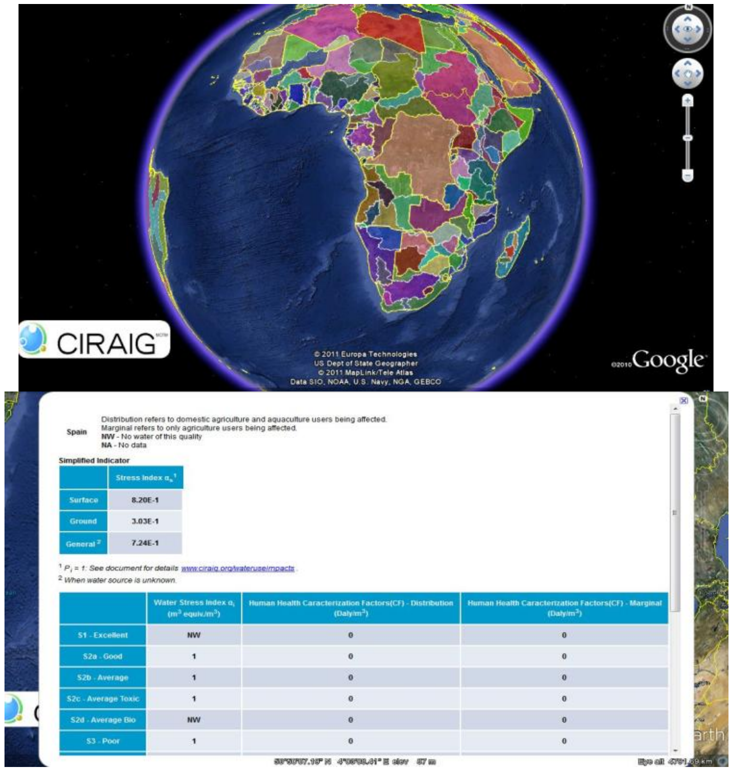

Figure 7-4 Aperçu de la couche Google Earth, disponible en ligne

7.4.2 Water Tool

Un des avantages de la méthode développée dans le contexte de ce projet est la flexibilité qu'elle offre à l'utilisateur. Celui-ci peut utiliser une version simplifiée avec pratiquement aucune donnée nécessaire outre le volume d'eau prélevée, rejetée et le pays, ou il peut pousser l'analyse aussi loin que ses données disponibles le permettent. Le Water Tool a été créé, également avec le support du stagiaire Assane Gueye, afin de supporter un utilisateur qui désire 1) caractériser des données de qualité d'eau pour obtenir le type d'eau correspondant, 2) faire une évaluation complète en utilisant des données d'inventaire et les données par défaut et obtenir les impacts totaux pour un ou plusieurs flux d'inventaire, et 3) remplacer les données de modélisation par défaut par ses propres données pour une évaluation plus spécifique. La figure 7-5 montre un aperçu des différentes fonctions de l'outil. Il est disponible en ligne à l'adresse suivante : http://www.ciraig.org/fr/watertool.php.

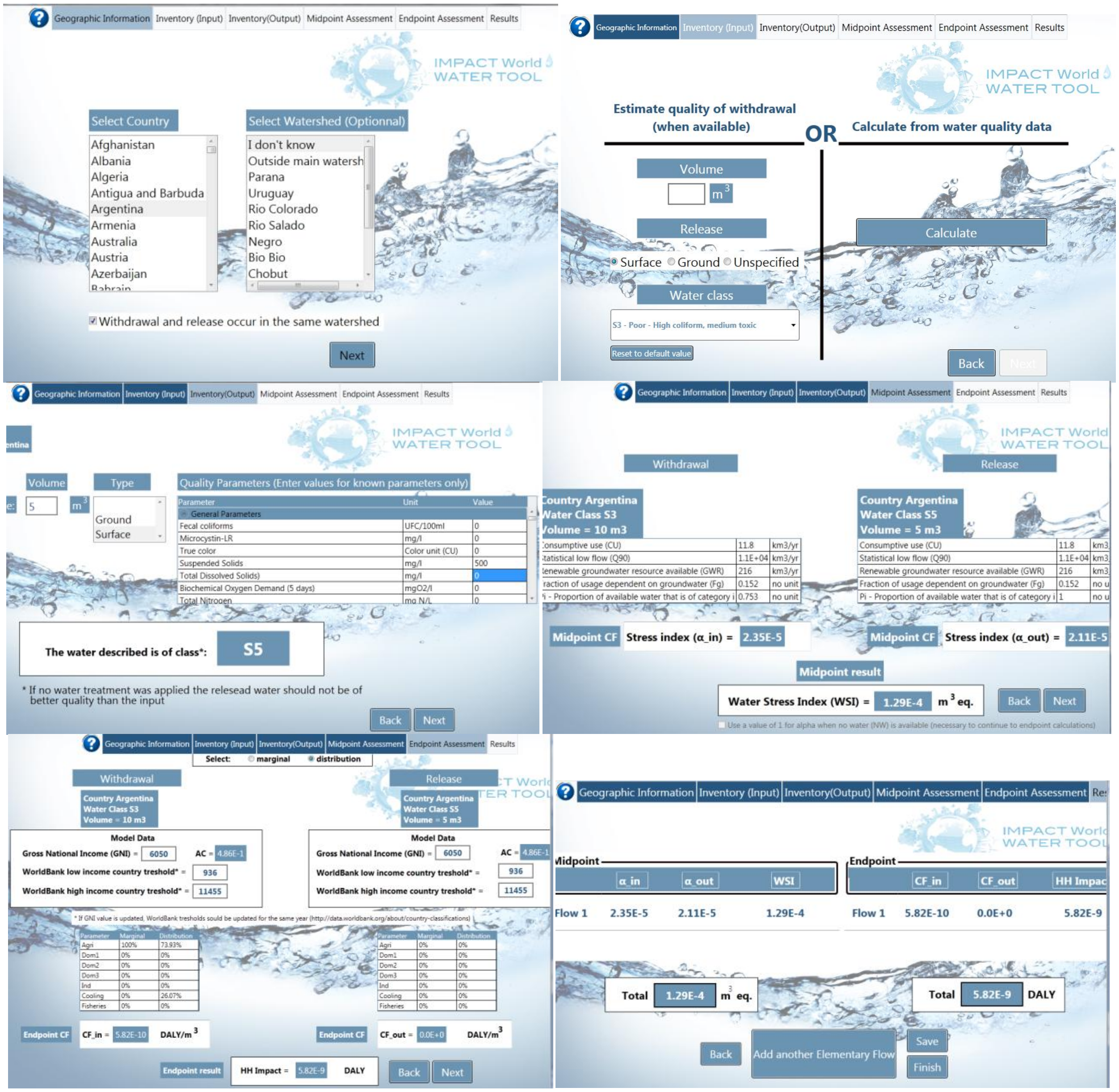

Figure 7-5 Aperçu des fonctions du Water Tool, disponible en ligne.

CHAPITRE 8 DISCUSSION GÉNÉRALE

8.1 Forces

Le modèle présenté aux chapitres 3 et 4 propose une évaluation exhaustive des impacts liés à une utilisation d'eau en ACV par l'intégration d'une différenciation entre l'eau de surface et souterraine, en considérant la qualité de l'eau prélevée et remise, en considérant tous les usagers humains *in-stream* et *off-stream*, et en évaluant la rareté d'eau basée sur la consommation d'eau et non les prélèvements. Inclure ces aspects dans la modélisation permet au model une meilleure représentation de la chaine cause-à-effet qui est décrite puisque chacun de ces paramètres représente un aspect de la problématique qui est modélisée. De plus, la plupart de ces aspects se sont révélés être influents sur les résultats en comparaison avec l'absence de ceux-ci, pour certaines, ou la majorité, des régions du monde tel que démontré aux chapitres 5 et 6. Seule l'inclusion des pêches/aquaculture s'est révélé être négligeable (section 5.3.3). Le facteur d'effet décrivant l'impact sur la santé humaine d'une privation d'eau pour les usagers agricoles s'est révélé être plus pertinent que le facteur comparable existant (Pfister et al. 2009), en décrivant de façon plus représentative la chaîne cause-à-effet. En effet, l'hypothèse que toute l'eau nécessaire à la production de nourriture pour un an par personne doit être consommée pour observer un cas de malnutrition est remplacée par une relation linéaire entre le manque de calorie et la malnutrition (voir section 5.3.3).

8.2 Faiblesses

La considération de la qualité de l'eau entrante et sortante entraîne un besoin de données supplémentaires pour la modélisation des facteurs de caractérisation qui décrivent la rareté associée à chaque catégorie d'eau. Des données par défaut qui proviennent du programme GEMStat (UNEP Global Environment Monitoring System (GEMS) Water Programme 2009) ont été utilisées, le plus exhaustif en termes de contaminants inclus dans la base de données, mais souvent inconsistant à plusieurs niveaux : fréquence des données reportées, nombre de points d'échantillonnage, liste des paramètres reportés et tests utilisés, etc. Ce manque de cohérence rend l'incertitude du paramètre P_i, qui décrit la fraction de l'eau disponible dans une région étant de qualité i ou meilleure (section 4.2), très élevée. Ces données sont également au cœur de l'identification de la catégorie d'eau disponible identifiée par région, en utilisant la médiane des valeurs de contaminants reportés (section 3.3.3), qui correspond à la donnée utilisée par défaut comme qualité d'eau prélevée lors de l'utilisation du modèle. Alternativement, une base de données potentiellement plus robuste existe, NEWS ("Oceanographic Commission UNESCO's Intergovernmental (IOC)" 2008), mais elle ne couvre que les paramètres d'azote et de phosphore. Il serait néanmoins intéressant de les comparer.

L'inclusion de la qualité dans l'indice de rareté amène également des questions de double comptage potentiel. Cet aspect est discuté dans les sections 5.3.2.1 et 6.4.1.9. Bien que souvent évité, la possibilité de double comptage est parfois discutable lorsque l'indicateur est utilisé en parallèle avec un indicateur de toxicité humaine, dans le cas ou le même contaminant qui rend l'eau non-fonctionnelle cause également des impacts en toxicité humaine, ce qui implique que l'eau fut utilisée. Mais même dans ce cas, les frontières temporelles et géographiques des deux types d'évaluations d'impacts entrent en jeux et une analyse approfondie au cas par cas doit être faite et le potentiel de double comptage identifié s'il y a lieu.

La section 5.3.2.2 a démontré que la modélisation de la rareté d'eau dépend largement de la relation donnée entre la fraction d'eau disponible utilisée (prélevée (WTA) ou consommée (CTA)) et la rareté résultante, ainsi que des seuils choisis pour définir ce qui constitue une rareté d'eau ou non. Des seuils liant des fractions d'eau disponible prélevées à des qualificatifs de *low*, *medium* et *high stress* sont souvent utilisés comme référence, mais leur équivalence en terme d'eau consommée est variable, et le seuil minimum reste discutable. En général ces choix demeurent quelque peu arbitraires puisque peu de directives ou de consensus existent à ce sujet et la vérification de ces choix reste difficile. Un consensus sur la problématique spécifique à qualifier par les termes de rareté et de stress devrait d'abord être atteint, afin que des paramètres agissant comme indicateurs de cette problématique puissent être identifiés et mesurés afin de paramétrer la relation mathématique et les seuils qui définissent ces indicateurs.

8.3 Perspectives pour l'empreinte eau

Dans un contexte d'empreinte eau, le modèle présenté permet l'évaluation des impacts liés à une utilisation d'eau sur un des deux sujets à protéger liés à l'utilisation de l'eau : la santé humaine, complémentant ainsi les méthodes d'évaluation d'impacts sur les écosystèmes. En ajoutant les impacts sur les écosystèmes en plus des impacts sur la santé humaine, on obtient un « water availability footprint », tel que décrit présentement dans le Draft International Standard ISO 14046. L'existence de cette méthode, et ma participation au groupe de travail a permis d'influencer le développement du standard en différenciant un « water availability footprint », où une baisse de disponibilité peut survenir suite à une dégradation ou une consommation d'eau, d'un « water scarcity footprint », qui prend seulement la consommation en compte pour quantifier la baisse de disponibilité. À ce jour, le modèle présenté ici est le seul permettant de calculer un *water availability footprint* au midpoint, en accord avec la norme.

Dans un cadre plus large, si l'objectif est une évaluation complète de la performance environnementale d'un produit, ou la comparaison de deux produits, une ACV s'impose pour éviter de déplacer les impacts d'une phase du cycle de vie à une autre, ou d'une catégorie d'impacts à une autre (ex : en dessalant de l'eau de mer à fort coût énergétique, et donc aussi d'impacts sur le réchauffement climatique, afin de préserver l'eau douce). La méthode développée s'intègre tout à fait dans les méthodologies ACV, ou encore à différents niveaux d'évaluation des impacts liés à l'eau, tel que montré par la figure 8-1. Tel des poupées russes, le *water scarcity footprint* évalue une problématique réduite par rapport au *water availability footprint*, qui est lui-même un aspect d'un *water footprint* exhaustif. Ce dernier considère tous les aspects liés à la ressource eau, incluant les catégories d'impacts associées à la pollution, i.e. acidification, eutrophication, etc. Finalement, on retrouve à un niveau plus haut encore, l'ACV complète, qui intègre tous les impacts sur l'environnement, et non seulement ceux liés à l'eau.

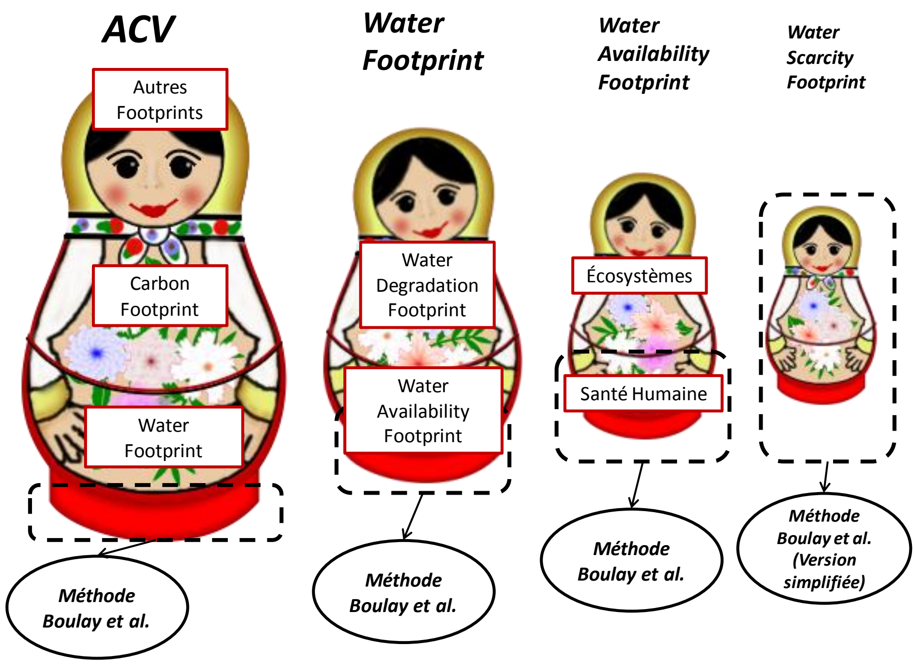

Figure 8-1 La méthode Boulay et al. mise en perspective dans un contexte de water footprint et d'ACV

Un problème en évaluation des impacts liés à l'utilisation de l'eau est la non-corrélation des modèles entre les indicateurs utilisés au niveau problèmes (i.e. rareté) et les résultats de modèles au niveau dommages. En effet, tant sur la santé humaine que sur les écosystèmes, la corrélation n'est pas constante. Pour la santé humaine, la différence de résultats entre un indicateur de rareté et les impacts sur la santé humaine vient du fait que les pays avec un GNI élevé peuvent présenter une rareté d'eau élevée sans qu'une utilisation d'eau se traduise en impacts liés à la malnutrition ou aux maladies liées à un manque d'eau potable. Dans ces pays, la baisse de disponibilité de l'eau pourra être comblée par l'utilisation de moyens compensatoires qui produiront également des impacts environnementaux, mas encore non inclus dans les modèles. Ceux-ci se doivent d'être inclus dans une ACV et un article en rédaction à ce sujet est présenté en annexe 7. Le résultat à ce jour est donc que les indicateurs de rareté sont corrélés aux dommages seulement pour les pays catégorisés comme « faible revenu », et sont partiellement corrélés pour les pays à « revenu moyen », alors que les impacts sont nuls au niveau dommage pour tous les pays à « revenu élevé » (section 4.2).

À ce jour aucun des modèles de caractérisation des impacts utilisent la rareté d'eau comme paramètre intermédiaire pour évaluer l'impact sur les écosystèmes de l'utilisation de l'eau. Ainsi, non seulement il n'y a pas de corrélation directe, mais on peut même se demander si au contraire, on ne prive pas davantage d'espèces si on diminue la disponibilité de l'eau dans une région humide, à faible rareté d'eau, qu'une région plus aride, à rareté élevé, ce qui entraînerait une corrélation inversée entre la rareté et les dommages sur les écosystèmes. Cette inconsistance force la communauté scientifique à se pencher sur la question et à fournir un indicateur de problèmes au moins partiellement corrélés avec les dommages évalués. L'industrie ayant déjà largement adopté les indicateurs de rareté comme métrique – du moins pour celles qui vont plus loin que les simple volumes d'eau prélevés, l'acceptabilité d'un nouvel indicateur ne risque pas d'être aisée. Un indicateur de problème, apparaissant comme un indicateur de rareté, qui prendrait en compte les besoins essentiels humains (ne pouvant pas être compensés) et écologiques, considérant la capacité d'adaptation et le type d'écosystème local, pourrait représenter un point plus éloigné sur la chaine cause-à-effet qui augmenterait potentiellement la corrélation avec les dommages.

Le travail effectué au cours de cette thèse m'a amenée à jouer un rôle souvent influent au sein de la communauté international. Pendant trois ans j'ai représenté le Canada dans le groupe de travail de la norme ISO sur le Water footprint, agissant souvent comme un des rares expertes scientifiques dans un comité formé principalement d'industries ou d'experts de d'autres domaines. Cette position m'a permis d'influencer la norme afin de l'aligner sur les développements scientifiques évoluant à l'intérieur du groupe de travail WULCA (Water Use in LCA) de la Life Cycle Initiative, auquel je participe également depuis quatre ans, d'abord comme membre, puis comme mandataire d'un projet de recherche (les articles présentés aux chapitres 5 et 6), et finalement comme Chaire du groupe. Ces deux expériences ont contribué non seulement à une évolution cohérente de ces deux groupes de travail et de ce projet de doctorat, mais elles m'ont également donné accès à une expertise, une communauté scientifique internationale et à des défis uniques et formateurs.

8.4 Ouverture et travaux futurs

Une augmentation de pertinence scientifique vient également avec une complexité plus élevée qui peut faire obstacle à la dissémination et l'adoption de la méthode par les industries. Pour contrevenir à ce problème, dans le cadre de ce doctorat j'ai élaboré des outils, d'information et d'accompagnement à l'utilisation aux usagers des méthodes développées (voir section 7.4). Parmi les aspects qui faciliteraient l'utilisation de la méthode, la création d'archétypes en est un que je recommanderais. Tant au niveau problèmes que dommages, les régions peuvent être classifiées en différentes catégories, et une valeur médiane par catégorie pourrait être représentative des impacts potentiels de ce type de région (ex : rareté élevée+ faible ressource économique, rareté moyenne+ressources économiques élevées, etc). Cette proposition est d'autant plus pertinente que l'incertitude est souvent plus élevée que les variations de résultats entre régions correspondant à un même archétype.

Un des piliers sur lequel repose le modèle et dépend la validité des résultats est la source de données pour les valeurs hydrologiques. Au moment de développer le modèle, WaterGap 2 présentait le seul modèle global fournissant des valeurs pour l'eau consommée et disponible. Il semble que d'autres sources de données ont maintenant émergé (ex : Aquaduct) et il est essentiel d'analyser en profondeur les différentes sources de données maintenant disponibles, leurs forces, faiblesses et ainsi statuer sur un modèle à privilégier.

Tel que mentionné ci-haut, les impacts associés aux processus compensatoires doivent être considérés pour évaluer de façon exhaustive les impacts liés à l'utilisation de l'eau dans les régions développées. Un cadre est développé dans la publication en cours de rédaction présentée en annexe 7. Un travail de compilation et d'opérationnalisation est par contre nécessaire afin de rendre accessible par les bases de données et logiciels d'ACV l'inclusion de ce type d'approche. Une autre façon de considérer l'effet des différentes capacités d'adaptation à un manque d'eau pour les usagers agricole est proposé par Motoshita et al. dans le concept du *trade effect (voir section 5.3.3).* Alors que l'approche présentée en Annexe 7 pré-suppose que l'adaptation à un manque d'eau pour l'irrigation se fera à un niveau local et visera à combler un manque d'eau par des sources alternatives, celle sous-jacente au *trade effect* implique qu'un manque d'eau pour l'irrigation se répercute par un changement sur le marché des imports/exports de nourriture, favorisant les pays à revenu plus élevés. Dans la réalité, les deux approches se côtoient probablement, la première représentant une diminution marginale de la disponibilité en eau, alors que la deuxième adresse potentiellement des changements de disponibilités en eau plus importants. Ces théories méritent d'être analysées, validées et opérationnalisées de façon harmonisée. De plus, les impacts liés à une adaptation des usagers ou un effet sur le système économique a déjà donné lieu à un débat (Weidema et al. 2005) qui a statué sur leurs inclusions à travers un inventaire additionnel de procédés, et non une modélisation au niveau des impacts.

Ce besoin d'harmonisation s'étend d'ailleurs à toute la méthodologie d'évaluation des impacts liés à une utilisation d'eau en ACV. Tel que décris au chapitre 5, plusieurs méthodes décrivent les mêmes chaînes cause-à-effet, utilisent des sources de données et des hypothèses différentes, sont calculées à des échelles différentes, etc. Ceci crée une situation difficile pour les praticiens mais également pour les règlementations et pour l'industrie qui souhaite faire référence à une méthode consensuelle. La prochaine étape afin d'augmenter l'acceptation auprès de l'industrie et des parties prenantes en générale est donc de trouver une méthode harmonisée qui permet de prendre les aspects importants et nécessaires de chaque méthode existante, en mettant à collaboration les développeurs de méthodes, et ainsi fournir une recommandation consensuelle.

CONCLUSION

Ce projet de thèse a permis de confirmer l'hypothèse de recherche qu'une méthodologie d'évaluation des impacts liés à l'utilisation d'eau en analyse du cycle de vie intégrant la qualité de l'eau entrante et sortante, la source et sa fonctionnalité pour des usagers humains spécifiques permet d'augmenter la complétude et le pouvoir de discrimination lors de l'évaluation des impacts potentiels d'un manque d'eau sur la santé humaine, causé par une consommation et/ou dégradation de la ressource. Ceci est attribuable à : 1) l'utilisation de la consommation au lieu des prélèvements d'eau dans l'indice de rareté, 2) la distinction de la source d'eau prélevée (eau de surface vs. souterraine), 3) la différentiation de la qualité d'eau prélevée et rejetée, 4) l'augmentation de la pertinence du facteur d'effet décrivant les impacts de la malnutrition causée par un manque d'eau, et 5) l'inclusion des usagers domestiques. La pertinence de ce dernier aspect demeurant toutefois discutable. Cependant, une différence importante a pu être observée de façon systématique principalement pour l'inclusion de la qualité et des usagers domestiques seulement. Les deux premiers aspects ont démontré une influence sur les résultats que dans des régions spécifiques du monde, et l'inclusion des usagers « in-stream », i.e. pêches/aquaculture, s'est révélée non-influente sur les impacts déjà quantifiés, et ce, pour le monde entier.

Un des défis de la recherche qui développe les modèles en ACV, est l'impossibilité fréquente de valider les modèles avec des données d'impacts correspondantes qui sont souvent manquantes pour décrire une chaine cause-à-effet spécifique. Bien qu'une validation a été présenté à la section 7.3, elle n'est que partielle. Ainsi, l'argumentation logique sur la problématique à modéliser demeure le meilleur moyen de validation d'un modèle, pour une science non expérimentale, quant à la pertinence de l'inclusion de chaque constituant individuel, suivi de l'observation d'une différence sur les résultats, telle que présentée au chapitre 5.

L'évaluation par le biais de la comparaison avec des modèles existants a permis de mettre en lumière l'importance des choix de modélisation suivants: 1) la résolution géographique à laquelle les facteurs sont originalement calculés, 2) les choix de modélisation entre le ratio d'eau disponible utilisée (CTA ou WTA) et le paramètre de rareté résultant, 3) le choix de la source de données pour les données hydrologiques et de distribution des usagers et 4) la résolution temporelle pour les prélèvements saisonniers ou spontanés. De plus, l'intégration des impacts environnementaux générés par une utilisation d'eau dans les pays capables de s'adapter à un manque d'eau, que ce soit par l'utilisation de technologies (Annexe 7) ou par un effet rebond sur les échanges commerciaux (section 5.3.3), est significative et il est important d'harmoniser la façon de l'adresser.

Finalement, l'application à l'étude de cas d'un produit de détergent à lessive a démontré que le modèle développé dans le cadre de ce projet de doctorat sur l'évaluation de l'utilisation d'eau peut directement être intégré dans le concept de l'empreinte eau tel que présenté dans la norme ISO DIS 14046 avec des catégories d'impacts complémentaires portant sur l'évaluation de la dégradation de l'eau. Les résultats ont permis de mettre en perspective les impacts potentiels dus à la consommation vs. la dégradation de l'eau et identifier les phases du cycle de vie et les lieux et dans le monde ou ces impacts sont potentiellement importants.

Malgré des incertitudes parfois élevées, un potentiel de surestimation des impacts dans certains pays, le besoin de données plus robuste et d'avantage d'effort dans opérationnalisation de ces modèles, ce travail contribue significativement à l'avancement des connaissances relativement à une meilleure compréhension des chemins d'impacts pour l'évaluation des impacts liés à l'utilisation de l'eau et au débat sur l'identification des meilleures pratiques.

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ANNEXE 1– ARTICLE 2 – INFORMATION SUPPLÉMENTAIRE

Cette annexe est partie intégrante du chapitre 4 et est publiée sous la forme d'information supplémentaire à l'article 2.

1 Water categories and users

The water categories represent the 17 possible elementary flows described by source (surface, ground or rain), water quality and user for which it is functional, as set out in Boulay *et al.*[1] and adapted here in Tables S1, S2 and S3.

Table S1: Water category functionalities per user (Source: Boulay *et al.*[1])*

Quality	**1**	**2a**	**2b**	**2c**	**2d**	**3**	**4**	**5**	**Rain**
Quality level	*Excellent*	*Good*	*Average*	*Avg-Tox*	*Avg-Bio*	*Poor*	*Very Poor*	*Un-usable*	
Contamina-tion	Low microbial low toxicity	Low microbial medium toxicity	Medium microbial medium toxicity	Low microbial high toxicity	High microbial low toxicity	High microbial medium toxicity	High microbial high toxicity	Other	N/A
Dom 1	√	**X**	**X**	**X**	**X**	**X**	**X**	**X**	√
Dom 2	√	√	√	**X**	**X**	**X**	**X**	**X**	√
Dom 3	√	√	√	√	√	√	√	**X**	√
Agriculture	√	√	√	√	√	√	**X**	**X**	√
Fisheries	√	**X**	**X**	**X**	√	**X**	**X**	**X**	√
Industry	√	√	√	**X**	**X**	**X**	**X**	**X**	√
Cooling	√	√	√	√	√	√	√	**X**	√
Surface water only (Groundwater not functional for in-stream users)									
Recreation	√	√	**X**	√	**X**	**X**	**X**	**X**	√
Transport	√	√	√	√	√	√	√	√	√
Hydro	√	√	√	√	√	√	√	√	√

√: Functional **X: Non-functional**

*****The distinction between Agri 1 and Agri 2 users in Boulay et al .is not explicitly considered here since the hypothesis is that irrigation occurs with locally available water and is therefore taken into account in the scarcity/stress calculation.*

Table S2: Definitions of water users (Source: Boulay et al.[1])

Water user	*Definition*
Domestic 1	Domestic user performing no treatment or simple chemical disinfection to the water prior to use
Domestic 2	Domestic user performing a conventional chemical-physical treatment (coagulation or precipitation, solid removal process, disinfection) or equivalent treatment to the water prior to use
Domestic 3	Domestic user performing an advanced treatment (i.e. conventional treatment and additional treatment (UV disinfection, adsorption, etc.)) , specific advanced treatment (e.g. reverse osmosis, nanofiltration, adsorption, ion exchange, desalination, etc.) or desalination to the water prior to use
Industrial	Industrial user (manufacturer) withdrawing available water and treating it at the required level
Cooling	Once through cooling water energy production
Agriculture 1	Agriculture that requires good quality irrigation water
Agriculture 2	Agriculture that only requires poor quality irrigation water
Fisheries	Freshwater aquaculture and catching
Hydropower	Hydroelectricity production
Transport	Transportation of goods through inland waters
Recreation	Recreational activities (e.g. swimming and water sports)

Table S3: Water category threshold values (selected parameters, source: Boulay *et al.*[1])

PARAMETERS	Symbol	Units	1	2a	2b	2c	2d	3	4	5
Fecal coliforms		UFC/100ml	20	200	2000	200	10000	10000	20000	
Microcystin-LR		mg/l	0.001	0.001	0.001					
True color		Color unit (CU)	15	50	50	100	100	100	100	
Suspended solids	SS	mg/l	25	25	100	25	40	100	300	
Total dissolved solids)	TDS	mg/l	500	500	500	2000	2000	2000	40000	
Biochemical oxygen demand (5 days)	BOD_5	mgO_2/l	5	5	5	5	5	20	20	
Total nitrogen		mg N/L	30	30	30	30	30	30		
Hardness		mg	500	500	500	700	7000	7000	7000	

		$CaCO_3$/l				0			
pH		pH unit	4.5 - 8.4	4.5 - 8.4	4.5 - 8.4	4.5 - 8.4	4.5 - 8.4	4.5 - 8.4	4.5 - 9
Hydrogen sulfide	H_2S	mg/l	0.4	2	2	2	0.4	2	
Total residual chlorine	Cl_2	mg/l	0.2	5	5	5	0.2	5	
Chloramines	NHn Cl(3-n) n=0,1 or 2	mg/l	0.1				0.1		
Chlorine dioxyde	ClO2	mg/l	0.1				0.1		
Aluminium	Al	mg/l	0.05	5	5	5	0.05	5	
Ammonia	NH_3	mgN/l	0.05	0.1	1	0.1	0.3	2	2
Antimony	Sb		0.02	0.02	0.02	0.2	0.2	0.2	0.2
Arsenic	As	mg/l	0.01	0.01	0.01	0.1	0.1	0.1	0.1

	(mg/l)								
Barium	Ba	mg/l	0.7	0.7	0.7	7	7	7	7
Beryllium	Be	mg/l	0.1	0.1	0.1	0.1	0.1	0.1	
Bicarbonate	HCO_3^-	mg/l	500	500	500	500	500	500	
Boron	B	mg/l	0.5	0.5	0.5	3	3	3	5
Cadmium	Cd	mg/l	0	0.003	0.003	0.03	0	0.03	0.03
$CaxSO_4$		$(mg/l)^2$	1000000	1000000	1000000	1000000	1000000	1000000	1000000
Chloride	Cl^-	mg/l	250	250	250	350	350	350	25000
Chromium (total)	Cr total	mg/l	0	0.05	0.05	0.1	0	0.1	0.5
Copper	Cu	mg/l	0.05	0.2	0.2	0.2	0.05	0.2	20
Cyanide	CN-	mg/l	0	0.07	0.07	0.7	0	0.7	0.7

	HCN								
Fluoride	F	mg/l	1	1	1	1	1	1	15
Iron	Fe	mg/l	5	5	5	5	5	5	10
Lead	Pb	mg/l	0	0.01	0.1	0.1	0	0.1	0.1
Manganese	Mn	mg/l	0.2	0.2	0.2	0.2	0.2	0.2	0.5
$MgxSiO_2$	$mgCaCO_3$ x $mgSiO_2/l^2$		35000	35000	35000	350 00	35000	35000	35000
Silica	SiO_2	mg/l	150	150	150	150	150	150	150
Mercury	Hg	mg/l	0.001	0.006	0.006	0.06	0.001	0.06	0.06
Molybdenum	Mo	mg/l	0.01	0.01	0.01	0.01	0.01	0.01	0.7
Nickel	Ni	mg/l	0.07	0.07	0.07	0.2	0.2	0.2	0.7
Nitrates	NO_3^-	mgN/l	50	50	50				

Nitrites	NO_2^-		3	3	3				
Chlorides/nitrites	Cl^- / NO_2^-	$mgCl^-$ / $mgN\text{-}NO_2^-$	< 17				< 17		
Phosphorus (total)	P tot	mgP/l	0.1	0.1	0.1	0.1			
Sulfur	S	mg/l	5	5	5	5	5	5	5
Selenium	Se	mg/l	0.01	0.01	0.01	0.02	0.02	0.02	0.1
Sodium	Na	mg/l	200	200	200	210	210	210	15000
Sulfate	SO_4^{2-}	mg/l	500	500	500	3000	3000	3000	3000
Uranium	U		0.015	0.015	0.015	0.15	0.15	0.15	0.15
Vanadium	V	mg/l	0.1	0.1	0.1	0.1	0.1	0.1	
Zinc	Zn	mg/l	0	2	2	2	0	2	

2 Water scarcity

Current methods considering potential impacts from water use in LCA are all based on the following hypothesis: the scarcer the resource, the higher the potential impacts[2-5] since competition among users increases with lower water availability. A relevant scarcity parameter is therefore important. However, the challenge in the search for an adequate indicator is finding the right balance between complexity and data availability. Rijsbeman[6] proposes a critical review of several water stress indicators. The most widely used indicator is the Criticality Ratio (CR) introduced by Alcamo *et al.*[7], also referred to as the withdrawal-to-availability ratio (WTA). This ratio links the total volume of human withdrawals to the volume of renewable water available for a specific geographic region. Water scarcity thresholds based on this ratio have been proposed.[8] However, while this indicator is easy to understand and data are readily available, it does not take seasonal variations or differentiate the source or quality of water into account. Pfister and colleagues[5] proposed an indicator, the Water Scarcity Index (WSI) based on the CR, which integrates a parameter accounting for seasonal variability according to WaterGap model data.[9] While the CR is a convenient index, it considers that all water withdrawals affect scarcity. Water that is withdrawn and later released (i.e. not consumed) should not, however, be considered in water scarcity. Therefore, consumed rather than withdrawn water was considered in this scarcity parameter. While releasing degraded water may contribute to water availability issues, this is considered through the use of different water categories, their respective scarcity and the associated losses in functionalities. The water availability parameters used (Q90 and GWR) to calculate the scarcity parameter are averages for the 1961-1990 time period, and the water consumption data (CU and fg) is for the year 2000.

Figure S1 shows the correlation between the withdrawal-based ratio and the consumption-based ratio (R^2= 0.63). On this basis, withdrawal-based stress thresholds of 10%, 20%, 40% and 80% are then converted to consumptive-based stress thresholds of 10%, 12%, 18% and 40%, respectively. This figure excludes outlier data and zero values, representing regions where withdrawals are very high (outlier) or irrelevant data showing consumption but no withdrawals (zeros). While the outlier data are relevant to support the choice of considering consumption rather than withdrawal, they are meaningless in the attempt to determine a trend linking withdrawals to consumption. These thresholds are then used in the modeling of the S-curve described in Equation S1, which served to obtain the stress index α from the scarcity parameter α^*, as presented in the main article.

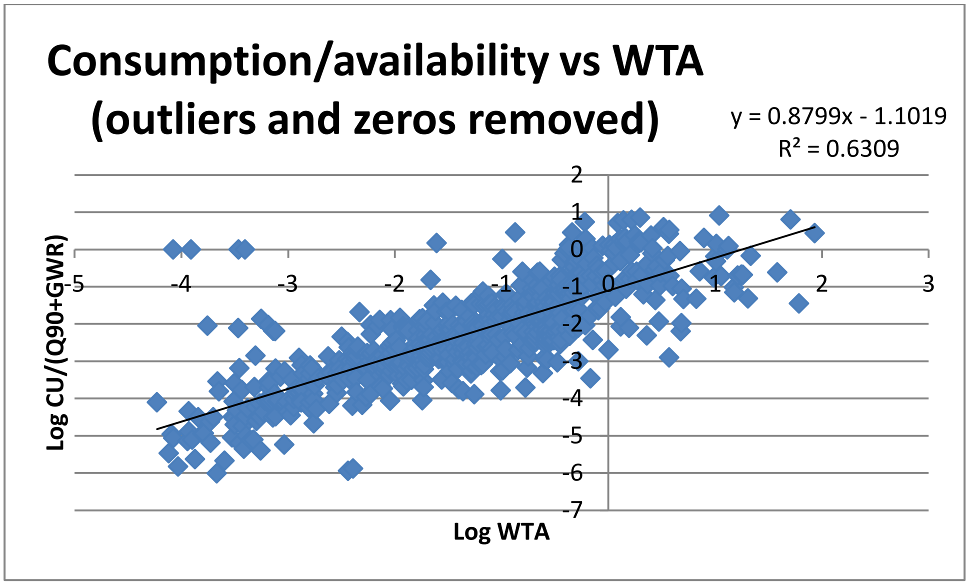


Figure S1: Correlations between withdrawal-to-availability ratio (or criticality ratio) and consumption-to-availability ratio

$$\alpha = \frac{1}{\left(1+0.331e^{-0.58(\alpha*-2.085)}\right)^{1/0.0039}} \qquad \text{Equation S1}$$

3 Special considerations for rain water and green water

A separate water category was created for rain water in order to allow the modeling of rain water harvesting. It describes water that is collected before it reaches the ground, and the scarcity is calculated like other scarcity parameters, as per equation S1, considering all consumptive uses and available surface and ground water. The parameter Pr represents the fraction of the best-quality water available in the region. α_{Rain} therefore assesses the local stress of the best available water category. When assessing the impacts of harvested water use, the functionality parameter is adjusted to account for the fact that a fraction of rainwater is naturally evapotranspirated and for the distribution between infiltration and run-off, which will lead to ground and surface water, respectively, and affect users differently. While rain water can serve all users, only part of it becomes functional as surface water from run-off (25%) or as groundwater from infiltration (25%), since part of it is lost through evapotranspiration (EPT, 50%).[10] Therefore 25% of rain water is considered functional for in-stream human use and 50% for off-stream users ($F_{rain,\ instream}$ = 0.25 and $F_{rain,\ offstream}$ = 0.50). These values may change from one region to another based on climate and vegetation and it may be relevant to adapt them from regional data on local run-off, infiltration and EPT. Resulting rain water scarcity and CF are available in the SI Excel document.

$$\alpha_{Rain} = \frac{CU}{(Q90+GWR)} \times 1/P_r \qquad \textit{Equation S1}$$

The impacts of the use of green water (rain water used in agricultural production) are not directly assessed by the methodology but may be assessed indirectly by the user by evaluating the change in water evapotranspiration caused by a given crop and incurring extra water consumption. The regional evapotranspiration reference state and the regional fraction of infiltration and run-off should be used for this purpose. Data from Siebert et al.[11] can be used to assess the evapotranspiration of different crops. The additional water evapotranspirated from the crop is equivalent to the water consumption to be assessed and should first be divided using this infiltration/run-off fraction into an equivalent of surface and ground water consumed (quality 1 or best available in the region). The methodology can then be used to assess impacts associated with these water consumptions.

4 User identification U_j

The fraction of total off-stream freshwater withdrawal used by the domestic and agricultural sectors, industry and cooling was obtained from WaterGap.[9] This fraction is further sub-divided among similar users (for Dom 1, 2 and 3) resulting in U_j, the proportion of water withdrawn for each user. For domestic users, because the proportion of municipal water treatment carried out in each region was not available, the following hypothesis was adopted: rural populations will generally be represented by Dom 1 users (not treating or only disinfecting their water), and urban populations will more likely correspond to Dom 2 (conventional water treatment). The fraction of available water from desalination[12] was used to evaluate the proportion of Dom 3 users, defined by desalination or extensive treatment by Boulay *et al.*[13] The latter was first deducted from the urban fraction. As it becomes accessible, data on the domestic water supply treatment carried out in each region will be a valuable addition to the model. Data on population distribution was obtained from the Earthtrend database.[14] Although Boulay et al. propose to distinguish two types of agriculture based on the quality of the water used, water was considered functional for agriculture if it meets the requirements of the lowest user (Agri 2). No distinction was therefore made at this level, implying that water use distribution for agriculture follows that of water quality in a country and is therefore accounted for in the Pi parameter of the scarcity index. That is, agriculture happens whether the available water is of good or bad quality and uses the resource proportionately to its availability.

For in-stream users, a change in availability of 1 m^3 of water will deprive each of the in-stream users proportionally to the intensity at which they use the surface water bodies. User distribution for fisheries was therefore calculated by dividing the total amount of water used for fisheries (aquaculture and capture) by total available surface water. Data on water used for freshwater fisheries can be obtained from the quantity of freshwater fish produced and captured per country[15] and the average amount of water needed to produce one kilogram of fish (0.57 to 11.76 litres for aquaculture; no value available for capture).[16]

5 Adaptation capacity

The GNI was chosen as the socio-economic indicator for its simplicity and correlation with relevant social development parameters such as access to improved drinking water sources and access to sanitation. Published in the third edition of the United Nation's *World Water Development Report*,[17] these correlations are reproduced below along with the World Bank thresholds for the adaptation capacity (AC) parameter, showing the acceptability of the thresholds as an indicator of a country's adaptation capcity with regards to water use.

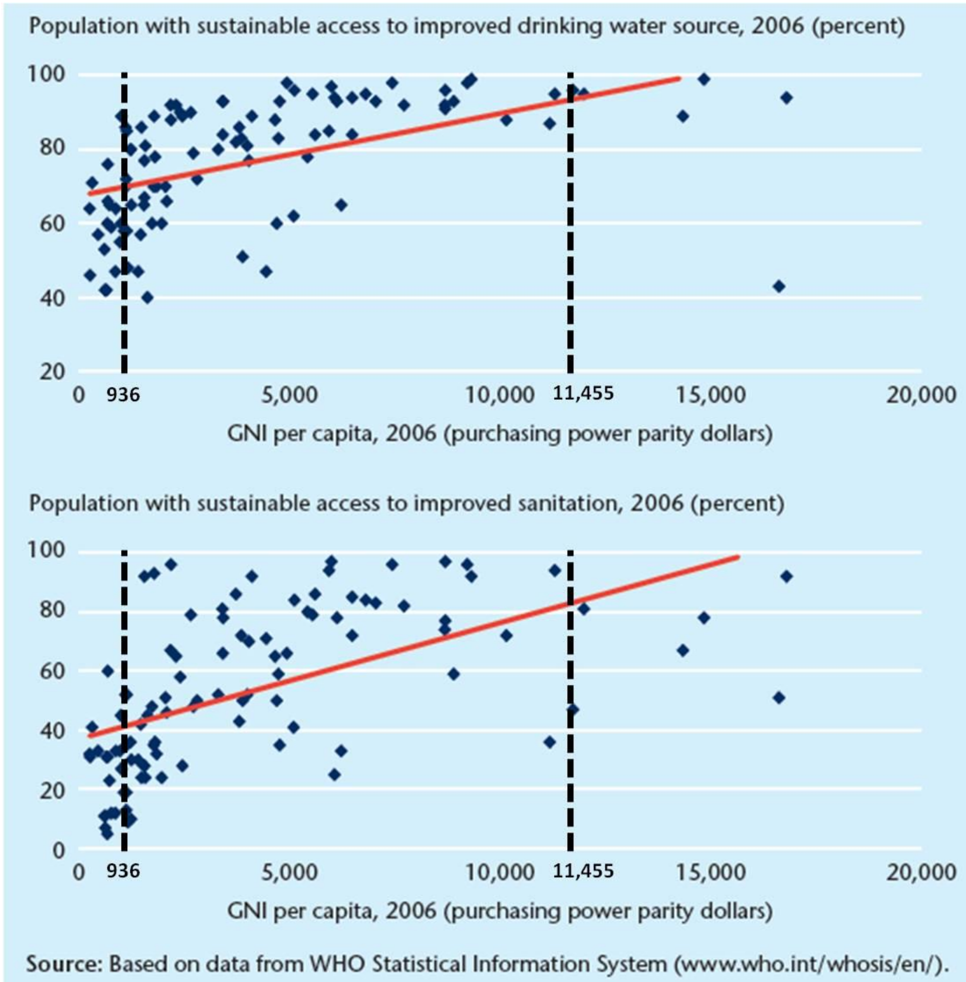


Figure S2: Relation of access to water and sanitation, income and adaptation thresholds from World Bank country classification (adapted from[16])

6 Effect factor

As discussed in the paper, impacts will be generated through a compensation process when a water deficit occurs for the remaining users (transport, hydro, industry, cooling and recreation) in countries able to compensate. The rationale relies on the fact that these uses are a means to increase quality of life but do not actually fulfill essential needs, as described in Maslow's pyramid of essential needs.[18] Although it can be argued that some industries fulfill essential needs (e.g. pharmaceuticals, building materials, etc.), the human health impacts of a water shortage for these industries has not been modeled. This is justified by the higher adaptation capacity at the industrial level as compared to the national level (including farmers, fishermen and domestic water supply). It is therefore reasonable to assume that these industries would first relocate or adapt their production if faced with a water shortage before anyone would suffer health consequences from losing access to their products. The effect factors were set out for agriculture, fisheries and domestic use. The details are presented here.

Human health impacts from water deficit in agriculture: To quantify the human health impacts caused by a water deficit in agriculture, adverse effects on human health (DALY) from a calorie deficit were linked to the amount of water required to produce 1 kcal food as per Equation S2.

$$E_{0,\text{agriculture}} = \frac{BHCM}{WRC} \qquad \text{Equation S2}$$

Where, $E_{0,\text{agriculture}}$ expresses the burden in DALY caused by the deficit of 1 m^3 water for agriculture, BHCM, the burden of health from calorie malnutrition (in DALY/kcal) and WRC, the water requirement per kcal (in m^3/kcal).

The daily nutritional requirement of 2600 kcal was used as a reference[19] to calculate the WRC. It is estimated that 2 000-5 000 l of water are required to produce this amount of calories.[16] The range accounts for the variety in diet throughout the world as well as the different water requirements depending on climate and agricultural practices. This translates into a range between 0.71 to 1.79 l per kcal produced. The average value of 1.25 l per kcal produced was used as the water requirement per calorie (WRC).

The BHCM was calculated using national data on the health burden caused by malnutrition along with the specific calorie deficit for the given country, as shown in Equation S3, resulting in $BHCM_c$ for each of the 54 low-income countries for which data were available. The calorie deficit per country (TCD_c) was obtained, as per Equation A3, from the undernourished population percentage[20], the specific food deficit of the undernourished population in a given country[21] and the total population. A geometric mean of the obtained $BHCM_c$ was then used to determine the BHCM used in Equation S2. The BHCM distribution is best represented by a log-normal fit with a p-value of 0.15 (acceptable distribution when above 0.05) justifying the choice of a geometric mean.

$$BHCM_c = \frac{HBM_c}{TCD_c} \qquad \text{Equation S3}$$

Where HBM_c represents the total health burden caused by calorie malnutrition in country c (in Daly/yr) and TCD_c is the total calorie deficit in country c (kcal/yr) calculated as per Equation S4.

$$TCD_c = PUN_c \times Pop_c \times FDUP_c \qquad \text{Equation S4}$$

Where PUNc represents the prevalence of undernourishment in the total population (in %)[20], Pop_c is the total population (in persons) per country and FDUPc is the food deficit of undernourished population (in kcal/person/day)[21] .

The HBM data were obtained using only 50% of the burden from protein-energy deficit malnutrition (calorie deficit) in DALY for each country[22] since the other 50% can be attributed to water, hygiene and sanitation issues, when, for example, parasites prevent the full absorption of ingested calories[16].

However, since about 40%[23] of the world's agricultural production is destined for livestock feed, a water shortage for agriculture destined for livestock production will not cause the same human health burden in DALY. An average of 7.22 kcal from grains are needed to produce 1 kcal of meat and dairy products based on a production-weighted average.[24,25] This is taken into account by correcting $E_{0,\text{agriculture}}$ accordingly in order to obtain the adjusted $E_{\text{agriculture}}$ as shown in Equation S5. Drinking water requirements for livestock were not taken into account since livestock was not considered as a separate user. This is justified by the hypothesis that the water needed to raise livestock is predominantly used for feed irrigation. Non-food agricultural production was not separated from these values. However, seeing as the world average is around 4% of total agricultural production[26], it is unlikely to make a significant difference.

$$E_{\text{agriculture}} = \left(0.6 \times E_{0,agriculture}\right) + \left(0.4 \times \frac{E_{0,agriculture}}{7.22}\right) \qquad \text{Equation S5}$$

Human health impacts from water deficit in fisheries: The health burden related to water deficit in fisheries was assessed in the same way as for agriculture (Equation S1). While the value of the BHCM remains unchanged, the WRC will vary depending on whether the kcal comes from agriculture or fisheries. The water needed to produce 1 kcal through fisheries ranges from 0.57 to 11.76 litres.[16] An average value of 6.165 l/kcal was used.

Human health impacts from water deficit for domestic uses: Assessing the health impacts of a water deficit for domestic use is not a simple task since several parameters are involved in the relationship between water availability, water use, sanitation and disease. Health impacts associated with water, hygiene and sanitation were considered in the model, including diarrheal diseases and nematode infections. In addition, 50% of the burden caused by malnutrition previously discarded because it was generated from water, hygiene and sanitation issues[16] was also considered. Although a lack of hygiene and sanitation is not necessarily linked to a lack of water use, it has been stated that "hygiene practices are, at least partly, correlated with water availability in regions where water is scarce."[27] The burden of these causes in DALY per country[22] was then related to the water deficit for domestic use in each country. This was calculated using the proposed 50 l per capita per day necessary to ensure low health concerns and cover the most basic needs, as evaluated by the WHO[27], resulting in a minimum water requirement for domestic use per country. The difference between this requirement and the actual amount of water used for domestic purposes[28] provided a deficit amount of water for domestic purposes per country. Only countries with water deficits were considered (Equations S6). The final value of $E_{Domestic}$ was obtained, as before, by the geometric average of all $E_{Domestic,c}$ values obtained from Equation S6, since this data fit a log-normal distribution with a p-value higher then 0.05 (0.0839).

$$E_{Domestic,c} = \frac{HBW_c}{DNeeds_c - DWithdrawals_c} \qquad \text{Equation S6}$$

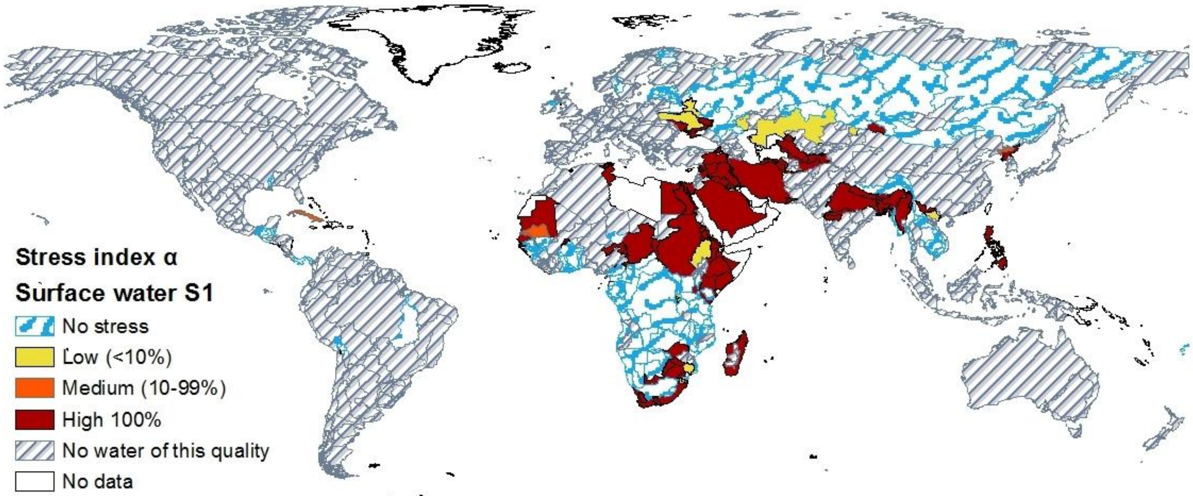
Where $E_{Domestic,c}$ represents the effect factor for domestic water deprivation in country c in DALY/m^3 and HBW_c is the health burden from water-related issues in country c in DALY/yr. The denominator represents the water deficit for domestic use in the country by the difference between $DNeeds_c$, the domestic needs for water in country c based on 50 l per capita in m^3/yr and $DWithdrawals_c$, the domestic water withdrawals in m^3/yr.

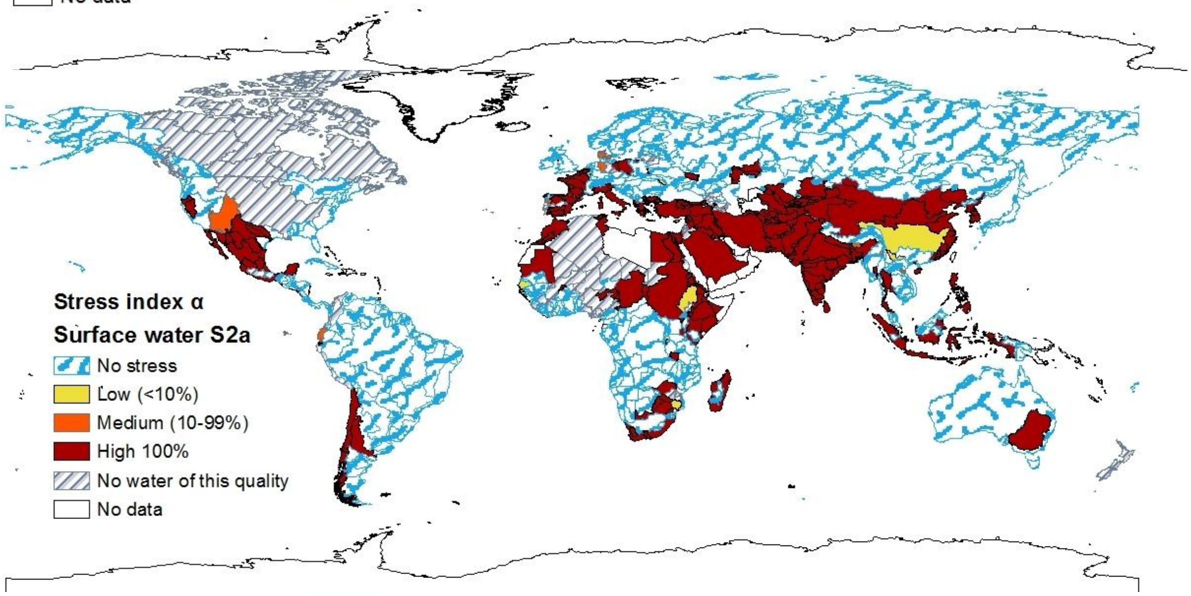
All results for these calculations are available in the Excel document.

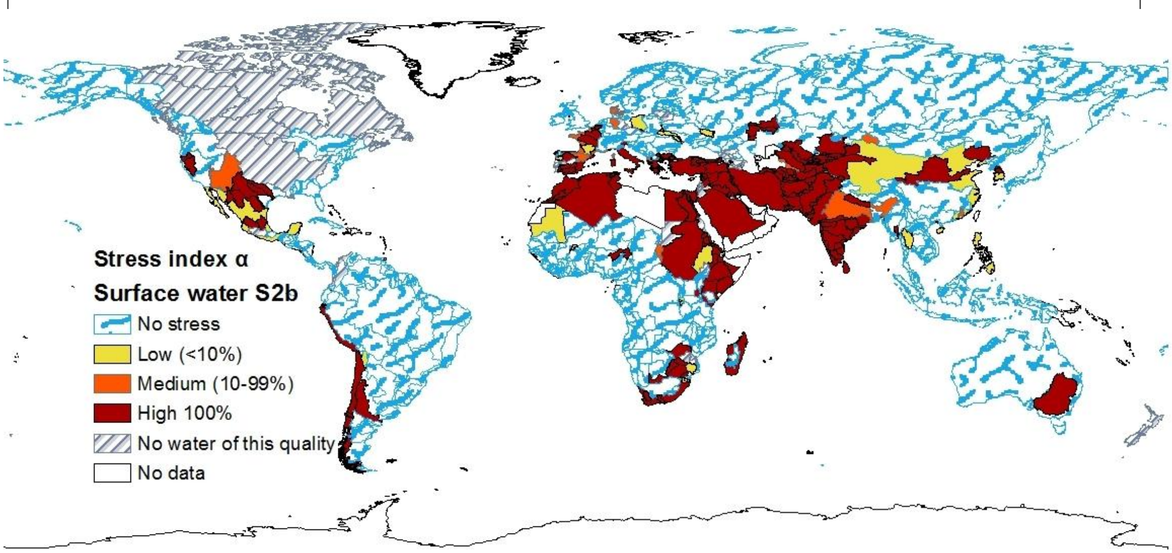
7 Results

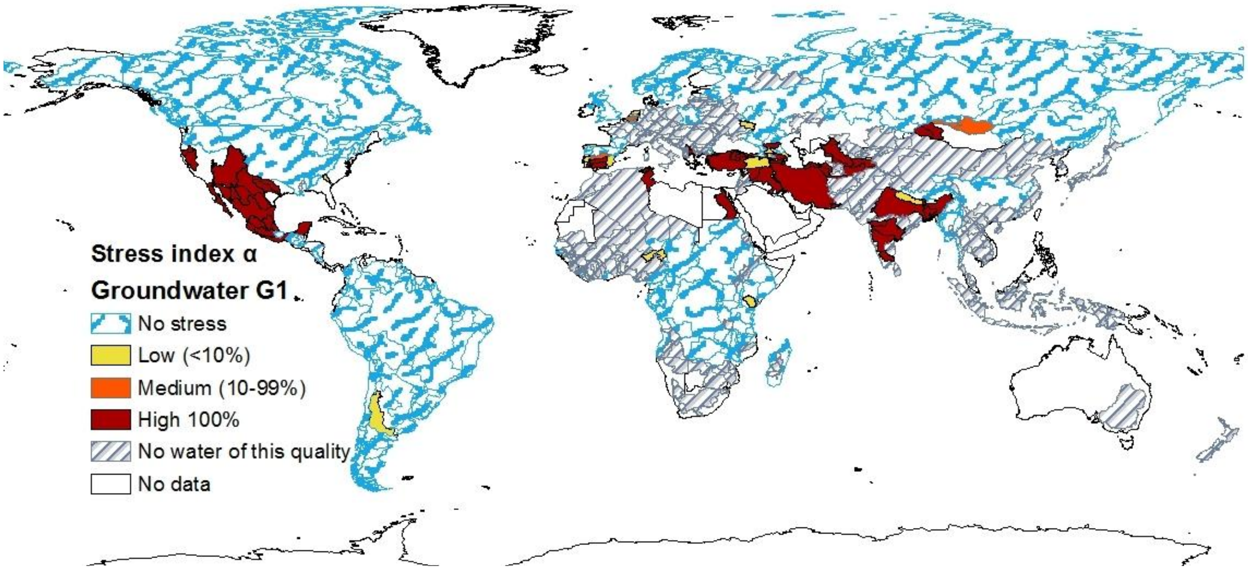
A - Scarcity

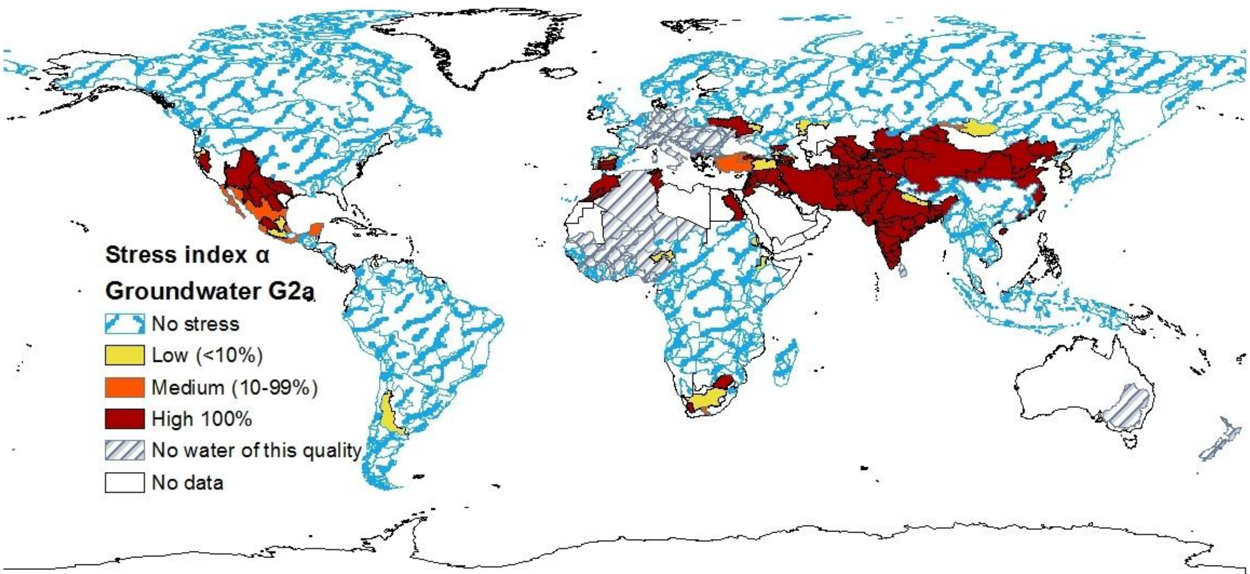
Water scarcity for surface categories S1, S2a, S2b, groundwater G1, G2a and rain are presented in Figure S3.


Stress index α
Surface water S1
No stress
Low (<10%)
Medium (10-99%)
High 100%
No water of this quality
No data


Stress index α
Surface water S2a
No stress
Low (<10%)
Medium (10-99%)
High 100%
No water of this quality
No data


Stress index α
Surface water S2b
No stress
Low (<10%)
Medium (10-99%)
High 100%
No water of this quality
No data


Stress index α
Groundwater G1
No stress
Low (<10%)
Medium (10-99%)
High 100%
No water of this quality
No data


Stress index α
Groundwater G2a
No stress
Low (<10%)
Medium (10-99%)
High 100%
No water of this quality
No data

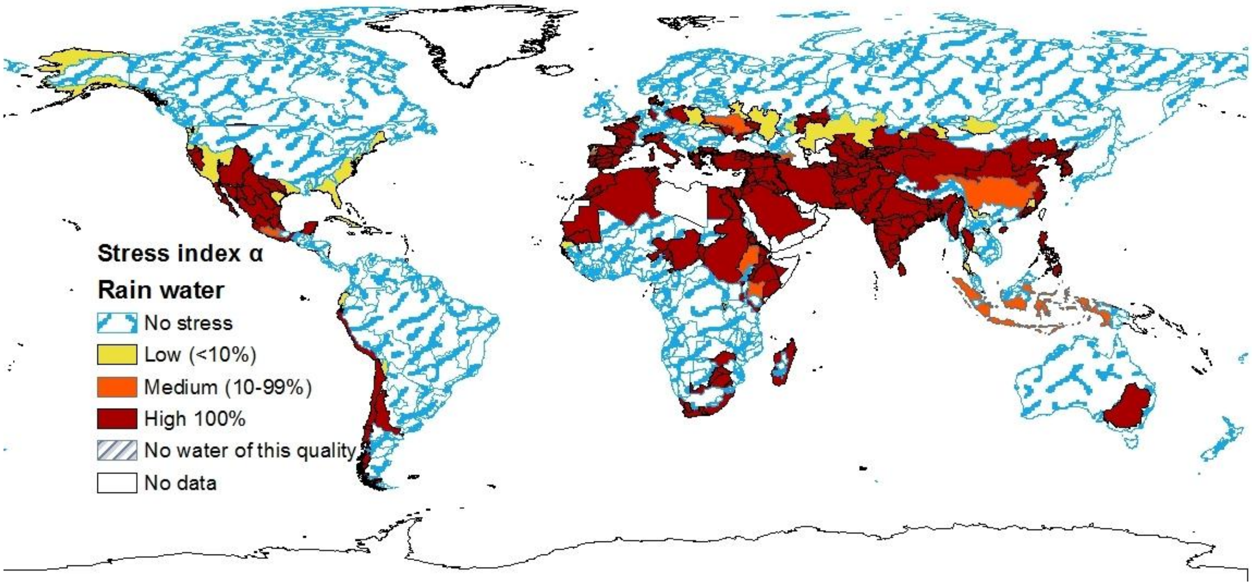
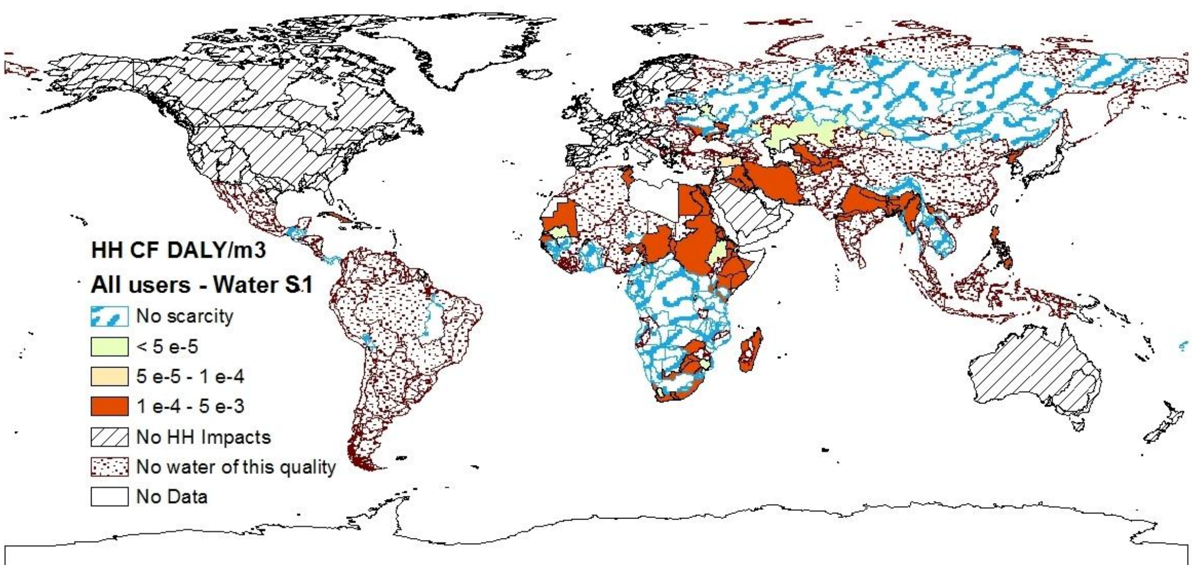
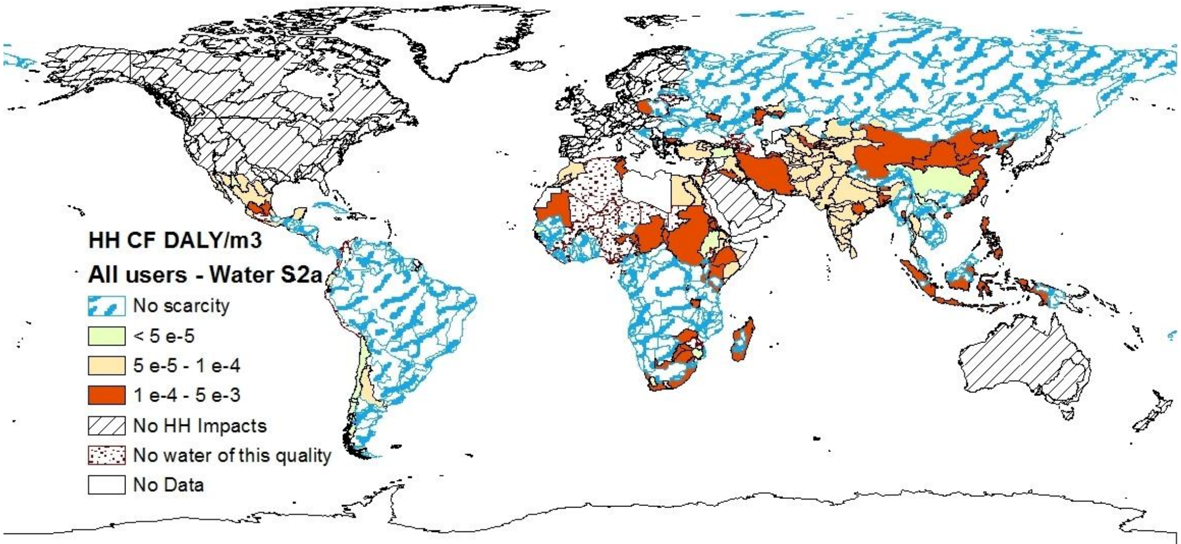


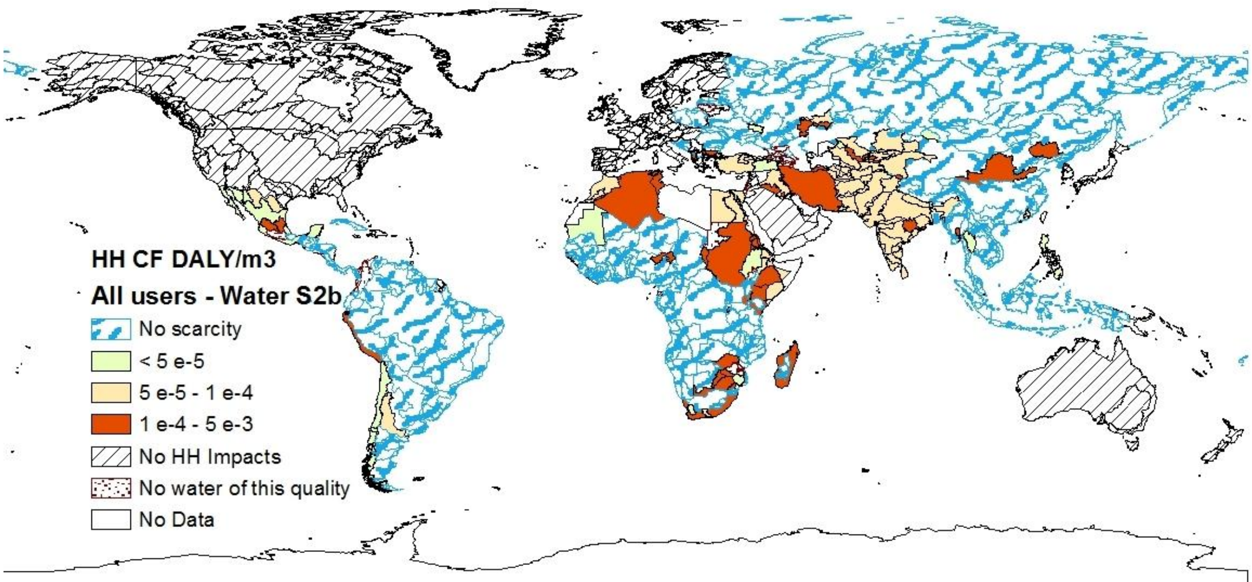
Figure S3: Water scarcity maps S1, S2a, S2b, G1, G2 and rain

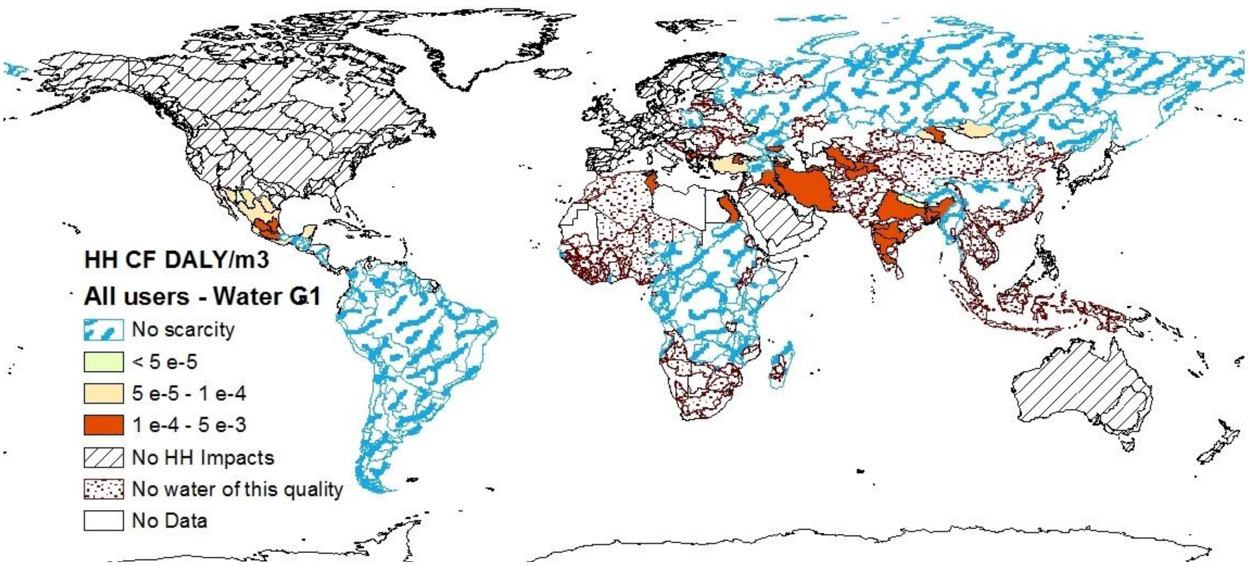
B - Human health CFs – Distribution hypothesis

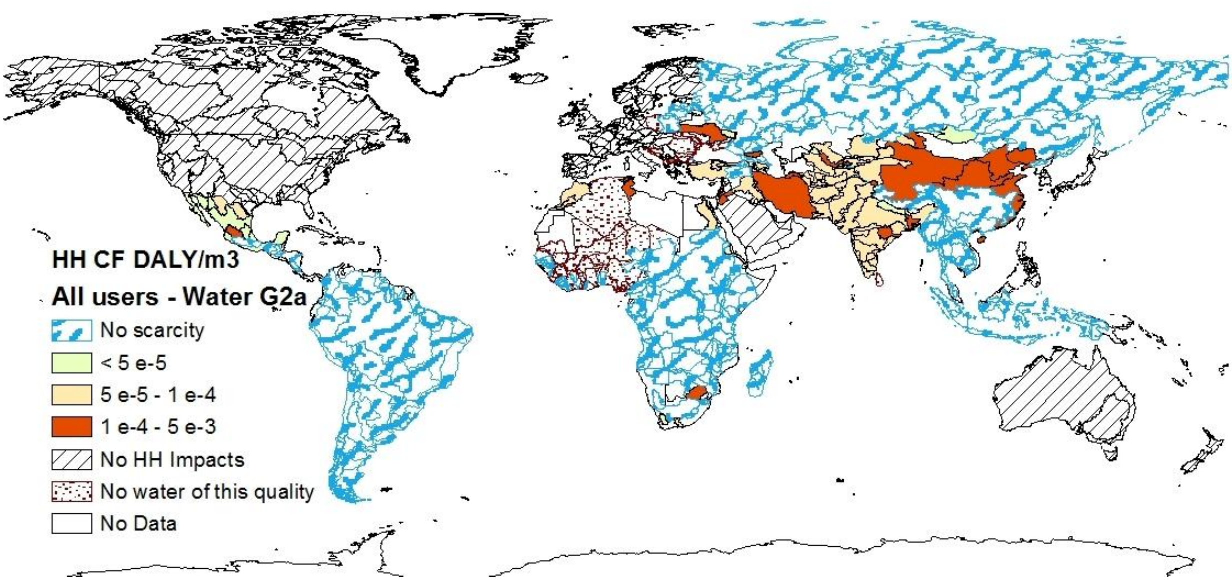
Results for human health characterization factors with the distribution hypothesis (including domestic users) are illustrated for water categories S1, S2a, S2b, G1, G2a and rain in Figure S4.




HH CF DALY/m3
All users - Water S2a
No scarcity
< 5 e-5
5 e-5 - 1 e-4
1 e-4 - 5 e-3
No HH Impacts
No water of this quality
No Data


HH CF DALY/m3
All users - Water S2b
No scarcity
< 5 e-5
5 e-5 - 1 e-4
1 e-4 - 5 e-3
No HH Impacts
No water of this quality
No Data


HH CF DALY/m3
All users - Water G1
No scarcity
< 5 e-5
5 e-5 - 1 e-4
1 e-4 - 5 e-3
No HH Impacts
No water of this quality
No Data


HH CF DALY/m3
All users - Water G2a
No scarcity
< 5 e-5
5 e-5 - 1 e-4
1 e-4 - 5 e-3
No HH Impacts
No water of this quality
No Data

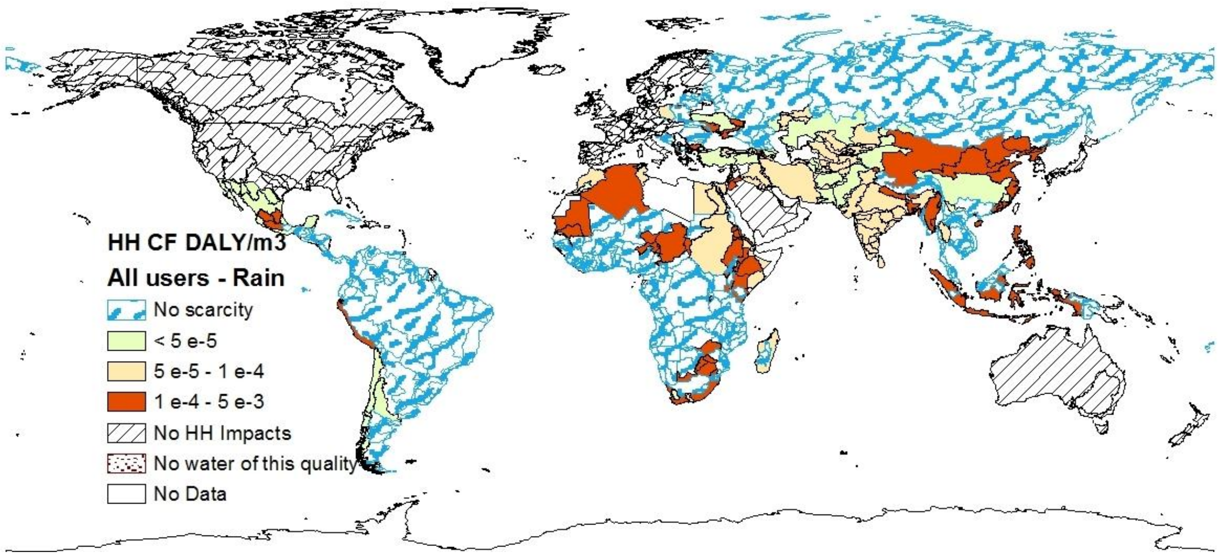
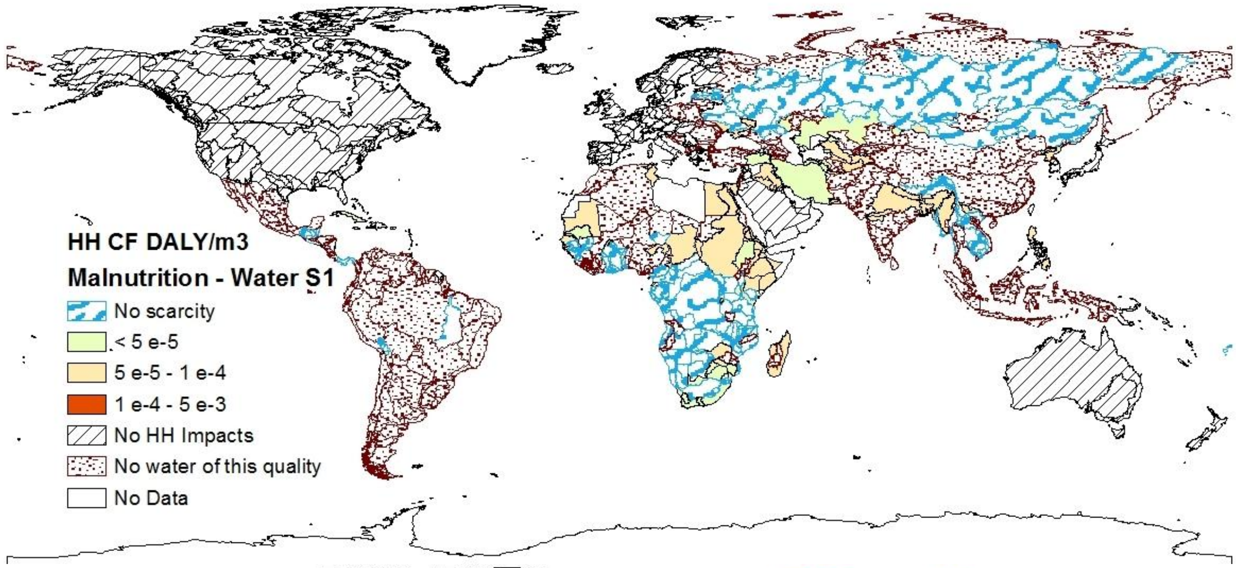
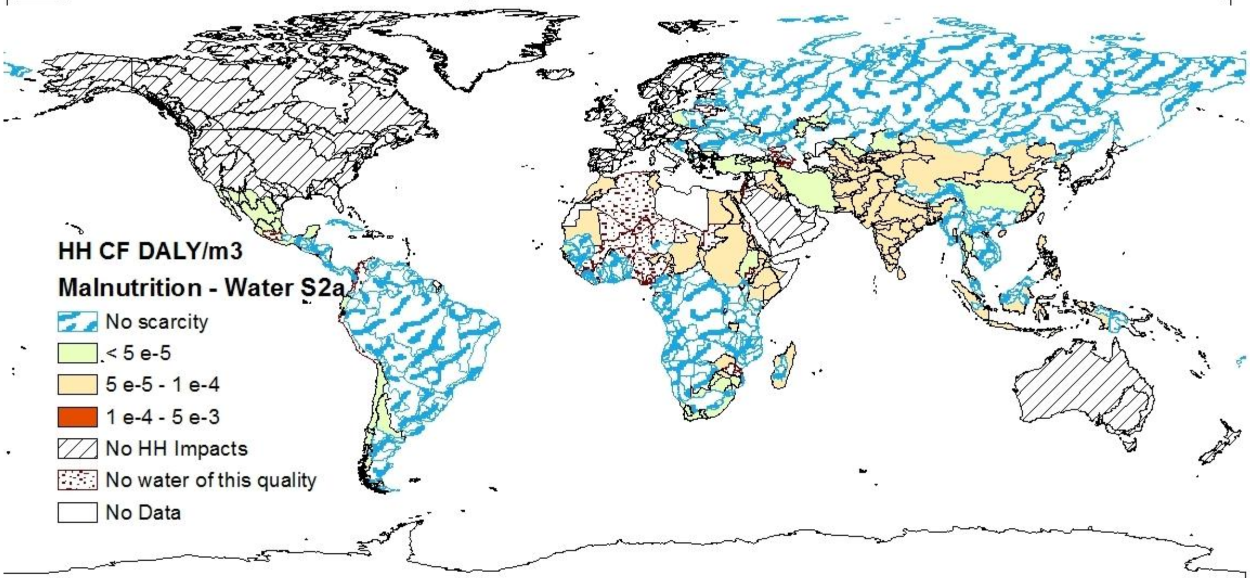


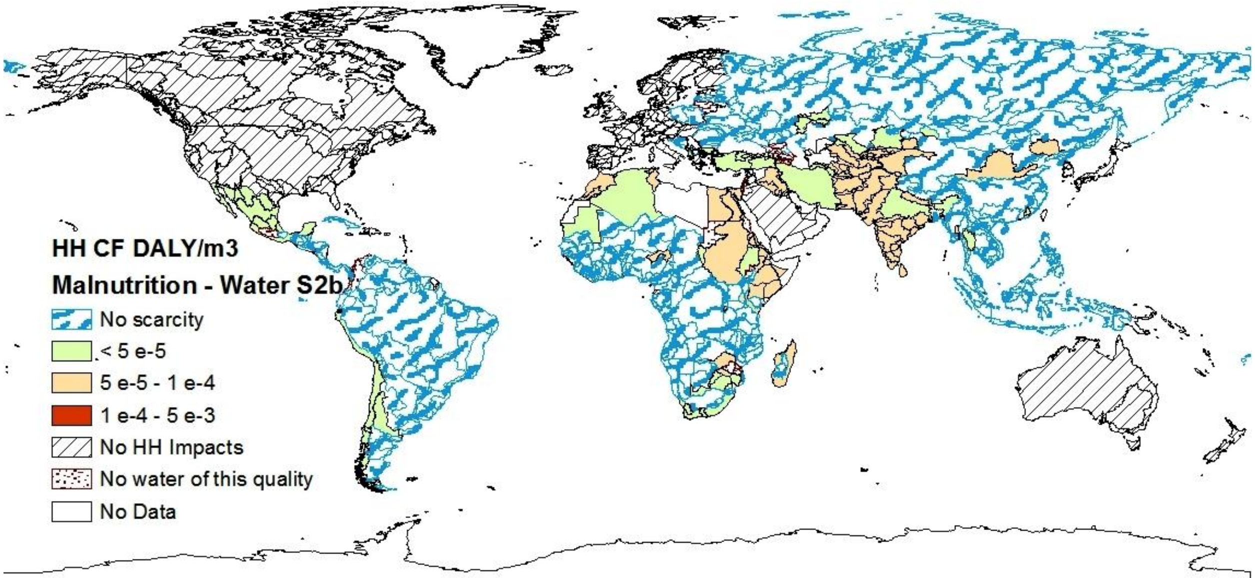
Figure S4: Human health impacts – distribution hypothesis - maps for water uses S1, S2a, S2b, G1, G2a and rain

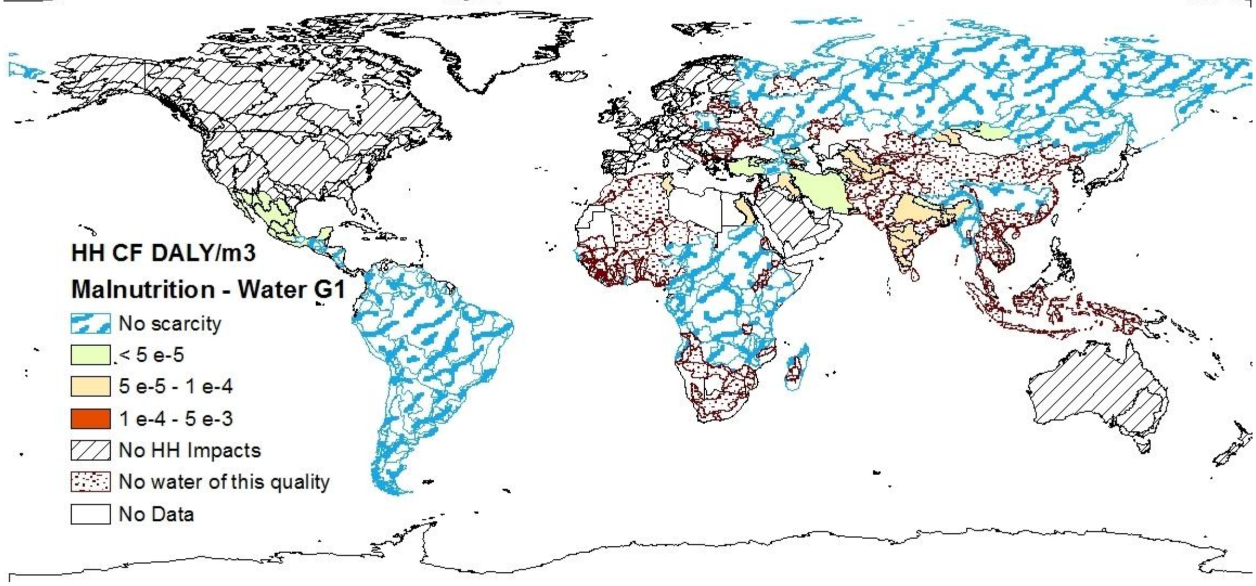
C - Human Health CFs – Marginal user hypothesis

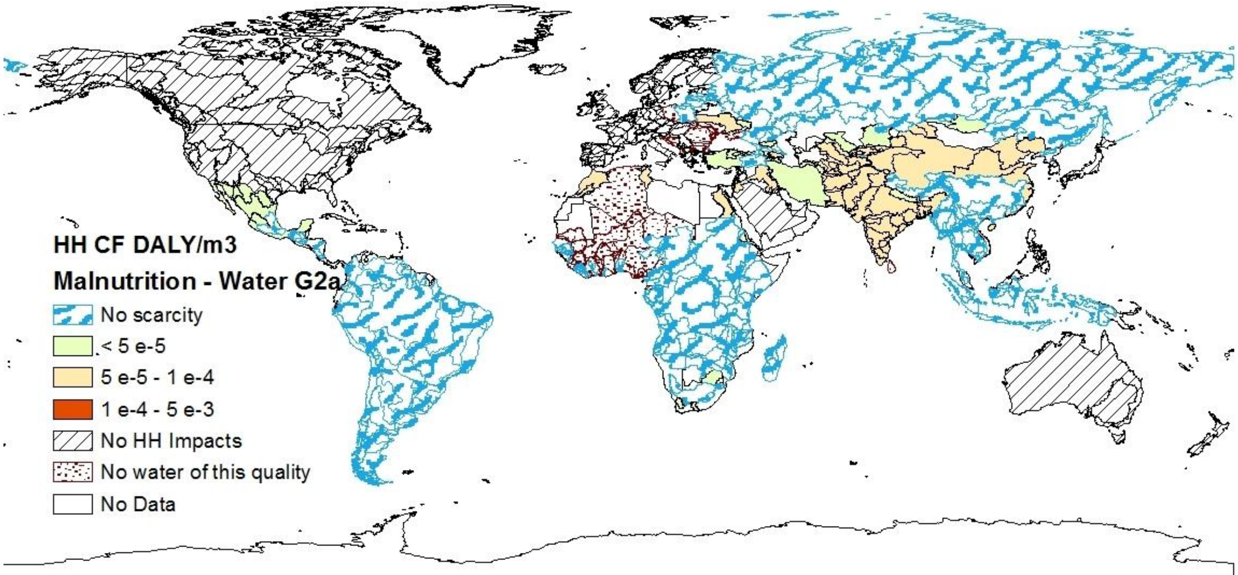
Results for human health characterization factors with the distribution hypothesis (including domestic users) are illustrated for water categories S1, S2a, S2b, G1, G2a and rain in Figure S5.


HH CF DALY/m3
Malnutrition - Water S1
No scarcity
< 5 e-5
5 e-5 - 1 e-4
1 e-4 - 5 e-3
No HH Impacts
No water of this quality
No Data


HH CF DALY/m3
Malnutrition - Water S2a
No scarcity
< 5 e-5
5 e-5 - 1 e-4
1 e-4 - 5 e-3
No HH Impacts
No water of this quality
No Data


HH CF DALY/m3
Malnutrition - Water S2b
No scarcity
< 5 e-5
5 e-5 - 1 e-4
1 e-4 - 5 e-3
No HH Impacts
No water of this quality
No Data


HH CF DALY/m3
Malnutrition - Water G1
No scarcity
< 5 e-5
5 e-5 - 1 e-4
1 e-4 - 5 e-3
No HH Impacts
No water of this quality
No Data



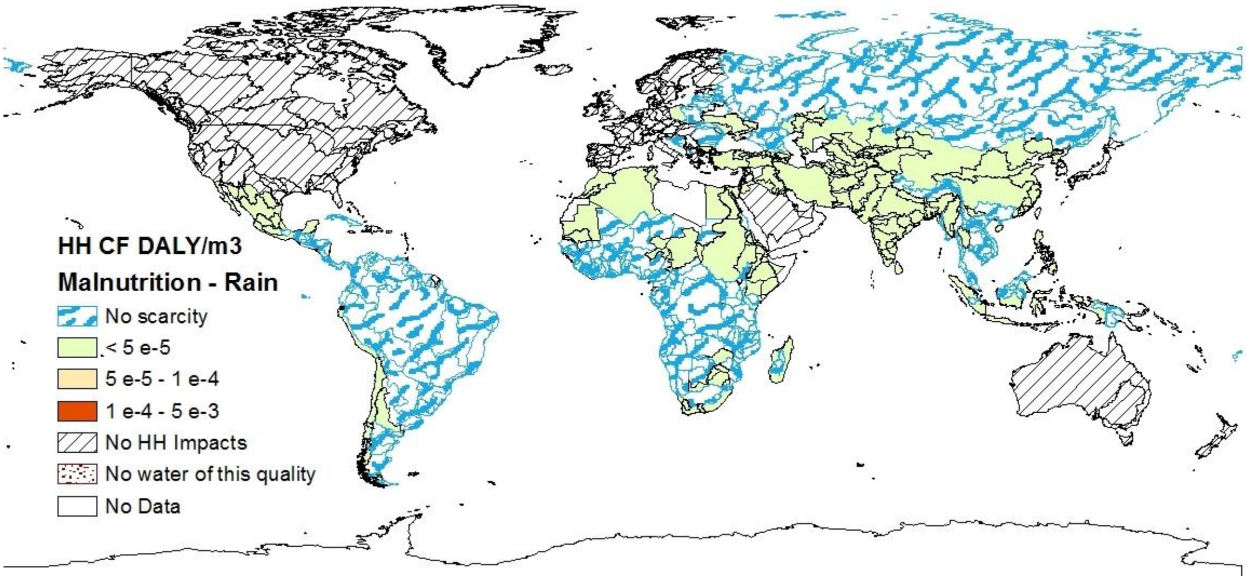


Figure S5: Human health impacts – marginal hypothesis - maps for water uses S1, S2a, S2b, G1, G2a and rain

8 Illustrative example

In the pulp and paper example, effluent data are used to assess water quality with the water category calculator provided by Boulay et al.[13] The effluent category is the lowest water category for which all parameter limits are respected (here, 2b). Table S4 shows the classification of the effluent with the available data.

Table S4: Effluent data and water classification of recycled cardboard production – O: Threshold respected, X: Threshold exceeded (Source: Cascades, represents industry averages)

Parameters	Units	Value	1	2a	2b	2c	2d	3	4	5
RESULT			X	X	O	X	X	O	O	O
General parameters										
Fecal coliforms	UFC/100ml									
Microcystin-LR	mg/l									
True color	Color unit									
Suspended Solids	mg/l	9	O	O	O	O	O	O	O	O
Total Dissolved Solids)	mg/l									
Biochemical Oxygen Demand (5 days)	mgO2/l	3	O	O	O	O	O	O	O	O
Total Nitrogen	mg N/L									
Hardness	mg CaCO3/l	412	O	O	O	O	O	O	O	O
pH	pH unit	7.3	O	O	O	O	O	O	O	O
Sodium Adsorption Ratio	meq/l									
Oil and grease (gravimetric method - exctraction with hexane)	mg/l	1.4	O	O	O	O	O	O	O	O
Inorganics										
Hydrogen sulfide	mg/l									
Total residual chlorine	mg/l									
Chloramines	mg/l									
Chlorine dioxyde	mg/l									
Aluminium	mg/l	0.03	O	O	O	O	O	O	O	O
Ammonia	mgN/l	1	X	X	O	X	X	O	O	O
Antimony	mg/l									
Arsenic	mg/l	0.001	O	O	O	O	O	O	O	O
Barium	mg/l									
Beryllium	mg/l									
Bicarbonate	mg/l									
Boron	mg/l	0.023	O	O	O	O	O	O	O	O
Cadmium	mg/l									
CaxSO4	(mg/l)2									
Chloride	mg/l									
Chromium (total)	mg/l	0.002	X	O	O	O	X	O	O	O
Copper	mg/l	0.0026	O	O	O	O	O	O	O	O
Cyanide	mg/l									
Fluoride	mg/l									
Iron	mg/l	0.31	O	O	O	O	O	O	O	O
Lead	mg/l	0.0014	X	O	O	O	X	O	O	O
Manganese	mg/l	0.15	O	O	O	O	O	O	O	O
MgxSiO2	0									
Silica	mg/l									
Mercury	mg/l	0.0001	O	O	O	O	O	O	O	O
Molybdenum	mg/l	0.0006	O	O	O	O	O	O	O	O
Nickel	mg/l	0.0057	O	O	O	O	O	O	O	O
Nitrates	mgN/l	5.9	O	O	O	O	O	O	O	O
Nitrites	mg/l									
Chlorides/nitrites	mgCl-/mgN-									
Phosphorus (total)	mgP/l									
Sulfur	mg/l									
Selenium	mg/l	0.001	O	O	O	O	O	O	O	O
Sodium	mg/l	9.7	O	O	O	O	O	O	O	O
Sulfate	mg/l									
Uranium	mg/l									
Vanadium	mg/l	0.0014	O	O	O	O	O	O	O	O
Zinc	mg/l	0.0855	X	O	O	O	X	O	O	O

9 Tailored CF

It can be useful to develop a tailored characterization factor (CF) if, for example, more specific data are available or if the water category results are too restrictive. Water classification can be restrictive and underestimate the quality and functionalities as presented for the Amazon Basin in Boulay *et al*.[13] However, case-specific CFs can always be set out by generating a specific set of functionality ($F_{i,j}$) parameters. These $F_{i,j}$ can be obtained based on quality data for a site-specific type of water used (or released) and the threshold values for each user.[13] This will result in 10 values for $F_{i,j}$, either 0 or 1. With this new set of $F_{i,j}$, a specific CF can be generated using intermediary model parameters (see Excel SI). The users affected (U_j) for a specific region can be found in the Supplementary Information Excel spreadsheet or modified according to any specific values on hand. In the "Spec CF" tab of this document, all values can be input (scarcity, GNI, U_j and $F_{i,j}$), either from the data provided or from new data. The spreadsheet uses the different model equations to calculate the resulting CF. These values can then be used with the inventory data to calculate impacts on human health.

10 Data availability

All parameters in the model are regionalized, except for the effect factor, which is a global parameter. Some parameters are relevant at the national level (e.g. adaptation capacity), whereas others are relevant at the watershed level (e.g. stress). However, data are also available at different scales. Even economic data could be relevant at a smaller scale for countries like China and India. Table S5 summarizes the data that was available and used at each scale in the model as well as the optimal scale for a specific parameter to produce the results at the scale used in this article.

A regionalization scale was generated using GIS software (ArcGIS) by overlapping the main watersheds of the world with the countries' borders. This resulted in 808 cells covering the entire world – 189 of which corresponded to regions in countries where no major watersheds are present. Maps of the world's main watersheds are available in PDF[14] and GIS[29] formats. Results were also calculated at the national level for cases in which no information on the specific location of the water withdrawal and release was available. A few regions could not be considered because of data gaps. All intermediary parameters are available in the SI, as well as midpoint WSI and endpoint CF for human health.

Table S5: Data availability and use in the model

Data	Smallest scale available	Scale used in the model	Optimal scale for cell result
Water consumption (CU, f_g)	0.5°x0.5° - from WaterGap model	Cell*	Cell*
Water availability (Q90, GWR)	0.5°x0.5° - from WaterGap model	Cell*	Cell*
Water quality data (Pi)	Various **	As available	Cell*
Socio-economic data (GNI, AC)	Country	Country	Country or smaller socio-political territory
General off-stream user distribution (agriculture, domestic industry and cooling)	0.5°x0.5° - from WaterGap model	Cell*	Cell*
Specific domestic user distribution (Dom 1, 2 and 3)	Country	Country	Cell*
In-stream user distribution (Hydropower, aquaculture)	Country	Country	Cell*

**Cell refers to the scale presented in this paper, which is the result of an intersection between country and main watershed – 808 cells.*

*** From GEMStat—See the Excel document for details.*

11 References

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ANNEXE 2 – ARTICLE 3 – INFORMATION SUPPLÉMENTAIRE

Cette annexe est partie intégrante du chapitre 5 et est soumise pour publication sous la forme d'information supplémentaire à l'article 3.

1- Introduction

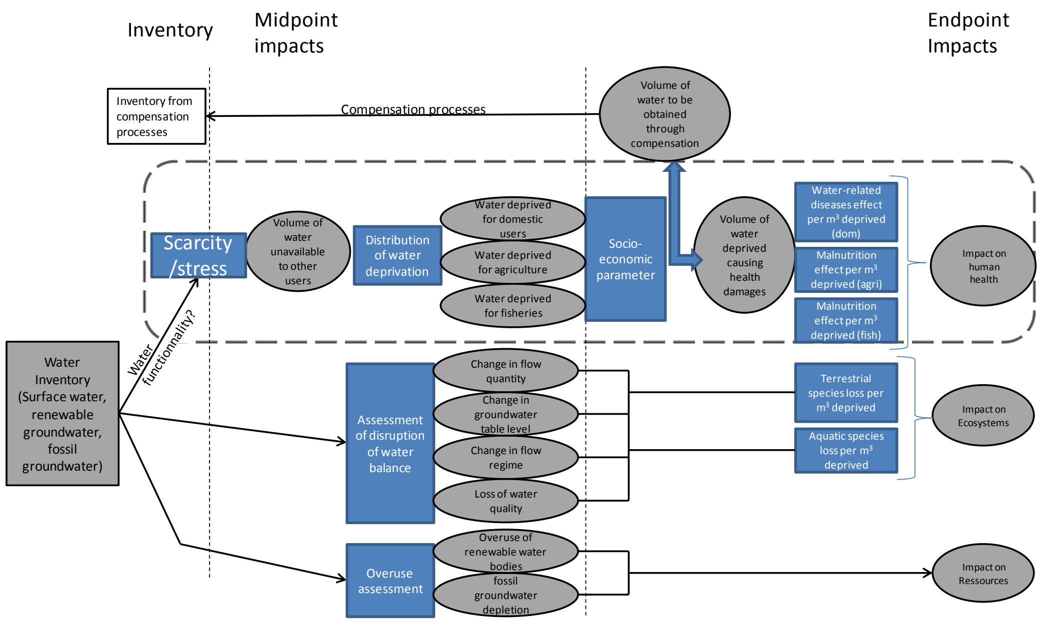


Fig.S1: General water use impacts framework (adapted from Kounina et al). The dotted rectangle highlights the impact pathway for which methods are compared in this paper.

Table S1: Summary of methods name and characteristics

Impact assessed	***Reference***	***Name***	***Details***
Scarcity	Frischknecht, 2008	M-SwissSc	Midpoint, scarcity, withdrawal-to-availability, power function
	Pfister, 2009	M- PfisterSc	Midpoint, scarcity, withdrawal-to-availability, logistic function
	Hoekstra, 2012	M-BWSSc	Midpoint, scarcity, consumption-to-availability, direct function

	Boulay, 2011	M-BoulaySc	Midpoint, scarcity, consumption-to-availability, logistic function
Availability	Boulay, 2011	M-BoulayAv	Midpoint, availability, consumption-to-availability (quality specific), logistic function
	Veolia, 2010	M-WIIXAv	Midpoint, availability, withdrawal-to-availability and distance to target for pollution
Human Health	Pfister, 2009	E-Pfister	As published, agricultural deprivation
	Motoshita, 2010a	E-Motoshita_dom	Effect factor as published, domestic deprivation, then combined with M-PfisterSc and distribution factor (DAU)
	Motoshita, 2010b	E-Motoshita_agri	Adapted from presentation, agricultural deprivation including trade effect , then combined with M-PfisterSc and distribution factor (DAU)
	Boulay, 2011	E-Motoshita_agri (no TE)	Adapted from presentation, agricultural deprivation excluding trade effect , then combined with M-PfisterSc and distribution factor (DAU)
		E-Boulay_marg	Simplified from publication (no quality), considers agriculture as off-stream user deprived (100%) and aquaculture as in-stream user deprived.
	Boulay, 2011	E-Boulay_distri	Simplified from publication (no quality), considers off-stream users to be deprived proportionally to their use (agriculture and domestic are included) and aquaculture as in-stream user deprived.
		E-Boulay_agri	Simplified from publication (no quality), represents a partial factor from E-Boulay_distri for comparison purposes, only for agricultural users.
		E-Boulay_dom	Simplified from publication (no quality), represents a partial factor from E-Boulay_distri for comparison purposes, only for domestic users.
		E-Boulay_marg_Q	As published, considers agriculture as off-stream user deprived (100%) and aquaculture as in-stream user deprived.
		E-Boulay_distri_Q	As published, considers off-stream users to be deprived proportionally to their use (agriculture and domestic are included) and aquaculture as in-stream user deprived.

		E-Boulay_agri_Q	Represents a partial factor from E-Boulay_distri for comparison purposes, only for agricultural users.
		E-Boulay_dom_Q	Represents a partial factor from E-Boulay_distri for comparison purposes, only for quality users.

2- Scarcity indicators

Fig.S2 plots the normalized scarcity indicators from all methods against the WTA index. The underlying data used to calculate this latter (WaterGap), differs from the ones used in M-SwissSc, which explains the observed inconsistency of the data series. Please note that all values equal zero (60% of values for M-Boulay-Sc), cannot be shown on the graph. The average values described in section 2.3 is also shown.

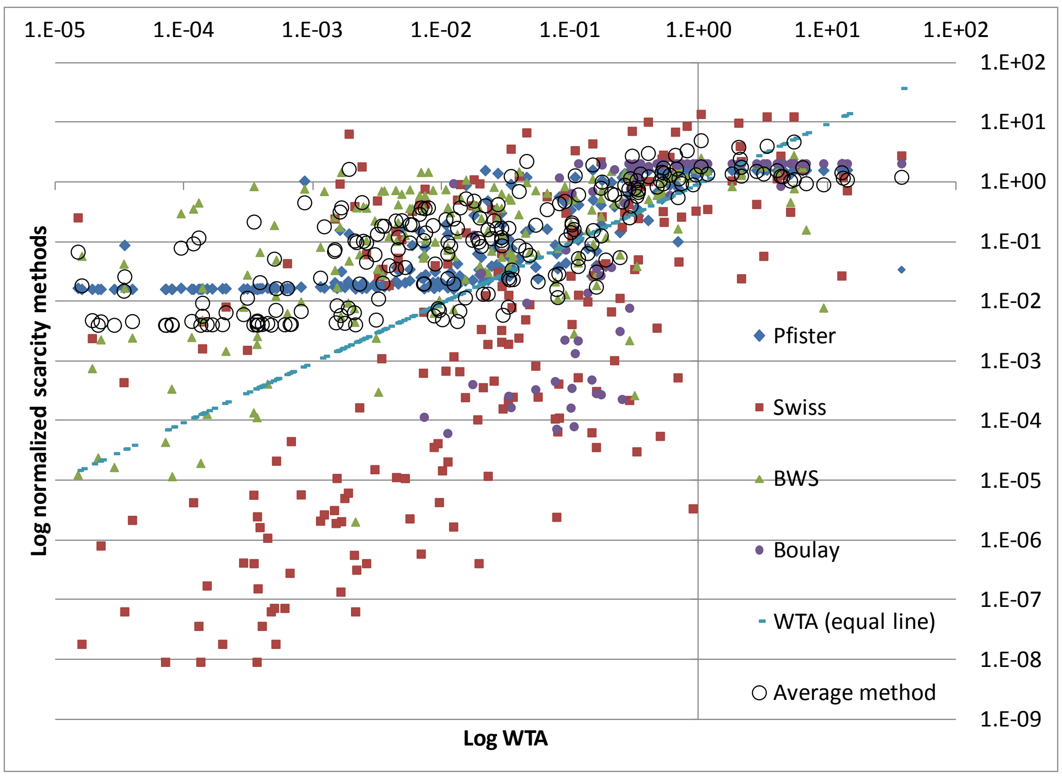
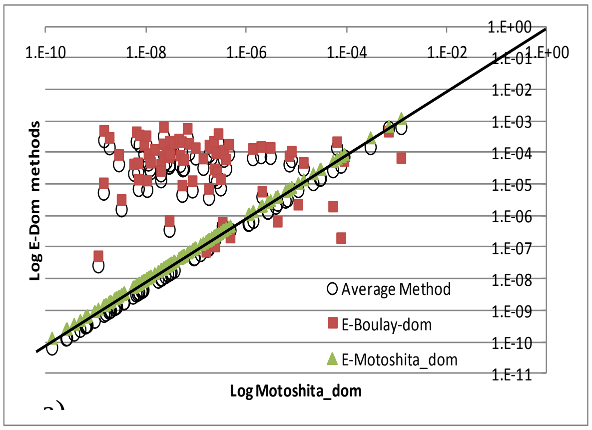


Fig.S2: Log graph of normalized scarcity methods against WTA for the main watersheds showing spread and differences among methods.

3- Human health: Domestic user deprivation



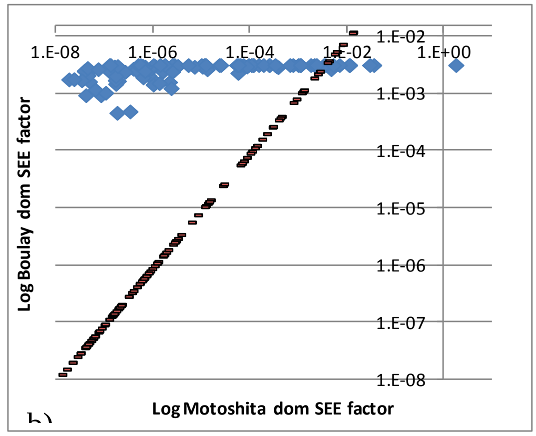


Fig.S3: Comparison of human health model outcomes from domestic water deprivation impact pathways using Boulay and Motoshita models. A) CF and b) socio-economic and effect factor (SEE), which excludes scarcity and distribution of affected users.

4- Human health: Agricultural user deprivation

While the consistency is higher between E-Motoshita_agri without trade effect and the two other models then it is with trade effect included, the mean difference between models decreases in comparison with E-Pfister but increases when comparing with E-Boulay_agri. This is because E-Boulay_agri shows higher results in general, and the addition of the trade effect increases the results as well in E-Motoshita_agri.

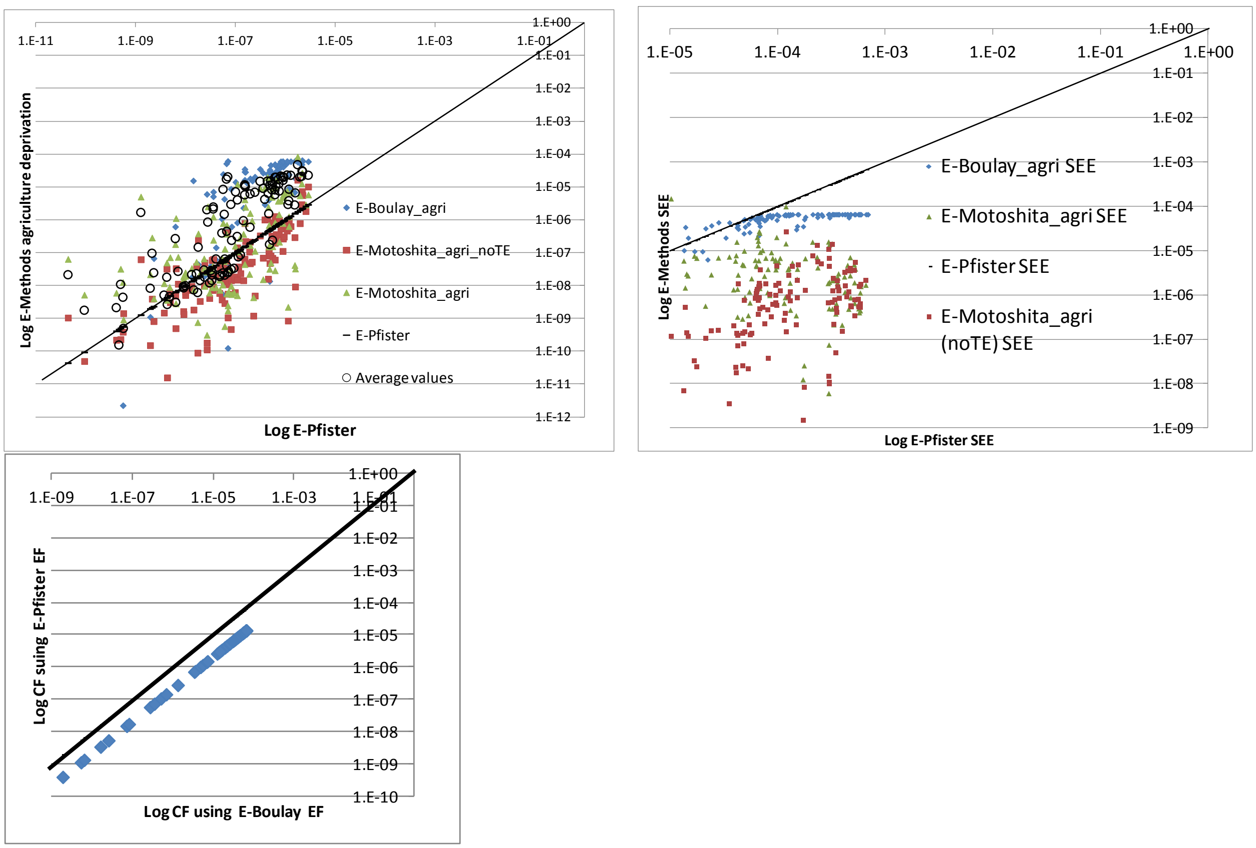


Fig.S4: Comparison of agriculture water deprivation impacts on human health. a) complete CF, b) Socio-economic and effect factors and c) effect factors only.

5- Inventory-related choices

Temporal variations

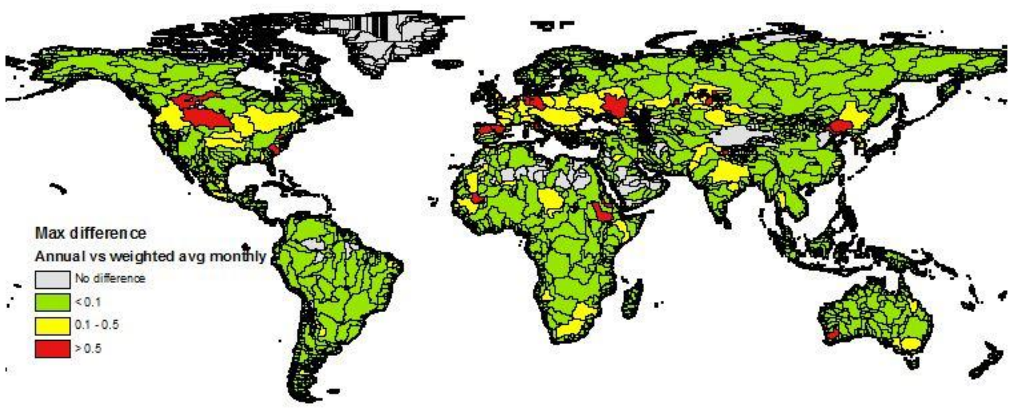


Fig.S5: Difference between annual scarcity indicators calculated from annual data vs. from a withdrawal-based weighted average of monthly data. Results are obtained with M-PfisterSc which scarcity indexes range from 0.01 to .1

Water Source

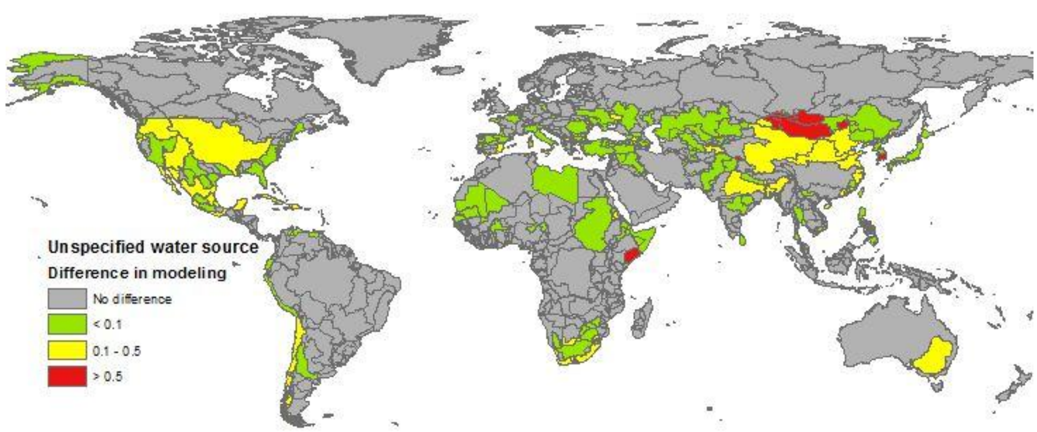
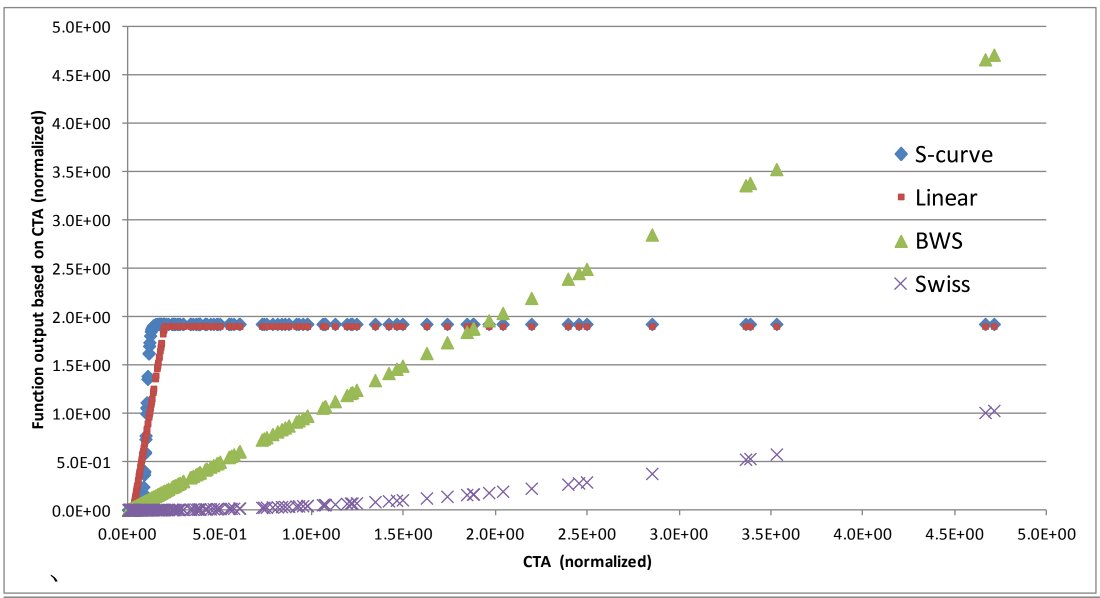


Fig.S6: Absolute difference in scarcity results between general scarcity indicators calculated using all water use and availability and weighted-average of surface and groundwater scarcities, based on intensity of groundwater withdrawals. Results are obtained using M-Boulay-Sc, which values range betwee 0 – 1.

Quality aspect

Out of the 600 regions of the world for which data were available (based on Boulay et al(Boulay et al. 2011a)), scarcity indicators are higher when consuming water of good quality then of unspecified quality in 42% of the cases. No difference is observed for the rest of the cases (58%), representing regions where general scarcity is already maximal (value of 1). Consuming water of poor quality results in higher stress in about 21% of the cases, corresponding to region where water quality is very poor. No difference is observed in 79% of cases, i.e. in regions where the average water quality is poor. Degrading good quality into very poor quality, will result in higher impacts than consuming general water (quality non-specified) in 39% of the cases, in countries with no general water scarcity, but with good quality water scarcity. These will present impacts when considering water of a certain quality but not when considering all water available.

6- Scarcity



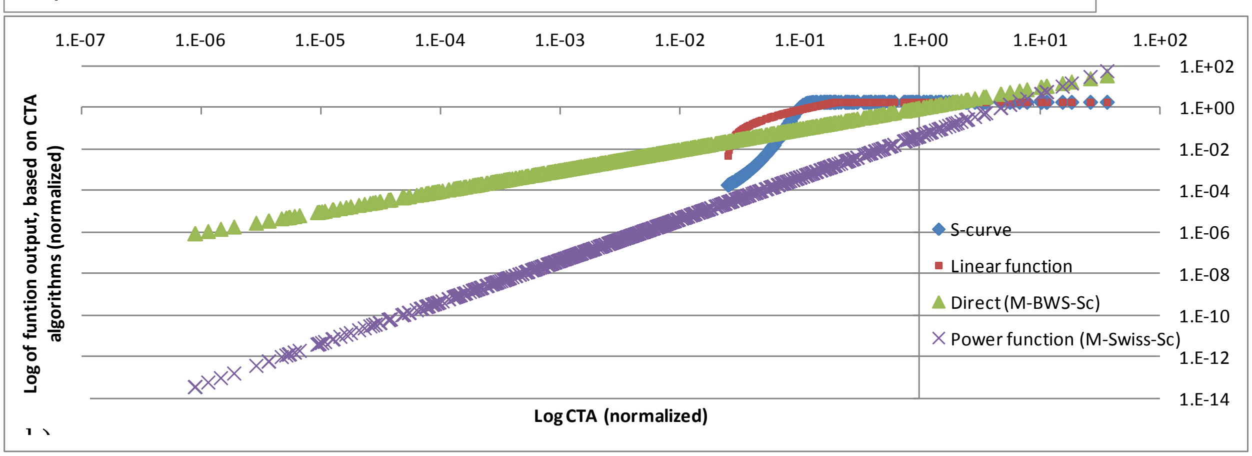


Fig.S7: Different model functions defining scarcity as a function of CTA a) normal scale and b)log scale.

ANNEXE 5 – ARTICLE 4 – INFORMATION SUPPLÉMENTAIRE

Cette annexe est partie intégrante du chapitre 6 et est publiée sous la forme d'information supplémentaire à l'article 4.

1- System Boundaries

The washing machine is excluded based on a cut-off criteria of 2% of total water consumed being attributable to this infrastructure. This is based on a coarse approximation of a 65 kg washing machine, made of steel, consuming 4 L/kg (taken from worldsteel.org). If the machine performs 3 loads per week for 10 years, the total water consumed is 0.17 l/load, less than 2% of the total water consumed (12 l). Moreover, the goal of this case study is to show method applicability and too much uncertainty is associated with the modeling of the machine (country of production, life span, amount of loads, transport distance, etc.) to justify its relevance for the goal of this study. If the machine is suspected to come from a high water scarcity region, then including it could be justified.

2- Regional Sensitivity Analysis

a. *Midpoint : Spain*

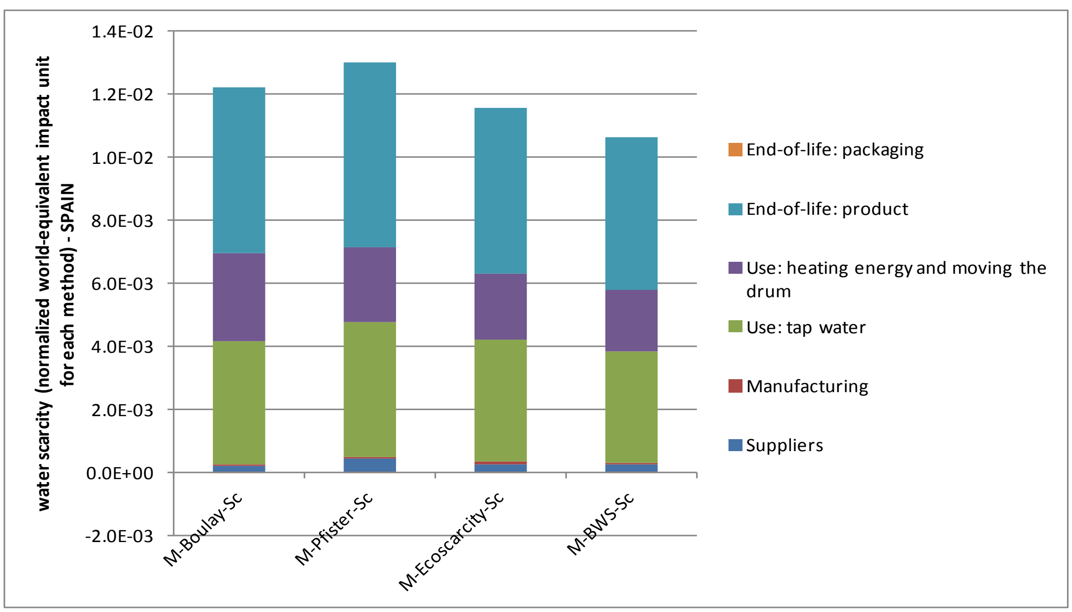


Fig.A: Sensitivity analysis on regional variation: scarcity results for a load of laundry with use and end-of-life in Spain. Results are normalized based on the world-weighted average of each method

b. Endpoint:India

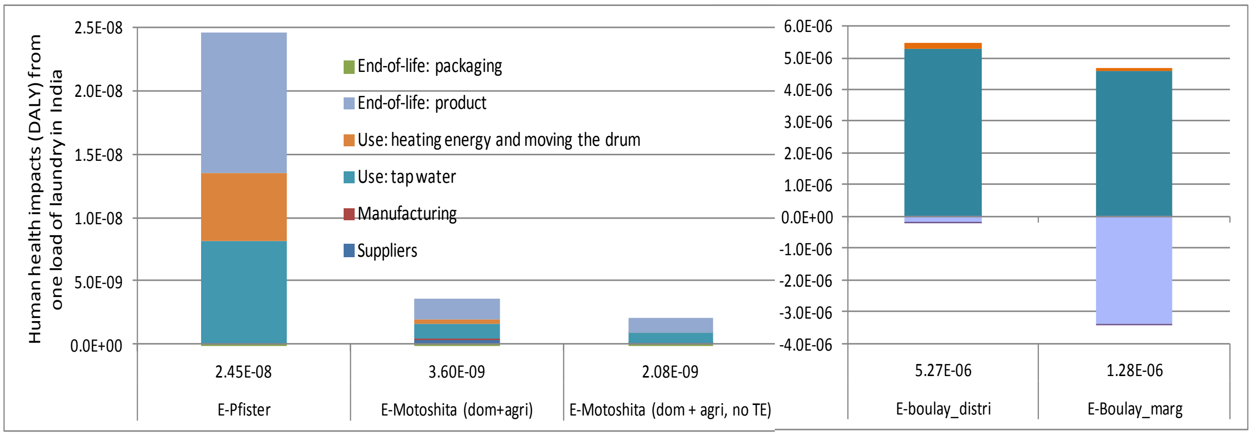


Fig.B: Sensitivity analysis on regional variation: human health impacts for a load of laundry with use and end-of-life in India.

c. __Domestic user deprivation in India__

In figure C, the difference between E-Boulay_distri and E-Boulay_marg refers to the inclusion or not of domestic user deprivation. As expected impacts are higher when domestic users are considered since one m^3 of water deprived from domestic users has more impacts on human health than if it is deprived from agriculture(Boulay et al. 2011b). In the final results, this leads to higher absolute impacts by a factor of about 2.6 for both cases, i.e. considering quality or not.

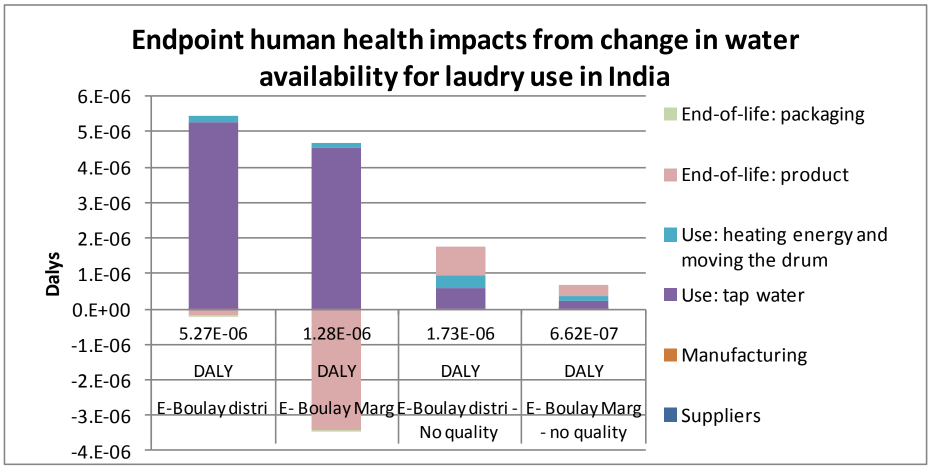


Figure C: Change in results caused by inclusion or not of domestic user deprivation (comparing distri and marginal) and of quality (comparing the original model with the version with no quality)