POLYTECHNIQUE MONTRÉAL
affiliée à l'Université de Montréal

Repenser l'évaluation des impacts du cycle du vie à travers le prisme des services écosystémiques : enjeux, cadre méthodologique et applications.

LAURA DEBARRE
Département de mathématiques et de génie industriel

Thèse présentée en vue de l'obtention du diplôme de *Philosophiæ Doctor*
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Cette thèse intitulée :

Repenser l'évaluation des impacts du cycle du vie à travers le prisme des services écosystémiques : enjeux, cadre méthodologique et applications.

présentée par **Laura DEBARRE** en vue de l'obtention du diplôme
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a été dûment acceptée par le jury d'examen constitué de :

Sophie BERNARD, présidente
Manuele MARGNI, membre et directeur de recherche
Montserrat NUÑEZ, membre
Claudio PETUCCO, membre externe

DÉDICACE

À mes parents, Thierry et Pascale

À ma famille, de sang et de cœur

À papy Gilou et ses histoires

À ceux qui refont le monde, via les mots ou le courage

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RÉSUMÉ

L'Analyse du Cycle de Vie (ACV) est un outil central pour évaluer la durabilité. Cette méthodologie, encadrée par des normes, se compose de quatre étapes. La troisième étape, appelée évaluation des impacts du cycle de vie (EICV), vise à convertir les résultats de l'inventaire (par exemple les émissions de substances toxiques) en impacts environnementaux potentiels. Cette conversion repose sur des facteurs de caractérisation (CF), calculés à partir de modèles de caractérisation qui simulent les chaînes de cause à effet reliant les éléments de l'inventaire à leurs impacts potentiels. Les modèles de caractérisation s'inscrivent dans un cadre conceptuel appelé "cadre EICV", qui identifie et structure de manière cohérente les différents chemins d'impact potentiels.

Ce cadre EICV et les modèles de caractérisation associés sont en constante évolution. Aujourd'hui, ils ne couvrent pas suffisamment les chemins d'impact des activités humaines sur les services rendus par la nature, appelés "services écosystémiques". Ces services jouent pourtant un rôle clé pour la société. Leur omission dans les profils d'impact de l'ACV représente alors un angle mort pour la prise de décision. Pour remédier à cette lacune et proposer des indicateurs d'impact sur les services écosystémiques, il est nécessaire de développer les modèles et les facteurs de caractérisation correspondants. Cependant, ces développements doivent rester cohérents avec les modèles existants, qui représentent des chemins d'impact sur d'autres types d'objets, tels que la santé humaine ou la qualité des écosystèmes. Or, ces derniers modèles intègrent souvent, de manière implicite, des concepts liés aux services écosystémiques tels que les fonctions écosystémiques. Pour développer des modèles de caractérisation sur les services écosystémiques cohérents avec les modèles existants, sans risque de double comptage des impacts, il est alors important d'identifier et d'expliciter ces interactions.

Dans ce contexte, cette thèse par articles propose de consolider l'EICV en repensant son cadre à travers le prisme des services écosystémiques.

Un premier article présente une revue critique de la littérature, croisant deux axes : une revue des concepts fondamentaux liés aux services écosystémiques et une analyse des tentatives d'intégration de ces derniers en EICV. Cette revue critique met en évidence la pertinence et les écueils de la prise en compte des services écosystémiques en EICV. Elle identifie des incohérences et un manque de clarté dans plusieurs propositions d'intégration des services écosystémiques en EICV, et appelle à l'élaboration d'un cadre EICV mieux adapté à l'intégration des services écosystémiques.

En réponse, un deuxième article propose un cadre EICV révisé, s'inspirant du "modèle en cascade" de la communauté des services écosystémiques. Ce nouveau cadre EICV révèle des liens étroits entre les concepts liés aux services écosystémiques et la majorité des modèles de caractérisation existants. Pour illustrer l'applicabilité et l'exhaustivité de la couverture des chemins d'impact potentiels au sein du cadre EICV révisé, l'ensemble des modèles de caractérisation de la méthodologie IMPACT World+ y est positionné.

Sur la base de ce cadre révisé, des modèles et des CF sont développés pour deux chemins d'impact. Dans un premier temps, un travail développe un modèle et ses CF évaluant les impacts de l'élévation du niveau des mers et de l'intensification des événements extrêmes sur les services écosystémiques côtiers. Les CF sont calculés en divisant la perte de potentiel de fourniture de services écosystémiques des écosystèmes côtiers susceptibles d'être inondés entre 2015 et 2100 par les émissions de gaz à effet de serre cumulées sur la même période. Dépendamment du scénario évalué, les valeurs des CF s'étendent de 16.7 à 66.1 $\$/\mathrm{tCO2}_{eq}$ comparables au coût social du carbone ($185\$/\mathrm{tCO}_{2\mathrm{eq}}$), ce qui valide leur ordre de grandeur. Des limites sont néanmoins identifiées, comme la difficulté à modéliser la capacité d'adaptation des écosystèmes aux événements extrêmes, ou à quantifier la demande en services écosystémiques à l'échelle mondiale.

Le second chemin d'impact du cadre EICV révisé porte sur les impacts potentiels sur la santé humaine résultant de l'utilisation et de la privation d'eau douce, exprimés en $\mathrm{DALY/m}^3$ consommé. Deux articles y sont consacrés. Le troisième article de cette thèse consolide un modèle de caractérisation développé en 2011, en révisant

ses facteurs de sort, d'exposition et d'effet pour proposer un nouvel ensemble de CF. Néanmoins, le travail souligne des limites importantes du facteur d'effet évaluant les impacts sur la santé humaine induits par la privation d'eau qui est une valeur globale ne permettant pas de rendre compte des disparités d'accès à l'eau au sein d'une population.

Pour pallier cette limite, un quatrième article propose une nouvelle méthode de calcul du facteur d'effet, fondée sur l'idée que les conséquences sur la santé induites par la privation d'1 m^3 d'eau varient selon la quantité d'eau quotidiennement utilisée. En distinguant différentes "classes d'utilisateurs", de nouveaux facteurs d'effet régionalisés à l'échelle nationale sont développés, puis intégrés aux autres composantes des CF mises à jour. Une démonstration empirique de la chaîne de cause à effet étudiée reste toutefois nécessaire pour consolider ce modèle.

L'utilisation du cadre EICV au travers des deux chemins d'impact étudiés démontre sa pertinence et son applicabilité. Néanmoins, son application pour le développement du modèle lié à l'élévation du niveau des mers souligne une limite de son opérationnalisation : la complexité de la modélisation de la demande en services écosystémiques.

Ce travail ne vise pas à proposer des lignes directrices strictes, mais à structurer les développements méthodologiques de manière cohérente et exhaustive, afin de produire des indicateurs pertinents pour la prise de décision. Ce cadre conceptuel est flexible et sa mise en oeuvre peut varier selon la chaîne de cause à effet étudiée. Il est néanmoins important de noter que l'identification des impacts jugés pertinents est un exercice subjectif ; des systèmes de valeur différents peuvent mener à des cadres différents.

Le cadre EICV proposé dans cette thèse offre aux modélisateurs un outil assurant la cohérence et la complétude de leurs modèles. Les CF développés dans cette thèse à partir de ce cadre sont quant à eux destinés aux praticiens ACV, leur permettant de fournir de l'information plus complète et plus robuste pour la prise de décision. L'ensemble de ces travaux, plus généralement, contribue à consolider l'EICV.

ABSTRACT

Life Cycle Assessment (LCA) is a key tool for sustainable management. This standardized methodology is structured into four main steps. The third step, called Life Cycle Impact Assessment (LCIA), aims to translate inventory data (e.g., emissions of toxic substances) into potential environmental impacts. This translation relies on characterization factors, which are derived from characterization models representing the involved environmental mechanisms. Characterization models are structured within an "LCIA framework", which is a coherent and comprehensive conceptual structure that identifies and organizes pathways of potential impact and relevant indicators for decision-making.

Over time, the LCIA framework and its associated characterization models have evolved. However, the current framework still insufficiently addresses the pathways by which human activities affect the benefits provided by nature, known as ecosystem services. Such services are essential to human well-being, and their omission from LCA impact profiles represents a significant blind spot for decision-making.

Bridging this gap requires the development of appropriate impact models and characterization factors assessing pathways of impacts on ecosystem services. Yet, such developments must remain consistent with existing LCIA models, which typically address other impact categories such as human health or ecosystem quality. Many of these existing models implicitly integrate concepts closely related to ecosystem services, such as ecosystem functions. Therefore, to ensure conceptual coherence and avoid double counting, it is important to identify and make explicit the interactions between ecosystem services concepts and traditional LCIA pathways.

In this context, the present article-based thesis proposes a revision of the LCIA framework through the lens of ecosystem services. Building on the revised framework, models and characterization factors are developed for two specific impact pathways.

The first article presents a critical literature review, combining two research works:

a review of the fundamental concepts relating to ecosystem services and an analysis of attempts to integrate ecosystem services into LCIA. The review highlights both the relevance and the challenges of incorporating ecosystem services into LCIA. It identifies inconsistencies and a lack of conceptual clarity in several existing proposals and advocates for a more suitable LCIA framework for ecosystem services integration.

In response, the second article introduces a revised LCIA framework built on the "cascade model" developed by the ecosystem services research community. This new framework reveals strong conceptual linkages between ES-related concepts and most existing LCIA models. To illustrate its comprehensiveness and applicability, all characterization models from the IMPACT World+ methodology are mapped onto this revised framework.

On the basis of this new structure, models and CFs are being developed for two impact pathways. The third study of this thesis develops a model and associated characterization factors to assess the impacts of sea-level rise and the intensification of extreme events on coastal ES. The characterization factors are calculated as the ratio between (1) the loss of ecosystem service supply potential in coastal zones projected to be flooded between 2015 and 2100, and (2) the cumulative greenhouse gas emissions over the same period. Depending on the scenario, characterization factors values range from 16.7 to 66.1\$/t$CO_{2,eq}$, aligning with the social cost of carbon (estimated at 185\$/t$CO_{2,eq}$), thus validating their order of magnitude. Nevertheless, limitations are acknowledged, including the challenges of modeling the capacity of coastal ecosystems to adapt to flooding and of quantifying global ecosystem services demand.

A second impact pathway of the revised framework explored in this thesis addresses the potential human health impacts from freshwater use and deprivation, expressed in DALYs/m^3 consumed. Two articles are dedicated to this topic. The fourth article of this thesis revisits and consolidates a characterization model developed in 2011, updating its fate, exposure, and effect factors to produce a new set of characterization factors. However, the work reveals a significant limitation of the effect factor, which estimates health impacts due to water deprivation. Being global in scope, the effect

factor does not account for disparities in water access within populations.

To address this shortcoming, the fifth article proposes a new effect factor calculation method, based on the premise that health consequences of losing 1 m^3 of water depend on the user's typical daily consumption. By distinguishing user classes, new country-scale effect factors are developed and integrated with the updated characterization factors components, yielding a consolidated set of characterization factors. Despite these improvements, the empirical demonstration of the causal chain from water deprivation to health outcomes remains a key research need.

The application of the revised LCIA framework across these two impact pathways demonstrates both its relevance and practical utility. However, the case study on sea-level rise also highlights a key limitation in operationalizing the framework: the complexity of modeling demand for ecosystem services.

The proposed framework is designed to be flexible and adaptable to different cause-effect chains. Rather than offering strict guidelines, this work aims to provide a coherent and comprehensive structure for methodological developments, ultimately enabling the creation of meaningful indicators for decision support. However, it is important to emphasize that the selection of relevant impacts is inherently subjective; differing value systems can lead to the development of alternative frameworks.

The revised LCIA framework proposed in this thesis serves as a conceptual tool for LCA modelers, supporting consistency and completeness in characterization models development. The characterization factors derived from it are intended for LCA practitioners, helping to generate more comprehensive and robust information for decision-making. More broadly, this work contributes to advancing and consolidating the field of LCIA.

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LISTE DES SIGLES ET ABRÉVIATIONS

ACV	Analyse du Cycle de Vie
AdP	Aire de Protection
FC	Facteur de Caractérisation
EF	Facteur d'effet
EICV	Évaluation des Impacts du Cycle de Vie
EQ	Qualité des écosystèmes
FF	Facteur de sort
GES	Gaz à Effet de Serre
SE	Services Écosystémiques
XF	Facteur d'exposition

AoP	Area of Protection
CF	Characterisation Factor
EF	Effect Factor
EFG	Ecosystem Functional Groups
EQ	Ecosystem Quality
ES	Ecosystem Services
ESL	Extreme Sea Level
FF	Fate Factor
GHG	Greenhouse Gas
HH	Human Health
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
XF	Exposure Factor

LECZ Low Elevation Coastal Zone

Introduction générale

Les rapports des experts nous mettant en garde quant à l'accélération des crises environnementales se succèdent (IPBES, 2019 ; IPCC, 2019). Face à leurs cris d'alerte, l'inaction n'est pas une option ; il faut agir. Mais par où commencer, à quoi s'attaquer, et comment changer nos habitudes ? L'intuition est un outil insuffisant, voire trompeur, quand il s'agit de durabilité. Nous avons besoin de mieux pour éclairer nos décisions.

L'Analyse du Cycle de vie (ACV) se présente alors comme un candidat idéal pour accompagner la prise de décision durable en évaluant les impacts environnementaux de tous types de systèmes (ISO, 2006). Ces impacts sont néanmoins nombreux et divers, et notre compréhension de ces derniers progresse au cours du temps. Mécaniquement, la méthodologie ACV et son cadre conceptuel d'évaluation des impacts évoluent avec lui.

Pour faciliter la prise de décision, les impacts environnementaux considérés en ACV sont agrégés au sein d'Aires de Protection, qui représentent les entités importantes pour la société, telles que la santé humaine ou la qualité des écosystèmes (Schaubroeck & Rugani, 2017). Néanmoins, cette liste d'objets importants n'est aujourd'hui pas complète ; certaines entités cruciales pour les sociétés ne sont pas encore représentées dans les profils d'impact offerts par l'ACV.

C'est le cas des supports de valeurs instrumentales, tels que les services rendus par la nature, dits "services écosystémiques". Pourtant, ces derniers nous sont précieux ; leur valeur économique dépasse annuellement le produit intérieur brut mondial (Costanza et al., 2014). Malheureusement, face aux pressions anthropiques, leur érosion accélère.

Pour éviter de prendre des décisions néfastes pour les services écosystémiques, l'ACV doit fournir des indicateurs traduisant l'impact des activités humaines sur la capacité des écosystèmes à fournir des services. Mais ces indicateurs, et leurs modèles sous-jacents, doivent être cohérents avec les indicateurs et les modèles actuels de l'ACV.

Cette thèse propose ainsi de repenser le cadre de l'évaluation des impacts en ACV afin de soutenir l'intégration cohérente d'indicateurs traduisant les impacts de nos activités sur la capacité des écosystèmes à fournir des services. Ces indicateurs permettront de compléter les profils d'impact de l'ACV, évitant les angles morts pour la prise de décision. Dans le contexte de cette thèse, ce nouveau cadre d'évaluation des impacts est mis en application pour le développement et l'amélioration de différents modèles d'impact.

CHAPITRE 1 L'ANALYSE DU CYCLE DE IE

Ce chapitre introduit les bases de l'analyse du cycle de vie. Il décrit l'outil (1.1) et son cadre méthodologique (1.2).

1.1 Introduction à l'analyse du cycle de vie

L'analyse du cycle de vie (ACV), également appelée « écobilan », est un outil central de la gestion environnementale (Hellweg et al., 2023). Elle repose sur une norme internationale (International Organization for Standardization, 2006), et permet de modéliser et d'évaluer les impacts environnementaux potentiels d'un système tout au long de son cycle de vie.

L'ACV évalue les impacts environnementaux d'un système en les rapportant à l'échelle d'une unité de fonction donnée, appelée unité fonctionnelle. (Jolliet et al., 2017). Le système étudié peut être un produit, un service, un processus, une organisation, une politique publique, ou encore un territoire (Hellweg et al., 2023 ; Jolliet et al., 2017 ; Loiseau et al., 2018 ; Martínez-Blanco, Inaba & Finkbeiner, 2016).

Un outil prenant en compte l'ensemble des étapes du cycle de vie. Lors d'une ACV, l'évaluation couvre l'ensemble des étapes du cycle de vie du système étudié : extraction des matières premières, processus de fabrication, distribution, utilisation, collecte et gestion de fin de vie (fig. 1.1). Cette approche holistique offre une vision globale des impacts environnementaux d'un produit, évitant les biais dans l'interprétation du résultat. Une vision partielle, focalisée sur une étape du cycle de vie, risquerait d'induire un "déplacement d'impact" vers une autre étape. Par exemple, une solution présentant de faibles impacts en phase d'utilisation pourrait impliquer des impacts plus importants en phase de fabrication, annulant la performance environnementale sur l'ensemble du cycle de vie.

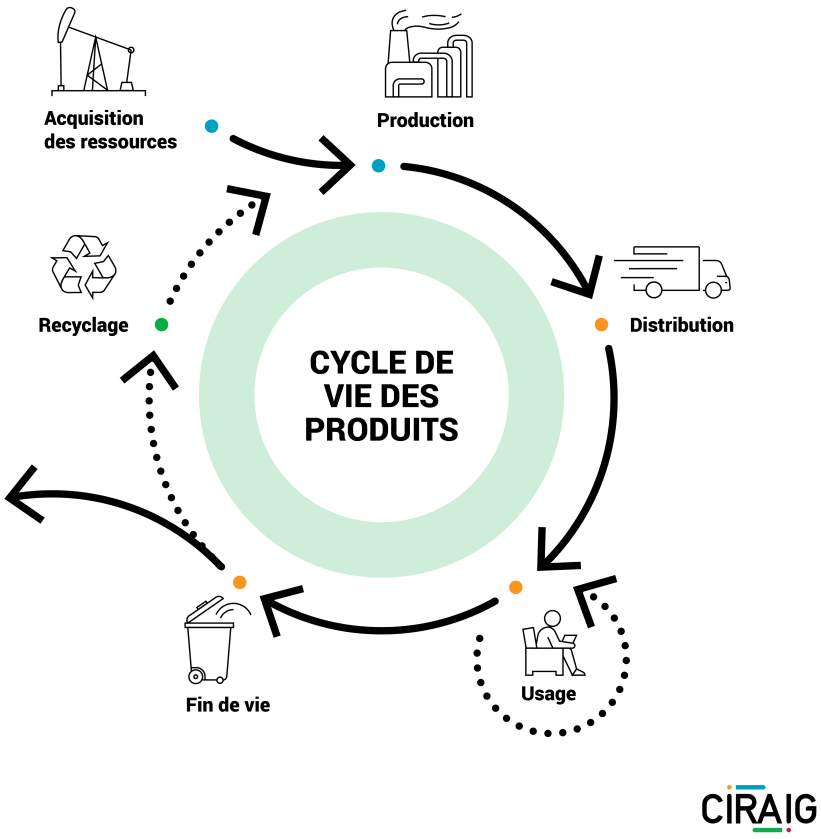


FIGURE 1.1 Schéma représentatif de l'analyse du cycle de vie

Un profil d'impact multicritère couvrant différents indicateurs d'impact environnemental. L'ACV permet d'éclairer la prise de décision en s'appuyant sur des indicateurs traduisant divers types d'impacts environnementaux. Son résultat prend la forme d'un profil environnemental multicritère, présentant des scores d'impact pour différentes catégories telles que le changement climatique, l'utilisation des sols, l'acidication, etc. (Jolliet et al., 2017). Cette approche globale contribue à éviter les déplacements d'impacts d'une catégorie à une autre.

1.2 Cadre méthodologique de l'analyse du cycle de vie

La réalisation d'analyses du cycle de vie est encadrée par la série de normes ISO14040. Les normes ISO 14040 et ISO 14044 définissent les principes et les lignes directrices générales de l'ACV (International Organization for Standardization, 2006 ; ISO, 2006). L'ACV est ainsi structurée en quatre étapes à suivre de manière itérative : la définition du champ de l'étude, l'inventaire, l'évaluation des impacts, et l'interprétation.

— **Définition et champ de l'étude**. Cette première étape vise à poser les bases de l'analyse en définissant clairement ses objectifs, les limites du système étudié, l'application envisagée et l'audience visée par l'étude. C'est à cette étape qu'est définie l'unité fonctionnelle du système, qui quantifie la fonction fournie par celui-ci. On y décrit également l'arbre des procédés et les frontières du système, qui offrent une représentation visuelle des processus pris en compte dans l'étude. Divers choix méthodologiques sont également introduits tels que les approches d'allocation privilégiées pour la gestion des processus multifonctionnels.

— **Inventaire du cycle de vie**. Cette étape consiste à identifier et quantifier l'ensemble des entrants et des sortants du système étudié, mis à l'échelle de l'unité fonctionnelle. Ces flux, appelés flux élémentaires, représentent les échanges opérant entre la technosphere (comprenant l'ensemble des activités humaines) et l'écosphere (qui représente l'environnement naturel).

— **Évaluation des impacts du cycle de vie**. Cette troisième étape permet de convertir les flux élémentaires quantifiés lors de l'inventaire du cycle de vie en potentiels impacts environnementaux. Le travail effectué dans le cadre de cette thèse porte principalement sur cette étape. Le chapitre 2 fournit davantage d'informations sur celle-ci.

— **Interprétation**. La dernière étape d'une ACV permet à la fois d'interpréter les résulats obtenus et d'évaluer les incertitudes liées à l'étude. Lors de cette étape, les points chauds du système (i.e. les plus gros contributeurs en termes d'étapes du cycle de vie ou de catégories d'impact) sont identifiés, et des pistes d'amélioration sont formulées. Des analyses de sensibilité, de scénario et des incertitudes permettent d'alimenter la discussion des résultats et des hypothèses ainsi que d'identifier les limites de l'étude.

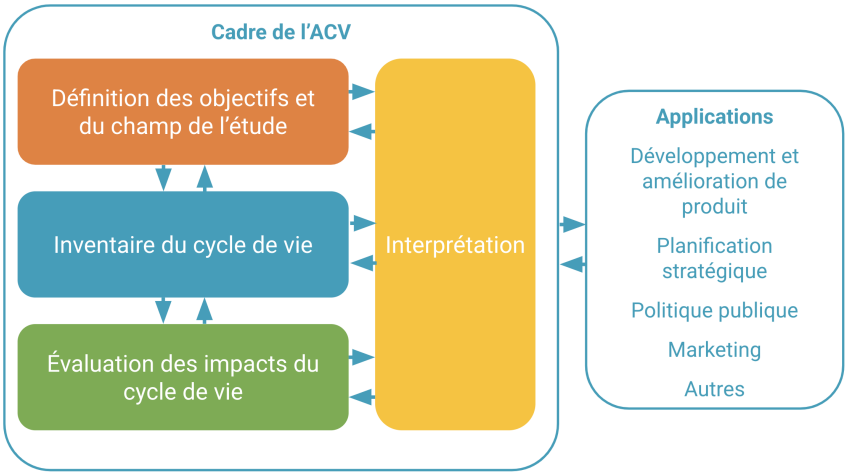


FIGURE 1.2 Cadre méthodologique et étapes de l'ACV d'après la norme ISO14040 (tirée du MOOC EPLFx: Introduction à l'analyse du cycle de vie - de la pensée à la pratique)

À retenir

L'ACV permet de quantifier les impacts environnementaux potentiels d'un système, considérant l'ensemble des étapes de son cycle de vie et un ensemble complet et diversifié d'impacts environnementaux. Elle est encadrée par la série de normes ISO 14000, et structurée selon 4 étapes à réaliser de façon itérative : la définition du champ de l'étude, l'inventaire, l'évaluation des impacts et l'interprétation des résultats.

CHAPITRE 2 ÉVALUATION DES IMPACTS DU CYCLE DE VIE

Ce chapitre présente les fondements de l'évaluation des impacts du cycle de vie. Il débute par une introduction à la terminologie associée (2.1), puis retrace l'évolution des pratiques de modélisation des impacts au fil du temps (2.2). La section 2.3 expose les principes de la caractérisation des impacts, notamment sa structure calculatoire. La section 2.4 dresse un état de l'art des initiatives visant à harmoniser les pratiques de modélisation des impacts en analyse du cycle de vie. L'une de ces initiatives a abouti à la proposition d'un cadre conceptuel d'évaluation des impacts, détaillé et illustré dans la section 2.5. Le travail de cette thèse s'appuie sur celui-ci.

L'évaluation des impacts du cycle de vie (EICV), troisième étape du cadre méthodologique de l'ACV, vise à soutenir l'interprétation des résultats (Udo de Haes et al., 2002). En effet, l'inventaire du cycle de vie contient un très grand nombre de flux élémentaires, exprimés en quantité de substances ou de ressources (e.g. en kg de CO_2 émis ou m^3 d'eau consommé), et dont la pertinence environnementale est largement variable. Un inventaire, bien que exhaustif, fournit ainsi une aide limitée pour la prise de décision. Dans ce contexte, l'EICV vise à traduire l'inventaire du cycle de vie en un profil d'impact contenant un nombre d'indicateurs limité mais permettant une vision complète des enjeux environnementaux (Hauschild, Michael Z and Huijbregts, 2015).

2.1 Terminologie

Pour traduire les flux d'inventaire en impacts environnementaux potentiels, les quantités de flux élémentaires i sont multipliées par un **facteur de caractérisation** (FC) pour chaque **catégorie d'impact** j considérée :

$$Score\,d'impact_{i,j} = Flux\,elementaire_i \cdot Facteur\,de\,caracterisation_{i,j} \qquad (2.1)$$

Les FC sont calculés sur la base de **modèles de caractérisation** qui visent à capturer, de manière simplifiée, l'ensemble des **mécanismes environnementaux** qui relient le flux élémentaire à l'impact étudié. Cette succession de mécanismes environnementaux est appelée **chaîne de cause-à-effet**. Celle-ci dresse les voies menant aux impacts inclus dans la catégorie d'impact.

Les **indicateurs d'impact** représentent un point de la chaîne de cause à effet. Différents FC peuvent traduire des conséquences plus ou moins loin sur la chaîne. On distingue alors les **indicateurs "orientés problème"** (situés quelque part le long de la chaîne de cause-à-effet) des **indicateurs "orientés dommage"** (situés au bout de la chaîne).

Lors de la réalisation d'une ACV, le praticien est amené à choisir une ou préférablement plusieurs **méthodologies EICV** pour traduire l'inventaire de l'ACV en profil d'impacts environnementaux. Ces méthodologies EICV sont composées d'un ensemble cohérent de modèles de caractérisation, et offrent les facteurs de caractérisation nécessaires pour fournir des scores sur différentes catégories d'impact.

Les modèles et facteurs de caractérisation d'une méthodologie EICV sont développés selon un **cadre conceptuel EICV** (e.g. figure 2.1), qui peut être spécifique à une méthodologie donnée. Ce cadre EICV est une structure *cohérente* guidant le développement de modèles de caractérisation des impacts du cycle de vie, via l'identification de *l'ensemble* des chemins d'impact potentiels et des indicateurs d'impacts *pertinents pour la prise de décision.*

2.2 Évolution de l'évaluation des impacts du cycle de vie

2.2.1 Prémisses et émergence de l'EICV

Dans les années 1970, l'industriel américain Coca-Cola initie les prémisses de l'EICV en développant un bilan énergétique visant à comparer la performance environnementale de différentes options d'emballage pour ses boissons. En 1984, la méthode

des volumes critiques (Bundesamt für Umweltschutz (BUS), 1984) introduit à son tour un cadre d'évaluation quantifiant les émissions de substances d'un système par compartiment (air, eau, sol), et les comparant à leur concentration limite définie par la réglementation environnementale suisse. Novatrice, cette approche ne repose néanmoins pas sur une chaîne de cause à effet mais sur la comparaison à une valeur seuil. De plus, elle ne prend pas en compte la persistance ni le devenir des substances toxiques dans les différents milieux, et ne permet pas d'analyser les interactions entre compartiments.

Avec le gain en popularité de l'ACV dans les années 1980 vient la nécessité croissante d'une meilleure interprétabilité des résultats. L'évaluation des impacts environnementaux associés à l'inventaire devient alors un enjeu central (Hauschild, Michael Z and Huijbregts, 2015).

Dans ce contexte, plusieurs méthodologies d'évaluation des impacts émergent, principalement en Europe. La méthodologie suisse Ecopoints (ou "Ecoscarcity") est développée en 1990 (Ahbe, Braunschweig & Müller-Wenk, 1990), suivie par la méthodologie hollandaise CML proposée en 1992 (Heijungs et al., 1992) et la suédoise EPS en 1993 (Ryding et al., 1993). Ces trois méthodologies découlent de courants de pensée différents, qui se reflètent dans les divers développements de méthodologies d'impact rythmant la deuxième moitié des années 90.

— La famille de méthodologies suisses Ecoscarcity (1991, 1997, 2006 et 2013) constitue l'un des premiers ensembles d'approches visant à évaluer la performance environnementale des systèmes à partir d'une **distance cible**[1]. À l'image de la méthode des volumes critiques, ces approches se distinguent des pratiques actuelles de l'EICV en ce qu'elles ne s'appuient pas sur des chaînes de cause à effet, mais sur la mesure de l'écart par rapport à un seuil ou objectif prédéfini.

1. Parmi les méthodologies structurées autour d'une comparaison à une valeur cible, JEPIX (Miyazaki, 2004) est adaptée au contexte japonais.

— Suivant la méthodologie CML (1992) (Heijungs et al., 1992), un deuxième courant de développement repose sur une **modélisation des impacts au niveau problème**. Au sein de ces méthodologies [2], les indicateurs d'impact sont calculés quelque part sur la chaine de cause-à-effet opérant entre l'émission de substances et les impacts associés. Ces différentes méthodologies comptent entre dix et vingt indicateurs et de catégories d'impact associées (Hauschild, Michael Z and Huijbregts, 2015).

— Finalement, la méthodologie EPS (Ryding et al., 1993) ouvre la voie à un ensemble de **méthodologies "orientées dommage"** dont les indicateurs sont situés au bout de la chaine de cause à effet, sur des entitées jugées importantes pour la société (voir 2.3). Précurseure dans l'évaluation au niveau dommage, EPS ne repose cependant pas sur les sciences naturelles pour quantifier les impacts mais plutôt sur l'évaluation économique et ses méthodes, telles que le consentement à payer. La méthodologie Ecoindicator 99 (Goedkoop & Spriensma, 2001), en revanche, est la première à modéliser les impacts jusqu'au dommages sur la base de mécanismes environnementaux. Dans la suite de ce manuscrit, seules les méthodologies reposant sur des mécanismes environnementaux sont appelées "méthodologies EICV".

2.2.2 Évolution vers des méthodologies problème-dommage

Plus un indicateur est loin dans la chaîne de cause à effet, plus le niveau d'incertitude lié à la modélisation des mécanismes environnementaux est grand (Bare et al., 2000). Ainsi, les méthodologies orientées problème présentent un plus faible niveau d'*incertitude de modélisation* que les méthodologies orientées dommages. En revanche, les indicateurs "problème" sont souvent plus intangibles (e.g. problème

2. Parmi ces approches orientées problème, on retrouve notamment EDIP97 (Wenzel, Hauschild & Alting, 1997), CML 2002 (Guinée et al., 2002), EDIP2003 (Hauschild & Potting, 2004), TRACI2003 (Bare, 2002), LUCAS (2007) (Toffoletto et al., 2007) et ILCD (JRC, 2011), qui devient EF dont la dernière version a été publiée en 2022 (Andreasi et al., 2023).

d'eutrophisation exprimé en kg PO_4^{3-}) et donc difficiles à appréhender par le décideur. Elles comportent donc une plus grande *incertitude d'interprétation*.

Le constat de ces forces et faiblesses des méthodologies orientées problème et dommage a donné l'impulsion à une quatrième école de pensée, qui propose de combiner ces approches (que l'on nommera "**méthodologies problème-dommage**") (Jolliet et al., 2004). Dans cette optique, la méthodologie IMPACT2002+ (Jolliet et al., 2003) combine pour la première fois les approches problème et dommages dans un *cadre cohérent* (voir fig. 2.1) et intègre des concepts provenant de l'analyse de risque au sein de l'ACV pour évaluer les dommages sur les écosystèmes et la santé humaine.

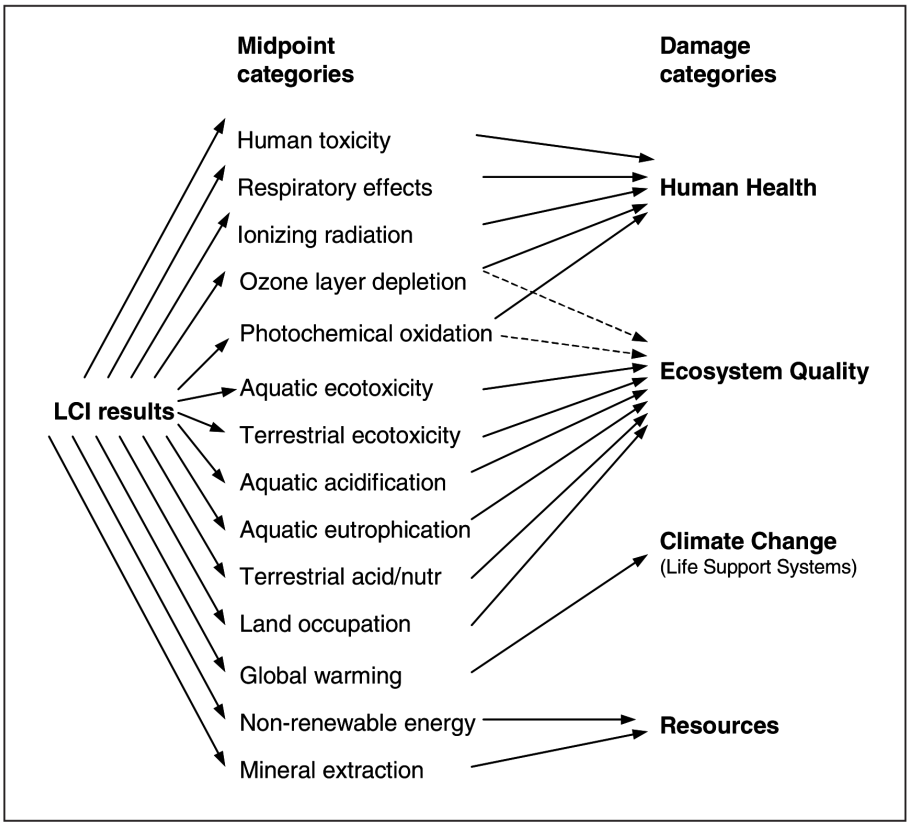


FIGURE 2.1 Schéma général du cadre conceptuel IMPACT 2002+, reliant les résultats de l'inventaire du cycle de vie aux catégories de dommages par l'intermédiaire des catégories problèmes, d'après Jolliet et al. (2003)

En 2019, la méthodologie d'évaluation des impacts régionalisée IMPACTWorld + (Bulle et al., 2019) a été proposée en intégrant les cadres EICV de IMPACT 2002+ et EDIP (Hauschild & Potting, 2004). Cette thèse porte principalement sur la méthodologie IMPACTWorld+, mais d'autres méthodologies problème-dommage ont été développées [3].

La figure 2.2 propose une représentation de l'évolution des différentes méthodologies EICV au cours du temps, et des liens entre elles.

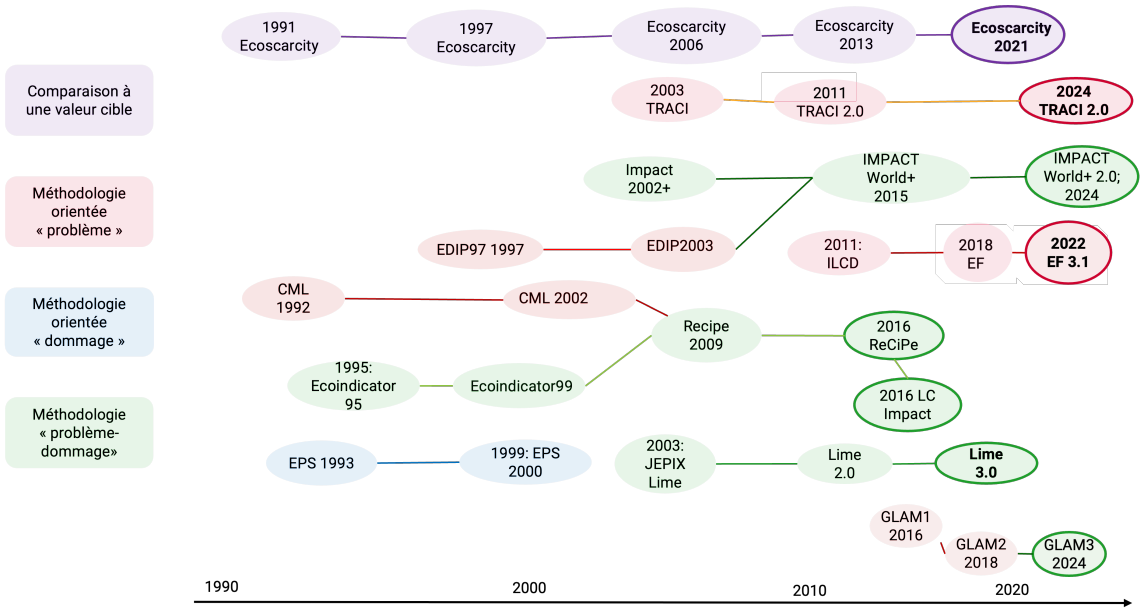


FIGURE 2.2 Évolution et liens des méthodologies EICV. Les méthodes les plus à jour sont représentées dans des ellipses à bords épais. Cette thèse est liée aux méthodologies IMPACTWorld + et GLAM3.

3. La méthodologie LIME (Itsubo & Inaba, 2003) propose une approche problème-dommage représentative du contexte environnemental japonais. Sur la base du guide ACV hollandais et de Ecoindicator99, ReCiPe (Goedkoop et al., 2009) est développée en 2008, puis mise à jour en 2016 (Huijbregts et al., 2017). En 2020, LC-Impact propose à son tour une méthodologie régionalisée combinant indicateurs orientés problème et dommage (Verones et al., 2020).

Conclusions de l'évolution de l'EICV

À retenir

L'évaluation des impacts du cycle de vie (EICV) repose sur le développement de modèles et facteurs de caractérisation. Ces derniers sont développés au niveau problème et/ou dommage pour chaque catégorie d'impact incluse dans la méthodologie EICV. L'EICV a largement évolué au cours des dernières décennies ; avec l'essor de l'ACV appliquée pour évaluer les impacts d'un spectre croissant de systèmes, le cadre de l'EICV évolue pour mieux accompagner la prise de décision ; les méthodologies EICV sont régulièrement repensées et mises à jour afin de garantir leur représentativité et leur pertinence.

2.3 Caractérisation des impacts du cycle de vie

Qu'elles soient orientées problème ou dommage, les méthodologies EICV et leurs modèles de caractérisation sont régis par un cadre calculatoire similaire.

2.3.1 Structure de calcul des facteurs de caractérisation

Les travaux de Udo de Haes et al. (2002) proposent un cadre calculatoire pour les FC. Ces derniers sont exprimés comme la multiplication d'un facteur de sort (FF), d'un facteur d'exposition (XF) et d'un facteur d'effet (EF) :

$$FC = FF \cdot XF \cdot EF \tag{2.2}$$

La signification attribuée à chacun de ces facteurs varie légèrement en fonction de la catégorie d'impact considérée (e.g. écotoxicité ou consommation d'eau).

La caractérisation des impacts du cycle de vie repose sur l'hypothèse de linéarité : l'impact est proportionnel à la quantité de substance émise, simplifiant la non-linéarité des mécanismes environnementaux (D. Li, Tsang & Kua, 2022). Cette hypothèse est posée pour son pragmatisme, mais elle représente une limite majeure

de l'EICV.

2.3.2 Aires de protection

Au niveau "dommage", les impacts se produisent sur des entités importantes que la société cherche à protéger, appelées **Aires de Protection (AdP)** (Schaubroeck & Rugani, 2017). La définition (ou la sélection) des AdP d'une méthodologie EICV est le résultat d'un choix, conscient ou inconscient. Elle dépend d'un jugement quant à ce qui est important, influencé par les perspectives et les parcours des chercheurs (Schaubroeck & Rugani, 2017). Ces choix sont porteurs de subjectivité et ne peuvent être considérés comme neutres (Freidberg, 2018). Ainsi, le développement de différentes méthodologies EICV peut donner lieu à la création de différentes AdP.

La figure 2.3 permet de visualiser les AdP considérées dans différentes méthodologies (problème-dommage et dommage) EICV. Elle permet d'illustrer les similitudes et les divergences entre celles-ci, aussi bien en termes de couverture que d'unités des AdP.

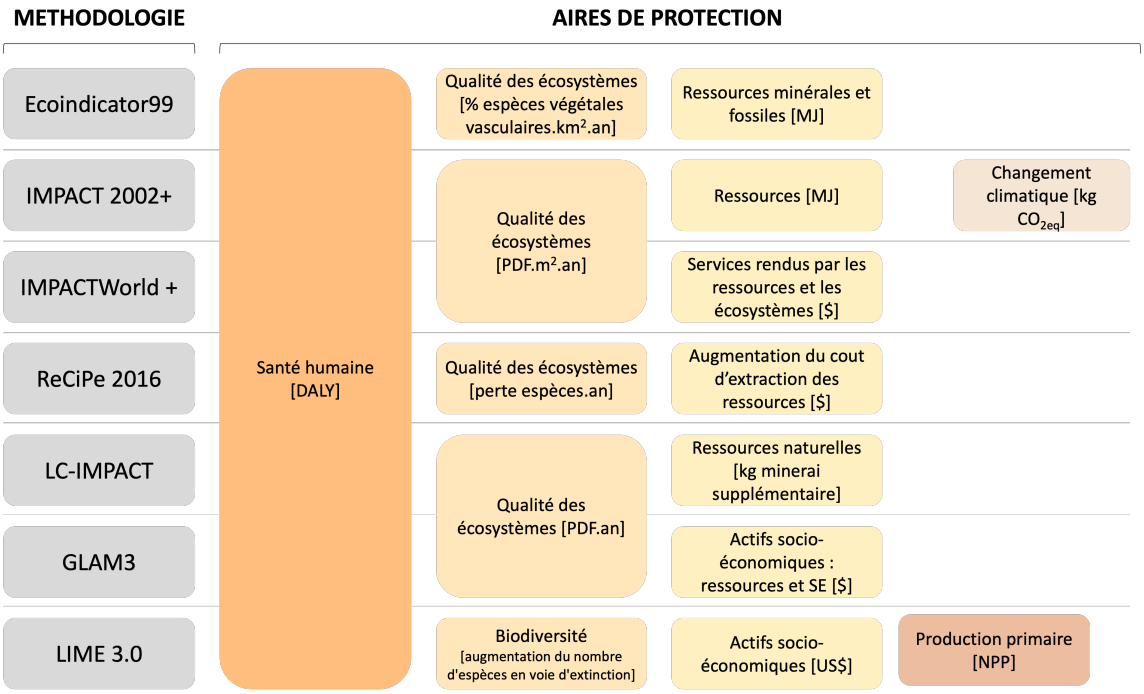


FIGURE 2.3 Aires de Protection et leurs unités considérées dans différentes méthodologies EICV.

La santé humaine est une AdP récurrente dans les différentes méthodologies, exprimée en DALYs (représentant des années de vie en bonne santé perdues). Un deuxième ensemble d'AdP peut être regroupé : les AdP portant sur la qualité des écosystèmes ou la biodiversité. Différentes unités y sont utilisées, traduisant des différences dans les chaines de cause à effet. Par exemple, certains modèles de caractérisation considèrent des extinctions d'espèces, tandis que d'autres évaluent des disparitions temporelles. Un troisième groupe d'AdP peut être formé, regroupant des AdP telles que les ressources (minérales, fossiles et/ou naturelles), les "services rendus par les écosystèmes" ou encore les "actifs socio-économiques". Au sein de ce troisième groupe, la diversité des intitulés et des unités des AdP traduit également d'importantes différences de pratiques et d'objectifs de modélisation entre les méthodologies.

2.4 Efforts d'harmonization

La diversité des méthodologies EICV offre une grande flexibilité au praticien ACV, mais peut être source de confusion quant au choix qui lui est le plus adapté. De plus, les résultats et conclusions d'une même étude peuvent changer dépendamment de la méthodologie utilisée, induisant une forme de scepticisme de la part des praticiens et des décideurs. Face à cette problématique, différents efforts d'harmonisation des pratiques EICV ont été menés.

2.4.1 Normes ISO

En 1993, en parallèle des premiers développements de méthodologies EICV, un processus de standardisation de la pratique de l'ACV est conduit sous l'égide de l'organisation internationale pour la normalisation (ISO). Ce processus aboutit à la publication d'un ensemble de normes, dont l'ISO 14040 (International Organization for Standardization, 2006). Bien que cette norme encadre la pratique de l'ACV, elle ne prescrit pas de méthodologie spécifique pour l'évaluation des impacts environnementaux. Le choix de la méthodologie reste à la discrétion du praticien, qui doit s'appuyer sur une approche scientifiquement reconnue et documentée, alignée avec les objectifs de son étude. En particulier, le praticien doit s'assurer que la couverture d'impacts environnementaux offerte par la méthodologie est pertinente pour l'étude.

2.4.2 L'Empreinte Environnementale (Environmental Footprint) par la Commission Européenne

Visant à la fois l'harmonisation, la transparence et la crédibilité des résultats des ACV en Europe, la Commission européenne recommande des méthodes d'empreintes environnementales spécifiques aux produits et aux organisations. Celles-ci reposent sur la méthodologie d'impact EF ("Environmental Footprint"), dont la version 3.1 a récemment été proposée (Andreasi et al., 2023).

Orientée "problème", EF inclut 16 catégories d'impacts, sélectionnées pour leur perti-

nence environnementale et leur niveau d'incertitude de modélisation bas. Au moment de cette sélection, les modèles orientés dommages sont jugés trop incertains, et le niveau problème est privilégié. Pour faciliter la prise de décision sur la base des 16 indicateurs au niveau problème, le PEF propose un schéma de pondération des impacts en un score unique, basé sur des enquêtes menées auprès d'experts EICV et d'individus européens (Sala, Cerutti & Pant, 2018).

Si la méthode est largement utilisée au sein de l'Europe, elle fait toutefois face à un certain nombre de critiques, notamment en se limitant à des indicateurs orientés problèmes (Finkbeiner, 2014 ; Pedersen & Remmen, 2022). De plus, de par son développement dans un cadre institutionnel, EF évolue selon des temporalités distinctes de celles de la recherche académique, ce qui limite l'intégration des modèles de caractérisation les plus récents.

2.4.3 PNUE/SETAC - Life Cycle Initiative

Portée par la SETAC[4] et le Programme des Nations Unies pour l'Environnement (PNUE), la Life Cycle Initiative rassemble de nombreuses parties prenantes dans le but de consolider les divers aspects de l'ACV, dont l'évaluation des impacts.

Au cours des années 2000, celle-ci atteint le consensus sur la nécessité de combiner l'évaluation des impacts orientée problèmes et dommages au sein de cadres cohérents (voir 2.2.1) (Jolliet et al., 2004), ainsi que sur les pratiques de modélisation des impacts liés à la toxicité et à l'écotoxicité (Rosenbaum et al., 2008).

4. La Society of Environmental Toxicology and Chemistry (SETAC) est une organisation internationale à but non lucratif composée de professionnels dédiés à l'étude et à la gestion des problèmes environnementaux. Elle regroupe divers experts travaillant sur des questions liées à la toxicologie environnementale, à la chimie et à l'analyse du cycle de vie.

2.5 Cadres conceptuels et méthodologies EICV

2.5.1 Travail de la PNUE-SETAC

En 2017, un groupe de travail de la PNUE-SETAC Life Cycle Initiative travaillant sur les "questions transversales" (Verones et al., 2017) propose **une révision de son cadre conceptuel EICV liant émissions et dommages** visant à l'harmonisation de l'EICV (voir fig. 2.4). Aligné avec la quatrième école de pensée introduite à la section 2.2, ce cadre combine de façon cohérente des indicateurs orientés problème et dommage.

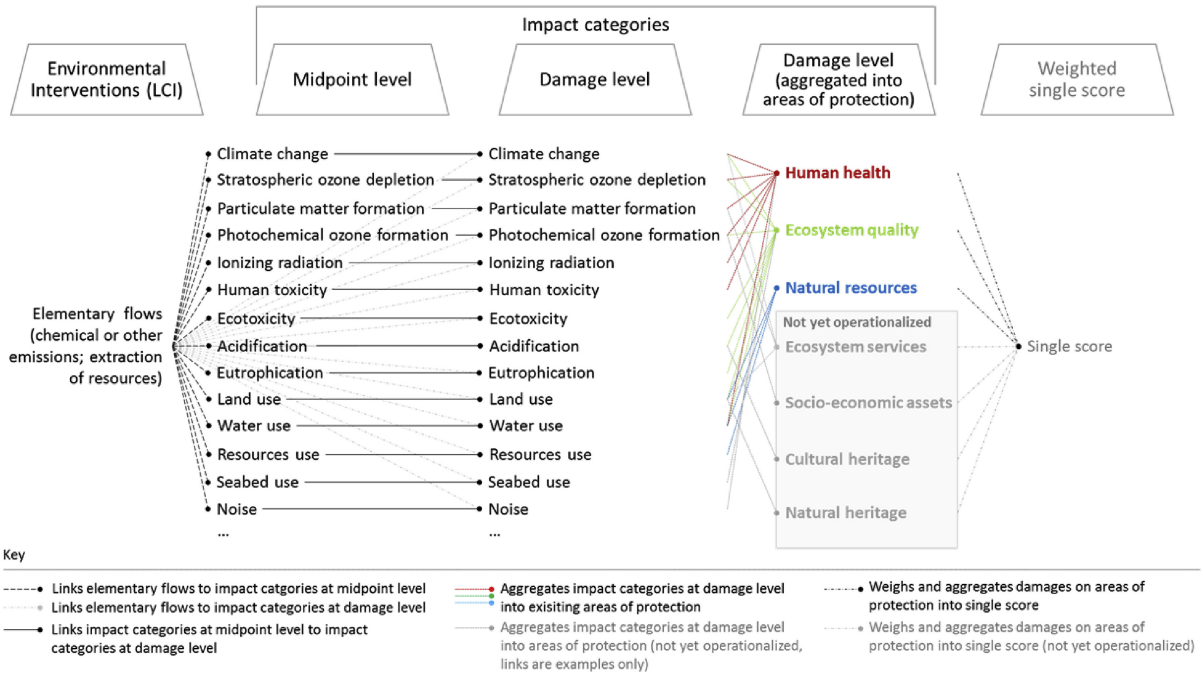


FIGURE 2.4 Cadre méthodologique de l'EICV combinant approches orientées problème et dommage tiré de Verones et al. (2017). Certains modèles de caractérisation sont limités au lien entre inventaire du cycle de vie et indicateur d'impact au niveau "problème" (2^{ieme} colonne, lignes noires en pointillés). D'autres lient résultat de l'inventaire au niveau "problème" puis au niveau "dommage" (3^{ieme} colonne, lignes noires pleines). Une troisième catégories de modèles de caractérisation lient directement résultat de l'inventaire et indicateurs d'impact au niveau dommage (3^{ieme} colonne, lignes grises en pointillés). Les impacts orientés dommages peuvent être agrégés au sein d'AdP. Une étape de pondération, qui peut inclure une normalisation, peut être réalisée pour combiner les résultats (au niveau dommage ou problème) en un score d'impact unique.

Ce cadre permet de représenter les liens existants entre l'inventaire du cycle de vie, les indicateurs orientés problème, et les indicateurs au niveau dommage. Les scores d'impacts au niveau dommages peuvent être agrégés au sein de différentes AdP.

Tel qu'illustré dans la section précédente (voir fig. 2.3), différentes méthodologies EICV considèrent différentes AdP. La création du cadre consensuel du PNUE-SETAC

est alors l'occasion de repenser et de clarifier la nature des AdP importantes à inclure dans une méthodologie EICV. Pour définir l'ensemble de ces AdP, Verones et al. (2017) proposent une vue d'ensemble des valeurs sociétales humaines et de la manière dont les dommages causés à ces valeurs sont ou pourraient être mesurés en ACV (Tableau 2.1).

TABLEAU 2.1 Vue d'ensemble des valeurs sociétales humaines et de la manière dont les dommages causés à ces valeurs sont ou pourraient être mesurés en ACV, d'après Verones et al. (2017). Les aires de protection considérées comme opérationnelles dans le cadre EICV de Verones et collègues sont marquées par une astérisque.

Valeur / **Système**	**Intrinsèque**	**Instrumentale**	**Culturelle**
Humain et fait par les humains	Santé humaine*	Biens socio-economiques	Héritage culturel
Naturel	Qualité des ecosystèmes*	Ressources & services écosystémiques*	Héritage naturel

À travers ce tableau 2.1, Verones et al. (2017) suggèrent le développement d'une AdP pour chaque type de système (humain ou naturel) et chaque type de valeur (intrinsèque, instrumentale et culturelle). Ainsi, la valeur instrumentale des systèmes naturels est couverte par une AdP "Ressources & Services Écosystémiques" [5].

Tel que visible sur la figure 2.4, seules 3 AdP sont actuellement opérationnelles : la santé humaine, la qualité des écosystèmes et les ressources naturelles. Elles représentent respectivement les impacts sur la valeur intrinsèque des systèmes humains, la valeur intrinsèque des systèmes naturels et (une partie de) la valeur instrumentale des systèmes naturels.

Sur la base de ce cadre EICV conceptuel et de ses AdP, la *Life Cycle Initiative*

5. Néanmoins, contrairement à ce qui est suggéré dans le tableau 2.1, la figure 2.4 sépare les ressources naturelles et les services écosystémiques en deux AdP distincts. Cette incohérence est discutée plus loin dans le manuscript.

développe le projet GLAM (Global Guidance for Life Cycle Impact Assessment Indicators and Methods) en 2013 dans l'objectif d'établir une méthodologie d'évaluation des impacts consensuelle et harmonisée. Le GLAM s'est développé en trois phases, dont la dernière (GLAM3) s'est tenue entre 2020 et 2024 et à l'issue de laquelle la première version de la méthodologie GLAM a été lancée [6].

2.5.2 IMPACT World+ (Bulle et al. 2019)

Les recommandations de la PNUE-SETAC Life Cycle Initiative ont également été suivies dans le développement de la méthodologie régionalisée problème-dommage IMPACT World+ (Bulle et al., 2019) [7]. La figure 2.5 représente son cadre conceptuel. 24 indicateurs orientés problème sont intégrés à sa version 2.1, ainsi que 46 indicateurs au niveau dommages et 2 AdP opérationnelles : santé humaine et qualité des écosystèmes. Une troisième AdP, "Services rendus par les ressources et les écosystèmes", est en voie d'intégration.

6. Une partie des travaux de cette thèse s'inscrit dans le cadre de GLAM3 ; le modèle de caractérisation présenté au Chapitre 8 a été intégré dans celle-ci.

7. La méthodologie IMPACTWorld + voit le jour en 2019 (Bulle et al., 2019), et sa deuxième version (actuellement v2.1) est partagée en fin d'année 2024 (Agez et al., 2024). Elle est le fruit d'une collaboration internationale, en grande partie soutenue par le Centre International de référence sur l'analyse du cycle de vie et la transition durable (CIRAIG). Cette thèse s'inscrit dans le cadre de l'amélioration continue et du développement de cette méthodologie.

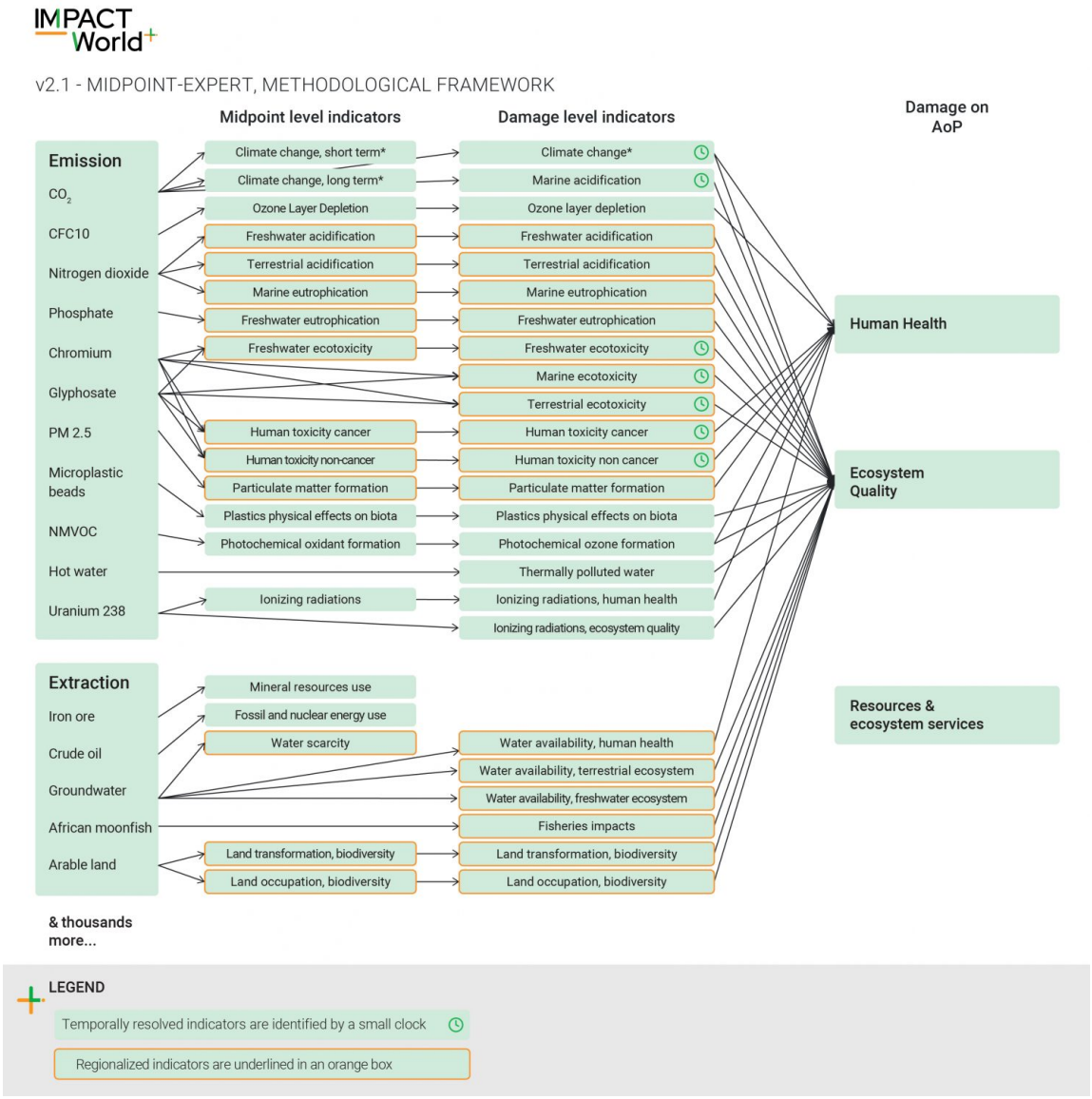


FIGURE 2.5 Cadre méthodologique de la méthodologie IMPACT World+ (Bulle et al., 2019) version 2.1, tiré du site internet de la méthodologie : https://www.impactworldplus.org/methodology/.

À retenir

Au niveau dommage, les scores d'impact environnementaux peuvent être agrégés au sein d'Aires de Protection (AdP) qui représentent les objets importants pour la société. Ces derniers sont subjectifs, et peuvent varier d'une méthodologie EICV à l'autre. Différents efforts d'harmonisation de l'EICV ont été menés. En particulier, un groupe de travail du Programme des Nations Unies pour l'Environnement a proposé un cadre conceptuel EICV problème-dommage, et clarifié ses AdP. Les méthodologies GLAM3 et IMPACTWorld + ont été développées sur la base de celui-ci, et en combinant, améliorant et complétant des méthodologies existantes.

CHAPITRE 3 LES SERVICES ÉCOSYSTÉMIQUES EN ÉVALUATION DES IMPACTS DU CYCLE DE VIE

Note au lecteur : Ce chapitre reprend une partie des éléments de l'une des publications scientifiques réalisées dans le cadre de cette thèse. Cette publication intitulée "Toward a harmonized integration of ecosystem services in LCIA : a critical review" a été soumise dans le journal The International Journal of Life Cycle Assessment. Il s'agit d'une revue critique discutant l'intégration des SE en évaluation des impacts du cycle de vie. Elle est structurée autour 1- d'une revue de la littérature des concepts clefs relatifs aux SE, et 2- d'une discussion critique sur les propositions qui ont été faites pour l'intégration des SE en EICV. À des fins de clarté et de fluidité de la thèse, cette publication y est présentée en annexes, et non pas dans le coeur du manuscript. Les principales notions et conclusions qui y sont présentées sont synthétisées au sein du présent chapitre. En particulier, la section 3.1 reprend les éléments clefs de la revue de littérature portant sur les SE, et la section 3.2 synthétise les discussions menées sur l'état de l'art de l'intégration des SE en EICV.

3.1 Les services écosystémiques

3.1.1 Émergence du concept de services écosystémiques

Le terme **« services écosystémiques »** (SE) désigne les avantages que les populations humaines dérivent des écosystèmes (Costanza et al., 1997). La Notte et al. (2017) les définissent comme les « services fournis par les fonctions et processus écologiques et contribuant au bien-être humain ».

Ils englobent, entre autres, l'approvisionnement en nourriture, en eau ou divers matériaux, la protection contre les maladies et les événements extrêmes, la régulation du climat et la pollinisation, ou encore des opportunités de récréation ou d'inspiration artistique. Ultimement, le concept de service écosystémique vise à mettre en évidence l'importance du bon fonctionnement des écosystèmes pour les sociétés humaines, et

la nécessité de leur conservation.

Si Platon (400 av. J.-C.) appréhende déjà de tels services dans ses écrits en soulignant notamment l'importance des forêts contre l'érosion (Platon, s. d.), le concept de SE se popularise dans les années 1990 grâce aux travaux de scientifiques, d'économistes et d'écologistes tels que Gretchen Daily, Robert Costanza ou encore Hal Mooney (Costanza et al., 1997 ; Daily, 1997 ; Mooney & Ehrlich, 1997).

En 1997, les travaux de Costanza et al. (1997) proposent une estimation globale de la valeur économique des services rendus par la nature. Celle-ci atteint un montant supérieur à 33 trillions US$ par an, une valeur dépassant le produit intérieur brut (PIB) mondial. Bien que l'estimation de Costanza et al. (1997) ait fait l'objet de nombreux débats par la suite, cette étude marque les esprits en soulignant l'importance économique de la biosphère, jusqu'alors exclue de l'économie conventionnelle.

En 2005, le Millenium Ecosystem Assessment (MEA) rassemble plus de 1300 experts et propose un cadre de classification pour ces « bénéfices que les humains tirent des écosystèmes [1] » (Millenium Ecosystem Assessment, 2005). Ce cadre offre une première structure permettant de classifier et d'évaluer les SE.

Dans les années qui suivent, les efforts se multiplient pour comprendre, évaluer et gérer ces «contributions directes ou indirectes des écosystèmes au bien-être humain» (R. De Groot et al., 2010). De nombreuses initiatives voient le jour, telles que la Plateforme intergouvernementale scientifique et politique sur la biodiversité et les services écosystémiques (IPBES), fondée par 94 gouvernements en 2012 et qui publie en 2019 son rapport de l'évaluation mondiale de la biodiversité et des SE (IPBES, 2019). De son côté, le Natural Capital Project naît d'un partenariat entre l'université de Stanford, The Nature Conservancy et le World Wildlife Fund, avec pour objectif

1. La convention sur la diversité biologique des Nations Unies (1992) définit un écosystème comme un complexe dynamique constitué de communautés de plantes, d'animaux et de micro-organismes interagissant avec leur environnement non vivant comme une unité fonctionnelle. Dans CICES (Haines-young, Paddocks & Lane, 2023), le terme "écosystème" est également utilisé pour désigner "les organismes vivants et leur environnement abiotique", reconnaissant la contribution des structures et processus biophysiques au bien-être humain.

une meilleure intégration des SE dans la prise de décision.

La notion de SE est aujourd'hui largement mobilisée au sein de la communauté scientifique, qui appelle à ce qu'elle gagne du terrain dans les systèmes de gestion de l'environnement et dans les processus de prise de décision (Guerry et al., 2015 ; Mandle et al., 2021 ; O'Higgins, Lago & DeWitt, 2020).

3.1.2 Services écosystémiques comme support de valeur instrumentale

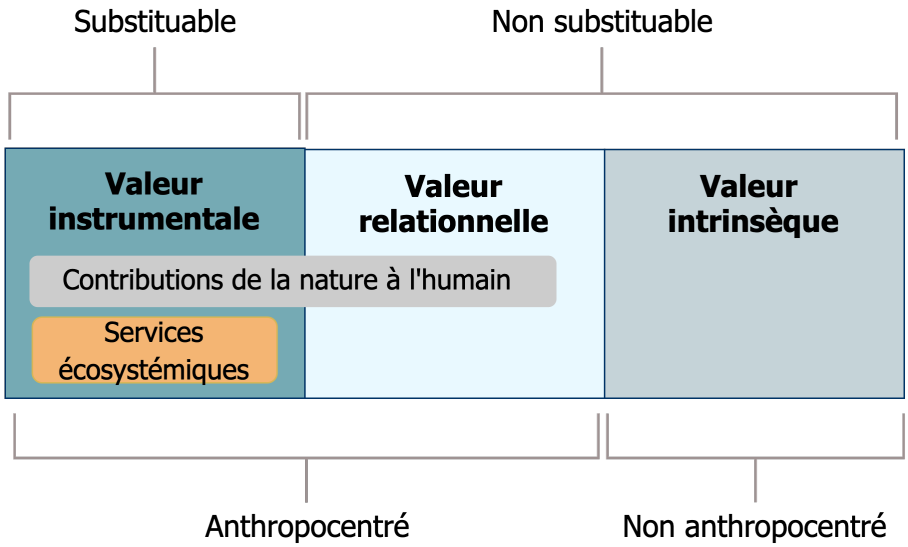


FIGURE 3.1 Positionnement du concept de service écosystémique par rapport aux différents domaines de valeur. Dans le cadre de cette thèse, le terme écosystème service est utilisé *per se*, pour désigner le support de valeurs instrumentales seulement. En ce sens, il se distingue du concept plus large de contributions de la nature à l'humain, mises en avant de manière croissante dans l'évaluation environnementale (notamment dans le cadre de l'IPBES).

Un même objet peut être porteur de trois types de valeur : intrinsèque, instrumentale et relationnelle (Arias-Arévalo, Martín-López & Gómez-Baggethun, 2017). Ces dimensions ne sont pas exclusives, mais reflètent des perspectives différentes sur la

manière dont une entité peut être considérée comme ayant de la valeur.

La **valeur instrumentale** fait référence à la valeur attribuée à un objet en fonction de son utilité, de son efficacité à atteindre un objectif ou à satisfaire un besoin (Arias-Arévalo, Martín-López & Gómez-Baggethun, 2017 ; Hill et al., 2021 ; Himes et al., 2024). Il s'agit d'un concept anthropocentré, qui traduit l'appréciation de l'humain. Étant donné que les valeurs instrumentales constituent des moyens pour atteindre une fin, Himes et al. (2024) suggèrent qu'une valeur instrumentale est, en principe, substituable, bien que cela ne soit pas toujours réalisable dans la pratique.

La **valeur intrinsèque** repose sur les caractéristiques inhérentes d'un objet, indépendamment de son utilité et de toute appréciation humaine (Arias-Arévalo, Martín-López & Gómez-Baggethun, 2017 ; Hargrove, 1992 ; R. K. Turner et al., 2003). Contrairement aux valeurs instrumentales, les valeurs intrinsèques sont non substituables.

Enfin, le concept de **valeur relationnelle** [2] renvoie aux liens significatifs et non substituables entre les êtres humains et la nature. Il englobe notamment des dimensions telles que le sens de la responsabilité envers la nature, le bien-être personnel lié à la nature (eudémonisme), ou le sentiment d'appartenance à un lieu (Chan, Gould & Pascual, 2018 ; Hill et al., 2021 ; Himes & Muraca, 2018 ; Himes et al., 2024). Contrairement à la valeur instrumentale, centrée sur l'utilité, la valeur relationnelle met l'accent sur les dimensions éthiques, culturelles et affectives des relations humain-

2. Le concept de valeurs relationnelles est de plus en plus reconnu comme un prisme pertinent pour appréhender les enjeux de durabilité environnementale. Il permet d'aborder des questions de justice environnementale, de prendre en compte la diversité des visions du monde et de mobiliser plus efficacement les publics autour des actions de conservation (Himes & Muraca, 2018 ; Himes et al., 2024 ; Pratson, Adams & Gould, 2023 ; Uehara, Sakurai & Hidaka, 2022). Les valeurs relationnelles gagnent ainsi en importance dans les approches internationales de la conservation. Elles occupent désormais une place centrale dans le cadre conceptuel et la typologie des valeurs proposés par la Plateforme intergouvernementale scientifique et politique sur la biodiversité et les SE (IPBES, 2019). Dans cette perspective, l'IPBES a introduit le concept de Contributions de la nature aux populations (Nature's Contributions to People, NCP) (Díaz et al., 2015, 2018). Bien que proche de la notion de SE, le concept de NCP propose une approche plus inclusive, en englobant un éventail plus large de relations et de bénéfices issus de la nature que celui de services écosystémique (voir fig. 3.1). Les NCP intègrent explicitement des visions du monde alternatives, ainsi que des systèmes de connaissances variés, notamment les savoirs autochtones.

nature.

Dans le cadre de cette thèse, **le terme SE est utilisé pour désigner le support de valeur instrumentale** uniquement.

3.1.3 La classification CICES

Depuis la publication du cadre du Millennium Ecosystem Assessment (MEA), d'autres classifications des SE ont été développées. Parmi les plus couramment utilisées figurent le cadre de l'Économie des Écosystèmes et de la Biodiversité (The Economics of Ecosystems and Biodiversity, TEEB) (R. De Groot et al., 2010) ainsi que la Classification Commune Internationale des Services Écosystémiques (Common International Classification of Ecosystem Services, CICES) (Haines-Young & Potschin, 2018), élaborée en 2009 par l'Agence européenne pour l'environnement.

Inspirée des classifications proposées par le MEA et le TEEB, CICES a progressivement acquis un statut de référence au sein de la communauté scientifique travaillant sur les SE (Finisdore et al., 2020 ; La Notte et al., 2017 ; Maes et al., 2016).

CICES répartit les SE en trois grandes catégories :

- **les SE d'approvisionnement** tels que l'approvisionnement en eau, bois, minéraux, biomasse, etc.
- **Les SE de régulation et de maintien** tels que la médiation des déchets, le contrôle de l'érosion ou l'atténuation des évènements extrêmes
- **Les SE culturels** tels que les activités récréatives ou les interactions spirituelles avec les systèmes naturels



FIGURE 3.2 Les catégories de SE selon la classification CICES

Ces trois catégories de SE reposent sur des processus écologiques tels que le cycle des nutriments, la formation des sols ou encore la production primaire [3].

Une classification holistique englobant les ressources naturelles et les services géophysiques. À l'origine, la classification CICES se limite aux SE reposant sur les systèmes vivants, dits "biotiques" (Haines-young, Paddocks & Lane, 2023).

Cette approche a toutefois été critiquée par une partie de la communauté scientifique, dans la mesure où elle néglige les interactions entre les composantes biotiques (vivantes) et abiotiques (non vivantes) des écosystèmes, interactions pourtant essentielles à la production et au maintien des SE (Baveye et al., 2018 ; Fox et al., 2020 ; Frisk et al., 2022 ; Gray, 2012 ; Gray, Gordon & Brown, 2013 ; Van Ree & Beukering, 2016).

La dimension biotique des premières versions de CICES a également été remise en question en raison de l'inclusion de services tels que l'approvisionnement en eau, qui ne relève pas exclusivement de processus biologiques.

En réponse à ces discussions, la dernière version de CICES (v5.2) clarifie son cadre.

3. Dans le cadre du Millenium Ecosystem Assessment (2005), ces processus écologiques étaient considérés comme une quatrième catégorie de services, qualifiés de services de soutien (ou intermédiaires). Cependant, cette catégorisation a fait l'objet de critiques, notamment en raison du risque de double comptages : elle peut en effet conduire à comptabiliser à la fois les services intermédiaires et les services finaux qu'ils rendent possibles. Pour remédier à ce problème, la classification CICES a fait le choix d'exclure les services dits "intermédiaires" de son cadre.

Elle explicite la nature des interactions biotiques et abiotiques à la base des services biotiques, et intègre une nouvelle section dédiée aux services géophysiques, permettant ainsi de prendre en compte explicitement les services d'origine abiotique. La table 3.3 présente un extrait de cette dernière version de CICES, incluant l'ensemble des sections désormais reconnues.

	Section	Division	Group	Class	5.2 Code	Class type	Simple descriptor
CICES (Biophysical ecosystem services)	**Provisioning (Biotic/ Biophysical)**	Biomass	Cultivated terrestrial plants for nutrition, materials or energy	Cultivated terrestrial plants (including fungi, algae) grown for nutritional purposes	1.1.1.1	Crops by type (e.g. cereals, root crops, soft fruit, etc.)	Any crops and fruits grown by humans for food; food crops
CICES (Ecosystem Services)	**Regulation & Maintenance (Biotic/ Biophysical)**	Transformation of biochemical or physical inputs to ecosystems	Reduction of nutrient loads and mediation of wastes or toxic substances of anthropogenic origin by living processes	Bio-remediation by micro-organisms, algae, plants, and animals	2.1.1.1	*By type of living system or by waste or subsistence type*	*Decomposing wastes or polluting substances*
CICES (Ecosystem Services)	**Cultural (Biotic/ Biophysical)**	Physical and experiential interactions with natural environment	Direct, in-situ and outdoor interactions with living systems that depend on presence in the environmental setting, i.e. broadly recreational activities	Elements of living systems that that enable activities promoting health, recuperation or enjoyment through active or immersive interactions	3.1.1.1	By type of living system or environmental setting	Using the environment for sport and recreation; using nature to help stay fit
CICES (Geosystem outputs)	**Provisioning (Abiotic/ geophysical)**	Water	Surface water used for nutrition, materials or energy	Surface water for drinking	4.1.1.1	By type or source	Drinking water from sources at the ground surface
CICES (Geosystem outputs)	**Regulation & Maintenance (Abiotic/ geophysical)**	Transformation of biochemical or physical inputs to ecosystems	Mediation of waste, toxics and other nuisances by non-living processes	Dilution or transport of wastes by freshwater and marine ecosystems	5.1.1.1	By type	Diluting or removing wastes
CICES (Geosystem outputs)	**Cultural (Abiotic/ geophysical)**	Physical and experiential interactions with biophysical environment	Direct, in-situ and outdoor interactions with geophysical systems that depend on presence in the environmental setting	Characteristics of geophysical systems that that enable activities promoting health, recuperation or enjoyment through active or immersive interactions	6.1.1.1	By type of geophysical characteristic or environmental setting	Using the environment for sport and recreation; using nature to help stay fit

FIGURE 3.3 Extrait de la classification CICES v5.2

3.1.4 Le modèle en cascade

Le modèle en cascade (Haines-Young & Potschin, 2010) des SE est un modèle conceptuel qui décrit les interrelations entre les écosystèmes et le bien-être humain. Il permet d'illustrer la manière dont les SE sont générés à partir des écosystèmes (leurs structures et processus) et dérivés en avantages pour les sociétés humaines.

Une première version du modèle en cascade est proposée en 2010 par Haines-Young et Potschin (2010) (voir fig. 3.4).

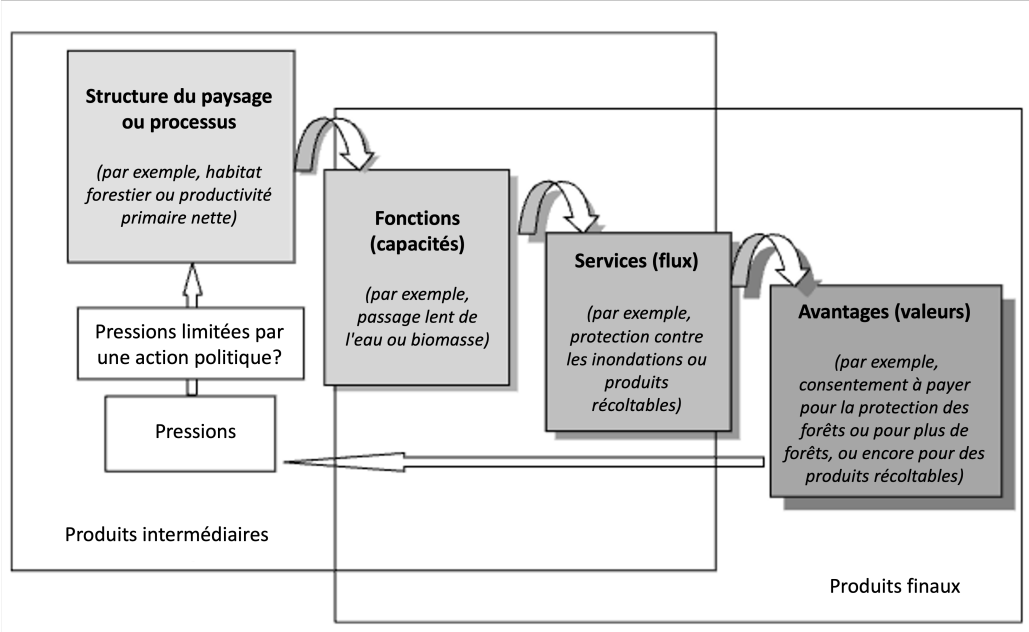


FIGURE 3.4 Modèle en cascade proposé par Haines-Young et Potschin (2010) (traduit de l'anglais).

Dans le modèle en cascade, la *structure* d'un écosystème dépend de ses composantes biotiques et abiotiques ainsi que de ses caractéristiques physiques, telles que le climat ou la production primaire nette. Les *processus* de l'écosystème quant à eux décrivent les interactions physiques, chimiques et biologiques qui lient les organismes et leur environnement. Ces interactions peuvent inclure, entre autres, les relations

prédateurs-proies, les flux d'énergie ou le cycle des nutriments. La *structure* et les *processus* d'un écosystème en soutiennent les *fonctions*, définies comme le "potentiel qu'ont les écosystèmes de fournir un service" (R. De Groot et al., 2010), ou "le rendement maximal hypothétique d'un service écosystémique" par un écosystème (Burkhard et al., 2014).

Une fraction de ce potentiel est utilisée par les humains : le *flux de service écosystémique*. En ce sens, il est important de noter que la notion de demande par les humains est une condition préalable à l'existence d'un flux de service écosystémique.

Finalement, la contribution d'un flux de service écosystémique au bien-être humain est comptabilisée comme un *avantage* (ou "bénéfice"), et peut être évaluée via différentes méthodes.

3.1.5 Impacts sur la capacité des écosystèmes à fournir des services

Au cours des cinquante dernières années, les écosystèmes naturels ont subi des transformations d'une ampleur et d'une rapidité sans précédent (IPBES, 2019). Ces bouleversements, induits par divers facteurs de changement, affectent profondément le fonctionnement des écosystèmes et compromettent leur capacité à fournir des SE. Parmi les principaux facteurs identifiés figurent les modifications dans l'utilisation des terres et des mers, l'exploitation directe des espèces, le changement climatique, la pollution et la prolifération d'espèces exotiques envahissantes (IPBES, 2019).

Dès 2005 le MEA attire l'attention sur la dégradation des écosystèmes et de leurs services. Il estime que plus de 60% de ces services ont été altérés ou transformés entre les années 1950 et 2000 (Millenium Ecosystem Assessment, 2005). Du côté des économistes, Costanza et al. (2014) estiment une perte de la valeur mondiale des SE entre 1997 et 2011 atteignant une fourchette de 4,3 à 20,2 billions de dollars américains par an ($\times 10^{12}$ US\$ $\cdot\, yr^{-1}$). Les analyses plus récentes de l'IPBES (2019) résonnent avec de tels constats et alertent quant à l'accélération du rythme de transformation des écosystèmes.

Pour limiter la dégradation des écosystèmes et des services qu'ils offrent à la société,

il est ainsi crucial d'intégrer les informations traduisant l'importance de leur intégrité au sein des systèmes de gestion environnementale et des processus de prise de décision (Mandle et al., 2021).

À retenir

Les services écosystémiques (SE) sont les avantages que les populations humaines dérivent des écosystèmes. Ils sont le support de valeurs instrumentales, par opposition aux valeurs intrinsèques et relationnelles. Le concept de SE a été démocratisé dans les années 90, et apparaît très pertinent pour la gestion environnementale. La classification de SE CICES est la plus utilisée. Sa dernière version réconcilie services biophysiques et géophysiques. Conceptuel, le modèle en cascade a été développé pour illustrer la séquence à travers laquelle les écosystèmes, par le biais de leurs fonctions et structures, fournissent des SE à la société. En altérant les écosystèmes, les activités humaines mettent en péril leur capacité à fournir des services. Pour limiter leur dégradation, il est important de mieux les considérer dans les systèmes de gestion environnementale et dans les processus de prise de décision.

3.2 Les services écosystémiques en évaluation des impacts du cycle de vie

Malgré des objectifs de durabilité communs, les communautés scientifiques travaillant sur les SE d'une part, et l'ACV de l'autre, ont évolué séparément, collaborant de manière limitée (Vanderwilde & Newell, 2021).

Au cours de la dernière décennie cependant, un nombre grandissant d'auteurs a réfléchi à la manière dont ces expertises peuvent être combinées. Certains proposent d'utiliser ACV et évaluation des SE indépendamment, en combinant leurs résultats, ou en intégrant l'une des approches au sein de la seconde (De Luca Peña et al.,

2022). Ainsi, certains travaux proposent d'intégrer des concepts liés aux SE au sein du cadre de l'ACV (De Luca Peña et al., 2022, 2024 ; Heitbrink, 2024 ; Maia de Souza et al., 2018 ; Rugani et al., 2019, 2022 ; Vanderwilde & Newell, 2021 ; Y. I. Zhang, Anil & Bakshi, 2010 ; Y. I. Zhang, Singh & Bakshi, 2010). Cette intégration peut être réalisée à différentes étapes de l'ACV, incluant la définition du champ de l'étude (e.g. Boone et al. (2019)), l'inventaire (Chaplin-Kramer et al., 2017) ou **l'évaluation des impacts du cycle de vie** [4].

En incluant des AdP couvrant les SE, le cadre méthodologique de l'EICV du PNUE-SETAC ainsi que des méthodologies IMPACTWorld+ et GLAM3 traduisent cette volonté de la communauté ACV d'**intégrer les SE en EICV**. Cette intégration **nécessite le développement de modèles et de facteurs de caractérisation permettant de quantifier l'impact potentiel de flux élémentaires, liés aux activités humaines, sur les SE.**

Différentes études ont discuté cet objectif (e.g. Alejandre, Bodegom et Guinée (2019), Maia de Souza et al. (2018) et Othoniel et al. (2016)) et/ou proposé des approches permettant de l'atteindre (e.g. Cao et al. (2015), Koellner et Geyer (2013) et Othoniel et al. (2019)).

Les premières propositions de modèles de caractérisation sur les SE datent des années 2010. Le projet LULCIA (Land Use Life Cycle Impact Assessment), conduit dans le cadre de la seconde phase de GLAM (voir 2.2), a notamment proposé un cadre pour l'évaluation des impacts liés à l'utilisation des terres, incluant des chemins d'impacts sur les SE (Koellner & Geyer, 2013).

Néanmoins, 12 ans plus tard, les AdP relatives aux SE demeurent inopérationnelles dans les cadres méthodologiques et les méthodologies EICV (voir figures 2.5 et 2.4) ; le développement et l'opérationnalisation de modèles de caractérisation des impacts sur les SE sont restés limités au cours de la dernière décennie. Notre revue critique a

4. La suite du manuscrit se concentre sur l'intégration des SE en EICV. Le lecteur est invité à se référer à l'article présenté en annexes pour en apprendre davantage sur les propositions d'intégration des SE en ACV.

permis de mettre en évidence des obstacles majeurs à cette avancée : **un manque de structuration, de cohérence et de clarté méthodologique** dans les approches existantes.

Les limites principales de l'EICV sur les SE incluent :

- **Un manque de clarté quant à la position des SE par rapport aux autres AdP.** Les AdP courantes (santé humaine et qualité des écosystèmes) et les modèles de caractérisation associés sont pour la plupart liés à des concepts relatifs aux SE. Ces liens ne sont pas suffisamment explicites.

Il existe des liens étroits entre les chemins d'impacts sur la biodiversité et les chemins d'impact sur les SE ; une transformation de la structure et/ou des processus d'un écosystème est susceptible d'entraîner des impacts sur ces deux AdP. La dégradation de la capacité des écosystèmes à fournir des services peut également entraîner des impacts sur la santé humaine : une réduction de la disponibilité d'eau douce (i.e. un changement de fourniture de SE d'approvisionnement) peut entraîner des enjeux de santé via une réduction de la production de nourriture ou une réduction des pratiques d'hygiène. Pour intégrer les SE au sein de l'EICV de façon cohérente, il est important de clarifier les interactions existant entre SE (et concepts reliés) et AdP et modèles de caractérisation existants.

- **Une distinction conceptuelle et méthodologique entre SE d'approvisionnement et ressources naturelles, faite par la communauté EICV.**

La version 5.2 de CICES réconcilie SE et services géophysiques dans un cadre commun, et offre un cadre cohérent qui englobe les "ressources naturelles" (aussi bien biotiques qu'abiotiques) comme le "résultat" d'un SE d'approvisionnement.

Cette approche contraste avec les pratiques courantes de la communauté EICV, qui tend à distinguer les "ressources naturelles" des "services écosystémiques". Notre analyse suggère que cette dissociation trouve principalement son origine dans une trajec-

toire historique : l'intégration des ressources naturelles dans les processus de décision s'est développée plus précocement que celle des SE, conduisant à une évolution en silos des communautés scientifiques mobilisées autour de ces deux objets.

Bien qu'une certaine proximité conceptuelle entre ressources naturelles et SE soit évoquée (en témoigne la création d'une AdP commune dans la grille de valeurs proposée par Verones et al. (2017) dans le tableau 2.1), l'examen des dynamiques de recherche révèle l'existence de deux communautés distinctes. Par exemple, dans le cadre du projet GLAM3, le groupe de travail "Natural Resources & Ecosystem Services" est structuré en deux sous-groupes distincts. De façon similaire, ces deux "types d'objets" sont dissociés en deux AdP dans le cadre conceptuel EICV proposé par Verones et al. (2017) (cf. figure 2.4).

- **L'absence d'harmonisation (en termes d'objectif et d'approche) et de complétude des modèles de caractérisation des impacts sur les SE.**

Différents travaux de recherche ont proposé des approches pour intégrer les SE en EICV. Ces approches sont variées ; elles découlent d'écoles de pensée différentes et la cohérence entre elles mérite d'être examinée.

Une première école de pensée s'intéresse à la relation entre **demande et fourniture** de SE. Il s'agit d'une approche courante au sein de la communauté des SE, dans laquelle l'utilisation de SE (tirée par la demande) est jugée soutenable dans la mesure où elle ne dépasse pas la capacité de l'écosystème à fournir ce service (X. Liu, Ziv & Bakshi, 2018a, 2018b). Certains auteurs utilisent notamment cette relation pour évaluer la durabilité absolue des systèmes étudiés (Xue & Bakshi, 2022) : l'impact des activités, exprimé sous la forme de demande en SE (pour atténuer les impacts ou liés aux ressources consommées), est comparé à la capacité de charge de l'écosystème dérivée du cadre des limites planétaires.

La mise en pratique d'approches "demande-fourniture" pour l'évaluation des impacts à l'échelle du cycle de vie rencontre d'importantes limites. Pour quantifier une demande en SE de régulation et maintenance pour l'atténuation d'impacts (par exemple

d'une substance toxique), une étape préliminaire de quantification de l'impact (sans atténuation) est nécessaire. Cette "évaluation en deux temps" n'est pas compatible avec le cadre de l'EICV actuel. De plus, la structure actuelle de l'ACV, qui nécessite l'évaluation des impacts potentiels de systèmes d'arrière-plan, dont la localisation est parfois inconnue ou distribuée à l'échelle mondiale, ne permet pas une représentation explicite de la demande de SE. Cette approche est donc difficilement généralisable à l'heure actuelle, et s'écarte des pratiques habituelles de modélisation des impacts du cycle de vie.

Une seconde école de pensée suit une approche plus courante de l'EICV et développe des modèles de caractérisation évaluant les **impacts potentiels des activités humaines sur la capacité des écosystèmes à fournir des services**. Deux chemins d'impact principaux sont explorés : 1- les impacts liés à une utilisation des terres (changement ou occupation des sols), et 2- les impacts induits par d'autres types de pressions environnementales induisant des changements dans la structure ou les fonctions de l'écosystème (e.g. l'émission de substances toxiques) (Hardaker et al., 2022).

Pour modéliser ces deux types de chemins d'impact, plusieurs approches sont suivies. Certains auteurs utilisent des modèles propres à l'utilisation des terres (Koellner & Geyer, 2013), explorent les liens entre pressions environnementales, impacts sur la biodiversité fonctionnelle et impacts sur les SE (Oginah et al., 2023), ou s'appuient sur des modèles dynamiques intégrés (Arbault et al., 2014 ; Othoniel et al., 2019). Au sein de ces différentes approches, le modèle en cascade est régulièrement mobilisé pour illustrer la séquence liant pressions environnementales, changements de structures et processus écosystémiques, et impacts sur les SE.

Malgré ces avancées néanmoins, le développement de modèles et facteurs de caractérisation évaluant les impacts des activités humaines sur la capacité des écosystèmes à fournir des services reste limité. Premièrement, le niveau de couverture des différents types de SE est faible (Alejandre, Bodegom & Guinée, 2019 ; De Luca Peña et al., 2022 ; Vanderwilde & Newell, 2021). Seul un nombre limité (principalement terrestres) de SE sont considérés (Alejandre, Bodegom & Guinée, 2019 ; Soldati et

al., 2023 ; Vanderwilde & Newell, 2021). De plus, l'accent a été mis principalement sur l'utilisation des sols comme pression clé sur les SE, tandis que d'autres pressions environnementales critiques sont sous-représentées (Hardaker et al., 2022 ; Rugani et al., 2022). Enfin, notre analyse met en évidence un manque de clarté terminologique concernant les concepts liés aux SE. La littérature fait fréquemment référence aux SE sans préciser s'il s'agit des fonctions écosystémiques, des flux de SE ou des bénéfices associés (Hardaker et al., 2022 ; Othoniel et al., 2016 ; Pavan & Ometto, 2018). Or, compte tenu des différences conceptuelles entre ces notions, soulignées par le modèle en cascade, l'absence de terminologie explicite nuit à la capacité de positionner les différentes approches proposées pour l'intégration des SE en EICV.

À retenir

L'intégration des SE en EICV a gagné en popularité au cours des dernières années. Différentes approches ont été proposées pour atteindre cet objectif. Malgré cet intérêt croissant, le développement de modèles et de facteurs de caractérisation évaluant les impacts potentiels des activités humaines sur les SE reste limité par son manque de structure, de clarté, de cohérence et d'harmonisation.

CHAPITRE 4 OBJECTIFS ET MÉTHODOLOGIE GÉNÉRALE

4.1 Problématique

Les méthodologies d'évaluation EICV et les pratiques de modélisation des impacts associées évoluent continuellement. Chaque année, de nouveaux modèles de caractérisation sont développés ou améliorés, en réponse aux avancées scientifiques et à l'évolution des besoins des décideurs en matière d'indicateurs d'impacts. Ces progrès contribuent à renforcer l'EICV et, en définitive, à affiner l'interprétation des résultats d'ACV.

Depuis le consensus international de 2004 promouvant les cadres d'EICV orientés vers les problèmes et les dommages (Jolliet et al., 2004) et la révision consensuelle de ce cadre en 2017 (Verones et al., 2017), la communauté scientifique manifeste un intérêt croissant pour l'intégration des SE dans ces derniers. Malgré cet intérêt, les efforts pour développer des modèles de caractérisation des impacts sur les SE, en particulier au niveau des dommages, demeurent limités et les approches proposées souffrent d'un manque d'harmonisation. Par ailleurs, de nombreux concepts liés aux SE sont déjà impliqués dans les chaînes de cause à effet traduisant des impacts sur des AdP courantes de l'EICV, telles que la santé humaine ou la qualité des écosystèmes.

Afin de garantir la cohérence entre les modèles de caractérisation et entre les AdP, de promouvoir une meilleure prise en compte des SE en ACV, et de poursuivre la consolidation des méthodologies EICV, il est essentiel de clarifier la place des SE dans le cadre méthodologique de l'EICV.

4.2 Objectifs de la recherche

4.2.1 Objectif principal

Dans ce contexte, cette thèse vise à :

Consolider l'évaluation des impacts du cycle de vie en repensant son cadre méthodologique à travers le prisme des services écosystémiques pour soutenir le développement de modèles caractérisant l'impact des activités humaines sur les services écosystémiques.

Plutôt que de proposer un cadre rompant avec les pratiques et modèles de caractérisation actuels, cette recherche vise à offrir *une nouvelle perspective* pour concevoir le cadre EICV : le "modèle en cascade". Sur cette base, l'évaluation des impacts du cycle de vie est consolidée : la cohérence de modèles de caractérisation existants par rapport au cadre révisé est évaluée, la robustesse de certains modèles de caractérisation est améliorée, et de nouvelles voies d'impacts sont développées.

4.2.2 Objectifs spécifiques

Pour répondre à son objectif principal, cette thèse poursuit quatre objectifs spécifiques (OS) (ou "sous-objectifs") :

Objectif spécifique 1 : Examiner l'intégration des services écosystémiques en évaluation des impacts du cycle de vie. Ce premier sous-objectif consiste à analyser les enjeux liés à la prise en compte des services écosystémiques dans la modélisation des impacts du cycle de vie, en dressant un état des lieux critique des propositions existantes.

Objectif spécifique 2 : Révéler le rôle des services écosystémiques dans les modèles de caractérisation existants. Il s'agit d'examiner et de mettre en évidence les liens pouvant exister entre les modèles de caractérisation existants

en ACV et les concepts liés aux services écosystémiques par une révision du cadre d'analyse des impacts du cycle de vie.

Objectif spécifique 3 : Illustrer l'application de ce nouveau cadre par le développement de modèles de caractérisation cohérents permettant d'évaluer les impacts potentiels des activités humaines sur les services écosystémiques. Ce travail propose d'appliquer le cadre révisé pour modéliser les impacts potentiels de l'élévation du niveau des mers sur la capacité des écosystèmes côtiers à fournir des services écosystémiques.

Objectif spécifique 4 : Renforcer l'évaluation des impacts du cycle de vie à travers l'amélioration des modèles de caractérisation liant à l'utilisation de l'eau douce aux impacts sur la santé humaine. Cet objectif spécifique vise à améliorer la modélisation des impacts potentiels de la diminution de la disponibilité en eau douce sur la santé humaine, de façon cohérente avec le cadre révisé.

4.3 Méthodologie générale

4.3.1 Schéma conceptuel de la thèse

Cette thèse aborde les **enjeux** liés à l'intégration des SE en EICV, propose un **cadre méthodologique** structurant pour accompagner cette intégration et consolider l'EICV, et explore ce dernier à travers une série d'**applications**.

Dans ce contexte, la thèse est structurée autour de 5 articles scientifiques [1], adressant les 4 sous-objectifs énoncés ci-dessus. La figure 4.1 schématise la manière dont la recherche et les différentes contributions sont articulées et présentées dans ce manuscrit.

1. Deux articles publiés, deux articles soumis pour révision, et un article sous la forme d'un chapitre de la présente thèse, visant a etre également soumis a un journal avec comité de relecture.

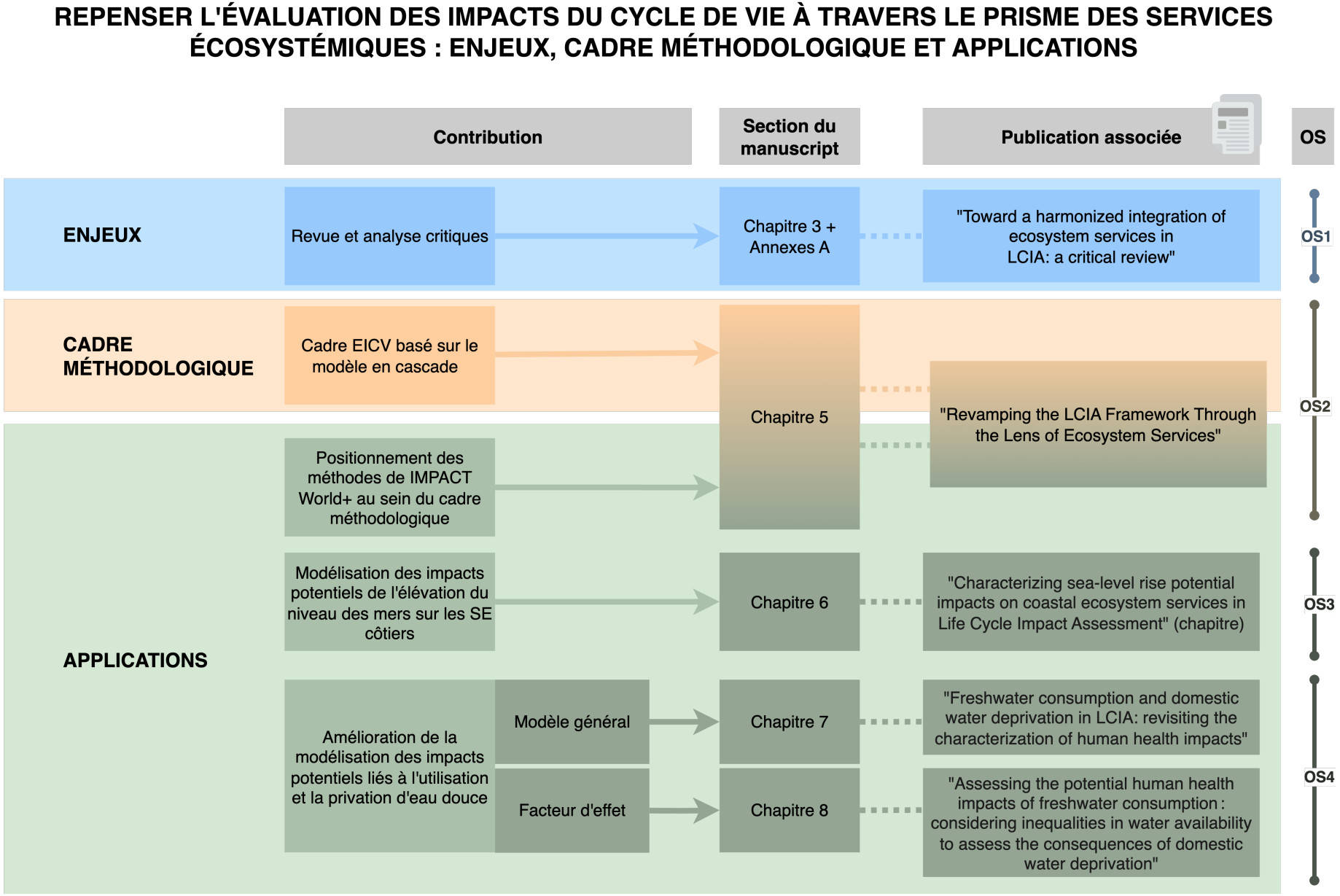


FIGURE 4.1 Représentation schématique de la thèse. OS : Objectif Spécifique.

4.3.2 OS1 : Enjeux - analyse critique de l'intégration des services écosystémiques en évaluation des impacts du cycle de vie

L'article *"Toward a harmonized integration of ecosystem services in LCIA : a critical review"* propose une analyse critique des enjeux liés à la prise en compte des services écosystémiques dans la modélisation des impacts du cycle de vie, en dressant un état des lieux critique des propositions existantes. Il établit un socle de connaissances fondamentales sur les SE, à partir duquel les propositions d'intégration en EICV sont analysées.

Cette revue critique met en lumière **un manque de structure, de cohérence et de clarté dans les propositions d'intégration des SE en EICV**. En identifiant ces limites et en clarifiant les concepts fondamentaux associés aux SE, l'article pose les bases pour le développement d'un cadre méthodologique structurant intégrant les SE en EICV.

Pour des raisons de clarté et de fluidité du manuscrit, une synthèse de cet article est présentée dans le chapitre précédent 3. Le lecteur est invité à consulter l'annexe A pour une lecture complète de l'article.

4.3.3 OS2 : Cadre méthodologique/Application - Mise en évidence du rôle des SE dans les modèles de caractérisation existant

Sur la base de l'OS1, l'article *"Revamping the LCIA Framework Through the Lens of Ecosystem Services"* présenté dans le chapitre 5 propose un cadre EICV complet revisité mettant en évidence le rôle des concepts liés aux SE dans la majorité des modèles de caractérisation existants. Construit autour du *modèle en cascade* des SE, il illustre l'ensemble des chaînes de cause-à-effet impliquées en EICV.

La section 5.2.3 de l'article positionne l'ensemble des chaînes de cause-à-effet des modèles de caractérisation de la méthodologie IMPACTWorld + (Bulle et al., 2019) au sein du cadre EICV revisité proposé dans l'article. Ce travail permet d'illustrer le fonctionnement du cadre EICV revisité et d'évaluer la cohérence et la complétude des modèles de caractérisation existants.

4.3.4 OS3 : Application - modélisation des impacts potentiels de l'élévation du niveau des mers

L'article *"Characterizing sea-level rise potential impacts on coastal ecosystem services in Life Cycle Impact Assessment"* présenté au Chapitre 6 applique le cadre EICV revisité en explorant l'une de ses chaînes de cause-à-effet ; il propose un modèle et des facteurs de caractérisation évaluant les impacts potentiels de l'élévation du niveau des mers et de l'accentuation des événements extrêmes sur la capacité des écosystèmes côtiers à fournir des services écosystémiques.

4.3.5 OS4 : Application - consolidation de la modélisation des impacts potentiels sur la santé humaine liés à l'utilisation et la privation d'eau douce

L'article *"Freshwater consumption and domestic water deprivation in LCIA : revisiting the characterization of human health impacts"*, présenté au Chapitre 7 propose de consolider la modélisation des impacts potentiels liés à la diminution de la disponibilité d'eau douce sur la santé humaine, de façon cohérente avec le cadre EICV revisité. Pour ce faire, l'article discute et améliore le modèle de caractérisation proposé par Boulay et al. (2011) ; les facteurs de sort, d'exposition et d'effet sont revisités. En plus des améliorations qu'il propose, cet article met en lumière les limites méthodologiques du facteur d'effet traduisant les impacts sur la santé humaine induite par la privation d'eau douce.

En réponse à ces limites, l'article *"Potential impacts of freshwater consumption on human health : considering freshwater use inequalities to assess domestic water deprivation"*, présenté au Chapitre 8 propose une nouvelle approche pour calculer le facteur d'effet. Un second jeu de facteurs de caractérisation intégrant ce facteur d'effet revisité est développé.

CHAPITRE 5 ARTICLE 1 : REVAMPING THE LCIA FRAMEWORK THROUGH THE LENS OF ECOSYSTEM SERVICES

Revamping the LCIA framework through the lens of ecosystem services

Laura Debarre, Titouan Greffe, Catherine Lalongé, Laure Patouillard, Benedetto Rugani, Jana Schluenss, Rutger de Wit, Cécile Bulle, Manuele Margni

submitted to the International Journal of Life Cycle Assessment in April, 22nd 2025

5.1 Présentation de l'article

La revue critique présentée en annexe A et résumée dans la section 3.2 souligne le manque de cohérence et de clarté méthodologique dans les propositions visant à intégrer les SE en EICV. En réponse à ce constat, cet article propose de repenser le cadre EICV pour structurer et soutenir le développement cohérent de modèles de caractérisation des impacts sur les SE. Pour ce faire, il mobilise le "modèle en cascade" pour révéler le rôle des SE dans la majorité des modèles de caractérisation.

Cet article a été co-écrit par Laura Debarre (première auteure), Titouan Greffe, Catherine Lalongé, Laure Patouillard, Benedetto Rugani, Jana Schluenss, Rutger de Wit, Cécile Bulle et Manuele Margni. Il a été soumis à l'*International Journal of Life Cycle Assessment* le 22 avril 2025. Le matériel supplémentaire associé à l'article est disponible en Annexe B.

Les résultats de ce projet ont été présentés à l'occasion de la *34^{ieme} réunion annuelle de la SETAC Europe* qui s'est tenue du 5 au 9 mai 2024 à Séville (Espagne).

5.2 Manuscript

Abstract

Purpose. The integration of ecosystem services into the impact assessment phase, through the development of characterization models and factors that reflect potential impacts on ecosystems' capacity to provide services, is gaining momentum within the life cycle assessment community. Progress remains limited due to the absence of a clear, standardized methodology for developing ecosystem services-related characterization models. A consistent life cycle impact assessment framework to support such integration is lacking. This research posits that ecosystem services concepts are already implicitly embedded within existing life cycle impact assessment models. To clarify their role and offer a consistent, structured basis for future model developments, this work proposes a revamped life cycle impact assessment framework through the lens of ecosystem services.

Method. The proposed framework involves a redefinition of the traditional areas of protection. Starting from the areas of protection proposed by Verones et al. (2017), the framework introduces a revised "Ecosystem and Resources services" area of protection, which aggregates potential impacts on the instrumental value of social-ecological systems. The Common International Classification of Ecosystem Services (CICES v5.2) is adopted to encompass a broad range of geophysical and biophysical ecosystem services.

Results and discussion. The revamped life cycle impact assessment framework is structured around the ecosystem services cascade model. It identifies causal pathways linking human activities to changes across different stages of the cascade and, ultimately, to the areas of protection. The framework considers the role of adaptation strategies in response to declining ecosystem services flows : adaptation strategies

induce adaptation costs and indirect impacts due to boundaries' expansion while non-adaptation strategies lead to a loss of services and potential impacts on human health. To assess the applicability of the framework, existing characterization models from the IMPACT World+ methodology are mapped within this new structure. This study initiates a broader reflection on the need to revisit ISO 14044, particularly regarding the delineation between the technosphere and ecosphere, and the characterization of elementary flows. It also calls for a re-examination of the definition and scope of areas of protection. The proposed framework helps identify conceptual and methodological gaps in current approaches, such as those in IMPACT World+, and offers a foundation for integrating ecosystem disservices into life cycle impact assessment.

Conclusion and recommendations. By reframing the life cycle impact assessment structure through the perspective of ecosystem services and related concepts, this work establishes a conceptual basis for the development of more coherent, comprehensive, and operational life cycle impact assessment models and methodologies.

5.2.1 Introduction

The concept of Ecosystem Services (ES) emerged in the late 70's but gained significant attention in 1997 when Costanza et al. (1997) defined it as "the benefits people derive from functioning ES". While several classification systems exist to identify and categorize ES, the Common International Classification of Ecosystem Services (CICES) (Haines-young, Paddocks & Lane, 2023) is one the most used in European-based ES studies (Costanza et al., 2017 ; Czúcz et al., 2018). CICES divides these benefits into three classes : provisioning ES (i.e. provisioning of biotic or abiotic resources used for various purposes), regulation & maintenance ES (e.g. dilution of waste in freshwater ecosystems or pest control) and cultural ES (e.g. opportunities for recreational activities, or for education and training). The ES concept underscores the dependence of human activities on natural capital and highlights the interconnection between human well-being and ecosystem functioning (IPBES, 2019). To illustrate how ecosystems provide ES that ultimately benefit human well-being, conceptual fra-

meworks such as the cascade model have been developed (Haines-Young & Potschin, 2010). The latter connects natural ecological processes with the social and economic systems that depend on them. The model illustrates the steps from ecosystems' processes and structures to ecosystem functions, that generate ES, and ultimately support human well-being. Human activities not only depend on ES but can also significantly impact their supply (Callesen, 2016 ; IPBES, 2019). The cascade model can be used to illustrate how human pressures on ecosystems can influence the provisioning of ES, and therefore the benefits people derive from them (Haines-Young & Potschin, 2010).

Since the ES notion was first introduced in the academic literature, additional frameworks have evolved, broadening the concept of ES *sensu stricto* (referring to instrumental values only, defined as "means to achieve human ends or satisfy human preferences" (Himes et al., 2024)) to incorporate *relational values* that emphasize relationships and connections between people and nature. Nature's contributions to people (NCP), therefore, offers a more comprehensive framework integrating diverse perspectives and values, including cultural, social, and emotional connections to the natural world (Díaz et al., 2018 ; Himes & Muraca, 2018 ; Pascual et al., 2023). In the rest of this article however, the focus is put on ES used *sensus stricto*.

Human pressures on ecosystems exert significant impact on the provision of ES (Costanza et al., 2014 ; Markandya, 2015). Recognizing and mitigating the influence of human activities on ecosystems' capacity to provide ES is essential to promote sustainable management practices (Natural Capital Coalition, 2016). It is, therefore, imperative to integrate information on ES into decision-making tools and processes to promote sustainable decisions (Daily et al., 2009 ; Othoniel et al., 2016 ; Value of Nature to Canadians Study Taskforce, 2017).

In this context, a growing number of scholars call for the inclusion of ES concepts within Life Cycle Assessment (LCA) (Vanderwilde & Newell, 2021). LCA is a widely adopted decision-support tool standardized under ISO 14044 (ISO, 2006) which

enables the assessment of the potential environmental impacts associated with various products and consumption systems. An LCA is iteratively conducted in four steps : the goal and scope definition, which outlines the purpose, boundaries, and functional unit of the study ; the inventory analysis, where all relevant inputs (e.g., raw materials, energy) and outputs (e.g., emissions, waste), called elementary flows, are quantified across the product's life cycle ; the impact assessment, which translates inventory data into potential environmental impacts ; and the interpretation, where the results are analyzed to draw conclusions, identify limitations, and provide recommendations for decision-making (ISO, 2006).

To conduct the impact assessment phase, characterization models and factors are developed by LCIA modelers to represent and quantify the potential environmental impacts induced by elementary flows on various impact categories (Hauschild, Rosenbaum & Olsen, 2018). These categories can be midpoint-oriented (expressed by an indicator selected at any point along the cause-effect chain prior to the end of it) or damage-oriented (expressed by an indicator measuring the damage at the end of the cause-effect chain). LCIA methodologies such as LC-IMPACT (Verones et al., 2020), ReCiPe (Huijbregts et al., 2017) or IMPACTWorld+ (Bulle et al., 2019), compile several coherent characterisation models and factors to provide a comprehensive coverage of impact categories.

The concept of ES can be incorporated into each phase of the LCA framework (Debarre et al., s. d.-b ; Rugani et al., 2023) [1]. The integration of ES in the impact assessment phase, through the development of characterization models and factors accounting for the ecosystems' capacity to provide services, is an increasingly prominent topic among the Life Cycle Impact Assessment (LCIA) community (Bulle et al., 2019 ; Debarre et al., s. d.-b ; Vanderwilde & Newell, 2021 ; Verones et al., 2017). However, few ES are currently covered in LCIA methods, and the different approaches that have been proposed lack consistency (Alejandre, Bodegom & Guinée, 2019 ; Debarre et al., s. d.-b).

1. Debarre et al. s.d.-b refers to the critical review provided in Annexe A

Without consistent characterization models that explicitly assess the potential impacts of human activities on ecosystems' capacity to deliver services, the profile of environmental impacts presented to decision-makers can be fragmented. Consequently, decisions may be biased toward options that inadvertently cause greater harm to the supply of ES, undermining sustainability goals (Chaplin-Kramer et al., 2017).

Various proposals have been done to integrate ES within LCIA (Debarre et al., s. d.-b). While some experts propose to highlight the sustainability of the balance between supply and demand for ES (e.g., X. Liu et Bakshi (2018)), other scholars suggest addressing the potential impacts of humans' activities on the capacity of ecosystems' to provide services, such as the costs to society to compensate for them (e.g., Cao et al. (2015)).

However, the assessment of potential impacts on ES within LCIA remains hindered by the absence of a clear, standardized approach to integrate ES into characterization models and, more broadly, by the lack of a related globally accepted LCIA framework (Debarre et al., s. d.-b). The critical review of Debarre et al. (s. d.-b) underlines four main limitations of the current proposals to integrate ES in LCIA.

The authors reported **(i) a lack of clarity of the position of ES in relation to the different Areas of Protection (AoP)**, which represent entities that are important to protect due to their high value to society (Schaubroeck & Rugani, 2017). Current common AoP include Human health, Ecosystem quality and Natural resources (Huijbregts et al., 2017).

They aggregate damage-oriented impact scores. Since ES depend on the integrity of ecosystems and condition human well-being by providing suitable living conditions and resources to support human activities, there are close links between ES concepts and the different AoP. However, these relationships are currently not sufficiently recognized in LCIA (Schaubroeck & Rugani, 2017).

Some authors have suggested developing an AoP specific to ES (Hardaker et al.,

2022), dividing ES into different AoP depending on their class (De Luca Peña et al., 2022 ; Rugani et al., 2019), or integrating ES and natural resources within a common AoP (Bulle et al., 2019 ; Verones et al., 2017), aggregating potential impacts on instrumental values. In this context, Bulle et al. (2019) define a "Resources and Ecosystem Services" AoP.

The absence of well-established links between ES and Natural resources has arguably contributed to **(ii) a conceptual and methodological divide between provisioning ES and natural resources within the LCIA community**. Although natural resources originate from provisioning ES, these aspects are typically addressed by two distinct LCIA research streams ; while water is currently considered natural abiotic resources in LCIA, it can be considered as the outcome of a water provisioning ES. In contrast, the ES community has made efforts to harmonize these elements within a coherent framework, notably through their integration into a unified classification system in the latest version of the Common International Classification of Ecosystem Services (version 5.2) (Haines-young, Paddocks & Lane, 2023). In not aligning with this trend, the conceptual separation of ES and natural resources among LCIA experts fosters the development of divergent objectives and inconsistent characterization models. The issues related to the diversity of modeling practices and objectives have already been encountered within the natural resource LCIA community (Debarre et al., s. d.-b ; Dewulf et al., 2015 ; Greffe, Margni & Bulle, 2022). However, the latter is evolving towards a consensus to characterize the potential impacts of resource dissipation, referring to the loss or degradation of the functionality of a resource, rendering it inaccessible or unusable for future use (Beylot et al., 2020 ; Helbig, Thorenz & Tuma, 2020)

Debarre et al. (s. d.-b)'s critical review also identifies **(iii) a lack of clarity and transparency regarding damage level metrics and modeling choices** proposed by LCIA modelers to value ES, as well as on the scope covered by these metrics. Various valuation methods have been explored, including physical approaches (e.g.

exergy, emergy (Y. I. Zhang, Anil & Bakshi, 2010)) and monetary approaches (Cao et al., 2015 ; Othoniel et al., 2019). Although monetary approaches offer the advantage of covering all types of ES and providing easily interpretable measures for the general public and decision makers (L. M. Brander et al., 2024 ; Kallis, Gómez-Baggethun & Zografos, 2013 ; Ruckelshaus et al., 2015), adaptation and damage costs represent the results of different cause-effect chains ; adaptation costs reflect the consequences in scenarios where the service is replaced or compensated, while damage costs (lost services) are faced in scenarios where the service is not restored. Therefore, it is essential for LCIA modelers to ensure transparency regarding their choice of valuation method and to maintain consistency between the selected monetary valuation approach and the cause-effect relationships underpinning their characterization models.

More generally, the authors underscore **(iv) the lack of harmonization and of completeness of ES integration in LCIA**.

Approaches aimed at assessing the potential impacts on ecosystem capacity to provide services are diverse and, at times, markedly different (Debarre et al., s. d.-b). Building on Hardaker et al. (2022), Vanderwilde et Newell (2021) and Rugani et al. (2023), these research proposals can be categorized into three main clusters : 1- the land use cluster, which focuses on developing ES-related characterization models and factors to assess the impacts of land use on ES, 2- the biodiversity and ES cluster, where scholars aim to link potential impacts on functional biodiversity to changes in the supply of ES and 3- the dynamic modeling group, which employs dynamic integrated models to quantify potential impacts on ES by accounting for the interactions between ecological and social systems. The cascade model proposed by Haines-Young et Potschin (2010) is frequently identified within the different clusters as a valuable framework for the development of characterization models (Alshehri et al., 2023 ; Lago-olveira, Moreira & Gonz, 2025 ; X. Liu et al., 2020 ; Maia de Souza et al., 2018 ; Pavan & Ometto, 2018 ; Rugani et al., 2019). These various approaches address different challenges associated with ES assessment, such as data availability, spatial and temporal dynamics, and are subject to different limitations, including

oversimplification, limited applicability, and high data requirements (Othoniel et al., 2016 ; Y. I. Zhang, Singh & Bakshi, 2010).

However, these proposals fail to provide comprehensive and consistent coverage of the assessment of human activities' impacts on ecosystems' capacity to provide services. Only a limited range of ES is addressed (Alejandre, Bodegom & Guinée, 2019 ; Vanderwilde & Newell, 2021), and a narrow set of environmental pressures is considered (Hardaker et al., 2022 ; Rugani et al., 2022). Current characterization models focus mainly on land use impacts, with an emphasis on a small subset of terrestrial ES (Alejandre, Bodegom & Guinée, 2019 ; Debarre et al., s. d.-b ; Hardaker et al., 2022 ; Rugani et al., 2019 ; Soldati et al., 2023 ; Vanderwilde & Newell, 2021).

In conclusion, the authors "call for the development of a comprehensive, consistent, and globally accepted framework to facilitate the integration of potential impacts on ecosystems' capacity to provide services at the damage level in LCIA" (Debarre et al., s. d.-b).

This research argues that ES concepts are already implicitly embedded in existing LCIA frameworks and models, even if they are not considered systematically or explicitly.

Building on this hypothesis, this study proposes a "revamped LCIA framework" based on the ES cascade model. This framework clarifies the interconnections between ES concepts and LCIA models and provides a comprehensive structure for the development of cause-effect chains and characterization models assessing potential impacts on ecosystems' capacity to provide services.

The specific objectives of this research are to : i) reconcile ES and natural resources within a revamped LCIA framework, ii) establish a clear and consistent terminology around ES concepts (e.g., 'ecosystem functions', 'ecosystem service potential', 'ecosystem service flow'), iii) illustrate the application of the revamped framework across all characterization models of the IMPACTWorld+ methodology (Bulle et al., 2019),

and iv) identify remaining gaps and research priorities in LCIA modelling through the lens of this novel framework.

5.2.2 Methods

The following method section presents the body of knowledge, from both the ES and LCA communities, used to understand the current LCIA framework through the lens of ES.

Reframing the existing LCIA AoP

Verones et al. (2017) shaped a consensus-based framework for LCIA methodologies.

More specifically, the authors suggest defining AoP based on types of systems (human and human-made versus natural) and types of values ("intrinsic, instrumental, cultural"). This latter value classification was further refined as 'intrinsic, instrumental, relational' in Debarre et al. (s. d.-b). Since different perspectives may lead to varying interpretations of what constitutes an AoP (Schaubroeck & Rugani, 2017), Verones et al. (2017)'s approach reflects a degree of subjectivity. Recognizing this inherent subjectivity, the present work seeks to build upon and further develop the framework initially proposed by Verones et al. (2017).

In Verones et al. (2017)'s proposition, the Human Health (HH) and Ecosystem Quality (EQ) AoP, respectively, seek to represent the intrinsic value of human and human-made systems, and of natural systems. This proposal, by suggesting impact indicators that reflect intrinsic value, implies that such value could be quantified. However, this assumption conflicts with the definition provided by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES), which characterizes intrinsic value as "the value inherent to nature, independent of human experience and evaluation, and thus beyond the scope of anthropocentric valuation approaches" (Díaz et al., 2015). The concept of intrinsic value remains a subject of extensive debate within the field of environmental ethics (Batavia & Nelson, 2017 ; Morrow, 2024), highlighting the need for a more nuanced discussion on its use within the LCIA

framework (Debarre et al., s. d.-b). Building on the analysis of Debarre et al. (s. d.-b) (supplementary information section A), a more in-depth exploration of this issue is presented in the Discussion section. For the remainder of this section, however, the term "intrinsic" will be used with an awareness of its conceptual limitations.

Tableau 5.1 Areas of Protection, defined by dividing types of components of the social-ecological system by types of values. Adapted from Verones et al. (2017) and Debarre et al. (in preparation). Social-economic systems combine both human/human-made systems and natural systems (McGinnis et al. 2014). The scope of the present work is limited to intrinsic and instrumental values only ; the integration of relational values in LCIA should be discussed in specific studies.

Value type / **Component of the social-ecological system**	**Intrinsic**	**Instrumental**	**Relational**
Human and human-made systems	Human health	Ecosystem and resources services (encompassing resources services and ecosystem services *sensus stricto*)	
Natural systems	Ecosystem quality		

As per Table 5.1, one can distinguish damage-oriented potential impacts on intrinsic, instrumental, and relational values (Debarre et al., s. d.-b). Relational values are still an emerging concept and their integration in the LCIA AoP requires further research efforts ; they fall outside the scope of this work.

ES *sensus stricto*, including the services provided by the resources (associated with provisioning ES), serve as the foundation of instrumental values. However, the types of entities possessing instrumental value are diverse and span both human-made and natural systems. Defining separate AoP to account for services derived from human-made systems on the one hand, and natural systems on the other, would share the same ultimate purpose : to protect the services that are provided to society. Therefore, such a separation may not enhance the decision-making process, as it could introduce unnecessary complexity without yielding additional practical benefits or clarity for decision makers. Given the difficulty in clearly categorizing these entities

as human-made or natural, it appeared more appropriate to consolidate them within a unified AoP, thus simplifying the framework and ensuring greater consistency in the assessment of instrumental values. In this context, an AoP "Ecosystem and resource services" was proposed to aggregate potential impacts on instrumental values of social-economic systems (combining human / human-made systems and natural systems (McGinnis & Ostrom, 2014)). The name of this AoP differs from the suggestions by Verones et al. (2017) and Bulle et al. (2019) ("[Natural] Resources and Ecosystem services") to emphasize the instrumental dimension of resources.

Selecting an ES classification reconciling natural resources and provisioning ES

Various ES classification systems have been developed in the last decade (Villamagna, Angermeier & Bennett, 2013). To align with the objectives of the proposed framework, the associated ES classification should be consensual within the ES community and effectively integrate natural resources within the broader ES framework.

The latest version of the CICES classification (v.5.2) (Haines-young, Paddocks & Lane, 2023) encompasses both geophysical and biophysical ES, defining provisioning services as including both biotic resources (e.g. plants, genetic material) and abiotic resources (e.g., water, minerals). By incorporating natural resources within the wider framework of ES, the CICES v5.2 classification bridges the conceptual divide between natural resources and ES. Given the widespread acceptance of CICES and the holistic nature of its latest version (v5.2), the latter was adopted in the rest of the paper.

Apprehending cause-effect chains and characterization models via the cascade model

Burkhard et al. (2014)'s cascading framework and application to LCIA cause effect chains. The cascade model (Haines-Young & Potschin, 2010) has been widely adopted by the ES community and is the basis of the CICES classification (Haines-young, Paddocks & Lane, 2023). The cascade model was declined in

various forms, including the conceptual model by Burkhard et al. (2014) (see figure 5.1).

In Burkhard et al. (2014)'s framework, the “functions” of an ecosystem (i.e. the natural processes and interactions within an ecosystem) result from the combination of its structure (physical and biological components) and processes (dynamic interactions among these components). The structure and processes underpin the 'ecological integrity' of the ecosystem, referring to its self-organization (Müller, 2005). According to Burkhard et al. (2014), the functions of an ecosystem "are strongly influenced by land cover and land use".

The concept of 'ecological integrity' has been widely debated (Roche & Campagne, 2017) for its vagueness, leading some ecologists to propose the term 'ecosystem state' as a clearer alternative (González, 2023 ; Hughes, 2019). The term 'ecosystem state' more effectively captures : (1) the 'ecological health' of the structure and processes of an ecosystem in a specific location, (2) the recognition that ecosystems can exist in multiple states, and (3) the dynamic transitions between states, governed by regime shifts and tipping points (González, 2023 ; Hughes, 2019).

The “functions” of the ecosystem support the 'supply' of ES that can be decomposed into three parts : (1) an ES potential (i.e., the 'hypothetic maximum yield of selected ecosystem services') that is 'comparable to natural capital stocks' (Burkhard et al., 2014). In a context of 'demand' for ES, the potential ES can be combined with (2) additional inputs (“social, human, financial and manufactured capital assets”) to generate (3) 'ES flows' (referring to 'de facto consumed goods and services in ecosystems'). For example, the provision of fish (ES flow) is made possible by the combination of a stock of fish (ES potential) and the deployment of material and human resources for fishing (additional inputs). Additional outputs and ES flows are relevant only if there is a demand for the ES. Then, ES flows contribute to human well-being benefits, which can be assessed using different valuation approaches.

With their conceptual model, Burkhard et al. (2014) acknowledge the influence of the state of an ecosystem on the functions it can provide. Due to the "cascading" na-

ture of the model, pressures on the ecosystem functions can jeopardize the potential supply of ES and, in a context of demand for this service, can result in a loss of ES flow.

Such interactions are highly relevant in cause-effect chains developed by LCIA modelers, used as supports for the development of characterization models and factors. Furthermore, the terminology proposed in Burkhard et al. (2014)'s model offers a clear distinction between key ES concepts, such as the ES potential and the ES flows, helping to clarify a common source of confusion among the LCIA modelers (Debarre et al., s. d.-b).

Therefore, in this article, Burkhard et al. (2014)'s model was used to examine (current and potential future) characterization models from an ES-related perspective. The revamped framework was developed acknowledging that human activities, in the form of elementary flows (or intermediate flows, see 5.2.4) can lead to a variety of environmental mechanisms linking up at different levels of Burkhard et al. (2014)'s cascade model. The following paragraphs describe these anchoring pathways.

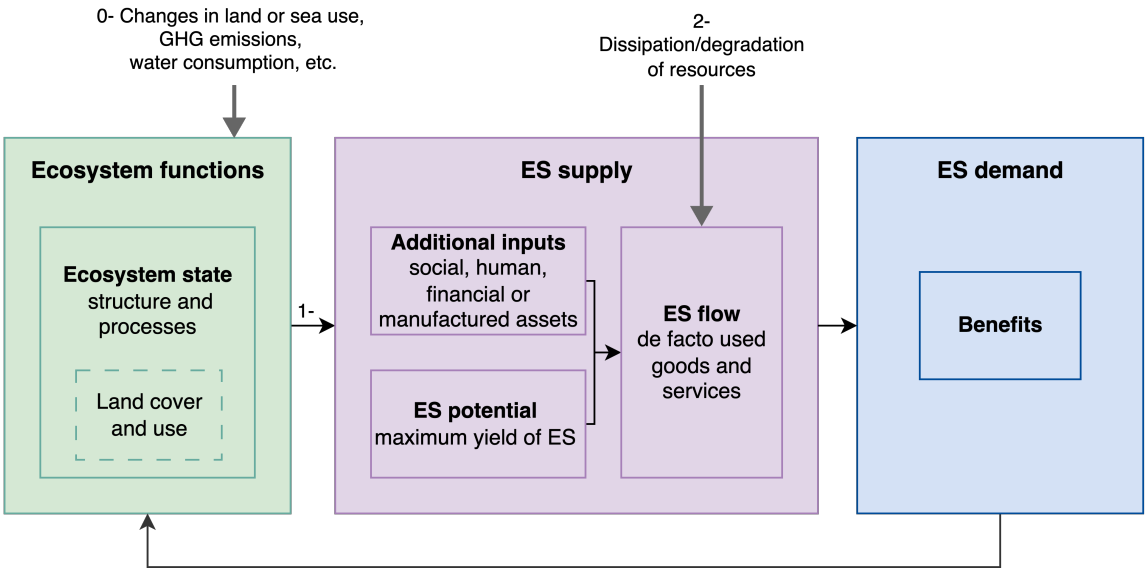


FIGURE 5.1 Conceptual cascading model linking ecosystem functions, ES supply and ES demand. Adapted from Burkhard et al. (2014)

Illustrating how human activities can alter the ecosystem functions. The revamped framework illustrates that human activities can alter the functions of an ecosystem through transformation or stress (De Luca Peña et al., 2024 ; Debarre et al., s. d.-b).

The transformation of an ecosystem, via a change of land use or cover, directly alters its state, and therefore its functions. The same mechanism can be applied to marine ecosystems by changing sea use or cover.

In addition, broader types of pressure (or "stress") can affect the ecosystem state by altering its structures and/or processes. Such pressures include various environmental interventions, such as greenhouse gas emissions that disrupt climate stability (Grimm et al., 2013), water overconsumption, or the release of substances or physical and biological agents (Müller & Burkhard, 2012). Similarly, the extraction of renewable resources, such as fish, timber, or water, can impact the state of the ecosystem when the extraction rate exceeds the natural regeneration capacity of these resources

(Crenna, Sozzo & Sala, 2018). For example, overfishing can destabilize food webs and disrupt predator-prey relationships (Stanford-Clark, Loiseau & Helias, 2024).

The proposed framework reports the influence of transformation and stress on ecosystem functions as per pathway 0 in fig.5.1. Due to the 'cascading' nature of the model, changes in ecosystem functions can propagate through the chain (path 1) and ultimately affect the ES supply. If there is a corresponding demand for the affected ES, a change in the supply results in a shift in the ES flow. In cases where demand is absent, we suggest considering changes in ES supply potential as potential impacts for future generations.

Illustrating how human activities can alter flow of services due to resources' loss of functionality. Another pathway of the framework has been defined to account for mechanisms linked to the loss of functionality of resources.

The loss of functionality of resource services occurs within the technosphere due to the dissipation or degradation of resources (Greffe, Margni & Bulle, 2022). This impact mechanism does not affect the functions and structure of the ecosystem, but directly alters the flow of services provided by resources.

Debarre et al. (s. d.-b) highlighted a frequent discrepancy between the typology of services provided by resources considered by LCIA method developers (e.g. a service of "mobility" supported by materials, such as aluminum in the engine of a vehicle) and the services categorized in the CICES classification ; at its most detailed level (the class), the CICES classification only refers to broad types of ES (e.g., "provisioning ES : mineral substances used for material purposes, including geophysical support (foundations)").

Therefore, to accurately characterize the loss of services provided by resources, LCIA modellers must develop a refined understanding of how specific provisioning ES -and their related resource services - are useful to society.

To address how human activities can alter the flow of services due to the loss of functionality of resources, the framework identifies a specific impact pathway that

links resource dissipation and/or degradation to ES flows (see pathway 2 in 5.1).

Defining the influence of adaptation or non-adaptation strategies following the decrease of ES flows

The revamped framework was developed based on the assumption that changes in an ES flow can trigger both adaptation and non-adaptation responses, each leading to consequences of varying nature (see figure 5.2). These responses may vary regionally. The term "adaptation" is usually used to refer to anticipatory or reactive measures implemented to adjust to environmental impacts (IPCC, 2022). In the context of our framework, the term "adaptation" specifically denotes reactive measures aimed at replacing or compensating for a lost ES flow. It does not include strategies designed to avoid or mitigate ES loss upstream.

The framework operates on the premise that if the population adapts to a change in ES flow, it must bear the associated economic adaptation costs (see path a, Figure 5.2). The degree of adaptation can vary, ranging from complete to partial. Furthermore, the framework assumes that adaptation measures implemented to cope with the loss of service may induce indirect impacts due to boundary expansion (pathway d). This concept refers to the potential impacts generated by the emissions or resource extraction (i.e. elementary flows) associated with adaptation activities intended to replace or compensate for the lost service. This mechanism can be interpreted as a “feedback loop” to elementary flows accounting, aligning with the approaches discussed in Boulay et al. (2011), Cao et al. (2015), or Weidema et al. (2018).

In contrast, if the population does not adapt to a change in ES flow, the instrumental values associated with the lost service (also referred to as a damage) are directly accounted for as an impact through pathway b. This impact represents the "adverse effects of [a stressor] that cannot be or have not been avoided through mitigation or managed through adaptation efforts" (Geest et al., 2019). Following this logic, if adaptation measures only partially avoid the loss of ES flow, residual damages must

still be considered through pathway b.

This impact should be interpreted as an integrated loss of ES flow over time, considering a relevant time horizon for decision making. The choice of this time horizon is a normative decision. For example, the Intergovernmental Panel on Climate Change (IPCC, 2021) often uses a 100-year horizon to guide policy makers on climate issues, similar to some characterization models within LCIA methodologies such as ReCiPe (Huijbregts et al., 2017) or IMPACTWorld+ (Bulle et al., 2019).

Whether through the valuation of the integrated loss of ES flow (in the case of non-adaptation) or the costs incurred to replace or compensate for it (in the case of adaptation), the framework describes pathways a and b of figure 5.2 as mechanisms for assessing impacts on instrumental values.

To facilitate the comparison and aggregation of service losses and adaptation costs, we recommend employing monetary valuation methods to assess service losses. Although the economic valuation of ES is subject to various limitations and criticisms (see SI for further discussion), economic metrics provide a standardized approach to expressing "the importance of changes in the provision of ES directly in terms of human welfare and uses a common unit of account (i.e., money) so that values can be communicated easily and directly compared across other goods, services, investments, and impacts in the economy" (L. M. Brander et al., 2024). In this regard, economic valuation serves as a pragmatic tool to convey the importance of ES to decision makers, to capture the full spectrum of ES categories, and to enable the aggregation of damages and adaptation costs (Pascual et al., 2010 ; Spash, 2013 ; Valverde et al., 2022). However, since adaptation costs and damage represent different cause-effect chains, their aggregation must be approached with caution and interpreted in full awareness of these methodological differences (Valverde et al., 2022).

Beyond impacts on instrumental values, the framework reflects the idea that the absence of adaptation to changes in ES flows can also induce impacts on human health (pathway c).

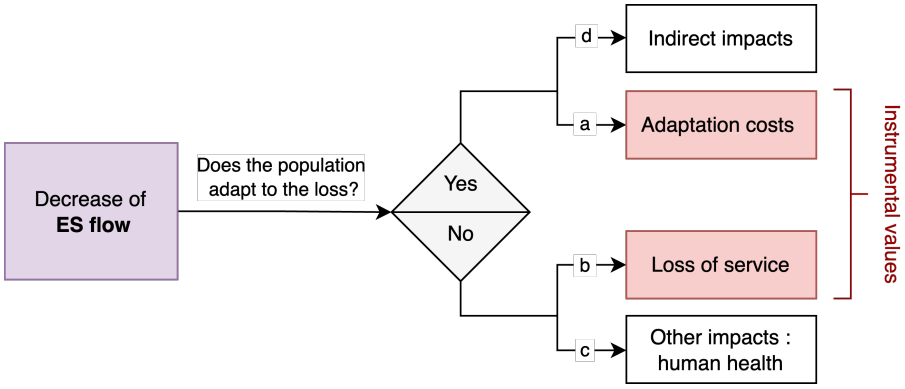


FIGURE 5.2 Adaptation strategy following a decrease of ES flow and its consequences.

Illustrating impact pathways associated with the HH and EQ AoP

Cause effect chains associated with the Ecosystem quality AoP already consider changes in ecosystem functions, albeit mainly implicitly. For example, marine acidification, a process related to a decrease in ocean pH (a structural change within the ecosystem), can adversely affect marine species, which are later captured within the EQ AoP framework (Azevedo et al., 2015). However, the modeling of such environmental mechanisms is mainly implicit in LCIA characterization models.

This framework therefore clarified such environmental mechanisms by identifying a pathway linking elementary flows to changes in ecosystem functions and to their potential impacts on the EQ AoP.

Similarly, an impact pathway was created to represent the environmental mechanisms linking inventory flows to changes in ecosystem functions that result in potential effects on human health.

Furthermore, efforts were made to illustrate environmental interventions and relative cause-effect chains that typically occur entirely within the technosphere, and have no influence on the ecosystem while impacting human health.

A revamped LCIA framework building on the cascade model and adaptation strategies

The revised cascade model (Figure 5.1) was combined with the adaptation strategies framework (Figure 5.2) to establish a comprehensive framework linking elementary flows to the Areas of Protection (AoPs) discussed in Section 2.1.

Positioning the IMPACT World+ (Bulle et al. (2019) characterization models within the revamped framework

To evaluate the operability and applicability of this framework, all characterization models associated with the 43 impact categories (both implemented and interim) from the IMPACT World+ methodology (Bulle et al., 2019) were mapped within the new framework. To this end, the publications of Bulle et al. (2019), their supplementary material, and the publications related to the different characterization models were reviewed. For each characterization model (e.g. climate change impacts on human health), the different impact covered (e.g. cardiovascular diseases, malaria, diarrhoea, floods and malnutrition) were identified. Building on the description of the mechanisms involved, pathways were identified building on expert knowledge from LCIA method developers to position the cause-effect chains within to the revamped framework.

5.2.3 Results

Figure 5.3 shows the comprehensive revamped LCIA framework.

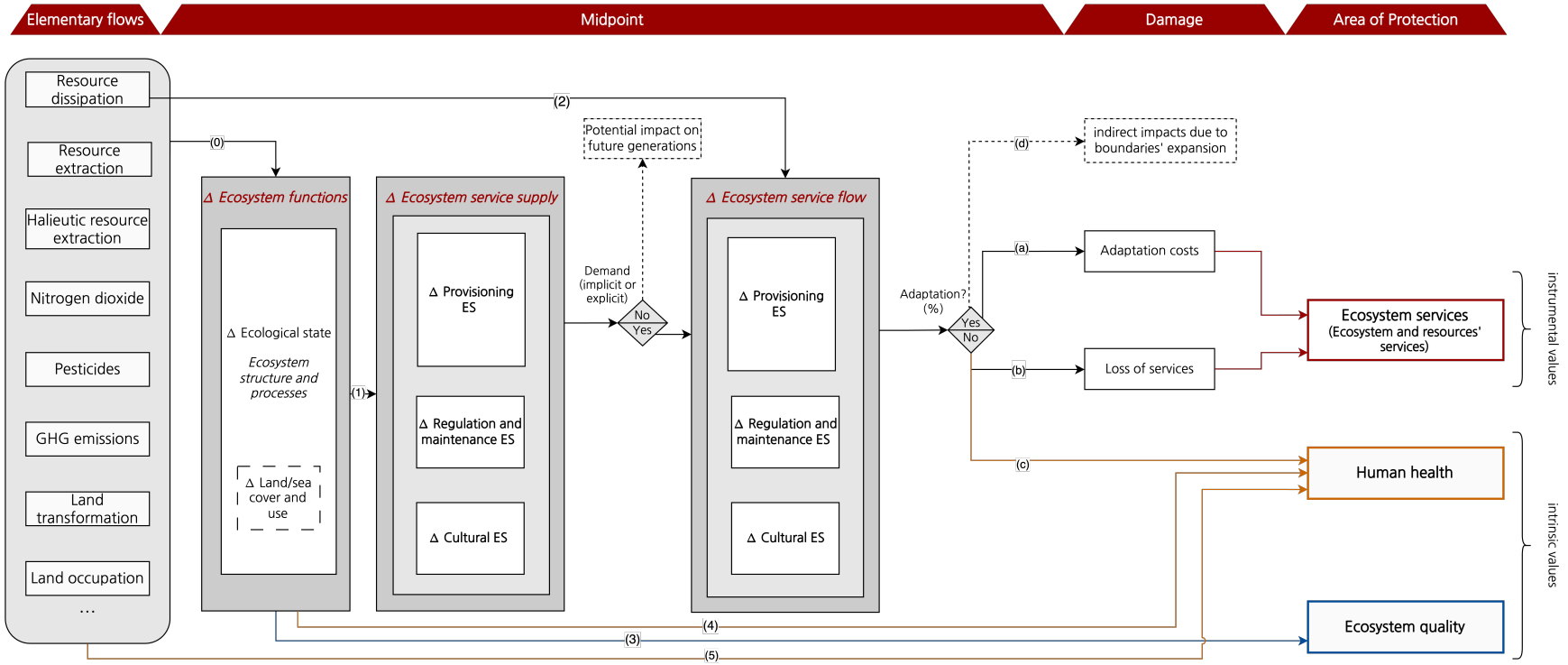


FIGURE 5.3 Revamped comprehensive LCIA framework that clarifies the role of ES concepts (including the services provided by resources) within current LCIA models. The framework is based on the ES cascade model of Burkhard et al. (2014) (grey boxes). The change in ecosystem functions (left grey box) represents a shift from an initial state, which is not necessarily pristine. Although not shown for simplicity, we recognize that the ES supply is often combined with additional inputs to generate ES flows. The revamped framework integrates the influence of adaptation measures on the cause-effect chains. The framework links elementary (and intermediate for resource dissipation, see section 5.2.4) flows to damages on AoP proposed in Verones et al. (2017) and refined in this work : human health, ecosystem quality, and a revised Ecosystem and resources AoP. The Ecosystem Services AoP aims to protect the services provided by ecosystems and resources to humans. The first decision box (diamond-shaped, on the left) accounts for ES demand, which can be explicit (e.g. food provisioning from cultivated fields) or implicit (e.g. mediation of toxic substances). The second decision box describes the selected adaptation strategy.

Illustration through the water consumption impact pathways. The impact pathways linked to water consumption are used as examples to illustrate some parts of the cause-effect framework. Water consumption can alter the ecological state of a watershed by reducing local water availability (Schmidt, Gonda & Transiskus, 2021) (path 0 in the figure 5.3). This reduction can potentially lead to adverse impacts on both human health (Boulay et al., 2011 ; Debarre et al., 2024 ; Motoshita et al., 2018 ; Pfister, Koehler & Hellweg, 2009) and ecosystem quality (E. Pierrat et al., 2023 ; Tendall et al., 2014 ; Zelm et al., 2011).

In terms of human health, a decrease in local water availability corresponds to a reduced water supply potential (path 1). When there is a human demand for water in the affected region, this reduction can impair water provisioning ES flow, potentially resulting in water shortages (Boulay et al., 2011 ; Debarre, Boulay & Margni, 2022). In response, populations with sufficient economic resources can implement various adaptation strategies, such as purchasing water or desalinating seawater for domestic use (Debarre, Boulay & Margni, 2022), or importing food that cannot be grown locally due to agricultural water shortages (Motoshita et al., 2018). However, these adaptation strategies involve associated costs (path 0-1-a) and can induce indirect environmental impacts due to boundary expansions, including the life cycle impacts of desalination plants, transportation, and other adaptation-related activities (0-1-d).

In contrast, if a population is unable to adapt to water shortages, it can face malnutrition (Motoshita et al., 2018) and lack of hygiene and sanitation (Debarre et al., 2024), reflected in the AoP for human health (0-1-c). In parallel, the value of the lost water provisioning ES flow should be accounted for within a "loss of services" indicator. Currently, no LCIA model covers this impact pathway.

Through a change in ecosystem structure and processes, a change in ecosystem functions (0) can directly lead to potential species disappearance or extinction, reflected in the EQ AoP (pathway 0-3). For example, water consumption can reduce river discharge and flow regulation, threatening freshwater species (E. Pierrat et al., 2023).

Illustration through the particulate matter emissions impact pathways. Moreover, a change in ecosystem functions (0) can make environmental conditions unsuitable for human health, without altering the supply of ES (pathway 0-4). For instance, outdoor emissions of particulate matter can modify ecosystem structures by altering the concentration of harmful substances in the environment (pathway 0) and impact human health through respiratory and cardiovascular issues (De Schryver et al., 2009).

Finally, certain environmental interventions, such as indoor particulate matter emissions, do not involve changes in ecosystem functions, as these emissions occur entirely within the technosphere. These interventions can directly impact human health, causing respiratory outcomes (Fantke et al., 2015) (pathway 5), without affecting ecosystems.

Application on the IW+ impact categories

Table 5.2 provides a sample of efforts undertaken to position the IMPACT World+ (Bulle et al., 2019) methodology's characterization models in relation to the revamped framework. This work illustrates that all impact pathways defined in the framework are currently applied in LCIA modeling practices. Additionally, it highlights that a single characterization model (e.g., climate change impact on human health) can capture multiple impacts (e.g. cardiovascular diseases, malaria, diarrhea, floods and malnutrition), each driven by distinct mechanisms following different impact pathways. For example, the increase in cardiovascular diseases related to climate change is driven by changes in ecosystem functions that directly affect human health (pathway 0-4) due to an increase in the frequency and intensity of heat waves. In contrast, the rise in the results of climate change-related malnutrition is attributed to the change in precipitation patterns that induces a decrease in water availability for agricultural users (Δ provisioning) and a decrease in crop yields to which the population has not adapted (pathway 0-1-c).

All characterization models assessing potential impacts on ecosystem quality (e.g., impacts of marine acidification) rely on cause-effect chains that involve changes in

ecosystem functions (pathway 0-3).

Some characterization models, where the cause-effect chain involves a loss of ES (e.g., impacts on ecosystems and resource services due to land transformation affecting soil mechanical filtration) represent adaptation costs (pathway 0-1-a). LCIA model developers have accounted for the costs incurred by the society to compensate for the ES losses, such as water purification expenses in response to changes in water provisioning (Cao et al., 2015). In contrast, other models based on ES flow loss (e.g., impacts of land transformation on soil biotic production) assume no adaptation strategies, leading to damage costs (e.g., the value of lost crop yields) (pathway 0-1-b) (Cao et al., 2015).

Both examples stem from the same environmental intervention, land transformation, but differ in the assumed adaptation strategies. However, the conceptual distinction between these two modeling approaches remains insufficiently explained in related models.

Tableau 5.2 Sample of positioning efforts of the IMPACT World+ methodology characterization methods within the revamped LCIA framework (Figure 5.3). The complete table is provided in the ESM. AoP : Area of Protection. HH : Human Health, EQ : Ecosystem Quality, ERS : Ecosystems and Resources Services. Reco. level : level of recommendation of the model by Bulle et al. (2019). R : Recommended, I : interim. The last column identifies the ES involved when the cause effect chain of the covered impact builds on a change in ES flow (pathways 0-1-a, 0-1-b, 0-1-c and 0-1-d)

Impact category (IMPACTWorld+)	**AoP**	**Reference**	**Reco. level**	**Covered impacts**	**Description of mechanisms**	**Pathway**	**Ecosystem service involved (CICES)**
Climate change	HH	De Schryver et al. (2009) ; Bulle et al. (2019)	R	Cardiovascular diseases	Δ frequency and intensity of heatwaves	0-4	
				Malaria	Δ mosquitos' distribution	0-4	
				Diarrhoea	Δ precipitation patterns, water availability	0-1-c	Provisioning of water
				Floods	Extreme events	0-4	
				Malnutrition	Δ precipitations patterns, water availab., crop yield	0-1-c	Provisioning of food
	EQ	De Schryver et al. (2009)	R	Species disappearance	Increased temperatures	0-3	
Marine acidification	EQ	Myhre et al. (2013) ; Azevedo et al. (2015)	R	Population affected	pH decrease, calcifying species	0-3	
Ozone layer depletion	HH	Struijs et al. (2008) ; WMO (2014)	R	Immune suppression ; skin cancer ; cataract	Δ ozone layer's capacity to absorb UV rad.	0-1-c	Protection from Sun's UV rad.
Water availability	HH	Boulay et al. (2011)	R	malnutrition (agricultural)	Agricultural and fisheries water deprivation	0-1-c	Provisioning of water
				Water-related diseases	Domestic water shortage, inadequate hygiene practices	0-1-c	Provisioning of water
	ERS	Boulay et al. (2011) Cao et al. (2015)	I	Adaptation costs	Costs to adaptat to water shortages	0-1-a	Provisioning of water
Land transf. - mech. filtration	ERS	Cao et al. (2015)	I	Adaptation costs	Δ soil filtration capacity, water purification costs	0-1-a	Mechanical filtration
Land transf. - biotic production	ERS	Cao et al. (2015)	I	Damage costs	Δ soil fertility, loss of crop yield	0-1-b	Biotic production
Mineral resource use	ERS	De Bruille (2014)	I	Adaptation costs	Loss of functionality due to dissipation, additional procurement efforts, substitution	2-a	Provisioning of mineral resource
Fossil energy use	ERS	Fatemi (2013)	I	Adaptation costs	Dissipation via combustion of fossil energy, backup technology	2-a	Provisioning of energy
...	...	...	...	...	...	...	...

5.2.4 Discussion

A flexible framework to support the structured development of characterisation models and factors

The proposed framework aims to provide a structured yet flexible foundation for developing characterization models and factors. Each grey box within the framework can be interpreted as a potential impact indicator, while the arrows connecting two boxes (consecutive or not) depict environmental mechanisms.

This work does not prescribe specific guidelines for modeling these relationships, which may be either mechanistic -based on causal relationships formalized through equations- or empirical -derived from observational data. The choice of modeling approach is left to the discretion of the LCIA model developer and may depend on factors such as the type of environmental intervention considered, the specific ES assessed, or the availability of data for modeling.

Implicit adaptation strategies in characterization models related to the HH and EQ AoP

Characterization models related to impact pathways *(0-3) : changes in ecosystem functions affecting ecosystem quality*, *(0-4) : changes in ecosystem functions affecting human health*, and *(5) : elementary flows directly impacting human health*, can also involve adaptation measures. For example, in the case of indoor particulate matter emissions within a factory (pathway 5), air pollution control technologies can be implemented to mitigate health impacts.

Consequently, additional decision nodes (diamond boxes) could be integrated into the framework to account for these adaptation strategies. For clarity and simplicity, these elements have not been explicitly represented in Figure 5.3.

Impact of one environmental intervention on several ES

Environmental interventions, such as changes in land use, can simultaneously affect multiple ES by altering ecosystem functions. The updated framework should be applied individually to each ES flow loss. When a single intervention impacts multiple ES, LCIA model developers should build upon the framework to establish distinct impact pathways for each affected ES.

However, ES are interconnected through complex relationships, including trade-offs and synergies (Cord et al., 2017) ; changes in one ES can influence the supply and quality of other ES, potentially amplifying or dampening overall impacts. In addition, ES are often subjected to multiple stressors, and their degradation may result from the cumulative effects of these pressures.

Despite the importance of modeling combined effects for the ES community, the computational structure of LCIA is currently inadequate to capture the interactions between stressors (Othoniel et al., 2016). The present framework does not provide specific guidance on managing the challenges associated with these interactions and calls for further research efforts to address them.

About the boundaries between ecosphere and technosphere

Our revised framework, which addresses the characterization of resource dissipation and, more broadly, the incorporation of ES in LCIA, opens the discussion on the need to revisit the ISO (2006) 14044 standard, particularly with respect to the boundary between the ecosphere and the technosphere and the characterization of elementary flows.

On the interpretation of ES definition and the acceptable degree of anthropization According to the current version of the ISO14044 standard (ISO, 2006), the characterization of potential impacts of a product system is restricted to elementary flows, defined as exchanges that occur at the boundary between the ecosphere and the technosphere. However, the delineation of this boundary remains

ambiguous and contested (Heijungs, 2024 ; Rosenbaum et al., 2015b ; Weidema et al., 2018), with inconsistencies in current practices.

For example, LCIA currently assesses the potential impacts of urban land occupation on ecosystem quality. However, from the perspective of the standard, given the anthropized nature of urban land, urban land occupation cannot be considered an elementary flow, and its characterization should technically fall outside the scope of LCIA. Similarly, emissions of toxic substances indoor, which do not enter the ecosphere directly, are characterized in LCIA models (Rosenbaum et al., 2015a). These practices reveal inconsistencies in the application of the standard.

A notable borderline case is agriculture, which crosses the boundary between natural and human-made environments. The use of pesticides exemplifies this issue. In 2015, a dedicated workshop was held to clarify the delineation between the 'product system model (life cycle inventory, technosphere) and the natural environment (life cycle impact assessment, ecosphere)' (Rosenbaum et al., 2015b). The workshop's conclusions underscored the context-dependent nature of this boundary, with experts refraining from recommending a fixed definition. Instead, the goal and scope of the LCA should guide this delineation. As a result, the classification of flows as elementary or intermediate remains context-specific, contributing to ongoing debates.

In this context, the interpretation of the definition of ES raises concerns. The CICES classification is recommended here as a starting point for defining ES. However, its level of detail is limited to the class level (e.g., 1.1.1.1 : *Cultivated terrestrial plants (including fungi, algae) grown for nutritional purposes*). Each LCIA method developer may further refine and interpret these classes, potentially introducing ambiguity in the definition of ES, particularly regarding the acceptable degree of human intervention for a service to be considered "ecosystemic". For example, concerns may arise when highly intensive agricultural systems, such as industrial hydroponic greenhouses, are classified as ES while largely processed by humans. This framework does not intend to establish guidelines specifying the allowed level of anthropogenic influence for ES definitions in LCA. Following the proposal to merge the instrumental values of the natural and human systems in Table 1, this work instead invites the rea-

der to question the decisional relevance of the human-nature boundary. Furthermore, we strongly encourage the developers of LCIA methods to transparently document their interpretation of the CICES classification and the level of anthropization they consider acceptable in the definition of ES.

On the potential impacts of resources' dissipation Furthermore, restricting impact characterization to elementary flows may overlook significant environmental impacts. For example, dissipation of elements within the technosphere, such as elements captured in alloys and lost due to low recyclability, results from exchanges from and to the technosphere (i.e., from intermediate flows). According to the current standard, these intermediate flows cannot be characterized. However, such dissipation processes result in the loss of functionality of resources, which is a relevant consideration under the ES Area of Protection (AoP) (Beylot et al., 2021, 2024 ; Greffe, Margni & Bulle, 2022 ; Lai & Beylot, 2023). LCIA models adopting this paradigm are emerging (Greffe, Margni & Bulle, 2022 ; Lai & Beylot, 2023), but cannot be applied consistently with the ISO 14044 standard.

Therefore, to account for the loss of instrumental value of resources, among others, characterization should be extended to some intermediate flows (Heijungs, 2024).

Two potential approaches can be proposed to revisit the ISO14044 standard :

— Introducing 'virtual' elementary flows with an "in technosphere" compartment in life cycle inventory (LCI) datasets. These flows would represent the quantity of intermediate flows to characterize.

— Extending the scope of the characterization to include selected intermediate flows. This would require modifying the characterization matrix (that is, the matrix containing characterization factors) to account for certain intermediate flows alongside elementary flows.

The first approach offers a pragmatic solution, as it requires no structural changes to LCA software and can be easily implemented. Further details of this approach are provided in the SI. In contrast, the second approach would require to extend the

characterisation matrix to include the characterization models of some intermediate flows.

Discussion on the AoP

Although the proposed framework builds on the AoP definitions introduced in Section 5.2.2, several conceptual and terminological aspects of these AoP warrant further discussion.

First, as elaborated in the supplementary information Debarre et al. (s. d.-b), there is an ongoing debate on whether potential impacts on intrinsic values can be adequately assessed within the ecosystem quality and human health AoPs, or more broadly, through LCIA impact indicators.

Intrinsic values denote the inherent worth of an entity, regardless of its usefulness to human societies (Arias-Arévalo, Martín-López & Gómez-Baggethun, 2017 ; Hargrove, 1992 ; R. K. Turner et al., 2003). Since intrinsic values exist regardless of human valuation (Himes et al., 2024), quantifying them poses significant challenges. Any attempt by humans to assess intrinsic value would inevitably be influenced by an anthropocentric perspective (Hargrove, 1992). Consequently, a quantitative assessment of intrinsic value remains highly contentious and, to date, has not been convincingly justified within the LCIA literature.

However, other scholars in environmental ethics argue that while intrinsic value may not lend itself to quantitative measurement, it can still be recognized or conceptually articulated (O'Connor & Kenter, 2019). In this context, one could consider that LCIA modelers have attempted to approximate a form of quantitative recognition of intrinsic values through the Ecosystem Quality and Human Health AoPs (Bulle et al., 2019 ; Verones et al., 2017). However, such indicators remain, at best, proxies and fall short of capturing the full ethical and metaphysical dimensions of intrinsic value.

A second critical consideration involves the substitutability assumption inherent in these AoPs. When potential impacts are aggregated into a single AoP, they are trea-

ted as substitutable, which means that the negative impacts (or benefits) associated with an environmental intervention can offset the positive impacts (or damages) of others within the indicator. However, according to Himes et Muraca (2018), the substituability assumption holds true only for instrumental values, whereas intrinsic and relational values embody unique and irreplaceable aspects of ecosystems.

Furthermore, beyond the inherent challenges of quantitatively assessing impacts on intrinsic values, fundamental questions remain regarding the very nature of the values - intrinsic or instrumental - that the ecosystem quality and human health AoPs are intended to protect.

In current LCIA methodologies such as ReCiPe (Huijbregts et al., 2017), IMPACT World+ (Bulle et al., 2019) or LC-Impact (Verones et al., 2020), indicators used to assess the potential impacts on EQ are typically expressed in terms of the fraction of disappeared species (or species loss). The loss of species could be considered as a measure of reduced existence value, which refers to the satisfaction individuals derive from knowing that nature and biodiversity are preserved (Pearce & Turner, 1990 ; Plottu & Plottu, 2007). However, from an environmental economic perspective, the existence value is instrumental, as it reflects human preferences rather than an intrinsic property of nature itself (Davidson, 2013 ; R. K. Turner et al., 2003).

Furthermore, some LCIA model developers justify the use of species disappearance (or loss) as an indicator of ecosystem quality by suggesting that it serves as a proxy for ecosystem integrity (Goedkoop et al., 2013). This implies that the primary objective may be to preserve the functional integrity of ecosystems, thus aligning the ecosystem quality indicator more closely with the support of instrumental values than intrinsic ones.

Similarly, the question of whether human health has intrinsic value is debated in philosophical and ethical discussions (Duncan, 2010). Although it is clear that human life has an intrinsic dimension, it can be assumed that the willingness to protect it is driven by instrumental considerations. Although the LCIA community has, in practice, classified human health as an object of intrinsic value, further reflection on

this point could be beneficial.

Finally, we acknowledge that our reliance on the revised social values proposed by Verones et al. (2017) represents a methodological choice, and alternative approaches to select AoP could be considered. In particular, Matuštík et Paulu (2025) recently suggested incorporating the Earth system functioning as a distinct AoP. Future research should investigate the connections between this proposed AoP and our revised LCIA framework.

In conclusion, we recommend a more thorough examination of the philosophical foundations underpinning these AoPs, which may include reconsidering and potentially renaming the type of values they seek to encompass.

In this paper, however, we propose to maintain the current classification of EQ and HH as AoP representing intrinsic values, acknowledging that this choice involves pragmatic considerations rather than strict adherence to philosophical definitions. We also emphasize that this classification remains open to further debate and revisions, especially as new insights into value frameworks and impact pathways emerge.

A More consistent Name for the AoP Representing Instrumental Values : "Ecosystem Services" ? The AoP that encompasses ES and the provisioning ES' subset of resource services could benefit from a more unified and descriptive title ; naming the AoP "Ecosystem Services" would create a more coherent and comprehensive classification.

However, to avoid this AoP being overlooked by stakeholders interested in resources but unfamiliar with the broader concept of ES, it may be preferable to retain a more explicit title that highlights both aspects.

We therefore proposed modifying the terminology used by Verones et al. (2017) and Bulle et al. (2019), who refer to this AoP as "[Natural] Resources & Ecosystem Services", by rearranging the order of terms to emphasize the functionality of resources within ecosystems : "Ecosystems and Resource Services".

5.2.5 Perspectives

A framework to identify gaps in current LCIA methodologies

The revamped framework can be used to identify gaps in the coverage of impact categories within LCIA methodologies such as IMPACT World+ (Bulle et al., 2019). For each impact category (e.g., climate change, acidification, depletion of the ozone layer), a review was conducted to assess whether existing models comprehensively capture potential impacts at the damage level across all relevant AoP (human health, ecosystem quality and ecosystem and resource services).

When there is no characterization model for a given AoP (for example, impacts of acidification on ecosystem and resource services), the relevance of integrating such impact pathway was evaluated based on expert knowledge (for example, high relevance due to the importance of impacts related to the loss of coastal protection induced by acidification). To support this analysis, examples of missing cause-effect chains were proposed. This analysis reveals the methodological avenues for improvement to be pursued. Its results are provided in Table 5.3 and further detailed in the SI. Future research efforts should prioritize the development of characterization models that address the potential impacts of acidification, ozone layer depletion, eutrophication, toxicity, photochemical oxidant formation, and ionizing radiation on ecosystem and resource services. Similarly, models that capture the impacts of ozone layer depletion and photochemical oxidant formation on ecosystem quality are needed. Although of comparatively lower priority, it would also be valuable to develop characterization models assessing the potential impacts of acidification, mineral resource use, and eutrophication on human health, as well as the effects of particulate matter emissions on ecosystem quality.

It is important to note, however, that the existence of a characterization model does not necessarily imply comprehensive coverage of all relevant cause-effect chains. Therefore, improving the completeness and robustness of existing impact categories remains an essential objective for advancing life cycle impact assessment methodologies.

Tableau 5.3 Coverage of damage level impact in the IMPACT World+ methodology Bulle et al. (2019). For each impact categories (per type of damage, to be aggregated into the AoP) the "Reco. level" column indicates the level of recommendation of the model by the authors : R (Recommended), I (Interim), and Absent (no model currently included in the IMPACTWorld+ methodology). The "Relevance of integration" column assesses whether the absence of a model warrants priority development efforts. NA (Not applicable) is used when a model is already included in the methodology. The relevance levels are indicated as follows : - (low environmental relevance of integration), + (moderate relevance), and ++ (high relevance).

Impact category	**AoP**	**Reco. level**	**Relevance of integration**	**Example of cause effect chain that could be investigated**
Climate change	HH	R	NA	–
	EQ	R	NA	–
	ERS	I	NA	–
Acidification	HH	Absent	+	Respiratory problems induced by exposure to acidic substances (Goyer et al., 1985)
	EQ	R	NA	–
	ERS	Absent	++	Habitat degradation and loss of coastal protection (Hall-Spencer & Harvey, 2019).
Ozone layer depl.	HH	R	NA	–
	EQ	Absent	++	Damage on plants DNA (Fuhrer et al., 2016)
	ERS	Absent	++	Damage on fish larvae DNA and impact on fisheries (Zagarese & Williamson, 2001)
Mineral resources use	HH	Absent	+	Loss of functionality of phosphorus impacting food supply leading to malnutrition issues
	EQ	Absent	-	–
	ERS	I	NA	–
Fossil energy use	HH	Absent	+	Energy shortage in healthcare services
	EQ	Absent	-	–
	ERS	I	NA	–
Eutrophication	HH	Absent	+	Harmful toxins produced by algal blooms (Hwang, 2020)
	EQ	R	NA	–
	ERS	Absent	++	Algal blooms impacting water quality and recreational value (Compton et al., 2011)
Toxicity	HH	R	NA	–
	EQ	R	NA	–
	ERS	Absent	++	Impacts on pollinators, pollination and food production (Chagnon et al., 2015)
Particulate matter formation	HH	R	NA	–
	EQ	Absent	+	Impact on plant morphology impairing growth and reproduction (Rai, 2016)
	ERS	Absent	+	Impacts on air quality and visibility affecting tourism and recreational activities
Photochemical oxidant formation	HH	R	NA	–
	EQ	Absent	++	Acute and chronic impacts on plants (Kley et al., 1999)
	ERS	Absent	++	Impact on forest vegetation and their services
Ionizing radiations	HH	R	NA	–
	EQ	R	NA	–
	ERS	Absent	++	Disruption of ecological relationships (Geras'kin, 2016) underpinning ES
Water availability	HH	R	NA	–
	EQ	R	NA	–
	ERS	I	NA	–
Land transformation	HH	I	NA	–
	EQ	R	NA	–
	ERS	I	NA	–
Land occupation	HH	I	NA	–
	EQ	R	NA	–
	ERS	I	NA	–

Application of the framework to ecosystem disservices

In contrast to ES, ecosystem disservices (EDS) are defined as "the ecosystem-generated functions, processes, and attributes that result in perceived or actual negative impacts on human well-being" (Shackleton et al., 2016). EDS encompasses various harmful agents including pests, diseases, allergens, or natural disasters (Guo, Song & Dong, 2022). The disservice should not be confused with the reduction of ES ; a reduction of water provisioning (ES) is not a disservice. Rather, an example of a disservice would be wetlands that provide habitat for malarial mosquitoes (Dunn, 2010). In recent years, growing interest has been shown in EDS since "investments into the management or reduction of ecosystem disservices may yield better outcomes for human wellbeing, or at a lower investment, than management of ecosystem services" (Shackleton et al., 2016). Since ecosystem disservices are correlated with ecosystem services (Nápoles-Vértiz & Caro-Borrero, 2024), they should be evaluated together under a common framework. Their integration in LCIA could be relevant for decision-making. Integration of EDS could help capture environmental issues such as invasive species in the LCIA framework.

The framework we propose can be used as a basis for the integration of EDS in LCIA (blue pathway in SI, Figure B.1. Burkhard et al. (2014)'s cascading model could be generalized to represent how an environmental intervention can alter the functions of the ecosystem and subsequently change its EDS supply. Since EDS represent adverse features of ecosystems, there is no demand for EDS ; instead, the population experiences undesired exposure to these disservices.

Populations may respond differently to an increase in EDS flow. In some cases, total adaptation is possible, albeit with economic costs. For example, populations exposed to increased air pollution can adapt by investing in air filtration systems, leading to financial burdens and inducing indirect impacts due to expansion of the boundaries. In contrast, populations with limited adaptive capacity are more likely to face direct impacts on human health, such as respiratory problems or cardiovascular diseases.

However, the integration of EDS into the proposed framework requires a compre-

hensive, complementary, and nonoverlapping classification of both ES and EDS. The current version of the Common International Classification of Ecosystem Services (CICES 5.2) is not adequately adapted to support this distinction. The CICES classification could be extended to offer a holistic framework considering both EDS and ES (Vaz et al., 2017).

5.2.6 Conclusion

Although necessary, the integration of the concept of ES into LCIA lacks momentum, hampered by the absence of a globally accepted framework, structured guidelines, and clear terminology.

This article addresses these issues and offers a fresh perspective to comprehend the LCIA framework through the lens of ES and related concepts. This holistic perspective reveals that ES are closely tied to a large number of existing characterization models, and acknowledging these deep links can foster the development of more coherent and comprehensive LCIA models and methodologies. This revamped framework is intended to support research efforts aimed at integrating ES into LCIA, and to promote consistency and comprehensiveness in the development of LCIA characterization models.

CHAPITRE 6 CHARACTERIZING SEA-LEVEL RISE POTENTIAL IMPACTS ON COASTAL ECOSYSTEM SERVICES IN LIFE CYCLE IMPACT ASSESSMENT

Characterizing sea-level rise potential impacts on coastal ecosystem services in Life Cycle Impact Assessment

Laura Debarre, Manuele Margni

6.1 Présentation du chapitre

Ce chapitre met le cadre EICV développé précédemment en application en étudiant l'une de ses chaînes de cause-à-effet. Il propose un modèle de caractérisation et les facteurs associés évaluant les impacts de l'augmentation du niveau des mers et de l'accentuation des événements extrêmes sur la fourniture de SE côtiers. La figure 6.1 positionne la chaîne de cause-à-effet étudiée par rapport au cadre EICV développé précédemment.

En raison des contraintes temporelles liées au projet de thèse, ce travail n'a pas été soumis à un journal avec comité de relecture. Une future soumission est considérée. Ce chapitre a été co-écrit par Laura Debarre et Manuele Margni, et les travaux ont été menés avec le soutien de Martin Dorber. Le matériel supplémentaire associé à ces travaux est disponible en Annexe C.

Les résultats de ce projet ont été présentés à l'occasion de la 33^{ieme} *réunion annuelle de la SETAC Europe* qui s'est tenue du 30 avril au 4 mai 2023 à Dublin (Irlande).

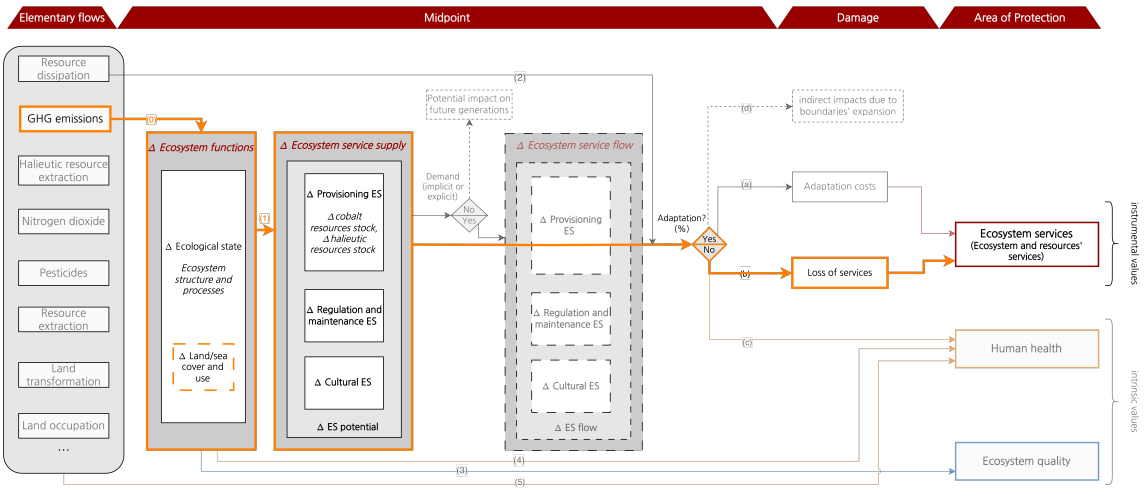


FIGURE 6.1 Positionnement de la chaine de cause-à-effet considérée au sein du cadre EICV révisé. Le modèle développé prend en compte le changement de fonctions écosystémiques dû aux inondations induites par l'élévation du niveau des mers et l'accentuation des événements extrêmes. Ces changements de fonctions induisent un changement de fourniture de SE, directement évalué en termes de pertes de SE. Contrairement à ce qui est suggéré dans le framework, le modèle ne traduit pas la perte de fourniture de SE en perte de flux de SE, en raison de la complexité de la modélisation de la demande globale en SE. Une approche conservative est suivie, dans laquelle on émet l'hypothèse que la demande en SE correspond à l'intégralité de la capacité de fourniture de SE de l'écosystème. Dans ce cadre, la perte de capacité de fourniture de SE est équivalente à la perte de flux de SE.

Keywords : Sea level rise, Characterization, Climate change, Ecosystem services

6.2 Manuscript

Abstract

Purpose. Ecosystems provide critical benefits to society, including coastal protection from storms, food production, or recreation opportunities. These benefits are referred to as "ecosystem services" (ES). Environmental impacts that affect ecosystems can

lead to a loss of ES. For example, coastal ecosystems and their services are threatened by sea level rise and the intensification of extreme events due to climate change. The integration of ES into life cycle impact assessment is gaining momentum. However, progress remains limited due to the absence of a clear, harmonized methodology for developing ES-related characterization models. To address these limitations, Debarre et al. (s. d.-a) proposed a revamped Life Cycle Impact Assessment (LCIA) framework revealing the role of ES within characterization models.

Method. Building on the LCIA framework of Debarre et al. (s. d.-a), this research aims to develop characterization models and factors assessing the potential impacts linked to greenhouse gas emissions and extreme sea level (ESL). The proposed characterization factors (CF), expressed in $/tCO2eq, quantify potential costs for society facing changes in coastal ES supply. CFs are calculated for RCP2.6, RCP4.5 and RCP8.5 and two adaptation scenarios (« No Additional Adaptation" (NAA) and "Adaptation Matching ESL Change" (AMEC)) by dividing the cumulative loss of global ES supply ($) value by the cumulative GHG emissions (tCO2eq) from 2015 to 2100. The cumulative loss of the global ES supply value is obtained by overlapping projections of ESL floodings with habitat maps. The monetary ES value of the flooded areas are evaluated using the Ecosystem Services Valuation Database (Brander et al. 2024). Cumulative GHG emissions are obtained from the RCP Database (2009).

Results and discussion. CF for RCP 4.5 (respectively, 2.6 and 8.5) reaches 28.0 (47.6 and 16.7) $/tCO2,eq under the AMEC scenario, and 37.4 (66.1 and 22) $/tCO2,eq under the NAA scenario. These values can be compared to the social cost of carbon, estimated at $185 by Rennert et al. (2022).

Conclusion and recommendations. The model (i) assumes that the ES supply losses are induced only by emissions from 2015 to 2100, neglecting the effects of prior emissions and their thermal inertia, and (ii) assumes a total loss of value for inundated habitats, despite their potential for adaptation or transition. However, this study strengthens the characterization of climate-related impacts in LCIA and serves as a proof of concept for Debarre et al. (s. d.-a)'s framework, consolidating

the integration of potential impacts on ES in LCA.

6.2.1 Introduction

Extreme sea level impacts on low elevation coastal ecosystems Coastal ecosystems are threatened by increasing direct and indirect anthropogenic pressures (Mehvar et al., 2018) including climate change (Barbier, 2017 ; Halpern et al., 2019). However, those ecosystems encompass various types of habitats such as sandy beaches and dunes, salt marshes, mangroves, sea-grass beds or lagoons (Barbier et al., 2011) and provide critical benefits to society, including habitat for biodiversity, food production, recreational opportunities, or carbon sequestration (Barbier et al., 2011 ; Drakou et al., 2017 ; K. Turner et al., 2014). Collectively referred to as "ecosystem services" (ES), these benefits are defined as the "benefits that people derive, directly or indirectly, from natural ecosystems" (Costanza et al., 1997). Using a benefit transfer method which estimates the value of ES per hectare based on similar land cover or habitat types (Plummer, 2009), Costanza et al. (2014) estimated the annual value of coastal ES at up to $27.7 billion per year (for 2011), underscoring their economic significance.

Low Elevation Coastal zones (LECZ) are defined as areas with elevations less than 10 meters above sea level (Barbier, 2015). The latter face sea level rise (SLR) and increased storm intensity and frequency (Gattuso et al., 2015 ; He & Silliman, 2019 ; Mehvar et al., 2019). Primarily driven by land-based ice mass loss and thermal expansion, SLR is not spatially uniform (Frederikse et al., 2020). Regional variations, known as Regional Sea Level Rise (RSLR), are notably influenced by ocean circulation patterns, gravitational effects, and land subsidence (Stammer et al., 2013). The interplay between RSLR and extreme events such as storm surges, tides, and waves results in Extreme Sea Levels (ESL) (Kirezci, Young & Ranasinghe, 2023). ESL can lead to permanent inundation, more frequent and intense temporary flooding, erosion, loss and transformation of coastal habitats, and soil and water salinization (Oppenheimer et al., 2019). These changes can significantly alter the structure and

function of coastal ecosystems and their capacity to provide services (Magalhães Filho et al., 2022).

The impact of ESL is highly dependent on the coastal defense infrastructure such as seawalls, dikes, and breakwaters, which mitigate damage (Kirezci, Young & Ranasinghe, 2023). Moreover, the projected impacts of ESL depend on the intensity of future greenhouse gas (GHG) emissions. To assess future climate scenarios, Representative Concentration Pathways (RCPs) have been developed to describe different GHG concentration trajectories based on varying levels of emissions mitigation (Meinshausen et al., 2011). RCP2.6 represents an ambitious mitigation scenario limiting global warming to well below 2°C, while RCP8.5 assumes continued high emissions, leading to severe climate impacts. Intermediate pathways, such as RCP4.5 and RCP6.0, represent moderate mitigation efforts. According to the latest Intergovernmental Panel on Climate Change (IPCC) report on the ocean and the cryosphere (Oppenheimer et al., 2019), global mean sea level is projected to rise between 0.43m (RCP2.6) and 0.84m (RCP8.5) by 2100. Although few studies have modeled SLR after 2100, the process will persist for thousands of years, with increasing divergence between emission scenarios (Oppenheimer et al., 2019).

Integration of ESL impacts in LCA Given the increasing frequency and severity of ESL-driven damages (Vitousek et al., 2017), it is crucial to incorporate their potential impacts into decision-making processes (Oppenheimer et al., 2019 ; Rasmussen et al., 2022).

Life Cycle Assessment (LCA) is a widely used methodology to guide sustainable decision making (Hellweg et al., 2023). Standardized and iterative, LCA follows four key stages : definition of the scope of the study, inventory, impact assessment, and interpretation (International Organization for Standardization, 2006). Multicriteria in nature, LCA generates impact profiles that include multiple environmental impact indicators.

There is a growing recognition of the need to integrate the potential impacts of human activities on ES within the Life Cycle Impact Assessment (LCIA) phase,

which quantifies the potential environmental impacts throughout the life cycle of a product (Alejandre, Bodegom & Guinée, 2019 ; Debarre et al., s. d.-a ; Maia de Souza et al., 2018 ; Othoniel et al., 2016 ; Rugani et al., 2019 ; Vanderwilde & Newell, 2021 ; Verones et al., 2017).

Within the LCIA step, elementary flows, referring to exchanges of emissions, energy, or resources between a product system and the environment, are converted into impact scores using characterization factors (CFs).

Debarre et al. (s. d.-a)'s LCIA framework [1] Debarre et al. (s. d.-a) proposed a comprehensive LCIA framework that clarifies the role of ES within LCIA models (see figure 6.1). A LCIA framework provides a structured approach for developing characterization models and factors, illustrating potential cause-effect chains, identifying relevant indicators, and defining appropriate Areas of Protection (AoP) which represent entities that society seeks to protect due to their perceived importance (Schaubroeck & Rugani, 2017).

Debarre et al. (s. d.-a)'s framework builds on the cascade model developed by Haines-Young et Potschin (2010) and refined by Burkhard et al. (2014), which conceptualizes the relationships between ecosystem functions (shaped by the state of the ecosystem and the type of land use), the flows of ES they generate, and the societal benefits derived from those flows.

Within this LCIA framework, Debarre et al. (s. d.-a) demonstrate how human activities, expressed as life cycle inventory flows, can alter ecosystem functions and, thereby, affect an ecosystem's capacity to provide ES flows. The resulting loss of ES flows can lead to adaptation costs or damage costs, depending on the strategy employed. Here, "adaptation" refers specifically to reactive measures taken after the loss of ES flows, distinguishing it from upstream adaptation strategies such as mentioned earlier in the text (adaptation measures against ESL, such as dikes).

Both adaptation and damage costs represent potential instrumental impacts on

1. Debarre et al. (s. d.-b)'s framework refers to the LCIA framework introduced in chapter 5. Debarre et al. (s. d.-a) refers to the critical review presented in the SI A.

social-ecological systems, which are aggregated under the "Ecosystem and Resources' Services" AoP. Additionally, adaptation strategies themselves can introduce indirect environmental impacts, as their life cycle expands the system's boundaries. In scenarios where no or only partial adaptation strategies are implemented, the framework also acknowledges the resulting impacts on human health.

Beyond ES flow disruptions, the framework highlights how degradation of ecosystem functions can compromise ecosystem integrity, leading to additional potential impacts on the Ecosystem Quality AoP.

Objective Building on Debarre et al. (s. d.-a)'s LCIA framework, this research aims to develop characterization models and factors assessing the potential impacts linked to GHG emissions and ESL.

The proposed Characterization Factors (CFs) assess the potential impacts of climate-change-induced ESL on coastal ES supply. They are expressed in $/tCO2eq.

This research contributes to the enhancement of impact assessment in LCA by providing a more comprehensive coverage of climate change impact pathways and operationalizing a LCIA framework that allows for a better integration of ES.

6.2.2 Method

6.2.3 Cause-effect chain

The characterization model and factors developed in this work build on the cause-effect chain represented in Fig. 6.2.

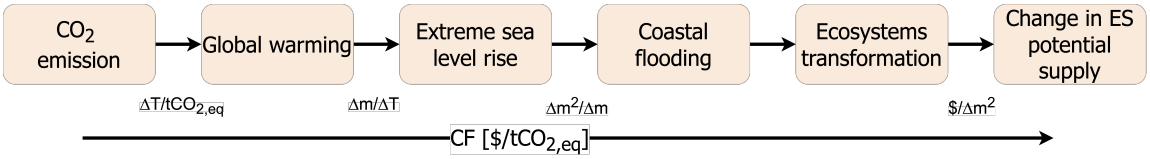


FIGURE 6.2 Cause-effect chain supporting the characterisation model of this work

Anthropogenic emissions of GHG into the atmosphere absorb and reemit infrared radiation, intensifying the natural greenhouse effect and driving global warming (Myhre et al., 2013). More than 90% of the excess heat generated by global warming is absorbed by the oceans (Cheng et al., 2017). As seawater warms, its density decreases, leading to thermal expansion (Church & Gregory, 2019). Simultaneously, rising global temperatures accelerate the melting of land-based ice reservoirs (Church & Gregory, 2019), which further contributes to mean SLR and to RSLR. These processes intensify ESL events by amplifying storm surges, tidal flooding, and wave setup, which can lead to coastal flooding (Kirezci et al., 2020). As a result, coastal flooding can transform coastal ecosystems (Chen et al., 2024) and alter ecosystems' capacity to provide services.

6.2.4 Positioning the cause-effect chain within Debarre et al. (submitted)'s LCIA framework

Through orange boxes and arrows, Fig. 6.1 illustrates the positioning of the presented cause-effect chain within Debarre et al. (s. d.-a)'s framework. The cause-effect chain links elementary flows of GHG emissions to changes in ES potential supply. It does not account for the resulting potential loss of ES flows, which requires an assessment of ES demand. Orange dotted lines and boxes illustrate the pathway yet to cover to reach the damage level and the Ecosystem and Resources' services AoP.

6.2.5 Characterization factors

Several CFs are developed in this research, differing by the scenario of GHG emissions and the scenario of coastal defenses considered.
Three scenarios of GHG emission are considered, corresponding to RCP2.6, RCP4.5 and RCP8.5. Those emission scenarios are declined into two coastal defense scenarios, based on Kirezci, Young et Ranasinghe (2023) : an "adaptation matching ESL change" (AMEC) in which coastal defenses are implemented to match the increased sea level, and a "no additional adaptation measures compared to the baseline" (NAA) scenario, where the baseline defenses are kept but no further defenses are implemen-

ted. Therefore, a total of six scenarios are considered : RCP2.6-NAA, RCP2.6-AMEC, RCP4.5-NAA, RCP4.5-AMEC, RCP8.5-NAA, and RCP8.5-AMEC.

Characterization factors (CFs), expressed in $/tCO2eq, quantify the average global potential ESL-driven impact on ES supply induced by the emission of 1 $tCO_{2,eq}$ for each scenario. For each of the four scenarios, these factors are calculated by dividing the cumulative loss of global ES supply value from 2015 to 2100 by cumulative GHG emissions over the same period, as per Eq6.2.5

$$CF_{i,j}\left[\frac{\$}{tCO_{2,eq}}\right] = \frac{\sum_{t=2015}^{2100} GESV_{i,j}(t) - GESV(t_{2015})}{\sum_{t=2015}^{2100} GHG_i(t)} \tag{6.1}$$

with $GESV_{i,j}(t)$: annual global value of ES provided by habitats under threat of flooding in scenario of emission i and coastal protection j for year t, in $/yr ; $GESV(t_{2015})$: annual global value of ES provided by habitats under threat of flooding in 2015 (baseline scenario) ; $GHG_i(t)$: global GHG emissions for year t in scenario i, in tCO2,eq/yr.

The following sections further describe the calculation of Eq1's terms. Figure 6.2.5 provides a graphical representation of the general methodology followed in this work.

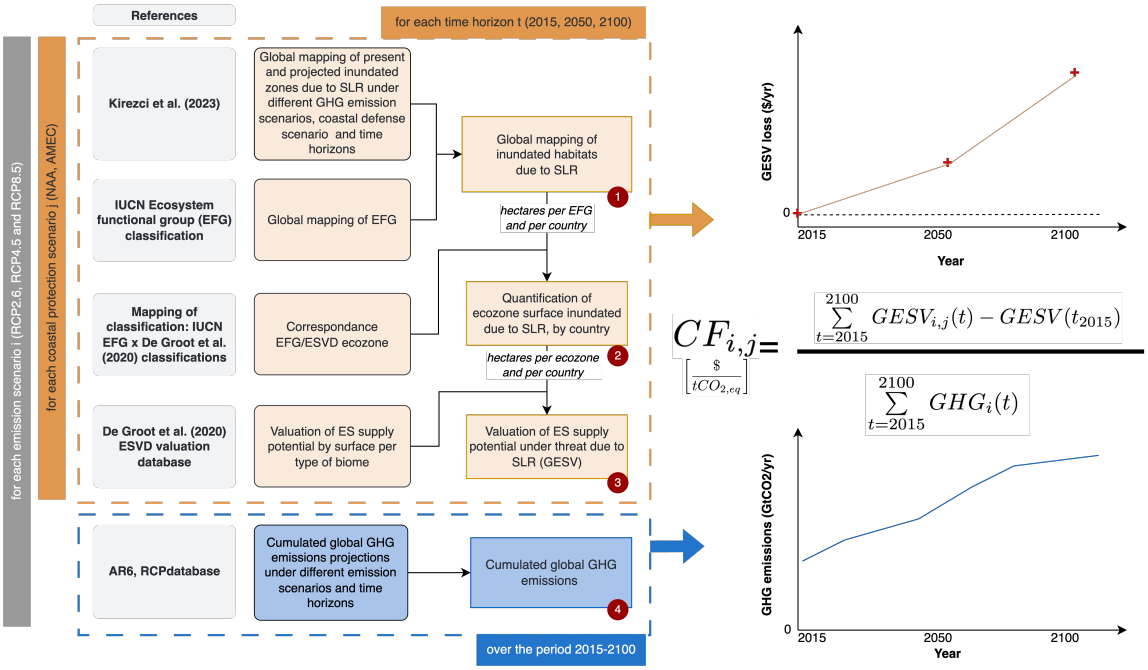


FIGURE 6.3 Methodology adopted in the present research work to derive $CF_{i,j}$ for emission scenario i (RCP2.6, RCP4.5 or RCP8.5) and adaptation scenario j (NAA or AMEC). The orange section of the figure describes the methodology conducted to quantify the cumulative loss of global ES supply value from 2015 to 2100. The blue section describes the methodology followed to quantify the cumulated global emissions of GHG over the period 2015-2100. Grey boxes provide the source of data used. Rounded rectangles indicate the data used. Rectangles indicate intermediate results. Italic texts indicate the units of intermediate results. Red dots represent 4 "steps" of the methodology further described in following sections. GESV : annual global value of ES provided by habitats under threat of flooding.

Assessment of the cumulated change in Global Ecosystem Services Value of habitats under threat of flooding

The numerator of Eq1 quantifies the cumulative global potential loss of ES supply due to flooding of coastal habitats from ESL between 2015 and 2100. This loss is calculated relative to a baseline year, set in 2015 on the basis of Kirezci, Young

et Ranasinghe (2023). The calculation is performed for each emission and coastal defense scenario.

Global mapping and quantification of ES supply under threat from ESL The valuation of ES supply at risk of flooding is based on the global mapping of present and projected coastal flooding by Kirezci, Young et Ranasinghe (2023). This dataset models the extent of flooding under different scenarios of greenhouse gas emissions (RCP 2.6, RCP 4.5 and RCP 8.5), two coastal protection scenarios (NAA and AMEC) and across three time horizons (2015, 2050, 2100).

To estimate ES loss, each flood map is intersected with global distributions of ecosystem functional groups (EFG) from the IUCN Global Ecosystem Classification (Keith et al., 2020). This intersection results in a global mapping of the EFG under threat of flooding. The total impacted area (in hectares) for each EFG is quantified at the national level, using the country boundaries defined by Sonderegger, Pfister et Oberschelp (2020) (step 1 in fig. 6.3).

A mapping exercise is conducted to align the IUCN EFG groups with the ESVD ecozone classification, providing estimates of at-risk surface area per ecozone and per country (step 2, Fig. 6.3). Following the ES-matrix approach proposed by Burkhard et al. (2014) which links habitat types to specific ES, ecozones serve as proxies for potential ES supply.

Monetary valuation of ES supply under threat from ESL The annual monetary value of the ES supply at risk of flooding (expressed in $/yr) is estimated assuming a constant annual ES value per hectare for each ecozone, based on estimates from the Ecosystem Services Valuation Database (ESVD) (L. M. Brander et al., 2024 ; Groot, Brander & Solomonides, 2020). For each scenario (and their related time horizon), the annual Global Ecosystem Services supply Value of habitats under threat of flooding (GESV) for year t is calculated as per Eq2 (step 3, fig. 6.3).

$$GESV(t)[\frac{\$}{yr}] = \sum_i S_{j,t}[ha] \cdot Value_j[\frac{\$}{ha.yr}] \tag{6.2}$$

Global Ecosystem Services supply Value of habitats under threat of flooding (GESV) for year t, with t : year ; S : coastal land surface inundated, in ha ; j : type of ecozone, $Value_j$: annual value of ES provided by the ecozone j, in $/ha/yr.

Cumulative loss of GESV. The cumulative loss in the global value of ES supply (that is, Eq1's numerator) is determined by computing the difference between GESV(t) at each step t and the baseline GESV (defined at t=2015) from 2015 and 2100. Due to data limitation, the difference between $GESV_{i,t}(t)$ and $GESV(t_{2015})$ (that is, the change in the ES supply) is calculated for only years 2015, 2050 and 2100. A linear increase in this difference is assumed between 2015 and 2050, as well as between 2050 and 2100.

Cumulated global annual GHG emissions (GHG) As per step 4 in fig. 6.3, yearly data for annual GHG emissions provided by the RCP Database (2009) were cumulated over the period 2015-2100 for the different RCP scenarios to obtain cumulated GHG emissions ($\sum_{t=2015}^{2100} GHG(t)$, i.e. Eq1's denominator). All GHG are converted into $CO_{2,eq}$ using their Global Warming Potential (GWP) with a 100-year horizon. Latter values are used from the IPCC AR6 (Smith et al., 2021). Further information on this calculation is provided in the SI.

6.2.6 Discussion

Sanity check

We evaluated the relevance of the computed CFs by comparing these latter to the Social Cost of Carbon (SCC), which "measures the monetized value of the damages to society caused by an incremental metric tonne of CO_2 emissions" (Rennert et al., 2022). The estimations of the SCC vary, building on several assumptions (e.g. discount rates) and models (e.g. socio-economic projections, climate models). Rennert

et al. (2022)'s SCC estimate reaches 185\$/tCO_2 at a near-term discount rate of 2%. The CF computed for RCP2.6-AMEC scenario (respectively RCP2.6-NAA, RCP4.5-AMEC, RCP4.5-NAA, RCP8.5-AMEC, RCP8.5-NAA) corresponds to 26% (respectively 36%,15%,20%, 9% and 12%) of this SCC estimation.

Nevertheless, one should keep in mind that the scope and the calculation approach of the SCC are not directly comparable to our estimation of the loss of coastal ES supply due to extreme sea levels. Rennert et al. (2022)'s SCC builds on damage functions addressing different climate change impact pathways including on agriculture, mortality, energy consumption and SLR. Furthermore, the SLR component builds on the CIAM model (Diaz, 2016) which estimates "the direct economic costs of SLR from the perspective of economic efficiency" by minimizing "the sum of adaptation costs (protection or retreat) plus residual damages due to inundation, wetland loss, and flooding". The scope of the CIAM model is therefore different than our assessment. The reader is invited to refer to SI C for further explanations. To assess the validity of the CFs, the latter could also be compared with historical data, computing CF with past observations of SLR-induced coastal floodings and GHG emissions.

Choice of CF

This study provides a set of CF, the selection of which is left to the discretion of the practitioner. This selection may be informed by the cultural perspective adopted by the latter (namely egalitarian, hierarchist or individualist) as described in Thompson et al. (1990). An egalitarian perspective, which prioritizes long-term global sustainability and human survival, may favor CFs based on the low-emission RCP2.6 scenario. A hierarchist perspective, which aims to balance the well-being of both current and future generations, could justify the selection of CFs corresponding to the RCP4.5. Conversely, an individualist perspective, which focuses on maximizing short-term welfare, may align with CFs derived from the high-emission RCP8.5 scenario. Alternatively, practitioners may prefer to select CFs based on the most plausible or empirically supported emissions trajectory.

Inertia of SLR : temporal delay between GHG emissions and SLR

SLR exhibits a significant delayed response to GHG emissions due to the long response times of oceanic and cryospheric systems (Portner et al., 2019). Even under scenarios of immediate and drastic emission reductions, ongoing thermal expansion of the oceans and the continued melting of ice sheets are expected to contribute to SLR for centuries to millennia. This temporal lag introduces a significant challenge in the development of our CF. By design, the developed CFs are expressed as a ratio, implicitly linking emissions and impacts within the same time frame (2015 to 2100). However, such an assumption oversimplifies the climate system's inertia. The SLR projected for the 2015–2100 period is not solely attributable to emissions occurring within that window ; it also reflects the legacy of emissions prior to 2015. Conversely, the long-term SLR consequences of emissions from 2015 to 2100 will extend beyond this century and are not captured within the CF.

Furthermore, because the CF is calculated as a ratio, it implicitly assumes an equal contribution of annual emissions to ES loss due to ESL over the defined period. This simplification overlooks the fact that emissions occurring earlier in the time frame, such as around 2015, have a greater impact on SLR by 2100 than those released closer to the end of the century. Dynamic LCA models such as the one developed by Levasseur et al. (2010) can be used to account for this temporal variability. Such models consider the timing of emissions and their evolving global warming potential.

ESL and coastal habitat transformation

This research was carried out under the assumption that ESL events result in a total loss of value of flooded habitats. This assumption has two key limitations : (1) it does not account for habitat transitions following ESL events, and (2) it does not differentiate between temporary and permanent inundation.

The model does not incorporate the potential for ecosystems to adapt or transition into alternative ecosystem types that may continue to provide services. However, ecosystems may undergo transitions following flooding and continue providing ES.

Wetlands, for example, can adapt to rising sea levels through surface elevation gains and lateral migration (Chen et al., 2024). However, modeling such transitions is complex, especially at the global scale ; transitions depend on various features such as submersion frequency, the distances to water tables, salinity level, or sediment supply, among others (Kuhfuss et al., 2016 ; Spencer et al., 2016).

Additionally, habitat transformations influence not only inundated areas but also adjacent habitats above the waterline. Many of these ecosystems have the capacity to reorganize and migrate landward, provided that adequate space and suitable conditions exist. Accurately modeling these dynamic movements requires detailed spatial information, such as sediment supply and lateral accommodation space (Chen et al., 2024). Such transformations are overlooked in the present characterization model.

The impact of ESL on habitat transformation is also contingent on the type and frequency of flooding. ESL events result from a combination of regional sea level rise, storm surges, tides, and wave action (Kirezci, Young & Ranasinghe, 2023), each of which influences coastal ecosystems in distinct ways. While some areas may experience permanent submersion, others may be subject to periodic or episodic inundation. The maps provided by Kirezci, Young et Ranasinghe (2023) are inadequate to distinguish these different mechanisms.

In this context, defining a single, uniform mechanism for predicting habitat transformations due to ESL events would be too simplistic. Further research is needed to better capture such dynamics.

Potential impacts of ESL on the ecosystem quality AoP In addition to the loss of ES, ESL can lead to potential impacts on the ecosystem quality. The CFs developed by Dorber, Kuipers et Verones (2020) evaluate terrestrial biodiversity losses from future land inundation. These could be combined with the inundation maps from Kirezci, Young et Ranasinghe (2023) to assess potential impacts of ESL on the ecosystem quality and to provide a broader picture of ESL potential impacts.

6.2.7 Conclusion

This study advances LCIA by introducing a novel impact pathway : the potential loss of coastal ES supply resulting from ESL events. CFs are developed across three different emission scenarios and two adaptation strategy scenarios. These CFs help provide a better coverage of climate-related impact pathways in LCA. In addition, this work paves the way for wider coverage of these impacts, by suggesting avenues for the development of models addressing ESL impacts on ecosystem quality.

This study represents a promising step forward in the integration of potential impacts on ES into LCA. It serves as a proof of concept for the LCIA framework developed by Debarre et al. (s. d.-a), built on ES concepts. The present work demonstrates the applicability and relevance of this revised LCIA framework to support the development of characterization models.

CHAPITRE 7 ARTICLE 2 : FRESHWATER CONSUMPTION AND DOMESTIC WATER DEPRIVATION IN LCIA : REVISITING THE CHARACTERIZATION OF HUMAN HEALTH IMPACTS

Freshwater consumption and domestic water deprivation in LCIA : revisiting the characterization of human health impacts

Laura Debarre, Anne-Marie Boulay, Manuele Margni

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7.1 Présentation de l'article

Le troisième article présenté dans ce chapitre vise à consolider la modélisation des impacts potentiels sur la santé humaine résultant de l'utilisation et de la privation d'eau douce. La figure 7.1 illustre cette chaîne de cause-à-effet au sein du cadre EICV précédemment élaboré.

Afin d'améliorer la caractérisation des impacts liés à la consommation d'eau, les recherches menées dans cet article actualisent, consolident et discutent le modèle de caractérisation initialement proposé par Boulay et al. (2011).

Cet article a été publié dans l'*International Journal of Life Cycle Assessment* le 23 avril 2022. Il a été coécrit par Laura Debarre, Anne-Marie Boulay et Manuele Margni. Le matériel supplémentaire associé à l'article est disponible en Annexe D.

Les résultats de cette étude ont été présentés lors de la 31^{ieme} *réunion annuelle de la SETAC Europe*, qui s'est tenue en ligne du 3 au 6 mai 2021.

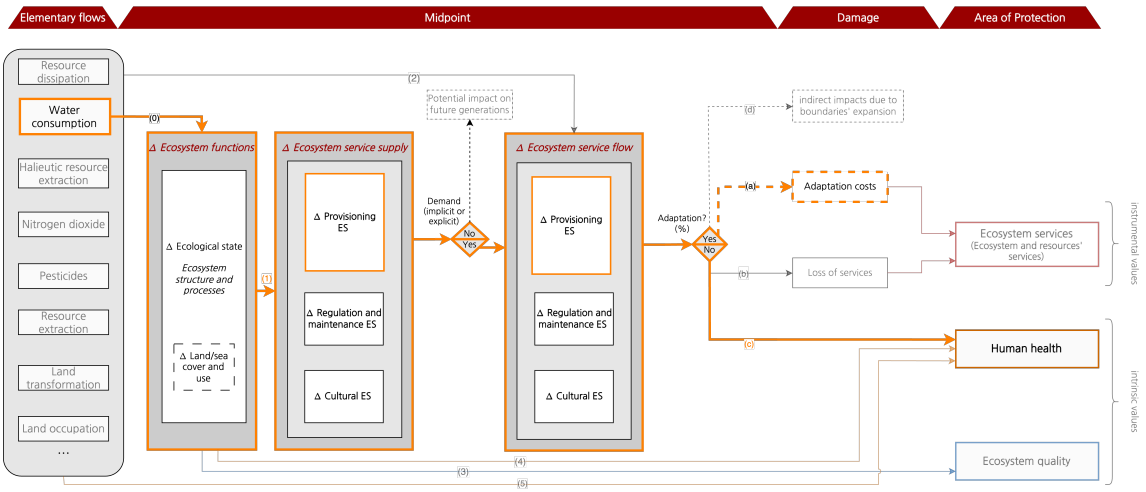


FIGURE 7.1 Positionnement de la chaine de cause-à-effet considérée au sein du cadre EICV révisé

7.2 Manuscript

Abstract

Purpose : An insufficient amount of available domestic water can lead to an increase in the occurrence of water-related diseases. No LCIA consensus has been reached on how to model the potential impacts on human health resulting from water use implying domestic water deprivation. Building on Boulay et al. (2011), this research work provides an updated and revisited characterization model and factors assessing the potential impact on human health induced along this impact pathway. **Method :** This work consolidates the cause effect chain linking water use to domestic impacts on human health. The revised fate factor aligns current water use assessment methods and includes information not only on the physical water scarcity but also on the level of population access to water in a region. Building on Boulay et al. (2011) the global effect factor is revised. The data source is updated, and a novel approach is developed estimating the domestic water deficit. Country scale exposure factors are updated, building on Boulay et al. (2011)'s proposal to rely on the gross national

income per capita as a proxy for a country's capacity to adapt to water shortages. **Results and discussion :** Compared to Boulay et al. (2011), the revised fate and exposure factors show lower values as a result of different methodological choices and of the overall increase of GNI per capita, respectively. The revised value of the effect factor is equal to 3.13E-3 DALY/m^3 which compares to the value of 3.11E-3 in Boulay et al. (2011). Revised characterization factors (CF) range from 0 DALY/m^3 (the potential impact on human health due to water use is null with respect to domestic water deprivation) to 3.13E-3 DALY/m^3. The distribution of the new CFs shows an order of magnitude decrease compared to the previous model. These CFs assess the consequences on human health induced by water use leading to short-term water deprivation. **Conclusion and recommendations :** This research work helps to better account for the impacts of water use at the endpoint level. However, it underlines significant limitations in the current calculation of the effect factor, in particular regarding current quantification of domestic water deprivation. These shortcomings prevent the model from considering a difference in vulnerability to health damages from the deprivation of 1m3 of domestic water. This research work argues for additional research efforts aimed at developing an alternative calculation method for this factor.

Keywords Water use ; Water consumption ; Human Health, Domestic water ; Water quantity ; LCIA ; Life cycle impact assessment ; Life Cycle Assessment, LCA.

7.2.1 Introduction

Water use, scarcity, sanitation, hygiene and health

Domestic freshwater is essential to human wellbeing and a minimum quantity is required to meet basic needs (drinking, cooking, hygiene and sanitation) and ensure fundamental health (Gleick, 1996). An insufficient amount of domestic water can lead to an increase in the occurrence of water-related diseases (Fry et al., 2010 ; Ketema et al., 2012 ; Mahande, Mazigo & Kweka, 2012 ; Nyemba et al., 2010 ; Stelmach & Clasen, 2015). Prüss-Ustün et al. (2019) estimate that in 2016 alone, 1.6 million

deaths and 105 million DALYs were attributable to quantifiable inadequate Water Sanitation and Hygiene (WASH). Freshwater consumption can result in a long- or short-term change of water availability at a local scale. Depending on specific conditions such as water availability and water accessibility (where availability refers to the amount of water that could be used and accessibility is linked to the level and quality of the infrastructure allowing to benefit from it), this can lead to water shortages and users' deprivation and incapacity to fulfill their basic requirements.

Water management and life cycle thinking

Sustainable Development Goal 6 calls for actions at the local level in order to promote universal access to safe water and sanitation for all (United Nations, 2015). While local actions and the development of water supply infrastructures are key, there is also a need to raise awareness among all stakeholders (consumers, industries, and decision-makers) on the indirect water use and its potential impacts along the value chain of consumed products and services (Hoekstra & Hung, 2005). Life Cycle Assessment (LCA) is a relevant tool to map all activities along the product life cycle, and assess the potential impact generated by their interaction with the environment in terms of resource consumption and pollutant emission (International Organization for Standardization, 2006). Applying the life cycle perspective is therefore useful to shed light on the indirect potential impacts due to water use, and this all along the value chain of products or services.

Human health impact of water use in Life Cycle Assessment

During the last decade, an increased interest was devoted in the assessment of water use in LCA (Bayart et al., 2010 ; Boulay et al., 2011, 2018 ; Kounina et al., 2013 ; Mikosch et al., 2021 ; Núnez et al., 2018 ; Pradinaud et al., 2019a). Consensus building and harmonization efforts have resulted in the publication of the ISO 14046 norm on water footprinting (ISO 14046 :2014, s. d.), designed to guide the evaluation of the potential impact of water use within the life cycle assessment framework. Since, the water use assessment framework in LCA is continuously evolving. With

the consensus-based life cycle impact assessment methodology AWARE (Boulay et al., 2018), the WULCA working group provides characterization factors for water use at a midpoint level.

Bayart et al. (2010) define water consumption as the "use of freshwater when release into the original watershed does not occur because of evaporation, product integration, or discharge into different watersheds or the sea". To date, to address the domestic water consumption' health issues in LCA, practitioners can use one of the two developed methodologies : Boulay et al., 2011 and Motoshita, Itsubo et Inaba, 2011. The two approaches' results reveal large variation, and a low rank correlation (26% (Boulay et al., 2015)). In these methods, freshwater use is assessed as a concern for the Human Health Area of Protection (AoP), without regard to the freshwater deprivation duration.

Despite regular development efforts for a better assessment of water use, still no consensus exists on how to characterize endpoint impacts on human health due to water use involving domestic water deprivation resulting in inadequate WASH practices (hygiene, hand washing behaviors, food hygiene etc.). In 2015, a consensus was built to assess human health impacts due to water use involving agricultural water deprivation, but no recommendation was provided regarding the approach or indicator to use for domestic water deprivation (Frischknecht et al., 2016). Two main issues were identified :

1. "High uncertainty in modeling the causality chain linking additional water consumption to water scarcity and human health impacts" (Frischknecht et al., 2016)
2. The challenge of characterizing damage on human health linked to domestic water use patterns that are not explicitly monitored worldwide and depend on local conditions, on water access infrastructures, e.g. "the distance of the population to the next well" (Pfister, Koehler & Hellweg, 2009) or on cultural habits such as lifestyle, religion and education (World Health Organization, 2013).

To develop a characterization factor expressing water use impacts on domestic human health issues, Motoshita, Itsubo et Inaba (2011) combine a water scarcity index developed by Pfister, Koehler et Hellweg (2009), a ratio of domestic over total human water use from Aquastat (FAO), and a socioeconomic and effect factor built as a multiple regression analysis describing water accessibility and damages from infectious diseases (in DALYs/m^3). Despite the relevance of the socioeconomic parameters integrated in the model, the resort to statistical tools such as the multiple regression used in Motoshita, Itsubo et Inaba (2011) raises the issue of transparency and verifiability of the mechanism at the root of the potential impact.

Boulay et al. (2011) developed two modeling alternatives. The first one, the *distribution modeling*, considers that all users are competing and impacted by an additional water use proportionally to their distribution share of water. The second one, the *marginal modeling*, assumes that only marginal users (agricultural users), less willing to pay for water (compared to domestic or industrial users), would be impacted. Potential impacts on domestic users are thus neglected in this second modeling. Therefore, the focus here is put on the first modeling alternative. Boulay et al. (2011) assume that the use of freshwater implies a change in freshwater availability when the resource is scarce. Freshwater can serve several users : ecosystems, industrial, agricultural or domestic users. We focus on the latter here. This change in availability could cause domestic water shortage, i.e. the incapacity to meet basic human water needs, which, in countries where the population is not able to adapt to water shortage, would lead to domestic water deprivation, resulting in inadequate hygiene and sanitation practices hence increasing the potential health effects.

The characterization factor CF of Boulay et al. (2011)'s results from the multiplication of a regionalized fate factor FF (to what extent will the use of 1m^3 of water deprive other users ?), a country-scaled exposure factor XF (to what extent will domestic users be impacted by this decrease in water availability and is the affected population able to adapt to water shortage ?), and a generic global effect factor EF (what are the consequences of domestic water shortages on human health ?)

Boulay et al. (2011)'s fate factor is calculated at a watershed scale on the basis of scarcity parameters, defined as the ratio of consumed water to renewable water. These parameters are used to calculate scarcity indicators using a logistic function (S-curve) to obtain values between 0 and 1 (m^3 deprived / m^3 consumed). The assumption that scarcity values follow a logistic relationship is "normative choice", while "little information is available" to support it (Boulay et al., 2018). Since this factor is defined as a ratio, a single value can represent two largely different realities : a value of 0.5 means that half of the available water is used by domestic users, but gives no information on the amount of water used, that could be 10 or 100000 m^3 (Boulay et al., 2018). This index is one of the several indices that have been developed to define water scarcity (Berger & Finkbeiner, 2010 ; Boulay et al., 2011 ; Hoekstra & Chapagain, 2007 ; Pfister, Koehler & Hellweg, 2009). Recently, the WULCA working group developed a fate factor FFagri, quantifying the "effect of the use of $1m^3$ of water in a watershed on the change of water availability for agricultural use" (Jolliet et al., 2018).

Boulay et al. (2011)'s exposure factor XF combines information on water distribution across users (the fraction of current domestic water consumption over the total human water consumption (Di)), with a factor addressing the capacity of a country to adapt to water shortage ("Adaptation Capacity", AC). XF is calculated at the country scale as XF=Di*(1-AC). It assumes that the richer a country is, the more likely its population is able to adapt to water deprivation by compensation mechanisms (such as desalination). Low-income countries are considered unable to adapt and are thus fully impacted (AC=0). Adaptation capacity of middle-income countries is assumed to be linearly proportional to Gross National Income per capita (GNI), and maximal adaptation capacity is assumed for high income countries (AC=1). That classification of countries by income per capita evolves with time and should be updated.

Boulay et al. (2011)'s effect factor is defined as the geometric mean of the ratio between the yearly water-related burden of disease (WRBD) (in DALYs/yr) and the yearly domestic water deficit per country (DWD) (m^3/yr). The WRBD is calculated as the sum of the burden of diarrheal diseases and nematode infections and 50% of the

burden due to malnutrition from the World Health Organisation (2009). That share of burden of malnutrition is included based on the assumption that water-related practices can trigger malnutrition issues e.g. when parasites cause low absorption of calories. The DWD is calculated as the difference between the national domestic withdrawals and the minimum human water requirements of 50 liters per capita/day (l/c/d), recommended by the World Health Organization (Gleick, 1996). To quantify the amount of deprived water, Boulay et al. (2011) consider an equal sharing of water resources among the population, defining domestic water use per capita as the ratio of total domestic water use divided by the population. However, water access and use are largely unequal across a country and an equal sharing does not provide a realistic picture of individuals' water consumption. Besides, the use of a global value for the effect factor implies that all individuals are suffering equally to the deprivation of $1m^3$ of domestic water, regardless of the country they live in (once economic adaptation has been considered in XF). Still, considering the high level of disparity between countries in terms of healthcare access and quality, this assumption could be revised.

Objective of the paper

Building on Boulay et al. (2011) this research aims to revisit the characterization model and factors of human health impacts through domestic water deprivation pathways due to water use in LCA. Specific objectives are :

1. Improve the description of the cause-effect chain linking water consumption, water scarcity, water deprivation and water-related burden of diseases.
2. Revisit intermediary models and factors (FF, XF, EF) and their spatial resolution for water consumption leading to domestic water deprivation, and provide revised characterization factors.
3. Discuss the relevance of modelling the human health impacts generated by water use and domestic water deprivation and define the limits of applicability of the model

7.2.2 Methods

Characterization model and factors

The cause-effect chain of water consumption leading to domestic water deprivation and health damages is presented in Figure 7.2. It can be expressed as per equation 1 according to the general life cycle impact assessment framework (Rosenbaum et al., 2008).

$$CF_{i,j} = FF_j \cdot XF_i \cdot EF \tag{7.1}$$

Where CF : characterization factor, FF : fate factor, XF : exposure factor, EF : effect factor. For river basin j, and country i. Below we describe how the fate and effect factors are revisited and the exposure factor is updated building on Boulay et al. (2011).

A marginal freshwater water consumption can lead to a local change in water availability. Called "physical depletion", this mechanism is part of the fate factor of this model. A change in water availability is likely to induce additional domestic water shortage only if, in addition to the resource being limited (i.e. physical depletion), the level of local water infrastructure is sufficient for people to consume as much available domestic water as physically available (a., Figure 7.2). If a country has insufficient water infrastructure to allow its population to access its water resources, a reduced availability might not imply additional domestic water shortage. This feature is characterised as "accessibility vulnerability". When affluent, a country might be able to adapt to a physical domestic water shortage by sourcing water elsewhere or by using "back-up technologies" (Boulay et al., 2011) (b., Figure 7.2). At the expense of economic costs, this can limit domestic water deprivation, where the population suffers from a water consumption below the minimum needs. This situation may put populations at risk of water-related diseases, as described by the "raw health effect" of domestic water deprivation. Depending on the local "health vulnerability" defined by local features such as the quality of the health care system, the final health burden

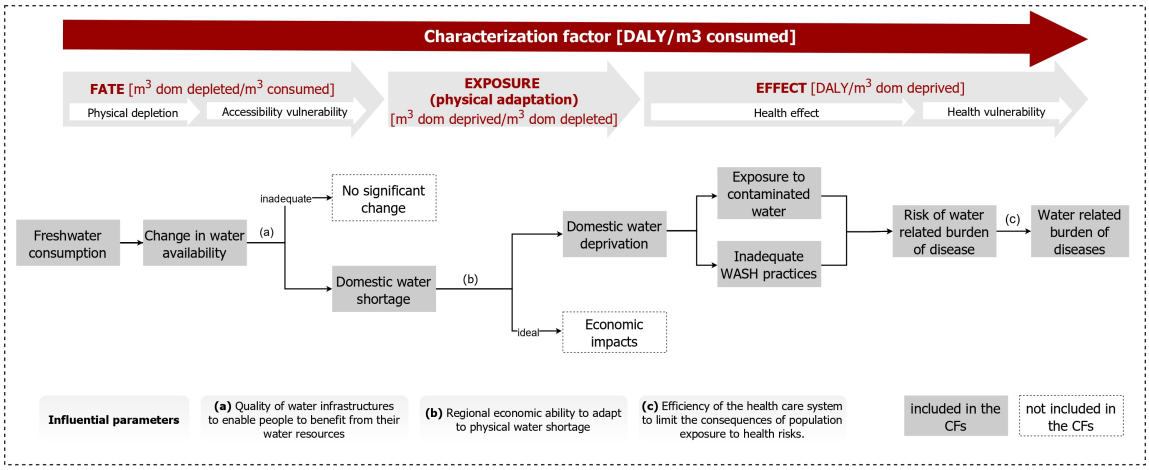


FIGURE 7.2 Impact pathway of water-related burden of diseases from freshwater use and domestic deprivation (adapted from Boulay et al. (2011)).

can vary (c., Figure 7.2).

Fate Factor

We develop a fate factor FF for domestic water scarcity in line with the WULCA working group's agricultural fate factor FF_{agri} (Frischknecht et al., 2016 ; Jolliet et al., 2018). As per Eq2, this fate factor results from the multiplication of a generic scarcity index with the proportion of a particular user's water consumption (agricultural in FFagri, domestic in FFdom) compared to total human water consumption. The generic scarcity index is calculated as the ratio of human water consumption (HWC) to the remaining water after human uses, defined as the Availability Minus human water Consumption (AMC). The simplification of identical terms allows to define the FF as the ratio of domestic HWC to AMC.

$$FF_{dom} = \frac{HWC_{total}[m^3]}{AMC[m^3]} \cdot \frac{HWC_{dom}[m^3]}{HWC_{total}[m^3]} = \frac{HWC_{dom}[m^3]}{AMC[m^3]} \tag{7.2}$$

$$AMC[m^3] = Availability[m^3] - HWC_{total}[m^3] \tag{7.3}$$

The resulting fate factor is limited to an upper limit of 1 building on the model of Frischknecht et al. (2016). WaterGAP (Flörke et al., 2013 ; Müller Schmied et al., 2014) provides data for HWC_{dom} and AMC at the watershed level.

Physical depletion and accessibility vulnerability As described by the white arrows inside the fate factor in Figure 7.2, the domestic water deprivation induced by water consumption and quantified by the fate factor results from two mechanisms : a physical depletion of water, influenced by the quantity of available water resources, and an accessibility vulnerability, which determines the extent to which this physical depletion is likely to impact populations given their current level of access to water.

Human Water Consumption (HWC) represents current water use. It is calculated as the multiplication of a national or regional water use intensity (in $m^3.cap^{-1}.yr^{-1}$, modeled through a sigmoid curve that increases with GDP per capita and is derived from historical data) with the population (Flörke et al., 2013). It therefore includes information on current water use quantities across different regions and on the level of water access and infrastructures.

A low fate factor can describe a situation in which the population does not suffer from water depletion since water availability and access are adequate, and water consumption is relatively low compared to water availability (low physical depletion potential). It can also express low water accessibility, that may lead to low water deprivation potential ; if local resources are abundant but water infrastructure is insufficient to provide water to the population (accessibility vulnerability is low), this latter is likely to use small quantities of water. Since water accessibility is the limiting parameter for these households' water consumption, a marginal use of water may have no direct impact on them. Households would not have been able to benefit from the resource independently of this additional water use. This situation is reflected by an associated low value of HWC_{dom}, and possibly fate and characterization factors. The authors consider that this aspect of the fate factor can address the concerns raised and mentioned above regarding the feasibility of linking water use flows from the inventory to human health damages due to domestic water deprivation in LCIA

(Frischknecht et al., 2016).

Exposure factor

The exposure factor used in this model focuses on the ease with which a region can adapt to a physical depletion of domestic water. It follows a linear relationship with the Adaptation Capacity of a country (AC), as per Eq4. The thresholds used to classify the adaptation capacity of countries to water deprivation (previously based on UNEP (2009)) are updated according to the latest classification of GNI per capita (Atlas method current $US, (The World Bank, s. d.)) : low-income (GNI per capita $\leq$ $1035), middle-income ($1035 $<$ GNI per capita $<$ $12536), and high-income countries (GNI per capita $\geq$ $12536). The adaptation capacity for low-income countries is zero, for high income countries 1 and for middle income countries, AC is calculated as per Eq5 :

$$XF = 1 - AC \tag{7.4}$$

$$AC = (8.696 \times 10^{-5} \times GNI) - 9.009 \times 10^{-2} \tag{7.5}$$

The User's Distribution factor Di as a part of the exposure factor in Boulay et al. (2011) becomes unnecessary : the extent to which 1m^3 use will deprive domestic users is here accounted for in the new fate factor via the ratio between human water consumption for domestic uses and the total human water consumption.

Effect factor

As described via the arrows in figure 7.2, the effect of domestic water deprivation on human health stems from a raw health effect modulated by the health vulnerability of a region to this latter. This raw health effect is the mean of the consequences induced by human exposure to contaminated water and to inadequate WASH practices, while the health vulnerability adjusts this value based on socioeconomic parameters such as the quality and accessibility of the region's health care system. Although

desirable, the developed EF factor does not capture this health vulnerability (see section Results). It is calculated as per Eq6, in accordance with Boulay et al. (2011).

$$EF = \frac{Water\ Related\ Burden\ of\ Disease,\ WRBD}{Domestic\ Water\ Deficit,\ DWD} \tag{7.6}$$

Water-related Burden of diseases WRBD The Subtotal drinking water, sanitation and hygiene DALYs attributable to inadequate WAter, Sanitation and Hygiene (WASH) generated by Prüss-Ustün et al. (2019) for the year 2016 are used to quantify the Water-Related Burden of Disease (WRBD). The term "inadequate" includes unsafe water and insufficient access. The data accounts for the fraction of burden of diseases attributed to inadequate WASH for diarrhoeal diseases, soil-transmitted helminth infections, respiratory infections, protein-energy malnutrition, trachoma and schistosomiasis. Expressed in DALY per type of disease and per country, this data is in line with Verones et al. (2017)'s recommendations to "continue using DALYs in LCIA for human health". Differences are observed in the comparison between this data and the initial model : while Boulay et al. (2011) integrate 50% of the burden of disease of malnutrition, Prüss-Ustün et al. (2019) estimate that 20% is attributable to inadequate WASH. The complete comparison between the drinking water, sanitation and hygiene DALYs calculated by Prüss-Ustün et al. (2019) and the WRBD as calculated by Boulay et al. (2011) for 2016 data is presented in the Supplementary Information (SI) C.

Domestic water deficit, DWD The domestic water deficit is defined as the sum of all individual water deficits at the country scale. Individual water deficit occurs when the amount of water used (in liters per capita per day, l/c/d) is inferior to the basic human water requirements (BHWR), defined as 50l/c/d (Gleick, 1996). This minimum quantity is still recognized by the World Health Organization (WHO, 2017).

To address the issue of water consumption inequality and quantify domestic water use -and domestic water deficit- we use the notion of access to private connection. Only

a fraction of the population within a country has access to water supply on premises (FPi) and their water consumption is generally higher compared to the population without access. This difference can vary between a factor 1.2 and 7, as found in the literature (Brown et al., 2013 ; Larson, Minten & Razafindralambo, 2006 ; Nunoo, Koomson & Orkoh, 2018).

A sensitivity analysis of values of p varying between 1 and 7 was carried out and the value of 6.3 was defined as the most representative of water use quantity inequality, based on Larson, Minten et Razafindralambo (2006) (see SI). Correlation is strong between the water-related DALY values per capita and the percentage of the population with access to water on premises (inversely proportional with R2 = 0.68, see SI), supporting the integration of this indicator in our calculations.

Considering that the total domestic water use in a country is the sum of domestic water use (DWU) of households with and without access on premises (respectively called Population With Access on Premises, PWAP and Population without Access on Premises, PwtAP and both expressed in percentage of total population) [Eq7], and that the former uses a quantity of water greater by a factor p than the latter [Eq8], we have :

$$DWU[m^3/cap] = PWAP \cdot DWU_{PWAP} + PWtAP \cdot DWU_{PWtAP} \tag{7.7}$$

$$DWU_{PWAP}[m^3/cap] = p \cdot DWU_{PWtAP}[m^3/cap] \tag{7.8}$$

$$DWU_{PWtAP}[m^3/cap] = \frac{DWU[m^3]}{p \cdot PWAP + PWtAP} \tag{7.9}$$

The total domestic water deficit (DWD) is calculated at the country scale as the sum of the domestic water deficit of the population with access to water on premises (PWAP) plus the domestic water deficit of the population without access to water on premises (PWtAP) as per Eq10 :

$$DWD[m^3_{deprived}] = \sum_{i=PWAP,PWtAP} (BHWR - DWU_i)[\frac{m^3_{deprived}}{cap}] \cdot FP_i[\oslash] \cdot Pop[cap] \tag{7.10}$$

Where BHWR stands for Basic Human Water Requirements (50l/c/d), DWU_i for Domestic Water Use of population i, FP_i for Fraction of the Population i (with or without access to water on premise) and Pop for the total population in the country. The total domestic water use (DWU) is estimated with the "municipal withdrawals" from FAO (2011). The percentages of the population with access to water on premises were provided from Prüss-Ustün et al. (2019) for the year 2016. The results of this study are compared to a "general" version of Boulay et al. (2011)'s model. This latter differs slightly from the published model as it excludes quality and source-specific aspects, making it comparable to our model. (see SI C for additional information).

7.2.3 Results

Fate Factor

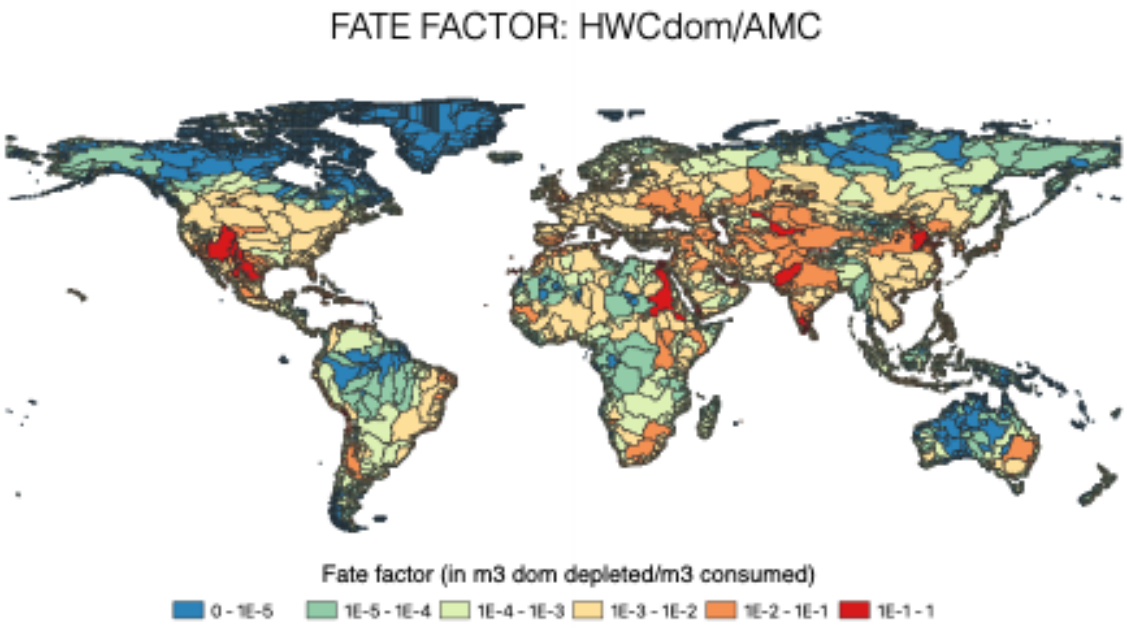


FIGURE 7.3 Revised fate factors [$m^3_{dom\ depleted}/m^3_{consumed}$]. Lower (respectively higher) values appear in blue (red). These fate factors account for areas where the potential domestic water deprivation caused by an incremental use of water from the current situation is low (respectively high).

The revisited fate factor is calculated for 12013 cells accounting for the river basin level (with the 34 world's largest river basins being divided into sub-basins). High fate factors (red cells with FF ranging between 0.1 and 1 m^3 domestic depleted water per m^3 of water used) in Figure 7.3 account for high physical water depletion potential in areas such as the Nile valley, California, Nevada or Arizona. In low population areas such as central Australia, northern Canada, the Amazon rainforest, northern Russia or Saharan Africa, low fate factors can be explained by low water demand. Low fate factors can also be explained by low accessibility vulnerability.

For instance, the Democratic Republic of the Congo (DRC) (where FF values are as low as 2.79E-7) is endowed with important freshwater resources, but governance choices and underinvestment in the water sector and infrastructures have seriously affected the population's water access (UNEP, 2011). DRC's mean fate factor (currently equal to 9.4E-5 m^3 deprived/m^3 consumed) would reach 7.0E-3 if the country's population could use 50lpcd, and 1.4E-2 if it could use 100lpcd. Details on the results of HWC_{total}/AMC and HWC_{dom}/HWC_{tot} ratios are presented in the SI.

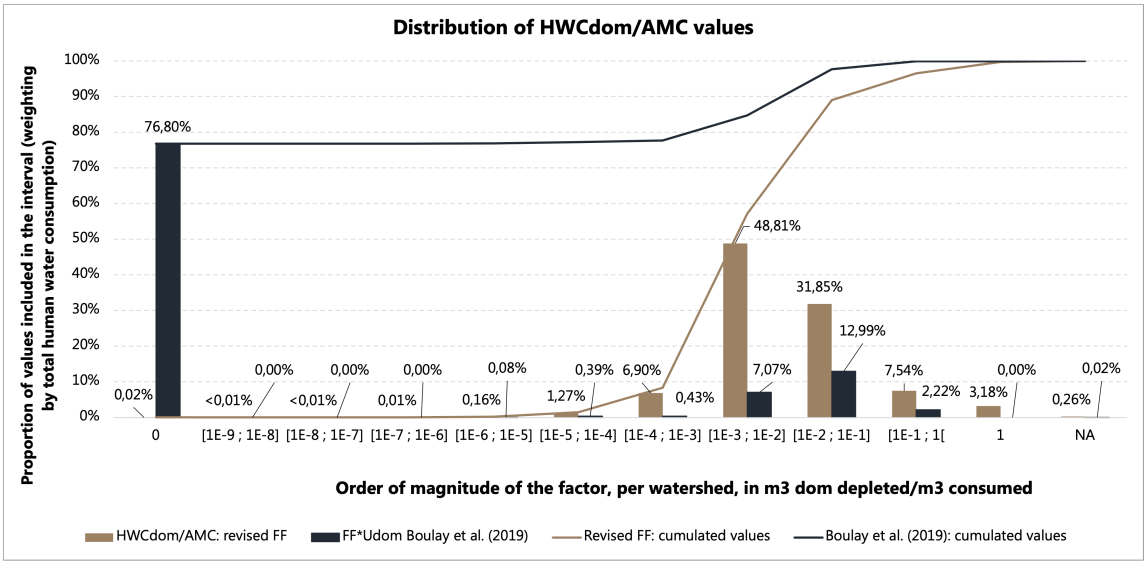


FIGURE 7.4 Water-use weighted distributions of fate factors in this model and in Boulay et al. (2011). The distribution is based on the factors' order of magnitude. To make the factors comparable, the revised fate factors are compared to the combination of Boulay et al. (2011)'s fate factors with the proportion of water use by domestic users (initially included in the XF in Boulay et al. (2011). For the revised (respectively Boulay et al. (2011)'s) model, brown (respectively dark blue) bars represent the percentage of total water use whose fate factors are included in corresponding intervals (expressed in m^3 depleted/m^3 consumed). Solid lines represent the cumulative distribution for each model. NA values account for missing values. The choice of a "general" version of Boulay et al. (2011)'s model as a reference for comparison can explain the large number of zero values.

Figure 7.4 illustrates the water-use weighted distribution of fate factors in this model and in Boulay et al. (2011). 95% of revised fate factors are included between 1E-4 and 1 m^3 dom depleted/m^3 consumed. Respectively, 95% of Boulay et al. (2011)'s non-zero values are included between 1E-3 and 1 m^3 dom depleted/m^3 consumed. Overall, the distribution curve of non-zero factors shows a larger distribution and is shifted to the left by about one order of magnitude in the revised model compared to the previous one. Additional results are presented in the SI.

Exposure Factor

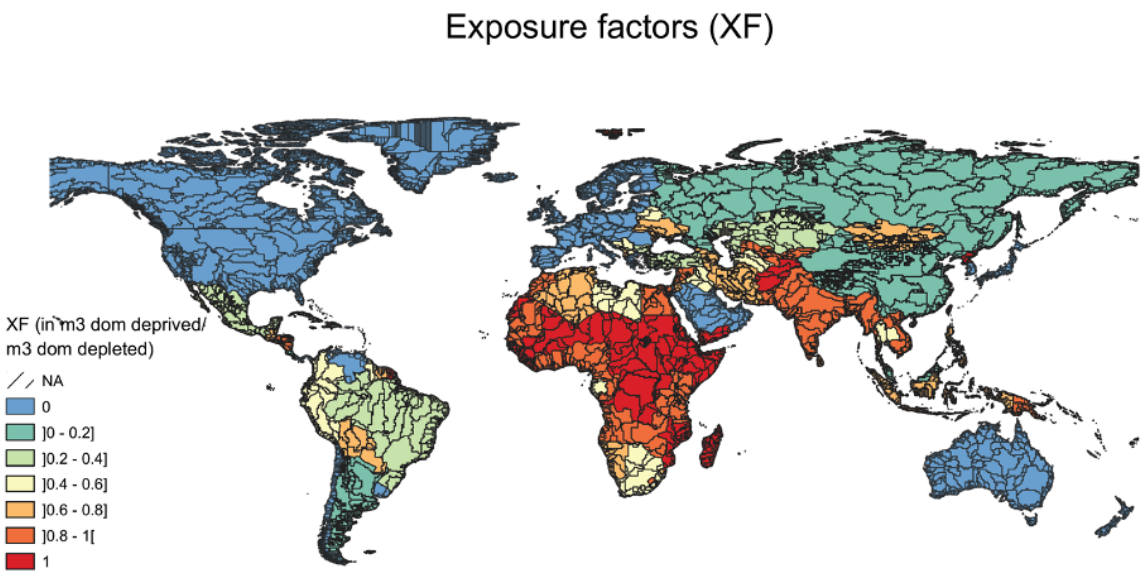


FIGURE 7.5 Graphical distribution of the updated exposure factors (in m^3 dom deprived/m^3 dom depleted). Building on Boulay et al. (2011), high value exposure factors are representative of countries that encounter difficulties to adapt to water shortage

Figure 7.5 provides a visual representation of the geographical distribution of exposure factors. Additional information on this distribution is presented in the SI. Blue, high-income countries can adapt to domestic water depletion (XF=0 and no

domestic water deprivation) while red, low-income countries are fully impacted and 1m^3 domestic water depletion implies 1m^3 domestic water deprivation (XF=1).

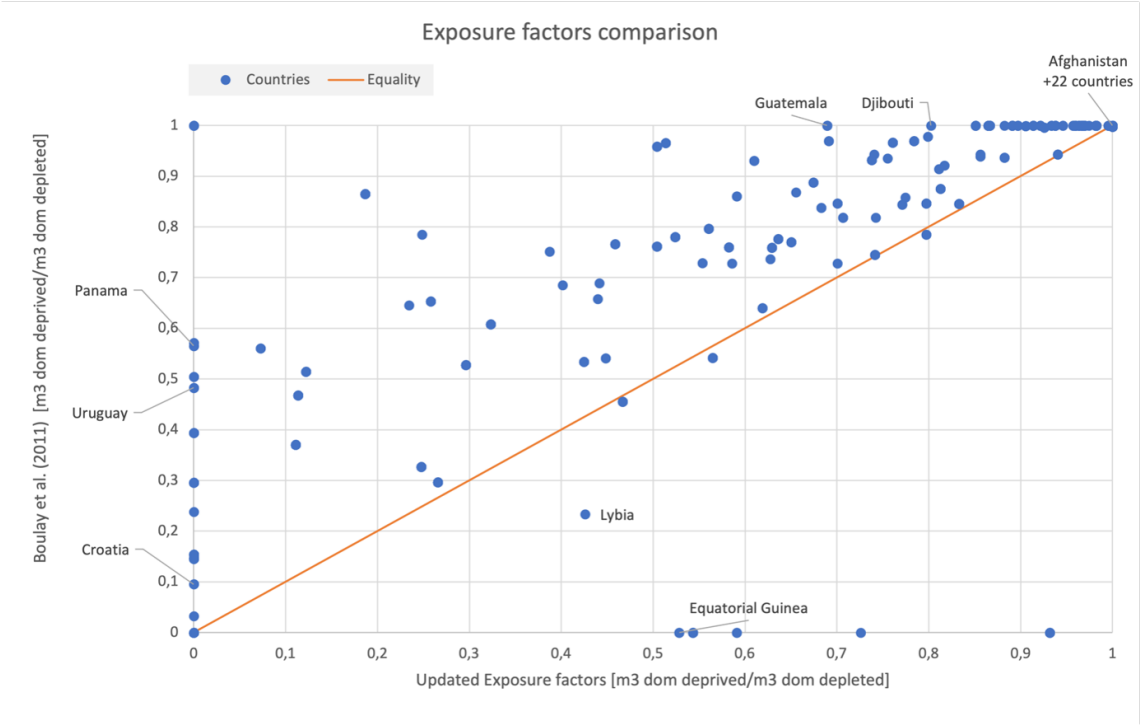


FIGURE 7.6 Comparison of the exposure factors calculated in Boulay et al. (2011) and this revised model. For comparison compatibility purpose, the revised exposure factors are compared with Boulay et al. (2011)'s exposure factors to which the proportion of water use by domestic users has been deducted. Each point corresponds to the exposure factor of a country in the revised model (abscise) and in the original Boulay et al. (2011) model (ordinate) in m^3 dom deprived/m^3 dom depleted. The orange line represents equality between both models.

Figure 7.6 shows the variation between Boulay et al. (2011)'s and the updated exposure factors. Of the 209 values calculated by Boulay et al. (2011), 84 remain identical (XF =0 in 61 countries, XF=1 in 23 countries). Most points are located above the line, showing a tendency of decreased exposure factors in the updated model. Accordingly, 27 countries are no longer considered unable to adapt, with exposure factors

evolving from 1 to inferior values (e.g., Guatemala, Djibouti, Papua New Guinea or Vietnam). This is consistent with the reduction of low-income economies from the year 2009 (reference year of Boulay et al. (2011)) to 2019 (present study), i.e countries in which the GNI per capita is below the threshold of 1035$ in 2019 (936$ in 2007) defined by the World Bank countries classification. Points on the Y-axis (updated exposure factors=0 vs. Boulay et al. (2011) $\neq 0$) describe countries whose economic group has changed and are no longer likely to suffer from domestic water shortage (e.g., Mauritius, Panama, Romania, Uruguay or Venezuela). Libya, which has suffered instability since its civil war in 2011, is one of the 8 countries whose exposure factor has increased since the calculations of Boulay et al. (2011) (from 0.23 to 0.43 m^3 deprived/m^3 depleted). Similarly, Equatorial Guinea's change of economic group from high to upper middle-income country between 2009 and 2019 explains the important change of its exposure factor (from 0 to 0.53).

Effect Factor

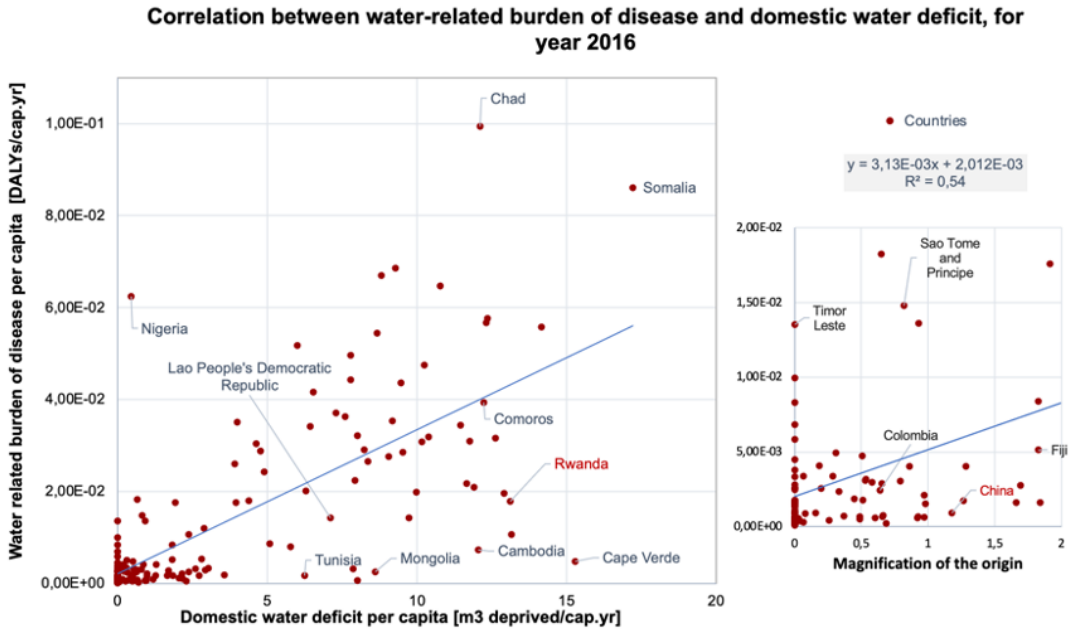


FIGURE 7.7 Correlation between domestic water deficit (x) and water-related burden of disease (y), for the year 2016. Blue points refer to countries. The continuous blue line is the linear regression between all points. The slope of the latter defines the global effect factor, expressed in DALYs/m^3 deprived.

Figure 7.7 shows the correlation between the domestic water deficit and the water-related burden of diseases (R2=0.59). The slope of the linear regression can be considered as the mean effect of water deprivation on human health, which has been called the "raw health effect". Vertical distance of points to the linear regression accounts for the local "health vulnerability", assessing the variability of health damage intensity due to water deprivation among countries.

Defining country-scale effect factors as the ratio of the WRBD and the domestic water deficit for each country would allow to capture this local health vulnerability. Different country features (health care system, education, cultural habits or climate) might influence the health consequences of an identical value of domestic water defi-

cit. For instance, Chad and Comoros both suffer from a 1.2E+1 m^3 domestic water deficit per capita (Figure 7.7), but the amount of DALYs related to water diseases per capita is 2.5 times superior in Chad (9.93E-2 vs 3.94E-2 in Comoros). Lower Human Development index (HDI) and Gross National income (GNI) per capita, along with fewer expected years of schooling in Chad than in Comoros (United Nations Development Programme) may explain this gap. Similarly, Nigeria and the Islamic Republic of Iran suffer from a 4.5E-1 m^3 deficit per capita, while Nigeria's amount of water-related DALYs is 34 times higher. The same trend between the two countries can be observed regarding the HDI, GNI and expected years of schooling (lower in Nigeria). A single value of country-scale effect factors can describe two different realities : while the ratio between the water-related burden of diseases and the domestic water deficit is the same for the Democratic Republic of Congo and Malaysia (4.67E-3), the numerators and denominators of these ratios are largely different, and so are socio-economic factors of the countries.

Important limitations in this calculation must be highlighted. Country-scale effect factors are highly sensitive to small changes of the numerator or denominator. This is especially true for small values. Although the work of Prüss-Ustün et al. (2019) has reduced uncertainty about the number of DALYs attributable to water-related diseases, the quantification of water deficit, necessary to calculate the effect factor, remains largely uncertain (see Discussion). These flaws can explain some unlikely country-wide EF values (e.g. Switzerland, EF=7.69E+2 DALY/m^3 deprived) and the unlikely difference between different values (up to 7 orders of magnitude).

To rule out absurd country scale effect factor values influenced by the high uncertainty of the water deficit data, it was decided not to discriminate between countries and only use a global value at this point. The effect factor is therefore defined as the slope of the linear regression and equal to 3.13E-3 DALYs/m^3 deprived, which compares to the value of 3.11E-3 in Boulay et al. (2011).

Characterization factor

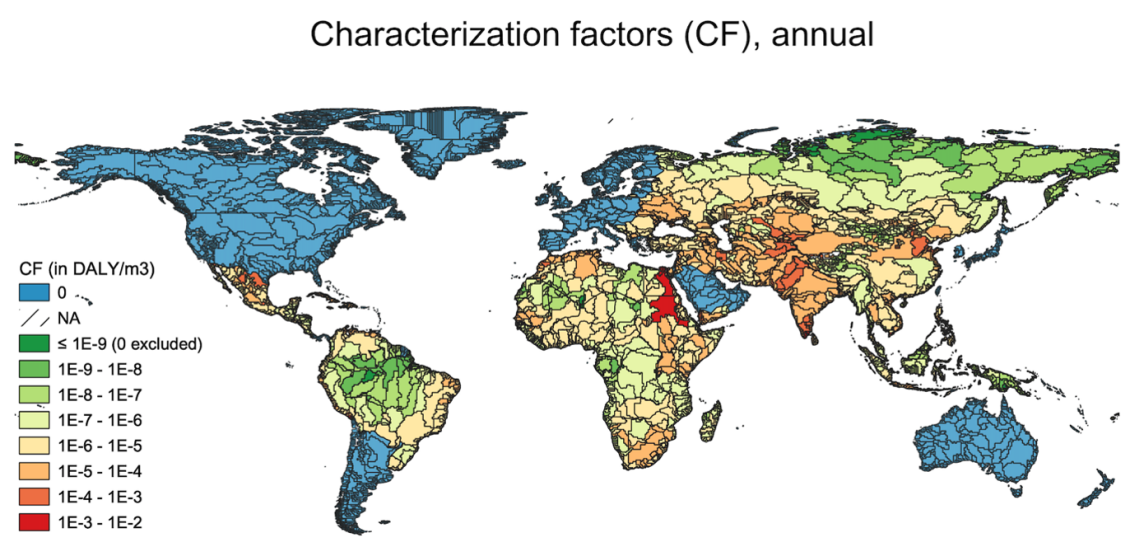


FIGURE 7.8 Spatial distribution of the updated characterization factors (in DALY/m^3 consumed)

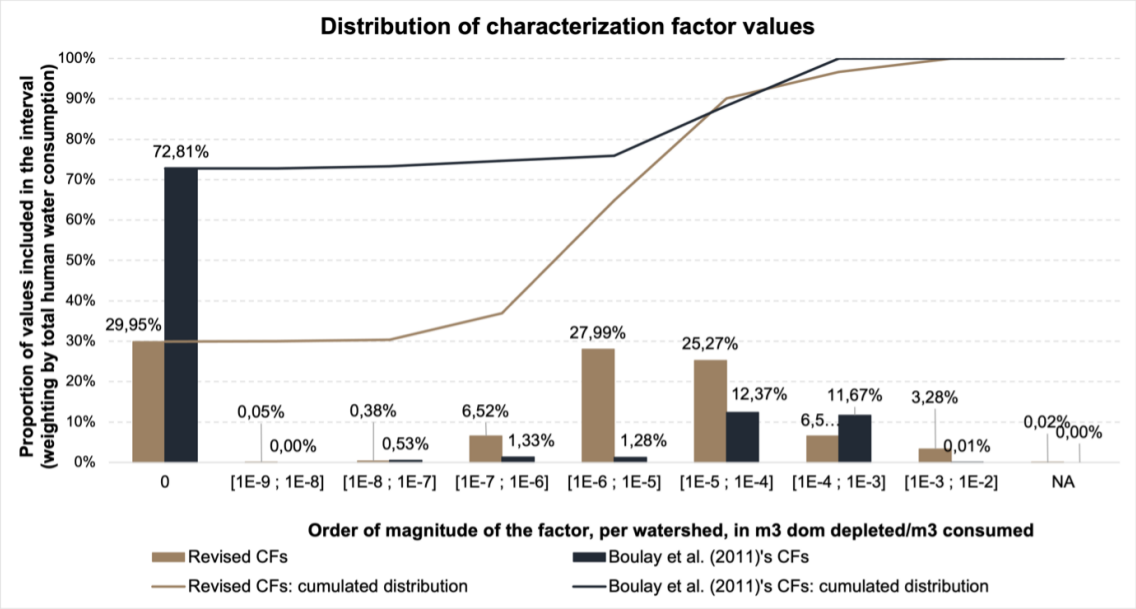


FIGURE 7.9 Water-use weighted distributions of revised (in brown) and of Boulay et al. (2011)'s (in dark blue) CF values (annual, in DALY/m^3 consumed).

Characterization factors are calculated for 17931 cells (unavailable data for 178 cells). Water-use weighted distributions of revised (in brown) and of Boulay et al. (2011)'s CFs are represented in Figure 7.9. 8487 revised characterization factors are equal to zero, accounting for 29.95% of the total annual human water consumption (Figure 7.9). Characterization factors range from 0 DALY/m^3 (the potential impact on human health due to water use is null with respect to domestic water deprivation) to 3.13E-3 DALY/m^3. The highest values are observed in Ethiopia, Sudan and Eritrea (3.13E-3), Senegal (3.02E-3) and Egypt (2.98E-3). Maximal (=1) fate and exposure factors in Eritrea, Sudan and Ethiopia explain these high CFs. In Senegal and Egypt, exposure factors are high (>0.95) and paired with maximal fate factors (=1). Relatively low values do not necessarily express the absence of water-related health issues or of water stress but describe the lower potential health damages caused by an incremental use of water from the current situation. For instance, Southern Algeria CFs are low (4.1E-10 to 8.6E-07 DALY/m^3) because of low fate factors related to low

domestic water consumption. This low water consumption can be explained by a low population density, characteristic of the Saharan desert. Low CFs can also be observed in the Democratic Republic of the Congo, where the exposure factor is maximal (=1), but low fate factors driven by low water use result in low CFs. Country scale CFs were calculated by weighting the factors by water use and are presented in the SI C, along with an additional comparison with Boulay et al. (2011)'s results.

7.2.4 Discussion

Applicability limits

This impact model relies on current water use and water availability data as provided by WaterGAP (Flörke et al., 2013) ; it provides no insight on long-term trends. In line with Pradinaud et al. (2019a)'s recommendations, we consider that the CFs developed in this work assess the consequences on human health induced by water use leading to short-term water deprivation.

Revised intermediary factors

Fate Domestic water use is hardly monitored worldwide and is here based on WaterGAP data. WaterGAP data was used in the calculation of the fate factor since offering consistency with current LCA water use assessment practices. It provides high geographical resolution (watershed), which presents interest since "national-level withdrawals information dilutes the usability of technical information as the country average value is not necessarily representative of any specific region in a country" (FAO, 2011). Nonetheless, it does not provide an exact picture of domestic water use since it does not provide insights on how water use is unevenly distributed among the population. WaterGAP defines the domestic water sector as "household use, small businesses and other municipal uses, which take high quality water directly from the municipal pipelines when it is available" (Alcamo et al., 2003). Data disaggregation to focus on households' water use solely is not possible.

Exposure Water access issues are complex and can not be assessed by economic parameters only. While the ISO 14044 norm describes environmental mechanisms as "systems of physical, chemical and biological processes", socioeconomic, geographical, cultural or political characteristics have to be taken into account to fully understand the mechanisms leading to potential water use impacts. Governance, political or investment choices have an important influence on the quality of water access within a country (The Lancet, 2014).

Effect Several limitations must be underlined in the calculation of the effect factor, whether it is about the numerator (WRBD) or denominator (DWD) of this ratio.

- WRBD : Boulay et al. (2011) calculate the WRBD as the sum of the burden of diarrheal diseases and 50% of the burden due to malnutrition. Prüss-Ustün et al. (2019) provide updated estimates of the burden of diseases due to exposure to inadequate drinking-water, sanitation, and hygiene behaviours. These latter estimates are 25% lower compared to Boulay et al. (2011)'s calculations (see SI) and are used in this work to strengthen the quantification of the WRBD. Nonetheless the WRBD provided by both avenues results from the contribution of more than the sole lack of domestic water use. While sparse epidemiological works point out the links between restrictions in water quantities and water-related health issues, a majority of studies link water related diseases to flooding, heavy rainfalls or temperature (Levy et al., 2016 ; Wang, Goggins & Chan, 2018). In this work we define the WRBD as the subset of water related diseases provided by Prüss-Ustün et al. (2019) linked with drinking-water, sanitation, and hygiene. This excludes the burden of diseases of malaria, which is linked to water resource management (e.g. elimination of stagnant water) rather than to water quantity. Nonetheless, additional efforts to identify the burden exclusively linked to water shortage are needed since current values are likely to still be overestimated.

- Domestic Water Deficit : The basic human water requirements and the domestic

water use are both highly uncertain.

Basic Human Water Requirement : The value of basic human water requirements of 50l/c/d, although recommended by the World Health Organization, is still debatable. According to Chenoweth (2008), 50l/c/d allow most of hygiene requirements to be met but 100l/c/d are needed to cover them all. Accordingly, the World Health Organization (Howard & Bartram, 2003) and the United Nations Development Program (Watkins, 2006) recommend a range of 50 to 100l/c/d of water to be piped into households for human development and to maintain adequate health. Human basic water requirements largely depend on characteristics such as standard of living, lifestyle, physical conditions, culture and/or climate. Well-off and urban households are likely to own more water-consuming appliances (lavatories, dishwashers, washing machines etc.) and a minimum of 50l would be insufficient for these households. Therefore, global economic development and urbanization could shake the current consensus about this value. The debate is far from being closed and some consider that the "effective use of water for hygiene purposes is more important than the quantity used" (Chenoweth, 2008). A rise of the value of the minimum water requirement would increase the domestic water deficit, leading to a decrease of the effect and characterization factor (cf. sensibility analysis in the SI).

Domestic water use : For country-scale effect factors, Aquastat's country scale "municipal withdrawals" data was preferred as a proxy for domestic water use since no higher geographical resolution was needed. Aquastat's "municipal withdrawals" represent water provided by public network means. They supply households but also other urban services (stores, markets, tourism centers, urban industry). The level of detail is insufficient to allow disaggregation of the values (FAO, s. d.). Aquastat's definition for municipal withdrawal does not clearly include or exclude informal water collection such as rainwater harvesting. Consequently, each country could either include it into its reporting or not (Gillet, 2020). In practice, few countries monitor such data. This is only the result of individual initiatives that are difficult to monitor, in particular in developing countries where consumption patterns have not

been well documented and investigated (Hussien, Memon & Savic, 2016). In general, assumptions and estimates have to be made to fill the gap of water consumption data (Cole et al., 2018). Accessibility is often used as a proxy for quantity but the "relationship between accessibility and the quantity of water collected and available for consumption in low and middle-income settings remains as an important literature gap" (Cassivi et al., 2019). Water access inequalities are not similar to income inequalities (Berthe, 2016) and many characteristics can explain variations in terms of water use quantity : location, education, wealth, age, gender, ethnicity, electricity access, water collection time or the number of adult female members in the households (Abubakar, 2019 ; Hussien, Memon & Savic, 2016). Water access and the proportion of water used by households with and without private access appeared to be the most suitable proxy for water use quantification at this stage. We introduced the concept of a factor p to account for water quantity inequalities between households with and without a private connection to water supply (p being the fraction of the domestic water use of the population within a country that has access to water supply on premises vs. the domestic water use of the population without access). However this is still insufficient to picture the extent of water use inequalities and to provide a robust enough domestic water use profile. The extent of quantity inequalities (factor p) can not be generalized among all countries and regions, and there is little evidence of the most suitable value to choose according to the studied region. Nonetheless, a change in this value showed to have no significant influence on the global value of the effect factor (see SI). These shortcomings in the estimation of water needs and consumption lead to significant uncertainty in the calculation of the domestic water deficit. This uncertainty is reflected in the effect factor for which a country-wide distinction does not seem feasible at this stage.

7.2.5 Conclusions and recommendations

In response to the lack of consensus on the best LCIA practice to characterize the domestic impacts on human health induced by water use, this research work provides updated and operationalizable factors to better account for the impacts of water use

at the endpoint level. Building on Boulay et al. (2011), we consolidate the cause effect chain linking water consumption to domestic impacts on human health, and we provide revised characterization factors ranging from 0 to 3.11E-3 DALY/m^3 consumed. We updated and revisited intermediary factors and highlighted a need for additional refinements. Fate factors have been adapted in line with current water assessment methods. Called water deprivation potential, they build on local circumstances and combine notions of physical water depletion potential and accessibility vulnerability to water shortage. Identifying the level of water infrastructures as a key parameter in populations' likelihood to suffer from water deprivation, this factor contributes to the consolidation of the cause effect chain and supports the integration of human health impacts due to domestic water deprivation into water use assessment. The choice of the GNI per capita as a proxy to assess the capacity of a region to adapt to physical water shortage within the exposure factor could be revised by considering other socio-economical parameters such as the economic component of the Inequality-Adjusted Human Development Index. This latter indicator is used in the AC to agricultural water deprivation in Frischknecht et al. (2016). Current modeling of the effect factor still fails to capture the share of water-related burden of diseases that can be directly linked to water shortage. Nor does it allow a realistic evaluation of domestic water deficit. This latter flaw induces a regrettable loss of information on local health vulnerability to water deprivation. We therefore argue for additional research efforts to develop an alternative method for calculating the effect factor.

CHAPITRE 8 ARTICLE 3 : ASSESSING THE POTENTIAL HUMAN HEALTH IMPACTS OF FRESHWATER CONSUMPTION : CONSIDERING FRESHWATER USE INEQUALITIES TO ASSESS DOMESTIC WATER DEPRIVATION

Assessing the potential humain health impacts of freshwater consumption : considering freshwater use inequalities to assess domestic water deprivation

Laura Debarre, Masaharu Motoshita, Stephan Pfister, Anne-Marie Boulay, Manuele Margni

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8.1 Présentation de l'article

Le chapitre 7 introduit un modèle révisé pour évaluer les impacts sur la santé humaine résultant de la consommation d'eau. Cependant, il met en lumière des limitations significatives dans l'élaboration de son facteur d'effet. Ce dernier est une valeur globale et ne permet pas de rendre compte des disparités d'accès à l'eau au sein d'une population. Pour pallier cette limite, l'article présenté dans ce chapitre propose une nouvelle approche pour le calcul du facteur d'effet, reposant sur l'hypothèse que les conséquences sur la santé associées à la privation d'un m^3 d'eau pour un utilisateur domestique dépendent de la quantité utilisée quotidiennement par celui-ci.

Cet article est le fruit d'une collaboration menée dans le cadre de la phase 3 du projet GLAM porté par la Life Cycle Initiative. Il a été coécrit par Laura Debarre, Masaharu Motoshita, Stephan Pfister, Anne-Marie Boulay et Manuele Margni, membres du groupe de travail "GLAM Water Scarcity Impacts - Domestic". Cet article a été publié dans l'*International Journal of Life Cycle Assessment* le 11 novembre 2024.

Les CFs développés dans le cadre de ce projet ont été intégrés aux méthodologies EICV GLAM et IMPACTWorld+. Le matériel supplémentaire associé à l'article est disponible en Annexe E.

8.2 Manuscript

Abstract

Abstract Purpose : The consumption of freshwater can increase local competition among ecosystems needs, agriculture, and domestic users. This competition can lead to reduced domestic water availability and subsequent inadequate hygiene practices leading to water-related diseases. Among the few attempts to develop endpoint-oriented characterization models and factors to assess this impact pathway, limitations still need to be addressed to improve the effect factor (EF) representing the marginal increase in health damage associated with 1 m^3 of water consumed. This research proposes a revised country-scale effect model assessing diarrheal diseases due to domestic water deprivation, considering different levels of water availability within a country.

Methods : The calculation of the EF is based on the principle that the probability of negative health consequences associated with depriving a domestic user of 1 m^3 of water depends on the quantity used daily by this user. Three classes of domestic water users are defined based on their range of daily water use. EFs are calculated for each class of domestic users building on a comparative risk assessment methodology and households' levels of access to drinking water and sanitation from the Joint Monitoring Program. Country-specific EFs are computed as the weighted

sum of class-specific EF proportionally to the population within each class. Revised country-specific EFs are used to overwrite the generic constant EF used in previous characterization models and to compute new characterization factors (CFs).

Results and discussion : Class-specific effect factors equal to 1.35e-3, 3.44e-4 and 7.53e-5 DALY/m^3 for the three classes of users, showing a 57, 89, and 98% reduction compared to previous characterization models. Country-specific EF values range from 7.5e-5 to 8.7e-4 DALY/m^3 deprived (M ; 2.5e-4 ; SD :1.8e-4, representing a reduction of 72.2 to 97.6% compared to previous models. New 11987 CFs were compiled ranging from 0 to 7.63e-4 DALY/m^3 consumed (M : 1.9e-6 ; SD : 2.1e-5). The global potential impact induced by water consumption over the year 2019 computed with our model reaches 2.77e+7 DALYs, corresponding to 50% of the water-related burden of diarrheal disease calculated by a recent epidemiology study, which confirms the plausibility of our results. Conclusions Unlike previous methods, our revised EFs acknowledge inequalities in domestic water consumption within a country. Revised EFs are calculated by country, going beyond the global resolution of previous models, compared to which they show a reduction of 72 to 98%. A sanity check confirms the plausibility of our CFs but does not rule out a potential overestimation. Future research is needed to provide empirical evidence supporting a causal link between water deprivation and diarrheal diseases and to assess uncertainties of the model results.

Keywords : Water consumption, Human health, Diarrhea burden of diseases, Freshwater, Domestic water deprivation, Water use inequalities, LCA, LCIA

8.2.1 Introduction

Safe drinking water is needed for domestic purposes that include hydration, food preparation, household, and personal hygiene (World Health Organization, 2022). The amount of safe freshwater used by households highly influences people's health (World Health Organization, 2022).

Domestic water use, which refers to the volume of freshwater withdrawn from sources,

varies considerably between and within countries (Cosgrove & Loucks, 2015). Studies show that daily freshwater use per capita is influenced by socioeconomic characteristics (Hussien, Memon & Savic, 2016 ; Matos et al., 2014 ; Nawaz et al., 2019), the number of women in the household (Hussien, Memon & Savic, 2016), and the level of freshwater accessibility (i.e., whether people are able to utilize the physically available water resources) (World Health Organization, 2022). Since data on domestic water use is scarce and difficult to monitor, the water access level is often considered as a "valid proxy for the quantity of water collected by households" (World Health Organization, 2022).

In a region facing high water scarcity (i.e., the "extent to which demand for water compares to the replenishment of water in an area" (ISO 14046 :2014, s. d.)), the consumption of freshwater (i.e., the use of freshwater without returning it to its source) may lead to a decrease in freshwater resource stocks and to water deprivation for other users (Boulay et al., 2011 ; Debarre, Boulay & Margni, 2022 ; Motoshita et al., 2018). A reduction in water availability (i.e., the "extent to which humans [...] have sufficient water for their needs" (ISO 14046 :2014, s. d.)) can result in health issues for local populations. There is an inverse correlation between the amount of water used by households and diarrheal diseases (see SI). By depriving domestic users, freshwater consumption can lead to insufficient coverage of households' daily needs (Boulay et al., 2011 ; Debarre, Boulay & Margni, 2022). Households that do not have sufficient access to adequate quality water may use unreliable water sources (Nyemba et al., 2010) and have poor hygiene practices (World Health Organization, 2022). The latter behaviors expose households to waterborne and water-washed diseases (Howard & Bartram, 2003).

Considerable research efforts have been made to provide information on freshwater issues for decision-making, to promote sustainable management of water (Prüss-üstün et al., 2002 ; Wolf et al., 2014, 2022). In the field of life cycle assessment (LCA), the willingness to better account for water-related issues in decision-making processes has resulted in the development of an ISO norm (ISO 14046 :2014, s. d.) and various

characterization models and frameworks (e.g., Bayart et al. (2010), Boulay et al. (2018), Kounina et al. (2013), Lathuillière, Bulle et Johnson (2018), É. Pierrat et al. (2023) et Pradinaud et al. (2019b).

Endpoint-oriented characterization models and factors (CF) have been developed to assess the potential impacts of water consumption on human health considering both agricultural (Boulay et al., 2011 ; Mikosch et al., 2021 ; Motoshita et al., 2018 ; Pfister, Koehler & Hellweg, 2009) and domestic users' (Boulay et al., 2011 ; Debarre, Boulay & Margni, 2022 ; Mikosch et al., 2021 ; Motoshita, Itsubo & Inaba, 2011) deprivation.

Boulay et al. (2011) developed two models for characterizing water use impacts on human health : (1) the *distribution model* in which every user competes for water resources and is affected by additional water usage proportional to their allocated share and (2) the *marginal model* focusing on the impact solely on marginal users, typically agricultural users with a lower willingness to pay for water compared to domestic or industrial users.

Building on Boulay et al. (2011), Debarre, Boulay et Margni (2022) revisited and updated characterization factors (CFs) of the distribution model, expressed at the watershed level in disability-adjusted life years (DALY)/m^3 consumed. The DALY is a unit representing the loss of the equivalent of one year of full health including both premature mortality and disability (WHO, 2013). Debarre, Boulay et Margni (2022)'s model operates under the premise that water sources and reservoirs within a specific catchment area are interconnected. Consequently, increased water scarcity resulting from additional resource exploitation affects the entire catchment area equally, diminishing the available water for all users. The CFs are designed to account for the potential impacts on human health caused by water consumption resulting in domestic water deprivation and inadequate hygiene and sanitation practices. The CFs follow the general LCIA framework (Rosenbaum et al., 2008) as per Eq. 1.

$$CF_{i,j} = FF_i \cdot XF_j \cdot EF \tag{8.1}$$

Equation 1 : general equation of CF in Debarre, Boulay et Margni (2022). CFi,j : characterization factor in watershed i and country j (DALY/m^3 consumed) ; FF : fate factor (m^3 depleted/ m^3 consumed), representing the level of potential freshwater deprivation for other users resulting from water consumption ; XF : exposure factor (m^3 deprived/m^3 depleted), representing the capacity of the country to adapt to a change in water availability ; EF : effect factor (DALY/m^3 deprived) assessing the magnitude of the effects on human health caused by a deprivation of water for the different users.

Debarre, Boulay et Margni (2022) adapted the fate factor (FF) developed for agricultural users by the Water Use in Life Cycle Assessment (WULCA) working group (Boulay et al., 2018) (Eq. 2) and updated the data used in Boulay et al. (2011) to calculate the exposure factor (XF) at the country level (Eq. 3).

$$FF_i[\frac{m^3\ depleted}{m^3\ consumed}] = \frac{HWC_{dom,i}}{Availability_i - HWC_{total,i}} \tag{8.2}$$

Equation 2 : fate factor for watershed i. HWCdom,i : human water consumption for domestic purposes in watershed i. HWCtotal,i : human water consumption for all purposes in watershed i.

$$\mathrm{XF}_j \left[\frac{\mathrm{m}^3_{\mathrm{deprived}}}{\mathrm{m}^3_{\mathrm{depleted}}}\right] = 1 - AC_j,$$
$$\text{with } AC_j = \begin{cases} (7.87 \times 10^{-5} \times GNI_j) - 8.94 \times 10^{-2} & \text{for } 1136 < GNI_j < 13{,}845 \\ 0 & \text{for } GNI_j \leq 1136 \\ 1 & \text{for } GNI_j \geq 13{,}845 \end{cases} \tag{8.3}$$

Equation 3 : exposure factor for country j, where GNIj is the gross national income per capita in country j in 2022. The values 1136 and 13845 correspond to the lower and upper thresholds, respectively, of the low-income and high-income country classes

provided by the World Bank for 2022 GNI per capita data.

Debarre, Boulay et Margni (2022) have revised the global effect factor (EF) calculation of Boulay et al. (2011) (equal to 3.11E-3 DALY/m^3 deprived) as per Eq4 and provide global EFs equal to 3.13E-3 DALY/m^3 deprived.

$$EF = \frac{Water\ Related\ Burden\ of\ Diseases[DALY]}{Domestic\ Water\ Deficit\ [m^3\ deficit]} \tag{8.4}$$

Although they provide an improved version, Debarre, Boulay et Margni (2022) highlight limitations in the calculation of the EF. While the revised EF relies on data of water consumption to quantify country-wide domestic water deficits, Debarre, Boulay et Margni (2022) recognize that the approach fails to account for disparities of water consumption within countries. Besides, the "water-related burden of disease" does not further distinguish the causes of water-related DALYs that can be generated by not only water deficit but also water excess (e.g., floodings). In addition, a constant EF without spatial resolution does not reflect the way in which the consequences of a lack of water can vary from one region to another. Therefore, Debarre, Boulay et Margni (2022) recommended exploring new approaches to calculate the EF, acknowledging water users' disparities in terms of water access and of consequences of water deprivation.

Burden of disease analyses (BoD) are used to provide information on the state of health of a population (Plass et al., 2022). BoD assessments attribute disease burden to a set of risk factors using methods such as the comparative risk assessment (CRA) methodology (Murray & Lopez, 1999).

A CRA assesses the changes in population health that may occur due to changes in population distribution of exposure to the risk factors under study (Murray et al., 2003). In a CRA, the different levels of exposure to risk factors in a population are paired to relative risks (RR). These latter RR reflect the extent to which the health

risk increases with the level of exposure (Murray et al., 2003). RR are calculated as the ratio of the probability of an outcome occurring in an exposed group divided by the probability of the outcome occurring in an unexposed group. The RR is used to calculate the "population attributable fraction" (PAF) (Eq. 5), defined as "the proportion of disease or death that could be prevented if exposures were reduced to an alternative [. . .] scenario, while other conditions remain unchanged" (Prüss-Ustün et al., 2019). This PAF is combined with the total burden of disease (e.g., diarrhea) in deaths and/or DALYs to calculate the share of this burden which is attributable to the risk factors under study as per Eq. 6.

$$PAF = \frac{\sum_{k=1}^{n} p_k(RR_k - 1)}{\sum_{k=1}^{n} p_k(RR_k - 1) + 1} \tag{8.5}$$

Equation 5 : Prüss-Ustün et al. (2019)'s equation of the PAF (unitless), where p_k : proportion of the population exposed to level of exposure k; RR_k : relative risk for exposure level k (unitless); and n : number of exposure levels.

$$B_r = PAF_r \times B \tag{8.6}$$

Equation 6 : total burden of disease B in a population attributable to the risk factors r (in death or DALYs); PAF_r : potentially attributable fraction for risk factors r; B : total burden of disease B in a population (in death or DALYs).

CRA studies have been used to quantify the burden of diarrheal diseases attributable to inadequate water, sanitation, and hygiene (WASH) (Prüss-Ustün et al., 2019). In these CRAs, the burden of disease is calculated for different levels of exposure to inadequate WASH practices, which depend on the level of access to improved source of water ranging from "no improved water supply and no basic sanitation" to "ideal situation, no water-related diseases". In the following, the expression "exposure levels" refers to this terminology used in the CRAs and is not related to the XF conventionally used in LCIA (see Eq. 3).

This work proposes a revised EF model assessing the diarrheal burden of diseases due to domestic water deprivation accounting for different levels of exposure to health

risks in a population. Provided at the country level, this revised EF takes better account of countries characteristics in terms of the intensity of the diarrheal effects they undergo when deprived of water. These EFs are used to update the characterization model and factors of Debarre, Boulay et Margni (2022).

8.2.2 Methods

We here propose revised EF calculated at the country scale. EF calculation builds on the rationale that in a country, the deprivation of 1 m^3 of water does not affect the whole population in the same way ; the lower the daily domestic freshwater availability to a domestic user, the higher the potential health consequences of depriving them from 1 m^3 of freshwater.

We define three categories or "classes" of domestic water users : (1) users using less than 20 L per day per capita (Lpcd) (identified by the index [< 20]), (2) users using between 20 and 50 Lpdc [20–50], and (3) users using more than 50 Lpdc [> 50]. The international community recommends a minimum volume of 50 Lpdc for drinking, cooking, bathing, and sanitation, while studies suggest that 20 Lpdc is the minimum volume required to ensure basic hand and face washing, with no other hygiene practices (Howard et al., 2020). The combination of these three classes covers the entire population of a country. We calculate generic global "class-specific effect factors" $cEF_{i,global}$ for the three domestic water users classes : $cEF_{[<20],global}$, $cEF_{[20–50],global}$, $cEF_{[>50],global}$. The $cEF_{i,global}$ represent the default (or "global") effect on human health due to the deprivation of 1 m^3 of water for each class i of domestic users (see section "Class-specific effect factors" for calculation details).

Country-specific EF

Country-specific EFs are computed as the weighted sum of $cEF_{i,global}$ by the proportion i of the population in each class of domestic water users for country j ($p_{i,j}$) (cf Eq. 7). Values of $p_{i,j}$ are derived from 2020 data from the Joint Monitoring Program (WHO/UNICEF, s. d.) (see SI). The proportion of population in each class $p_{i,j}$ was

not available for 21 countries for which sub-regional mean value of EF has been selected as a proxy (see SI).

$$EF_j = \sum_i cEF_{i,global} \times p_{i,j} \tag{8.7}$$

Equation 7 : country-specific effect factor. Subscript j refers to the country being considered and subscript i refers to the different classes of domestic water users.

Class-specific effect factors cEF

Building on the assumption that potential impact on human health due to 1 m^3 domestic water deprivation per class of users is uniform across countries, we define a generic $cEF_{i,global}$ as the arithmetic mean value of country-specific $cEF_{i,j}$ for each class i (averaged over countries j). The key aspects of this hypothesis, including its rationale and implications, are discussed in the following sections. The following paragraphs describe the calculation of country-specific $cEF_{i,j}$.

To calculate country- and class-specific effect factors $cEF_{i,j}$, we combine the work of Fry et al. (2010) with an adaptation of the CRA method of Fewtrell et al. (2007). We acknowledge that these studies are longstanding but to our knowledge no recent study provides updated similar data. These studies are used with time-consistent data sources to calculate class- specific effect factors assumed to be time independent.

We use Fewtrell et al. (2007)'s levels of exposure to the burden of diarrheal diseases associated with drinking water, sanitation, and related diarrhea occurrence within a population. These exposure levels depend on households' level of access to drinking water and sanitation (Table 1).

In the following, exposure levels are identified with subscript "x". RR defined by Fewtrell et al. (2007) are assigned to each level of exposure, along with the corresponding percentage $p_{x,j}$ of the population provided by the same source (Table 2). Daily water use amounts are also associated to each level of exposure, based on estimates from Fry et al. (2010). For each level of exposure, the PAF is calculated by

Exposure Level	Description	Quantity of water consumed (Lpcd)
II	Improved drinking water and sanitation in countries where more than 98% of the population has access to those services	110
IV	Improved drinking water and improved sanitation (< 98%)	50
Vb	Improved drinking water but no improved sanitation	20
VI	No improved water supply and no basic sanitation	10

TABLEAU 8.1 Exposure levels to the burden of diarrheal diseases associated with drinking water, sanitation and related diarrhea occurrence, adapted from Fewtrell et al. (2007) and Fry et al. (2010). Seven levels of exposure are developed in Fewtrell et al. (2007) and Fry et al. (2010) ranging from I to Vb (see SI). Only exposure levels whose associated water consumption represents a threshold of the water classes (e.g., 20 Lpcd or 50 Lpcd for class [20–50]) have been listed here. See SI **??** and Table 3 for additional information on how water classes are linked to exposure levels

adapting the equation proposed by (Prüss-Ustün et al., 2019), as per Eq. 8. Specific to a level of exposure, Eq. 8 slightly differs from Eq. 5 where the numerator is a sum.

$$PAF_{x,j} = \frac{p_{x,j}(RR_x - 1)}{\sum_k p_{k,j}(RR_k - 1) + 1} \tag{8.8}$$

Equation 8 : population attributable fraction for level of exposure x in country j. $p_{x,j}$ *refers to the percentage of the population of country j at level of exposure x ; RRx : risk ratio for level of exposure x ; k : range of exposure levels, i.e., (II, IV, Vb, VI).*

The annual burden of diarrheal diseases per capita attributable to each level of exposure x in country j ($\mathrm{BoD}_{x,j}$) is calculated as per Eq. 9. Population in each level of exposure is estimated building on country population data and values of $\mathrm{p}_{x,j}$ from Fewtrell et al. (2007) (see Table 2).

$$BoD_{x,j}[\frac{DALY}{capita.yr}] = \frac{BoD_j \times PAF_{x,j}}{p_{x,j} \times population_j} \tag{8.9}$$

Equation 9 : burden of diarrheal diseases per capita attributable to level of exposure x in country j (in DALY/cap.yr) ; BoD_j : annual burden of diarrheal disease in country j (DALY/yr) ; PAFx,j : population attributable fraction of diarrheal diseases for level of exposure x in country j. $p_{x,j}$: proportion of the population of country j in level of exposure x ; $population_j$: total population in country j.

TABLEAU 8.2 Levels of exposure to burden of diarrheal diseases. Sums in the denominators of the PAF are made on subscripts II, IV, Vb, and VI (exposure levels)

Level of exposure	Description	Risk Ratio RR [unitless]	Water use [L/ capita.day]	Proportion p of the population	PAF - WASH diarrhoeal DALY [unitless]	Diarrhoeal burden of diseases (BoD) per capita [DALYs/cap]
Reference	Fewtrell et al. (2007)	Fewtrell et al. (2007)	Fry et al. (2010)	Fewtrell et al. (2007)	This work, derived from Fewtrell et al. (2007)	This work, derived from Fewtrell et al. (2007)
II	Improved drinking water and sanitation in countries where more than 98% of the population has access to those services	2.5	110	p_{II}	$\frac{p_{II}(RR_{II}-1)}{\sum_I^{VI} p_k(RR_k-1)+1}$	$PAF_{II} \times \frac{TOTAL_{DiarrhealDALYs}}{p_{II} \times population} = \frac{p_{II}(RR_{II}-1)}{\sum_I^{VI} p_k(RR_k-1)+1} \times \frac{TOTAL_{DiarrhealDALYs}}{p_{II} \times population}$
IV	Improved drinking water and improved sanitation (<98%)	6.9	50	p_{IV}	$\frac{p_{IV}(RR_{VI}-1)}{\sum_I^{VI} p_k(RR_k-1)+1}$	$PAF_{IV} \times \frac{TOTAL_{DiarrhealDALYs}}{p_{IV} \times population} = \frac{p_{IV}(RR_{IV}-1)}{\sum_I^{VI} p_k(RR_k-1)+1} \times \frac{TOTAL_{DiarrhealDALYs}}{p_{IV} \times population}$
Vb	Improved drinking water but no improved sanitation	8.7	20	p_{Vb}	$\frac{p_{Vb}(RR_{Vb}-1)}{\sum_I^{VI} p_k(RR_k-1)+1}$	$PAF_{Vb} \times \frac{TOTAL_{DiarrhealDALYs}}{p_{Vb} \times population} = \frac{p_{Vb}(RR_{Vb}-1)}{\sum_I^{VI} p_k(RR_k-1)+1} \times \frac{TOTAL_{DiarrhealDALYs}}{p_{Vb} \times population}$
VI	No improved water supply and no basic sanitation	11	10	p_{VI}	$\frac{p_{VI}(RR_{VI}-1)}{\sum_I^{VI} p_k(RR_k-1)+1}$	$PAF_{VI} \times \frac{TOTAL_{DiarrhealDALYs}}{p_{VI} \times population} = \frac{p_{VI}(RR_{IV}-1)}{\sum_I^{VI} p_k(RR_k-1)+1} \times \frac{TOTAL_{DiarrhealDALYs}}{p_{IV} \times population}$

$cEF_{i,j}$ are calculated per water use class i of the population and per country j by dividing the difference of annual burden of diarrheal diseases per capita ($BoD_{x,j}$) at levels of exposure x that correspond to the thresholds of the class, by the difference in annual water use amounts at these thresholds (see Eqs. 10, 11a, and 11b for calculations). Table 3 provides additional information on thresholds by class and the associated exposure levels.

The cEF for the class of population consuming between 20 and 50 Lpcd is calculated as per Eq. 10 :

$$cEF_{[20-50]}[\frac{DALY}{m^3}] = \frac{BoD_{Vb} - BoD_{IV}[\frac{DALY}{cap.yr}]}{365 \times (50-20)/1000[\frac{m^3}{cap.yr}]} \tag{8.10}$$

where BoD_{Vb} (respectively BoD_{IV}) represents the annual burden of diarrheal diseases per capital at the lower (higher) threshold of the class, i.e., at level of exposure Vb (IV), in which population benefit from 20 Lpcd (50 Lpcd). Similarly, $cEF_{[<20]}$ and $cEF_{[>50]}$ are calculated as per (based on Tables 1 and 2) :

$$cEF_{[<20]}[\frac{DALY}{m^3}] = \frac{BoD_{VI} - BoD_{Vb}[\frac{DALY}{cap.yr}]}{365 \times (20-10)/1000[\frac{m^3}{cap.yr}]} \tag{8.11}$$

$$cEF_{[>50]}[\frac{DALY}{m^3}] = \frac{BoD_{IV} - BoD_{II}[\frac{DALY}{cap.yr}]}{365 \times (110-50)/1000[\frac{m^3}{cap.yr}]} \tag{8.12}$$

The choice of using generic rather than country-scale cEF_i results from a lack of data. To assess the influence of this modeling choice on final results, a sensitivity analysis integrating country-j-specific $cEF_{i,j}$ within characterization factors is provided in the SI.

Integration within characterization factors

Country-specific EFs obtained from Eq. 7 are used to overwrite the generic EF used in the characterization model of Debarre, Boulay et Margni (2022) to compute CFs

Class of users	Water consumption range thresholds (in Lpcd)	Levels of exposure paired to calculate the cEF
[<20]	10 (lowest value) and 20	Level Vb [20 Lpcd] and VI [10 Lpcd]
[20–50]	20 and 50	Level Vb [20 Lpcd] and IV [50 Lpcd]
[>50]	50 and 110 (highest value)	Level IV [50 Lpcd] and II [110 Lpcd]

TABLEAU 8.3 Levels of exposure paired to calculate the cEF per class of domestic water users. cEFs per class depend on the annual burden of diarrheal diseases and the annual water consumption per capita for the high and low thresholds of water consumption in each class. Both thresholds are associated to a level of exposure defined in Table 1. For instance, the class [20–50] is bounded by the thresholds 20 and 50, respectively, associated with exposure levels Vb and IV. The lowest (respectively highest) threshold of class [< 20] (> 50) is assumed to be 10 (110) Lpcd since it is the lowest (highest) value of water consumption estimated by Fry et al. (2010).

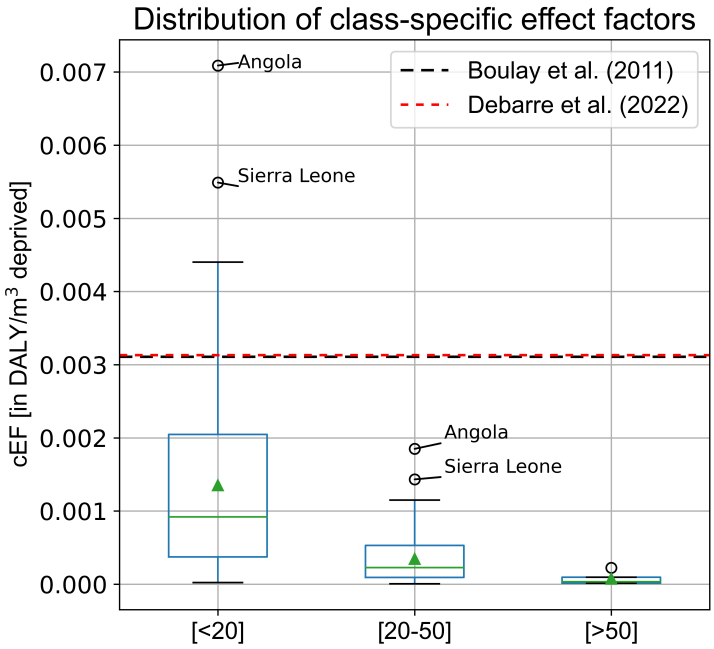
as per Eq. 1. Fate factors were computed as per Eq. 2 and XF were updated using 2022 values of GNI per capita (The World Bank, s. d.) in Eq. 3. CFs were computed combining maps of FF (defined at the sub-river basin level), XF (defined at the country level), and EF (defined at the country level). CFs are also computed at the country, regional, and global levels by aggregating CFs by water consumption using data from the WaterGAP v2.2d model (Müller Schmied et al., 2020) (see SI **??** for further details on the aggregation method).

8.2.3 Results

Class-specific effect factors : $\mathbf{cEF}_{i,j}$

Figure 8.1 shows the distributions of the class-specific effect factors $cEF_{i,j}$ among countries. Distributions are provided for classes of users benefiting from different daily quantities of domestic water. We provide 107 country-specific values of $cEF_{[<20],j}$, 110 $cEF_{[20–50],j}$, and only for 4 $cEF_{[>50],j}$ depending on data availability.

The red and black dotted lines represent the EF values calculated by Debarre, Boulay et Margni (2022) and Boulay et al. (2011) reaching 3.13e-3 and 3.11e-3 DALY



	<20	20-50	>50
mean	1.35e-03	3.44e-04	7.53e-05
count	107	110	4
std	1.33e-03	3.45e-04	1.00e-04
min	2.28e-05	5.95e-06	1.36e-05
max	7.09e-03	1.85e-03	2.23e-04

FIGURE 8.1 Distribution of the class-specific EF ($cEF_{i,j}$) per country, shown per class of users : " <20", "20–50", and " >50" classes of population consuming "less than 20 Lpcd," "between 20 and 50 Lpcd", and "above 50 Lpcd", respectively. Descriptive statistics are provided in the table on the right : Mean, standard ("std"), minimum ("min"), and maximum ("max") values are expressed in DALY/m^3 deprived. Green triangles indicate the arithmetic mean ($cEF_{i,global}$) and green lines indicate the median value.

per m^3 deprived, respectively. Mean class-specific effect factors, $cEF_{i,global}$, equal to 1.35e-3, 3.44e-4, and 7.53e-5 DALY/m^3 for the three classes of users, respectively. $cEF_{[<20],global}$ shows a 57% reduction compared to the EF of Debarre, Boulay et Margni (2022). Accordingly, $cEF_{[20–50],global}$ and $cEF_{[>50],global}$ show an 89% and 98% reduction. Angola, followed by Sierra Leone, shows the highest value of $cEF_{i,j}$ in the different classes of population due to high estimates of diarrheal DALYs per capita (1.0e-1 and 7.8e-2).

Country-specific EF

Figure 8.2 provides a visual representation of global distribution of country-specific EFs. EF values range from 7.5e-5 to 8.7e-4 DALY/m^3 deprived (M : 2.5e-4 ; SD : 1.8e-4), with highest values in Papua New Guinea (8.7e-4 DALY/m^3 deprived), Madagascar (7.3e-4), and Chad (7.2e-4). This represents a reduction of between 72.2 and 97.6% in DALYs per m^3 of water deprivation relative to the EF by Debarre, Boulay et Margni (2022). As a comparison, the EF calculated by Boulay et al. (2011) for the agricultural impact pathway reaches 6.53e-5 DALYs.

Lowest values are observed in countries where more than 99% of the population has access to safely managed water services (where EF = 7.5e-5 DALY/m^3 deprived). A map of country-specific EFs building on country-specific $cEF_{i,j}$ is provided in the SI.

Aggregation of EF at the regional and sub-regional scales

Figure 8.3 provides insights on the distribution of country-specific EFs, grouped by regions. Regional mean of EFs is highest in sub-Saharan Africa (M : 4.9e-4 DALY/m^3 deprived ; SD : 1.5e-4), followed by South Asia (M : 3.5e-4 ; SD : 9.7e-5), East Asia and Pacific (M : 2.5e-4 ; SD : 1.8e-4), Latin America and Caribbean (M : 2.2e-4 ; SD : 1.2e-4), Middle East and North Africa (M : 1.8e-4 ; SD : 1.3e-4), North America (M : 1.1e-4 ; SD : 6.3e-5), and Europe and Central Asia (M : 1.1e-4 ; SD : 5.9e-5). Since EFs result from the weighted sum of generic $cEF_{i,global}$ by the proportion of the population in each class of users, values of EF are driven by the level of access

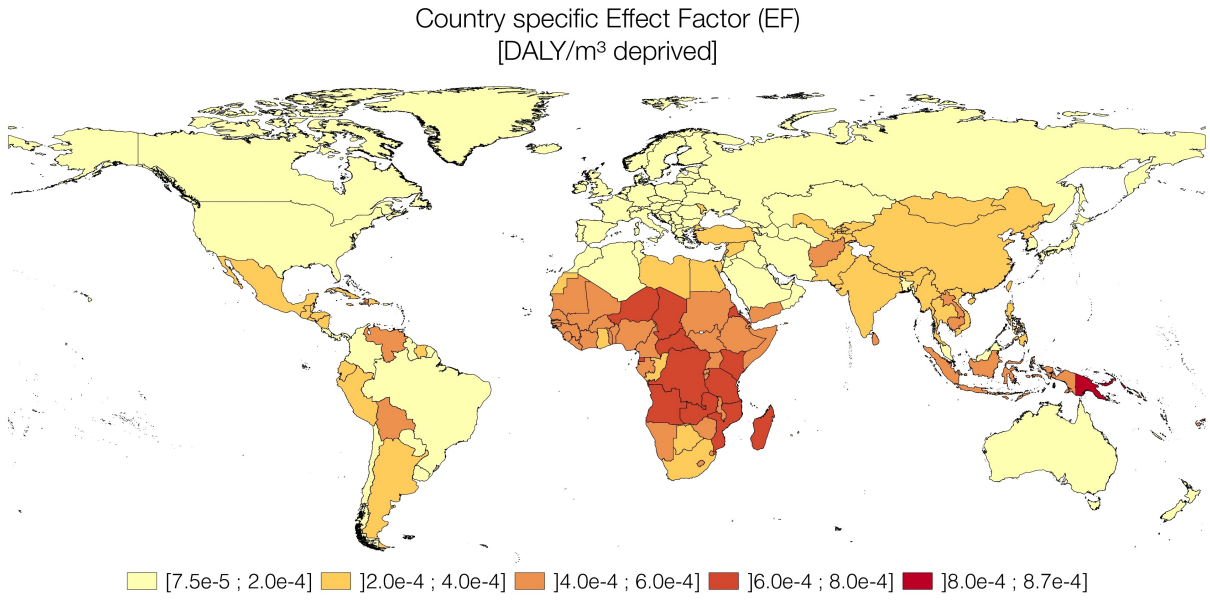
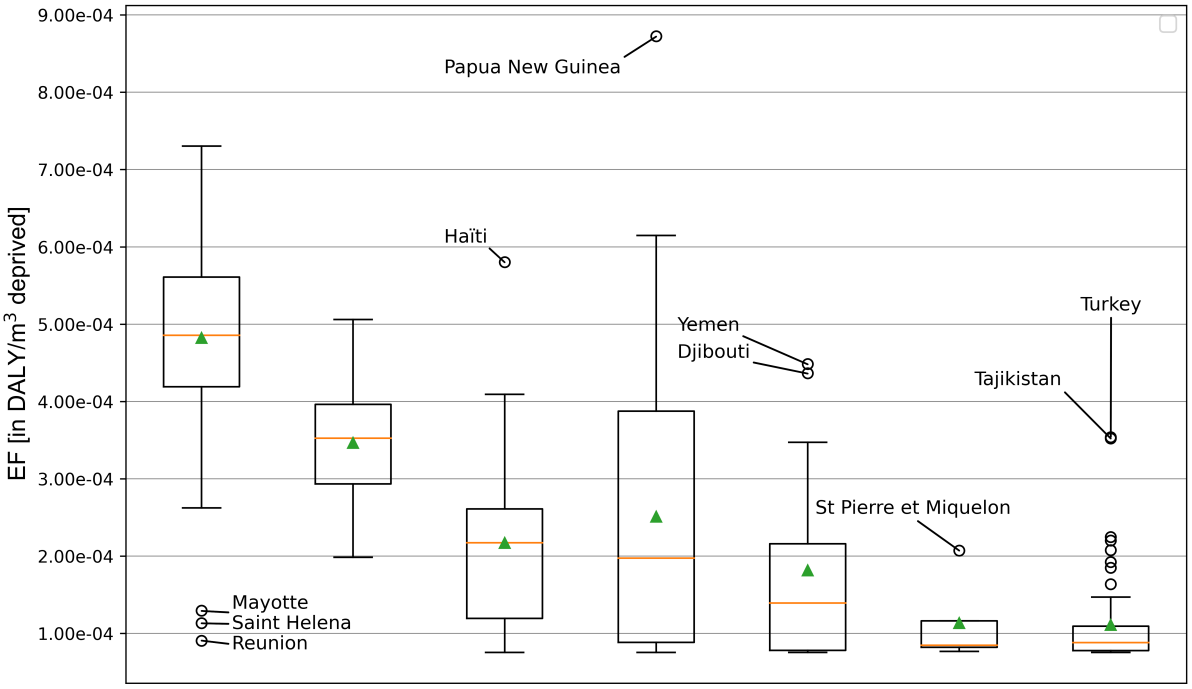


FIGURE 8.2 Country-specific effect factors EF_i

to water facilities in each country. Outliers (e.g., Papua New Guinea) stand out due to a lower level of access to water compared to other countries in the region.

Integration within characterization factors

Figure 8.4 shows the distribution of sub-river basin CFs. We compiled 11987 CFs ranging from 0 to 7.63e-4 DALY/ m^3 consumed (M : 1.9e-6; SD : 2.1e-5). The highest value is observed in a sub-river basin of Papua New Guinea (7.63e-4) followed by Ethiopia (5.48e-4) and Eritrea (4.85e-4). CFs are equal to zero in high-income countries (XF = 0) and/or in a sub-river basin where the fate factor is equal to zero. CFs per country and regions are provided in the SI **??** along with comparisons of the CFs with previous models (domestic and agricultural water deprivation).



	Sub-Saharan Africa	South Asia	Latin America and Caribbean	East Asia and Pacific	Middle East and North Africa	North America	Europe and Central Asia
mean	4.82e-04	3.46e-04	2.17e-04	2.51e-04	1.81e-04	1.13e-04	1.11e-04
count	51	8	46	41	21	4	60
std	1.46e-04	9.66e-05	1.18e-04	1.81e-04	1.25e-04	6.27e-05	5.89e-05
min	9.06e-05	1.98e-04	7.53e-05	7.53e-05	7.53e-05	7.65e-05	7.53e-05
max	7.30e-04	5.06e-04	5.80e-04	8.72e-04	4.48e-04	2.07e-04	3.54e-04

FIGURE 8.3 Boxplots of country-specific EFs, grouped by region. Green triangles indicate arithmetic mean. Orange line indicates the median value

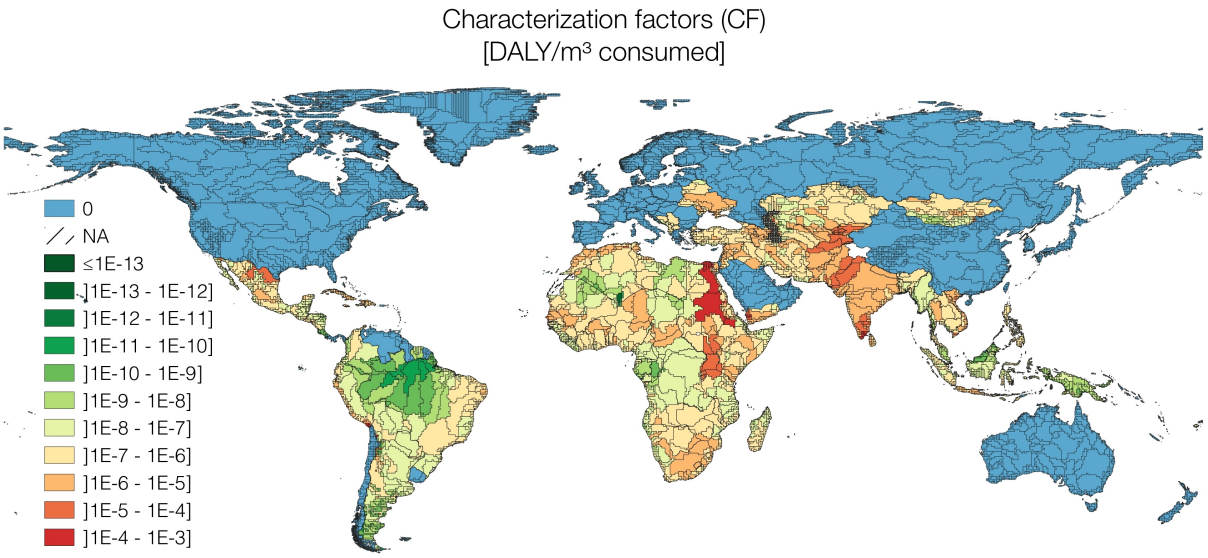


FIGURE 8.4 Sub-river basin characterization factors (in DALY/m^3 consumed)

Sanity check : potential total impact

The global potential impact induced by water consumption over the year 2019 is computed as the product of water consumption in each sub-river basin (Müller Schmied et al., 2020) and the corresponding CF calculated in this research work, summed for the globe. This global potential impact reaches 2.77e+7 DALYs for the year 2019.

According to Wolf et al. (2023), about 5.5e+7 DALYs are attributable to WASH diarrhea burden in 2019. This value encompasses the various components of unsafe WASH, which are not limited to water supply but also include other dimensions such as the access to sanitation facilities or to the adoption of adequate hygiene practices. Our results reach 50% of the burden of diarrhea disease calculated by Wolf et al. (2023). The order of magnitude of our result appears to be plausible, but we acknowledge a possible overestimation of the burden of diarrhea due to domestic water deficit. During floods and water-related disasters, water can be contaminated due to mixing of quality and contaminated waters. Pathogens (e.g., bacteria, viruses, parasites) can infiltrate water sources used for domestic purposes and cause outbreaks

of diarrhea diseases. Besides, extreme events can displace populations, increasing the exposure to diarrheal diseases.

8.2.4 Discussion

This paper focuses on the improvement of the effect factors within of Debarre, Boulay et Margni (2022) characterization model that assesses the potential impacts of water consumption on human health. We generate new characterization factors with the improved EF while updating exposure factor with more recent data and keeping the same fate factors of Debarre, Boulay et Margni (2022)'s characterization model. The following paragraphs provide a discussion of each of the factors in the characterization model.

Effect factor

Water-related diseases extend beyond diarrhea, encompassing conditions such as trachoma, schistosomiasis, and soil-transmitted infections (World Health Organization, 2022).

The EF model proposed in this paper focuses on diarrhea disease only and is inadequate to assess the burden due to other water-related issues. However, in the assessment of the contribution of four health outcomes (diarrhea, acute respiratory infections, undernutrition and soil-transmitted helminthiasis) to WASH-attributable burden of diseases, (Wolf et al., 2023) point to diarrheal diseases as the main source of WASH DALYs, with a contribution reaching 74%. This result supports the relevance of focusing on diarrheal diseases in a first step.

In addition, the burden of disease due to deficit of domestic water extends beyond direct water-related diseases (World Health Organization, 2022) ; domestic water deficit is a threat for productivity at the workplace, economic development, and, but not limited to, gender equality. The burden of water collection, mostly carried by women and girls, jeopardizes their physical health since water collection puts the water carriers at "risk of damage to the skeleton and muscular system" (World

Health Organization, 2022).

To enhance the accuracy of water consumption patterns within a population compared to previous models based on country-wide values, we have introduced specific classes of water users. While more accurate, this revised profile still provides a coarse picture of water consumption patterns. Estimates of consumption by specific classes are based on the work of Fry et al. (2010) which, to our knowledge, is the only attempt to estimate the daily amount of water use per capita associated to each level of exposure. The unique and possibly outdated nature of this study and the absence of quantified uncertainty in its values constitute a limitation for our model. Besides, the methodological choice of using generic rather than country-scale values for cEF_i is an important limitation in capturing regional specificities, and the extent to which the same class of water users from different countries may be more or less vulnerable to the burden of diarrheal disease. Other factors can influence the vulnerability of water users, such as the level of education, medical care, or standard of living that are implicitly reflected in the country-specific burden of diseases at each exposure level. If more data become available, using country-scale cEFs could help capture some of these specificities. However, it should be kept in mind that although the final cEF_i per class are generic (global), they are calculated as the average of the $cEF_{i,j}$ per country. Those values are calculated building on two paired exposure levels and the burden of diarrheal diseases per capita in each of these exposure levels, which is specific to each country.

Despite these limitations, the sanity check supports the plausibility of lower EF factors obtained in this paper compared to Debarre, Boulay et Margni (2022).

Fate and exposure factors

Fate Even though we did not update fate factors, it is worth to discuss some of the shortcomings of the factors proposed by Debarre, Boulay et Margni (2022). Population growth, urbanization, and economic growth are expected to accentuate water-related issues (scarcity in some places, floods in others, and increased temporal disparities) in the coming years (Howard et al., 2016). According to the UN, demand

for water might be 40% higher than supply by 2030 if water management practices are not improved (Vairavamoorthy et al., 2016). More than 5 billion people may suffer from water shortages in 2050 (WWAP (United Nations World Water Assessment Programme)/UN-Water, 2018). This underlines the sensitivity of the fate factor in our model, which needs to be updated regularly to provide accurate information on water scarcity.

Climate change is also likely to increase water scarcity in some regions by altering the hydrological cycle and increasing both the water demand and supply (Gleick & Cooley, 2021). In addition, sea level rise is likely to pose risks of salinization of groundwaters and rivers, jeopardizing the possibility of using those water sources for domestic purposes. These phenomena can also have an influence on the fate factor and must be accounted for into the model.

Exposure The new model recognizes disparities in exposure to diarrheal diseases within a country, due to socioeconomic inequalities. This raises questions about the legitimacy of applying a uniform adaptation capacity in a country (influencing the XF) segmented into several socioeconomic classes. The integration into XF of parameters other than the GNI per capita could also be explored, such as the Human Development Index (with or without inequality adjustments) or the GINI index. This exploration is beyond the scope of this paper.

Other factors underlying diarrheal disease, and the importance of an empirical demonstration linking the quantity of domestic water and diarrhea

Compared with other factors such as the level of hygiene knowledge and practices, the availability of piped water on premise, or water quality, the link between the quantity of domestic water and water-related diseases has received little attention in the scientific literature (Stelmach & Clasen, 2015). An empirical demonstration and a quantified evaluation linking water quantity and diarrheal diseases are lacking. However, the World Health Organization recognizes that “poor hygiene may in part be caused by insufficient domestic water” (Howard et al., 2020). Accordingly, a syste-

matic review on household water quantity and health suggests “a positive association between water quantity and health outcomes” which depends on how the water is used but showing that “increased water consumption [is] generally associated with reduced gastrointestinal infection and diarrheal disease” (Stelmach & Clasen, 2015). Our model is based on the rationale that a reduction in the amount of water available can exacerbate each of the potential vectors of diarrheal disease. A reduction in the quantity of water available can (1) lead to a reduction in hygiene practices, to concentrate on vital practices (drinking, cooking) ; (2) alter the quality of the running water service, leading to an intermittent service ; and (3) encourage households to turn to water sources that they would not otherwise have used due to poor quality.

There is no empirical evidence to support the link between water use, user deprivation, and diarrheal disease. While correlations between the amount of water used and diarrheal diseases can be observed (see SI), the demonstration of a causal link is lacking, which opens the door to potential future research.

8.2.5 Conclusion

In response to the current limitations in quantifying the potential health impacts associated with domestic water deprivation, a new calculation method is proposed. Revised EFs account for diarrhea impacts due to domestic water deprivation. Unlike previous methods, those revised EFs acknowledge inequalities in domestic water consumption within a country. Revised EFs are now calculated by country, going beyond the sole global resolution provided by previous models. They show a reduction of 72 to 98% compared to Debarre, Boulay et Margni (2022)’s EFs. A sanity check compares the overall impact of water consumption (and domestic water deprivation) using our revised effect factor against the total burden of diarrheal disease due to WASH for the year 2019. This sanity check estimates a contribution of 50% of our impact pathway to the total burden due to WASH, which confirms the plausibility of our revised CFs. Nevertheless, it does not rule out a potential overestimation due to the limitations of the model discussed in this paper. Future research is needed to provide further empirical evidence to support a causal link between water

deprivation and diarrheal diseases and assess uncertainties of the model results.

CHAPITRE 9 CONTRIBUTION ADDITIONNELLE : IMPACT WORLD+ V2.1

Les FC évaluant les impacts potentiels sur la santé humaine liés à la consommation d'eau présentés au chapitre 8 ont été intégrés à la version 2.1 de la méthodologie IMPACTWorld + (Agez et al., 2024 ; Bulle et al., 2019).

Ces facteurs évaluent les impacts liés à la privation d'eau pour les usagers domestiques (soumis à une potentielle restriction de leurs pratiques d'hygiène). Ce chemin d'impact est l'un des chemins liant consommation d'eau à un impact sur la santé humaine. Dans leur modèle, Boulay et al. (2011) proposent un second chemin d'impact, qui traduit les impacts de la privation des usagers agricoles, induisant une réduction d'eau pour l'irrigation et entraînant des enjeux de malnutrition.

Dans la pratique, la méthodologie IMPACTWorld + fournit un seul FC (régionalisé) évaluant les impacts sur la santé humaine liés à la consommation d'eau. Les FC développés pour les voies d'impacts domestique d'une part, et agricole de l'autre, sont donc additionnés dans un facteur commun.

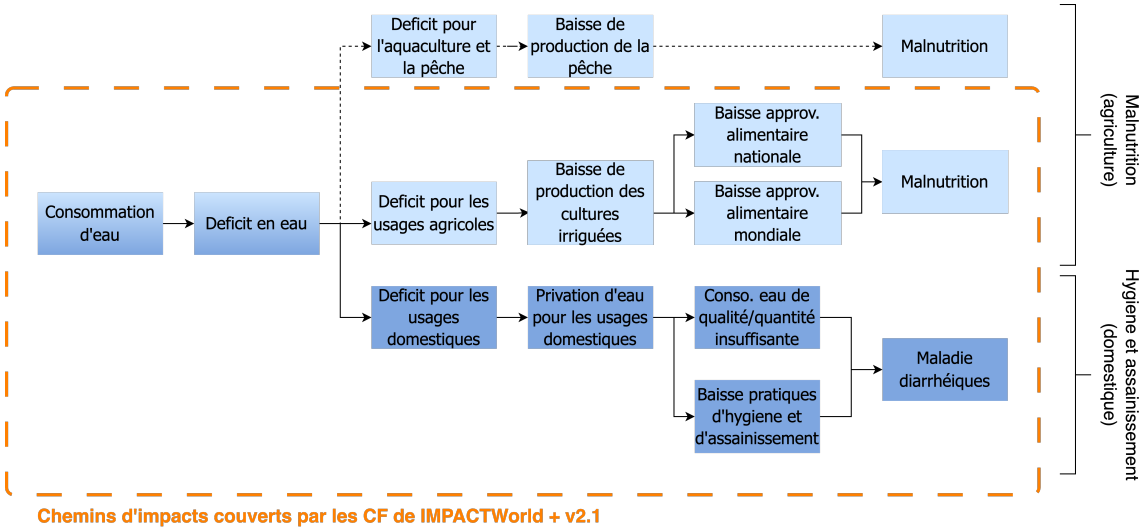


FIGURE 9.1 Chemins d'impacts sur la santé humaine liés à la consommation d'eau et leur couverture par les facteurs de caractérisation fournis dans IMPACTWorld + v2.1.

Un second ensemble de FC a donc été développé, pour représenter le chemin d'impact agricole. Celui-ci a été développé en cohérence avec le modèle domestique. Sa structure générale de calcul est similaire au FC domestique ; il est calculé comme le produit d'un facteur de sort, d'un facteur d'exposition et d'un facteur d'effet.

Facteurs de sort Le facteur de sort a été adapté pour les usagers agricoles :

$$FF_{agri}[\frac{m^3 depleted}{m^3 consumed}] = \frac{HWC_{agri,i}}{Availability_i - HWC_{total,i}} \tag{9.1}$$

avec $\text{HWC}_{agri,i}$: Consommation humaine pour les usages agricoles dans le bassin versant i, Availability : disponibilité d'eau dans le bassin versant i, et $\text{HWC}_{total,i}$: Consommation humaine totale pour tous types d'usages dans le bassin versant i.

Les facteurs de sort sont limités à 1 ; les résultats de l'équation 9.1 excédant cette valeur sont égaux à 1. Ce facteur de sort adapté pour le chemin d'impact agricole ne diffère du facteur de sort domestique que par son numérateur, qui représente la

consommation d'eau pour les usages agricoles (vs. la consommation d'eau pour les usages domestiques).

Facteurs d'exposition Le facteur d'exposition pour les usagers agricoles est similaire au facteur d'exposition pour les usagers domestiques. On considère que la capacité de ces usagers à s'adapter au manque d'eau est proportionnelle au Produit Intérieur Brut de leur pays.

Facteurs d'effet Le facteur d'effet spécifique aux usagers agricoles est tiré de Boulay et al. (2011), et égal à 6.53E-5 DALY/m^3.

CHAPITRE 10 DISCUSSION

Dans ce chapitre, nous proposons de prendre du recul pour analyser et discuter des différentes contributions de cette thèse, leurs liens et leurs limites. Des recommandations sont proposées en réponse à ces dernières, et des pistes pour de futurs travaux de recherche sont suggérées.

10.1 Atteinte de l'objectif de recherche

Cette thèse a été réalisée avec l'ambition de **consolider l'évaluation des impacts du cycle de vie en repensant son cadre méthodologique à travers le prisme des services écosystémiques pour supporter le développement de modèles caractérisant l'impact des activités humaines sur les services écosystémiques**.

L'article *Toward a harmonized integration of ecosystem services in LCIA : a critical review* (Annexe A) a permis de poser les bases du travail en réalisant une revue critique de la littérature sur l'intégration des SE en EICV. Il a identifié les limites principales à celle-ci, dont l'absence de cadre structurant permettant le développement cohérent de modèles de caractérisation des impacts sur les SE.

En réponse, l'article *Revamping the LCIA framework through the lens of ecosystem services* (chapitre 5) a proposé de **repenser le cadre méthodologique de l'évaluation des impacts du cycle de vie à travers le prisme des services écosystémiques**. Ce faisant, celui-ci a révélé le rôle de concepts liés aux SE (tels que les fonctions écosystémiques ou le potentiel de fourniture de services) dans la majorité des modèles de caractérisation. Il offre un outil exhaustif et cohérent pour le futur développement de modèles de caractérisation représentant les mécanismes d'impact sur les objets importants pour la société.

Ce cadre EICV révisé a permis de **supporter le développement de modèles caractérisant l'impact des activités humaines sur les services écosysté-**

miques, tel que le modèle présenté au chapitre 6 évaluant les impacts de l'élévation du niveau des mers et de l'accentuation des événements extrêmes sur la capacité des écosystèmes côtiers à fournir des services.

En parallèle, la modélisation des impacts sur la santé humaine induits par une diminution de la disponibilité d'eau potable pour les usagers domestiques a été **consolidée** de façon cohérente avec le cadre EICV révisé, à travers les articles présentés dans les chapitres 7 et 8.

Ainsi, cette thèse a permis d'atteindre son objectif en identifiant les **enjeux** liés à l'intégration des SE en EICV - aussi bien en termes de difficultés que de pertinence - en proposant un **cadre méthodologique** révélant la place des SE dans les modèles de caractérisation, et en mettant ce dernier en **application** à travers différentes chaînes de cause-à-effet.

10.2 Discussion générale : un nouveau cadre EICV

Le cadre EICV révisé présenté au chapitre 5 peut être considéré comme le "squelette" de ce projet de thèse. Cette section propose une réflexion sur celui-ci, et sur ses liens avec les modèles de caractérisation développés dans cette thèse. Le lecteur est également invité à se référer à la section "Discussion" du chapitre 5 pour des détails supplémentaires.

10.2.1 Un effort de clarification

Un cadre EICV offre une structure cohérente guidant le développement de modèles de caractérisation des impacts du cycle de vie, via l'identification de l'ensemble des chemins d'impact potentiels et des indicateurs d'impacts pertinents pour la prise de décision. Il s'agit d'un outil pour les développeurs de modèles de caractérisation (ou "modélisateurs EICV") visant la cohérence et la pertinence de leurs travaux.

Le cadre EICV révisé présenté au chapitre 5 s'inscrit dans cet objectif. Il permet de clarifier à la fois 1-les objectifs de modélisation des impacts, en menant une réflexion

sur les Aires de Protection, et 2- les enjeux de cohérence dont les développeurs de modèles de caractérisation doivent être conscients, en révélant l'existence de liens étroits entre ces derniers.

Un nombre limité de travaux a mené ce type de réflexion sur le cadre de EICV (Frischknecht et al., 2016 ; Jolliet et al., 2004 ; Verones et al., 2017). Cette thèse permet de les consolider.

10.2.2 Une nouvelle façon de concevoir l'EICV

Le cadre EICV révisé offre une nouvelle façon de concevoir l'EICV, sans rompre avec l'existant. Il ne suggère pas de réviser l'ensemble des modèles de caractérisation actuels, mais invite à les regarder sous un autre angle, révélant leurs potentiels liens et enchevêtrement avec les concepts liés aux SE.

Cette nouvelle approche offre ainsi un angle différent pour concevoir certains concepts de l'EICV tels que les facteurs d'exposition (XF) utilisés en EICV [1]. À travers le nouveau cadre EICV, on peut concevoir un facteur d'exposition aussi bien à travers la *demande* en SE qu'à travers la *capacité d'adaptation* à la perte de flux de SE. Sans *demande* en SE, il n'y a pas de *flux* de SE. Par conséquent, une perte de *potentiel de fourniture* de SE n'entraîne pas nécessairement d'impact s'il n'y a pas de *demande* en SE. De même, si face à une perte de flux de SE, un utilisateur a une *capacité d'adaptation* suffisante, certains impacts (e.g. sur la santé humaine) peuvent être évités. Ainsi, ces deux facteurs traduisent une forme d'*exposition*.

Ce cadre permet ainsi aux modélisateurs EICV de définir plus précisément les facteurs de leur modèle de caractérisation (tels que le facteur d'exposition) en les positionnant explicitement par rapport à sa structure.

1. Dans le cadre d'impacts liés à des émissions, les facteurs d'exposition représentent *le contact de substances avec une cible sensible telles que les animaux, les plantes, les écosystèmes ou les êtres humains (Rosenbaum et al., 2017)*. Dans le cadre d'impacts liés à l'extraction de ressources, les facteurs d'exposition représentent un *changement dans la quantité disponible, la qualité ou la fonctionnalité d'une ressource, et la potentielle concurrence entre utilisateurs (Rosenbaum et al., 2017)*.

10.2.3 Un cadre conceptuel

L'applicabilité et la pertinence du cadre EICV révisé ont été validées en y positionnant les modèles de caractérisation de la méthodologie IMPACTWorld+ au chapitre 5. Néanmoins, il est important de rappeler qu'il s'agit d'un cadre *conceptuel*. Il vise à proposer une structure flexible pour le développement de modèles et de facteurs de caractérisation cohérents, sans fournir de lignes directrices strictes. Sa mise en application peut se heurter à différentes difficultés et peut être discutée.

La question de la demande

Le cadre méthodologique révisé permet de mettre en évidence que la *demande en SE* est un prérequis pour faire le lien entre un *potentiel de fourniture de SE* et un *flux de SE*. Le cadre suggère ainsi que, lors du développement d'un modèle de caractérisation, le modélisateur EICV prenne en compte la demande en SE pour quantifier rigoureusement une perte de flux de SE.

Or, cette exigence n'est pas strictement opérationnalisée dans le modèle de caractérisation des impacts de l'élévation du niveau des mers sur les SE côtiers présenté au chapitre 6. Dans ce dernier, la perte de potentiel de fourniture de SE est directement évaluée en termes monétaires pour évaluer une "perte de services", sans considération explicite de la demande en SE. Cette simplification méthodologique est motivée par la difficulté technique de quantifier la demande (actuelle et future) en SE à l'échelle globale et pour différentes catégories de services. Elle met ainsi en lumière une **limite opérationnelle** du cadre EICV, notamment lorsqu'il est appliqué pour développer des modèles de caractérisation à des échelles spatiales très larges.

Néanmoins, en supposant une demande en SE équivalente à la totalité du potentiel de fourniture de SE, l'approche suivie dans le modèle de caractérisation du chapitre 6 peut être interprétée comme une approche conservatrice, cohérente avec la nature de l'ACV qui vise à évaluer des impacts *potentiels*. Dans ce contexte, la perte de potentiel de SE correspond à une potentielle perte de flux de SE.

La subjectivité dans le concept de SE

L'évaluation de la demande est limitée à la fois par la complexité de sa représentation spatiale et par le fait que la demande en SE varie selon l'utilisateur. En effet, un même service peut présenter une utilité ou une importance différente selon l'utilisateur et le contexte. Selon le type d'acteur, ses valeurs culturelles et spirituelles, ou encore le contexte socio-économique dans lequel s'inscrit la demande, deux utilisateurs peuvent s'intéresser à un même SE pour des usages distincts (par exemple, l'approvisionnement en eau pour des usages agricoles ou domestiques). Ils peuvent également lui attribuer une valeur différente en fonction de leur perception ou de leur capacité à payer, entre autres facteurs. Ce travail de thèse n'établit pas de lignes directrices pour intégrer ces aspects, mais encourage une réflexion sur ces questions dans de futures recherches.

Multiplicité des impacts liés à un changement de fonctions écosystémiques

Le changement de fonctions écosystémiques lié à une intervention environnementale (un flux élémentaire) peut induire une multitude d'impacts potentiels. En effet, un changement de fonctions écosystémiques (chemin 0 du cadre 5.3) peut générer des chemins d'impacts différents, pointant vers des AdP différentes (e.g. changement des fonctions de l'écosystème menant aux chemins 0-1-a représentant des coûts d'adaptation à une perte de SE, 0-3 représentant les impacts sur la qualité des écosystèmes, ou 0-4 représentant des impacts sur la santé humaine).

De la même façon, un même changement de fonctions écosystémiques peut induire plusieurs impacts empruntant le même chemin du cadre mais selon plusieurs chaînes de cause à effet différentes. Par exemple, un changement des terres peut induire simultanément une perte de SE d'approvisionnement, une perte de SE de filtration des sols et une perte de SE de contrôle de l'érosion.

Le modélisateur peut modéliser chacune de ces voies en s'appuyant sur le cadre EICV (et considérant des stratégies d'adaptation éventuellement différentes). Néanmoins, les SE sont liés entre eux via des relations complexes, incluant synergies et arbitrages

(Cord et al., 2017). Une modélisation indépendante de chaque mécanisme est ainsi susceptible de sous-estimer les conséquences globales. Des travaux complémentaires sont nécessaires pour réfléchir à la pertinence et à la façon de prendre ces interactions en considération.

10.2.4 La pertinence de l'interdisciplinarité pour le futur développement de modèles cohérents avec le cadre EICV

Utilisation de modèles de la communauté des SE

L'utilisation de modèles et d'outils développés par la communauté des SE, tels que inVEST (Tallis & Polasky, 2011), ARIES (Villa et al., 2009) ou MIMES (Boumans et al., 2015), est une piste à explorer pour prendre en compte les synergies et les arbitrages évoqués ci-dessus. Bien que cette thèse ne propose pas de stratégies concrètes pour l'utilisation de tels modèles par rapport au cadre EICV proposé, elle appelle à explorer ces pistes et à encourager l'interdisciplinarité pour développer des modèles de caractérisation cohérents avec le cadre.

Liens entre la biodiversité et les services écosystémiques

La boîte (ou indicateur d'impact) "changement de fonctions écosystémiques" du cadre EICV est liée à l'ensemble des AdP, traduisant une interaction forte entre les mécanismes d'impact menant aux AdP "Qualité des écosystèmes" et "Services rendus par les ressources et les écosystèmes'. Cette interaction, explorée notamment dans les travaux de Oginah et al. (2023) et T. Li et al. (2023), constitue un axe de recherche particulièrement pertinent à approfondir pour une meilleure prise en compte des impacts sur les SE en ACV.

10.2.5 Un cadre méthodologique représentatif d'un système de valeurs

Un cadre EICV permet de structurer le développement de modèles de caractérisation dans l'optique de fournir des indicateurs pertinents pour la prise de décision. La

sélection des AdP en tant qu'indicateurs pertinents pour la prise de décision est déterminante pour la construction d'un cadre ; elle conditionne à la fois la manière dont il est articulé et les informations qu'il permet de fournir aux décideurs. Or, cette sélection d'objets "importants pour la société et qui doivent être protégés" (Schaubroeck & Rugani, 2017) est subjective, et le résultat d'un choix traduisant une perspective, ou un système de valeurs.

Dans le cadre de ces travaux, le choix a été fait de s'appuyer sur la table de valeurs sociales de Verones et al. (2017) pour définir un ensemble d'AdP cohérent. Il s'agit d'un choix méthodologique fort, dont l'influence est centrale dans la construction du cadre. Pour utiliser le cadre EICV, il est important de reconnaître et d'embrasser ce choix méthodologique et la vision qui le soutient. Cela implique par exemple de reconnaître l'existence *distincte* de valeurs intrinsèque, instrumentale et relationnelle d'un même objet, et de concevoir les AdP à travers cette typologie.

Il est important que le modélisateur EICV partage la même définition des AdP que celle adoptée dans cette thèse pour développer des modèles de caractérisation cohérents, sans double comptage. Si, par exemple, un modélisateur EICV conçoit implicitement ou inconsciemment l'AdP "qualité des écosystèmes" de manière instrumentale - en considérant que l'on cherche à limiter les impacts sur cette AdP afin de préserver le fonctionnement des écosystèmes car il est utile pour la société -, il risque de développer un modèle incohérent avec le cadre EICV proposé, susceptible d'induire des double comptages avec des modèles évaluant des impacts instrumentaux (pointant vers l'AdP Services rendus par les ressources et les écosystèmes).

D'autres perspectives quant aux objets importants pour la société auraient pu être explorées, induisant une sélection d'AdP et un cadre EICV différents. Par exemple, Schaubroeck et Rugani (2017) adoptent une perspective purement instrumentale et proposent une AdP unique représentant le bien-être humain. Cette proposition rompt avec le cadre EICV ici proposé. De leur côté, Matuštík et Paulu (2025) suggèrent d'ajouter une AdP représentant le fonctionnement du système Terre aux côtés des AdP existantes. Cette proposition découle d'une adaptation de la table des valeurs de Verones et al. (2017) à laquelle un troisième type de systèmes est ajouté aux

côtés des systèmes humains et naturels : le système Terre. Cette proposition semble compatible avec le cadre EICV ici proposé. Des travaux additionnels pourraient permettre d'évaluer la pertinence et la cohérence de leur combinaison.

10.3 Discussions spécifiques sur les modèles de caractérisation développés

Les modèles de caractérisation des impacts développés dans cette thèse, portant sur deux chemins d'impact distincts, ont été conçus en cohérence avec le cadre EICV proposé. Toutefois, leur portée dépasse celle d'une simple application du cadre : ils fournissent des FC utiles et pertinents pour consolider l'EICV et éclairer la prise de décision.

10.3.1 Impacts de l'élévation du niveau des mers sur les SE côtiers

Le chapitre 6 propose le premier modèle de caractérisation étudiant la chaîne de cause à effet liant émission de gaz à effet de serre, élévation du niveau des mers et impacts sur les SE en ACV. Ce développement méthodologique contribue à 1- élargir la gamme des chemins d'impact des émissions de gaz à effet de serre considérés en EICV, et à 2- renforcer l'évaluation des impacts sur les SE, insuffisamment couverts dans les méthodologies existantes. Ainsi, ce travail améliore la couverture des impacts environnementaux pris en compte en ACV.

Malgré ces avancées, le modèle présente différentes limites, discutées ci-bas.

Inertie des systèmes climatiques et océaniques. Tout d'abord, le modèle développé ne permet pas de capturer l'inertie des systèmes climatiques et océaniques. Les FC sont calculés à travers le ratio entre les pertes de potentiel de fourniture de services et les émissions cumulées sur une même période (2015–2100), faisant l'hypothèse implicite que les pertes observées durant cette période résultent uniquement des émissions ayant lieu sur ce même intervalle. Or, cette hypothèse omet le rôle des émissions antérieures à 2015, tout comme elle ignore les potentiels effets différés des

émissions postérieures à 2100.

Une approche de modélisation différente pourrait être explorée pour lier les émissions de GES aux inondations côtières. Un modèle climatique pourrait être utilisé pour simuler l'évolution au cours d'une certaine période des inondations des zones côtières dans le cadre d'un scénario donné, dit de référence. Ce scénario pourrait par la suite être modifié par l'ajout d'une salve d'émissions de GES en début de période. La comparaison des étendues d'inondations entre le scénario de référence et le scénario modifié permettrait ainsi d'isoler la contribution de la salve d'émission aux inondations totales. Sur cette base, il serait possible de développer des FC propres à différents scénarios de référence (e.g., différents RCPs). Néanmoins, cette approche repose sur l'utilisation conjointe de modèles climatiques et océaniques et de modèles topographiques globaux à très haute résolution, nécessitant une importante puissance de calcul.

Capacité d'adaptation des écosystèmes. Ensuite, le modèle ne permet pas de considérer la capacité d'adaptation des écosystèmes côtiers face aux inondations provoquées par des événements extrêmes, ni de distinguer les conséquences des inondations permanentes de celles des inondations temporaires. En supposant que tout écosystème inondé, quelle que soit la durée, subit une perte totale de sa capacité à fournir des SE, le modèle tend à surestimer les impacts associés.

Afin de mieux intégrer la capacité des écosystèmes à s'adapter aux impacts des inondations, de futurs efforts de recherche pourraient développer des archétypes d'écosystèmes, caractérisés par des comportements ou des réponses similaires aux niveaux extrêmes de la mer. Leur définition reposerait sur une combinaison de facteurs à identifier, tels que la distance au trait de côte, le degré de salinité, le régime climatique, la topographie, ou encore le niveau de sédimentation.

10.3.2 Impacts sur la santé humaine induite par l'utilisation et la privation d'eau

Les chapitres 7 et 8 contribuent à renforcer la modélisation des impacts sur la santé humaine liés à l'utilisation et à la privation d'eau. Ces travaux permettent ainsi d'améliorer la robustesse des FC et des profils d'impacts de l'ACV.

Ces travaux de recherche prennent pour point de départ le modèle de caractérisation initialement proposé par Boulay et al. (2011), distinguant deux voies d'impact (voir 9) : les effets sur la santé humaine dus à la privation d'eau pour les usagers (1) agricoles et (2) domestiques. Les ordres de grandeur des FC relatifs aux usages domestiques étaient suspectés d'être surestimés. Dans ce contexte, un des objectifs de cette thèse a été d'évaluer la robustesse du modèle domestique et de ses différents facteurs, et de proposer des pistes d'amélioration. Les trois composantes des facteurs de caractérisation (c-à-d le sort, l'exposition et l'effet) ont ainsi été analysées et consolidées.

Un contrôle de cohérence ("sanity check") de l'ordre de grandeur des FCs obtenus dans cette thèse a été réalisé au chapitre 8. Pour ce faire, la consommation en eau par sous-bassin versant, pour l'année 2019, a été multipliée par le FC associé au même sous-bassin versant [2]. Cette multiplication a permis d'obtenir une valeur d'impact potentiel global pour cette chaîne de cause à effet, atteignant 2.77e+7 DALY. Cette valeur a par la suite été comparée à l'estimation de la charge de morbidité associée aux maladies diarrhéiques liées à l'eau, à l'hygiène et à l'assainissement (WASH) en 2019, égale à 5.5e+7 DALY (Wolf et al., 2023). Ainsi, les impacts potentiels évalués à travers les FCs développés dans cette thèse atteignent 50% de la charge de morbidité évaluée par l'Organisation Mondiale de la Santé, ce qui valide la plausibilité des résultats et justifie la révision des facteurs de Boulay et al. (2011).

Malgré ces efforts de consolidation, des pistes d'amélioration de ce modèle de caractérisation peuvent encore être suggérées.

2. L'évaluation a aussi été faite en multipliant la consommation en eau mondiale, pour l'année 2019, par le FC mondial, et a mené à la même conclusion.

Le chapitre 8 propose une approche révisée pour calculer le facteur d'effet du modèle et tenir compte des disparités de consommation d'eau au sein d'une population. Il permet ainsi de mieux capturer des réalités différentes au sein et entre pays. Néanmoins, il ne couvre que les maladies diarrhéiques et ne permet pas de quantifier les impacts induits par d'autres types de maladies. Bien que les maladies diarrhéiques représentent le plus gros contributeur à la charge de morbidité liée à l'eau (Wolf et al., 2023), une prise en compte des autres maladies permettrait de renforcer davantage le facteur d'effet. Par ailleurs, le calcul de ce nouveau facteur d'effet s'appuie largement sur les estimations de consommation en eau par "classes d'utilisateurs" proposées par Fry et al. (2010) et sur les facteurs de risques (RR) tirés de Fewtrell et al. (2007). Afin de consolider le modèle, il serait pertinent d'évaluer la robustesse de ces valeurs.

Le facteur d'exposition est calculé à partir du produit national brut par habitant, utilisé comme proxy pour évaluer la capacité d'un pays à s'adapter à un manque d'eau. Cette valeur, définie à l'échelle nationale, ne permet pas de prendre en compte des capacités d'adaptation variables au sein d'un même pays. L'utilisation d'un indice capturant ce type de disparités, tel que l'indice de développement humain ajusté aux inégalités, ou l'indice de GINI pourrait être explorée.

Le facteur de sort révisé intègre à la fois des informations sur la rareté physique de l'eau et sur le niveau d'accès à cette ressource. Étant calculé comme le ratio entre la consommation d'eau domestique et la disponibilité en eau, ce facteur peut être influencé tant par une augmentation des besoins en eau (due, par exemple, à l'évolution des modes de consommation) que par une diminution de la disponibilité des ressources hydriques. Dans ce contexte, les effets du changement climatique sur la disponibilité en eau peuvent avoir un impact significatif sur les facteurs de sort. Une actualisation régulière de ces derniers apparaît dès lors pertinente afin de maintenir la robustesse et la représentativité des modèles de caractérisation.

Enfin, certaines critiques soulignent l'absence de démonstration empirique établissant un lien direct entre une quantité insuffisante d'eau et l'incidence des maladies diarrhéiques. Bien que des études suggèrent qu'une augmentation de la consommation d'eau au sein des foyers est généralement associée à une diminution des infections

gastro-intestinales et des maladies diarrhéiques (Stelmach & Clasen, 2015), la manière dont cette eau est utilisée et le niveau de pratiques d'hygiène pratiqué au sein d'un foyer ont une forte influence sur l'occurrence de maladies diarrhéiques (Howard et al., 2020).

Le modèle proposé repose sur l'hypothèse qu'une réduction de la quantité d'eau disponible peut exacerber plusieurs facteurs de risque associés aux maladies diarrhéiques. En particulier, une telle réduction peut : (1) entraîner une baisse des pratiques d'hygiène au profit des pratiques vitales telles que la consommation ou la préparation des aliments ; (2) dégrader la qualité du service d'eau potable, en provoquant notamment un approvisionnement intermittent ; et (3) inciter les ménages à recourir à des sources d'eau de moindre qualité qu'ils auraient autrement évitées.

À ce jour, aucune preuve empirique directe ne permet d'établir un lien causal entre la privation d'eau et l'occurrence de maladies diarrhéiques. Des corrélations entre consommation d'eau et incidence de ces maladies ont été observées (voir annexe **??**), mais l'établissement d'une relation causale reste un défi méthodologique majeur, ouvrant ainsi la voie à de futures recherches dans ce domaine.

10.4 Contributions spécifiques et public visé

En outre de leur contribution à l'atteinte de l'objectif principal de cette thèse, chaque article développé dans ce projet offre une contribution indépendante à la communauté ACV.

• L'article *Toward a harmonized integration of ecosystem services in LCIA : a critical review* (A) propose une synthèse critique pertinente pour les **modélisateurs EICV** qui souhaitent mieux comprendre la manière dont les différentes propositions pour intégrer les SE en ACV et en EICV s'articulent, et qui désirent lever le voile sur les incohérences et le manque de clarté qui freinent cette intégration.

• L'article *Revamping the LCIA framework through the lens of ecosystem services* (chapitre 5) est un outil sur lequel les **modélisateurs EICV** peuvent s'appuyer pour développer des modèles de caractérisation. En révélant le rôle des SE dans les

chaînes de cause à effet, ce cadre souligne également la pertinence de combiner les connaissances des communautés EICV et des SE pour construire des modèles de caractérisation.

• L'article *Characterizing sea-level rise potential impacts on coastal ecosystem services in Life Cycle Impact Assessment* (chapitre 6) offre un ensemble de facteurs de caractérisation pouvant être utilisés par les **praticiens ACV** [3] afin de fournir de l'information sur les impacts potentiels des GES sur les SE côtiers dans les profils d'impact d'ACV.

• De même, les articles *Freshwater consumption and domestic water deprivation in LCIA : revisiting the characterization of human health impacts* (chapitre 7) et *Potential impacts of freshwater consumption on human health : considering freshwater use inequalities to assess domestic water deprivation* (chapitre 8) offrent aux **praticiens ACV** un ensemble de facteurs de caractérisation régionalisés permettant une meilleure évaluation des impacts liés à la consommation d'eau des systèmes étudiés.

3. Le terme "praticien ACV" est utilisé pour désigner les personnes réalisant des ACV, suivant l'ensemble des étapes énoncées par la norme ISO14044 (définition du champs de l'étude, inventaire, évaluation des impacts, interprétation).

CHAPITRE 11 CONCLUSION ET RECOMMANDATIONS

L'ACV est un outil central de l'évaluation environnementale. Avec la progression des connaissances et des besoins d'indicateurs pour la prise de décision, le cadre conceptuel de l'EICV, qui structure le développement de modèles de caractérisation, évolue. La volonté de fournir de l'information sur les impacts des activités humaines sur les SE progresse. Pour intégrer ceux-ci dans les profils d'impact fournis par l'ACV, il faut développer des modèles de caractérisation spécifiques, mais cohérents avec les modèles existants.

C'est dans ce contexte que s'inscrit cette thèse, dont l'objectif principal est de consolider l'évaluation des impacts du cycle de vie en repensant son cadre méthodologique à travers le prisme des services écosystémiques pour soutenir le développement de modèles caractérisant l'impact des activités humaines sur les services écosystémiques.

Plusieurs objectifs spécifiques ont été atteints pour répondre à cet objectif principal.

— **L'intégration des services écosystémiques en EICV a été examinée**, à travers une revue critique de la littérature. Celle-ci a permis de poser les bases du travail de cette thèse, en identifiant la pertinence, les écoles de pensée, et les difficultés liées à la prise en compte des services écosystémiques en EICV. Elle a permis de développer un socle de connaissances fondamentales sur les concepts liés aux services écosystémiques, permettant d'adopter une terminologie adaptée et de maîtriser les concepts clefs.

— **Le rôle des services écosystémiques dans les modèles de caractérisation existants a été révélé**, en mettant en évidence des liens étroits entre le modèle en cascade et les modèles de caractérisation des impacts sur la santé humaine et sur la qualité des écosystèmes. Ceci a inspiré une révision du cadre EICV, sur la base du modèle en cascade.

— **Un nouveau cadre EICV a été développé et mis en application**. Un nouveau cadre EICV a été construit sur la base du modèle en cascade, et

les liens de ce dernier avec des entités importantes à protéger -qui ont été clarifiées- ont été identifiés. Ce cadre a été illustré et mis en application en développant un modèle de caractérisation évaluant les impacts de l'élévation du niveau des mers sur les services écosystémiques côtiers.

— **L'évaluation des impacts du cycle de vie a été renforcée** à travers l'amélioration du modèle de caractérisation des impacts sur la santé humaine liés à la consommation et la privation d'eau domestique, dont la chaîne de cause à effet à été positionnée par rapport au cadre. Ceci a permis de consolider la modélisation de ce chemin d'impact en EICV.

Le cadre EICV proposé est un outil sur la base duquel nous invitons les modélisateurs EICV à s'appuyer pour développer des modèles de caractérisation cohérents. Il s'agit d'un outil conceptuel, et des efforts de recherche complémentaires sont nécessaires pour fournir des lignes directrices plus spécifiques aux différentes chaînes de cause à effet. L'utilisation d'outils de la communauté des services écosystémiques et l'étude des liens entre biodiversité fonctionnelle et services écosystémiques représentent des avenues de recherche pertinentes pour le développement de modèles de caractérisation s'inscrivant dans ce cadre EICV.

Bien que les différentes contributions de cette thèse répondent collectivement à son objectif principal, elles peuvent également intéresser des communautés distinctes, telles que les modélisateurs EICV et les praticiens ACV. Chaque modèle de caractérisation développé dans cette thèse présente certaines limites. La modélisation des impacts de l'élévation du niveau des mers sur les services écosystémiques côtiers pourrait être consolidée par une meilleure prise en compte de l'inertie des systèmes climatiques et océaniques, ainsi que de la capacité des écosystèmes à s'adapter à l'élévation du niveau des mers. La modélisation des impacts sur la santé humaine induits par la consommation et la privation d'eau domestique, quant à elle, pourrait être consolidée via la démonstration empirique de la chaine de cause-à-effet étudiée.

En prenant du recul pour analyser l'ensemble des contributions de cette thèse, on peut en dégager un objectif plus général : **la consolidation de l'EICV**.

Bien que le prisme des services écosystémiques ait servi de point d'ancrage méthodologique, les apports de cette recherche dépassent ce seul domaine. Le cadre et les facteurs de caractérisation proposés visent à renforcer la robustesse, la cohérence et l'exhaustivité des profils d'impact produits par l'ACV. À ce titre, cette thèse s'adresse à l'ensemble de la communauté EICV et ACV, en apportant des outils conceptuels et opérationnels pour des résultats d'ACV plus robustes, cohérents, et exhaustifs.

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ANNEXE A – ARTICLE 4 : TOWARD A HARMONIZED INTEGRATION OF ECOSYSTEM SERVICES IN LCIA : A CRITICAL REVIEW

Toward a harmonized integration of ecosystem services in LCIA : a critical review

Laura Debarre, Titouan Greffe, Camille Chabas, Catherine Lalongé, Jérôme Lavoie, Laure Patouillard, Benedetto Rugani, Jana Schluenss, Rutger de Wit, Cécile Bulle and Manuele Margni

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Abstract

Purpose. The integration of ecosystem services in life cycle assessment has gained increasing attention. However, its operationalization remains limited and inconsistent, particularly within the impact assessment phase. This study hypothesizes that the lack of harmonized and structuring life cycle impact assessment modeling guidelines hinders the development of consistent models for the assessment of ecosystem services. This research work aims to lay the foundation for a comprehensive and structured life cycle impact assessment framework that enables a coherent integration of ecosystem services.

Method. This study critically examines the integration of ecosystem services into life cycle impact assessment through two complementary literature reviews : (i) a foundational review of ecosystem services concepts, informing (ii) a critical analysis of existing approaches for incorporating ecosystem services into life cycle impact assessment.

Results and discussion. The current integration of ecosystem services in life cycle impact assessment lacks clarity, coherence, and structure. Various proposals suggest incorporating ecosystem services at the damage level either as a standalone Area of Protection or within an existing one, yet no consensus has emerged. This fragmentation raises concerns about consistency among proposals, with existing Areas of Protection and with ecosystem services knowledge. Different approaches have been suggested for the assessment of ecosystem services in life cycle assessment, but they follow different schools of thought with no harmonized guidelines to unify the efforts. The links between ecosystem services, natural resources, and other protected areas remain underexplored. Finally, damage-oriented ecosystem services metrics are often vague, with insufficient discussion or justification for their selection.

Conclusion and recommendations. Progress in integrating ecosystem services into life cycle impact remains hampered by conceptual, methodological and terminological inconsistencies. This critical review underlines the need for a comprehensive, harmonized, and globally accepted framework to facilitate the integration of potential impacts on ecosystems' capacity to provide services at the damage level in life cycle impact assessment.

Introduction

Nature is essential for human well-being and activities, which depend on the stock of natural assets collectively known as "natural capital" (IPBES, 2019). Recognition of the significance of nature for humans led to the development of the concept of "ecosystem services" (ES), defined by Costanza et al. (1997) as the "benefits populations derive, directly or indirectly, from ecosystem functions" and highlighting the

link between human well-being and ecosystem functioning within social-ecological systems (Costanza et al., 1997 ; McGinnis & Ostrom, 2014). Natural Capital Accounting (NCA) methods, such as the Natural Capital Protocol (Natural Capital Coalition, 2016), highlight not only the dependency of human activities on functioning ecosystems but also the influence of our activities on the ability of ecosystems to provide benefits to people.

Due to increasing environmental pressures, the ability of ecosystems to provide services is at risk, threatening human well-being (IPBES, 2019 ; Millenium Ecosystem Assessment, 2005). To prevent further decline, integrating ES into policy and decision making is crucial to support management practices that ensure the sustainability of these services (European Commission, 2020). Numerous assessment tools and models are available to assess ES, chosen based on the context and objectives of decision making (K. G. Turner et al., 2015).

Life cycle assessment (LCA) is a prominent tool for supporting sustainable decision making and policy development (Hellweg et al., 2023). This methodology assesses the potential environmental impacts of human activities on subjects that are important to society and need to be protected, referred to as Areas of Protection (AoP) (Schaubroeck & Rugani, 2017 ; Verones et al., 2017). Current common AoP include human health, ecosystem quality, and natural resources (Huijbregts et al., 2017).

Although LCA scholars have acknowledged the societal importance of ES (Bulle et al., 2019 ; Verones et al., 2017), their operational integration in LCA remains limited (Vanderwilde & Newell, 2021). However, such inclusion in the LCA methodology is essential to comprehensively capture "the hidden costs or benefits to society of [human activities] degrading or supporting functional ecosystems" (Othoniel et al., 2016).

Different approaches have been proposed for the integration of ES in LCA, which attempt to overcome both conceptual and practical challenges (Rugani et al., 2023 ; Vanderwilde & Newell, 2021). However, a critical gap remains in the Life Cycle Impact Assessment (LCIA) phase, where ES concepts have not yet been fully in-

corporated into existing impact characterization models and indicators (Alejandre, Bodegom & Guinée, 2019 ; Othoniel et al., 2016).

This study hypothesizes that damage-oriented ES assessment in LCIA is hindered by the absence of harmonized, transparent, and consistent modeling guidelines, which are essential for developing consistent impact characterization models. We further hypothesize that this lack of methodological clarity has contributed to confusion about the integration of ES among LCIA method developers and LCA practitioners.

To investigate the validity of these research hypotheses, this critical review seeks two specific objectives. The first objective is to establish a foundational understanding of ES to provide a knowledge base for evaluating existing LCIA ES models and guiding the development of future ones. This includes clarifying key concepts such as instrumental value (Arias-Arévalo, Martín-López & Gómez-Baggethun, 2017 ; Arias-Arévalo et al., 2018 ; Schroeder, 2024), cascading models (Potschin-Young et al., 2018) and valuation methods (IPBES, 2022). The second specific objective aims at critically assessing existing proposals for integrating ES into LCIA. This involves reviewing and discussing proposed methods in relation to ES knowledge, identifying conceptual and terminological misalignments, methodological gaps, and areas for improvement.

The findings of this article aim to harmonize the development of LCIA models, fostering the integration of ES concepts into LCA and decision making processes. This critical review serves as a foundation for the proposal of a revamped and comprehensive LCIA framework, as developed in (Debarre et al., s. d.-a).

Methodology

This critical analysis is based on two distinct literature reviews to support a comprehensive understanding of ES and their integration into LCIA. The first review focused on foundational concepts within the ES field, while the second review critically assessed existing approaches to incorporating ES within LCIA. These reviews were conducted using distinct methods, tailored to meet the specific objectives of

each.

The first literature review examined seminal and widely recognized publications in the field of ES. Given that this review sought to capture fundamental concepts rather than exhaustively catalog all literature on ES, the review focused on key sources widely cited within the ES community. The foundational texts included The Millennium Ecosystem Assessment (MEA) (Millenium Ecosystem Assessment, 2005), influential works by Daily (1997), R. S. De Groot, Wilson et Boumans (2002), Potschin-Young et al. (2018) and Costanza et al. (1997, 2017), key reports from the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES, 2019), and other significant frameworks and classification systems, such as the Common International Classification of Ecosystem Services (CICES) (Haines-young, Paddocks & Lane, 2023). To expand this initial set of sources, relevant studies were identified through forward and backward citation tracking, enabling the identification of influential works frequently referenced in the ES field. This approach allowed for an in-depth understanding of ES concepts, classifications, and valuation frameworks, forming a theoretical basis for evaluating the integration of ES into LCIA.

The second literature review critically examined the integration of ES in LCIA, with a focus on characterization models and methods. The search was performed in the Compendex and Google Scholar databases. The following keywords were used in various combinations to capture the relevant literature : 'ecosystem services', 'ecosystem functions', 'ecological functions', 'life cycle impact assessment', 'LCIA', 'life cycle assessment', 'LCA', 'characterization', 'impact', and their variations. The initial search results were selected based on the relevance of the topic, specifically focusing on studies that discussed methods, frameworks, or models for incorporating ES into LCIA.

The insights gained from the first literature review informed a critical evaluation of the findings of the second review. Building on foundational knowledge from the ES community, key concepts such as ES classifications, terminology, valuation methods, and their underlying assumptions were used as reference points to assess the methodological rigor and coherence of ES integration approaches in LCIA litera-

ture. This critical synthesis allowed for the identification of areas where ES concepts might be misinterpreted or underdeveloped within LCIA, providing a foundation for recommendations on improving ES integration.

Section A provides the review of the foundational concepts of ES, while Section A examines the integration of ES within LCA, and section A presents the results of the critical analysis of existing proposals to incorporate ES into LCA.

Fundamental understanding of ecosystem services

The CICES ecosystem services classification

Bringing together more than 1300 experts, the Millenium Ecosystem Assessment (2005) has catalysed the interest of scientists and decision-makers in ES and proposed the first ES classification. Following this initiative, other classifications have been proposed (for example, R. De Groot et al. (2010) et B. Fisher, Turner et Morling (2009)). The most common classifications include The Economics of Ecosystems and Biodiversity (TEEB) framework (TEEB, 2012), and the Common International Classification of Ecosystem Services (CICES) (Haines-Young & Potschin, 2018) which was developed in 2009 by the European Environment Agency. Based on the MEA and TEEB typologies and regularly updated, CICES has become a reference for the ES community (Finisdore et al., 2020 ; La Notte et al., 2017 ; Maes et al., 2016).

The CICES classification categorizes ES into 3 main sections : 1-provisioning ES (e.g. provisioning of water, timber, minerals, biomass, etc.), 2-regulation and maintenance ES (e.g. mediation of waste, erosion control, or hazard mitigation) and 3-cultural ES (e.g. recreational activities or spiritual interaction with natural system). These 3 ES categories rely on ecological processes, such as nutrient cycling, soil formation, or primary production, which were considered as supporting/intermediate services in the MEA and are currently defined as the "underpinning structures and processes that ultimately give rise to ecosystem services" in CICES.

Initially, "CICES has focused on defining [...] ecosystem services that depend on living systems" (Haines-young, Paddocks & Lane, 2023). This exclusive biocentric focus has been criticized by some authors as it ignores the benefits offered by geodiversity ("diversity of geological structures and processes, including rocks and minerals ; geomorphology, including land forms and topography ; sediments and soils, including formation processes ; and hydrology, including marine, surface, and subsurface waters" (Fox et al., 2020)), and overlook synergies among biotic and abiotic domains (Baveye et al., 2018 ; Fox et al., 2020 ; Gray, 2012 ; Gray, Gordon & Brown, 2013 ; Van Ree & Beukering, 2016). These "geosystem services" can include the provision of rare-earth metal, water or the regulation of thermal flows, and are often referred to as "abiotic" ES (Fox et al., 2020 ; Gray, 2012). The authors underscore that what is often referred to as "biotic" ES is largely dependent on biophysical processes (i.e., based on a combination of biotic and abiotic processes) and not only biological processes (Fox et al., 2020 ; Frisk et al., 2022 ; Gray, 2012). In response to these criticisms, the latest version of CICES (v5.2) combines the "biophysical ecosystem services" and "geophysical services" sections within the same ES classification. Hence, each section of the v5.2 classification (provisioning, regulation & maintenance and cultural) includes bio- and geophysical services.

The three CICES categories of ES, segmented into divisions, groups, and classes, intend to represent the final services offered by nature used by the social-economic system. The CICES classification, even if more detailed than most of the other ES classifications, remains limited to relatively coarse levels of detail in terms of service provided. For example, the provisioning ES of mineral substances in CICES is limited to the provisioning of substances 1-for nutritional purposes, 2-for material purposes, and 3- as energy source (codes 4.2.1.1, 4.2.1.2, 4.2.1.3 in CICES v5.2). Within each of these categories, flow of ES can be used for more specific final purposes (e.g., minerals ingested by humans like salt, or minerals used as crop fertilizers), which are not detailed for practical reasons. For specific, context-dependent assessments, practitioners may require more refined classifications such as Whiting, Carmona et Carrasco (2022), which detail the services that can be provided through the provision

of natural resources.

Ecosystem services frameworks

Like most ES classifications, CICES is based on the "cascade model" proposed by Haines-Young et Potschin (2010). The latter illustrates the linkages between an ecosystem's structure and processes which, through ecosystem functions, provide the support for human well-being (Burkhard et al., 2014 ; R. De Groot et al., 2010 ; Haines-Young & Potschin, 2010 ; Kandziora, Burkhard & Müller, 2013 ; La Notte et al., 2017) (Figure A.1). The structure of an ecosystem depends on its biotic and abiotic components as well as its physical features (e.g., climate or nutrient distribution) while ecosystem processes are defined as the physical, chemical and biological interactions linking organisms and their environment (e.g., food web dynamics or nutrient cycling). The structure and processes of an ecosystem underpin their functions, defined in the TEEB (2010) as the "potential that ecosystems have to deliver a service" (R. De Groot et al., 2010). Burkhard et al. (2012) define the ES potential as the "hypothetical maximum yield of selected ecosystem service". The ES flow (real supply) is the fraction of this potential used by humans. The notion of demand by humans is a prerequisite for the existence of an ES which does "not exist in isolation from people's needs" (Haines-Young & Potschin, 2010). The contribution of an ES to human well-being is accounted for as a benefit, and a value can be assigned to it.

Spangenberg, Haaren et Settele (2014) highlighted the societal conditions required for certain ecosystem services to exist, such as legal frameworks that allow the ex-

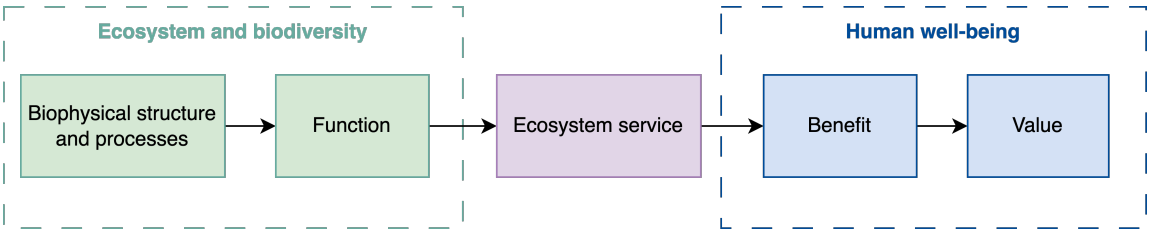


FIGURE A.1 ES cascade model, adapted from De Groot et al. (2010).

traction of natural resources, including wood harvesting and mineral extraction. Kalt et al. (2019) proposed an energy service cascade which incorporates both the "socio-economic system's structures and processes" alongside the "biophysical structure and processes", building on the ES cascade model. Kalt et al. (2019) framework focuses on energy services, such as lighting or temperature-controlled living spaces, but does not extend to material services, which are nonenergy-related services provided by resources. Whiting, Carmona et Carrasco (2022) addressed this limitation by proposing a comprehensive "resource service" cascade model that combines ecosystem, energy, and material services and includes a classification of resource services.

These recent frameworks underscore the importance of socio-economic system's structures and processes in supporting certain ES. A prominent example is the provisioning service of food ; as pointed out by Barot et al. (2017), food provisioning ES arises from the co-production of both biophysical structures, such as soil, and socio-economic system structures, such as agricultural infrastructures.

Ecosystem services models

Although the cascade model (and its derivatives) serves as a conceptual framework for understanding the flow from ecosystem functions to human benefits, various operational ES models have been developed to quantify, map and assess these benefits (K. G. Turner et al., 2015). Among the most well-known, without ranking, are :

i) the Integrated Valuation of Ecosystem Services and Tradeoffs (inVEST) series (Tallis & Polasky, 2011), widely used for mapping and valuing ES across landscapes, ii) the Integrated Model to Assess the Global Environment (IMAGE) (Stehfest et al., 2014), which focuses on large-scale, long-term environmental assessments, iii) the Multiscale Integrated Models of Ecosystem Services (MIMES) model, which examines the dynamics associated with ES functions and human activities across spatial and temporal scales (Boumans et al., 2015), iv) the ARtificial Intelligence for Ecosystem Services (ARIES) modeling platform (Villa et al., 2009), which leverages AI to dynamically map and value ecosystem services, v) the Ecopath with Ecosim (EwE) modeling software suite (Christensen, Walters & Pauly, 2005), designed to model

marine ecosystems and evaluate the impacts of fishing and environmental changes, and vi) the Global Unified Metamodel of the Biosphere (GUMBO) (Boumans et al., 2002), which simulates the interactions between human and natural systems by linking ecological, economic, and social dynamics at a global scale.

These models help translate ecological processes into services that can be evaluated, often for decision making, conservation, land use planning, or economic valuation. However, they vary in their goals, target audiences, and areas of application (Agudelo, Bustos & Moreno, 2020 ; Posner et al., 2016). The choice of which model to use depends on the scale of analysis, the type of ES that is being evaluated, and the intended use of the results, whether for scientific research, policy development, or practical management decisions.

Sustainable management of ecosystem services

The balance between ES supply and demand, called the "supply-demand" relationship, is crucial for sustainable management (Burkhard et al., 2012 ; X. Zhang et al., 2024). This concept emerged alongside the idea of ecological carrying capacity (X. Zhang et al., 2024), defined as "the maximum population level that a given environment can support given finite resources" (Monte-Luna et al., 2004).

An ES flow is considered sustainable when it meets demand without compromising future provision (Villamagna, Angermeier & Bennett, 2013). In contrast, it becomes unsustainable when current capacity cannot meet demand or when fulfilling demand reduces other services or the future availability of the same service (Villamagna, Angermeier & Bennett, 2013). Therefore, ES sustainability declines when demand increases or when the ecosystem's capacity to provide services (that is, ES supply) decreases. The next subsection discusses changes in the ES supply.

Changes in ES supply

Various drivers of change can affect the functioning of ecosystems and their ability to provide ecosystem services (IPBES, 2019). One key driver is land use change,

which directly alters ecosystems and their capacity to provide services by impacting essential ecological processes such as energy flows, soil erosion, water cycles and biogeochemical cycles (Burkhard et al., 2012 ; Fu et al., 2015 ; Hasan et al., 2020). Other factors, such as changes in sea use, resource extraction, pollution, invasive alien species, and climate change, can also influence the delivery of ES (Hevia et al., 2017 ; IPBES, 2019). Prolonged exposure to these pressures can alter the state of the ecosystem and can permanently change its capacity to deliver ES (Villamagna, Angermeier & Bennett, 2013).

The effects of these drivers are closely related to the ecosystem's *resilience*, defined as its capacity to absorb disturbances while maintaining essential functions and structures (Monte-Luna et al., 2004). Regulating services within the ecosystem play a crucial role in mitigating the impacts of ecological pressures (Villamagna, Angermeier & Bennett, 2013). For long-term sustainability, the demand for goods and services must remain within the ecosystem's ecological *carrying capacity* (Goodland & Daly, 1996).

Different dimensions of the values of nature

Valuation debates have recently evolved towards acknowledging three different value dimensions of nature to describe the ways in which nature and ecosystems are important to people. To account for this pluralism of values, scholars distinguish between intrinsic, instrumental, and relational values (Arias-Arévalo, Martín-López & Gómez-Baggethun, 2017 ; Himes et al., 2024 ; Kadykalo et al., 2019 ; Pereira et al., 2020). Value types refer to different aspects of the importance of an entity. However, an entity (e.g. an object, a place, an animal, an ecosystem, etc.) can support different types of value. **Intrinsic values** of nature are described as values inherent to a natural entity simply by virtue of its existence and independently of its utility for human societies (Arias-Arévalo, Martín-López & Gómez-Baggethun, 2017 ; Hargrove, 1992 ; R. K. Turner et al., 2003). **Instrumental values** refer to entities that are useful to people and are described as "means to achieve human ends or satisfy human preferences" (Arias-Arévalo, Martín-López & Gómez-Baggethun, 2017 ;

Hill et al., 2021 ; Himes & Muraca, 2018 ; Himes et al., 2024). They are strongly and explicitly associated with anthropocentrism and utilitarianism (Himes et al., 2024). Unlike its intrinsic value, the instrumental value of an entity is substitutable (Himes & Muraca, 2018) : other entities can be used to achieve the same ends.

The concept of **relational values** seeks to move beyond the traditional dichotomy of intrinsic and instrumental values, which has long dominated the discourse on sustainability (Pascual et al., 2023). **Relational values** refer to meaningful and non-substituted relationships between humans and nature, such as the relations of responsibility and care toward nature, eudemonism or place-based value (Chan, Gould & Pascual, 2018 ; Hill et al., 2021 ; Himes & Muraca, 2018). Himes et al. (2024) defined relational values as "values of meaningful and often reciprocal human relationships -beyond means to an end- with nature (often specified as a particular landscape, place, species, forest, etc.) and among people through nature".

Defined from an anthropocentric point of view, the concept of ES supports instrumental values only. However, in response to the emergence of the notion of relational values, the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) has broadened the concept of ES by introducing the concept of "nature's contributions to people" (NCP) (Hill et al., 2021 ; IPBES, 2019). The NCP concept is broader and more inclusive than ES, and acknowledges that our appreciation of nature's benefits to people is multidimensional and shaped by context, culture and worldviews, including indigenous and local knowledge (Díaz et al., 2018 ; Hill et al., 2021 ; Marcos et al., 2021 ; Pascual et al., 2023).

To avoid any ambiguity in terminology, this article uses the term "ecosystem services" sensu stricto, i.e. restricted to the instrumental value of nature only, and the term "nature's contributions to people" as a concept encompassing both the instrumental and relational value of nature (see Fig.A.2).

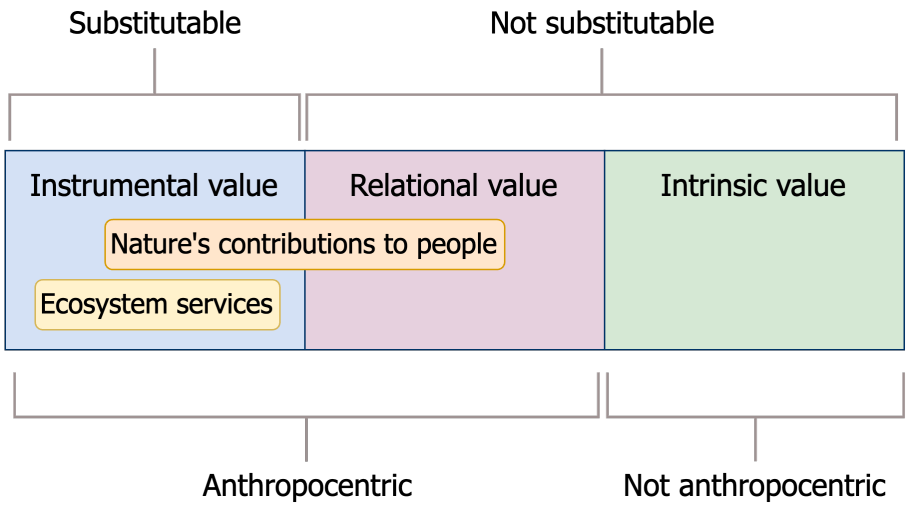


FIGURE A.2 Domains of values of nature. Adapted from Himes and Muraca (2018). The figure positions ES and nature's contributions to people in the scope of this paper. Nature's contribution to people extends the scope of ES to encompass relational values and acknowledge more diverse conceptualizations of human-nature relationships.

Instrumental use and non-use values

Instrumental values (and therefore ES) encompass both the *use value* and the *non-use value* (Himes et al., 2024 ; R. K. Turner et al., 2003) (see Figure A.3). Use value encompasses actual value (current use) and option value (expectation of future use). Actual value is linked to both the direct and indirect use of nature. Direct use can be consumptive (e.g. collection of berries) or nonconsumptive (e.g. birdwatching). Indirect use value mostly relies on regulation and maintenance of ecosystem functions, which people can benefit from without interacting directly with ecosystems. For example, indirect use values can be attributed to carbon storage or pollination benefits.

Non-use values are derived from knowledge of the continued existence of (a part of) nature (Davidson, 2013). They encompass the warm glow value (valuing benefits for other people or future generations) and the existence value (satisfaction with knowing that nature exists) (Davidson, 2013).

A common mistake lays in the confusion between existence and intrinsic values. The existence value is "related to the satisfaction that people may derive from the mere knowledge that nature exists" and can be classified as a cultural ES (Davidson, 2013). However, the concept of intrinsic value breaks away from an anthropocentric judgment and "lie[s] outside the scope of the wide palette of ES" (Davidson, 2013).

Valuation metrics

Different valuation metrics have been proposed to valuate ES including eco-energetic, thermodynamic, economic, or social values (Felipe-Lucia, Comín & Escalera-Reyes, 2014 ; Groot, Brander & Solomonides, 2020 ; Vihervaara, Franzese & Buonocore, 2019). Economic value is the most common method, allowing one to express ES "in terms of human welfare [...] measured in a common unit, allowing values to be aggregated across ecosystem services and directly compared with the values of other goods and services in the economy" (L. Brander & Guisado Gõni, 2023). However, this practice has been subject to debate, with some scholars highlighting -among

other limits- the risks of commodification of nature (De Luca Peña et al., 2022 ; Jax et al., 2013). Economic valuation often uses monetary units (dollars, euros, etc.) as the common units and includes revealed preferences techniques (e.g. hedonic prices or travel costs), stated preferences techniques (e.g. contingent valuation techniques or choice experiment) or market-based methods (e.g. market prices or replacement costs) (J. M. Salles, 2011).

As mentioned here-above (see section 2.4), ES can encompass different types of instrumental values. The Total Economic Value (TEV) (Figure A.3) has been developed to recognize this plurality and to provide a comprehensive economic valuation (L. Brander & Guisado Gõni, 2023). The various economic valuation methods are adequate to account for the whole or parts of the TEV ; while the contingent valuation method covers use and non-use values, travel costs cover direct use value only. Although it is not a mistake from an economic point of view, it is necessary to recognize these differences in coverage when aggregating the results of different valuation methods.

ES in life cycle impact assessment

Life Cycle Assessment

LCA is a prominent methodology designed to support sustainable decision-making by assessing the potential environmental impacts of human activities across all stages of a system's life cycle (ISO, 2006).

LCA operates in 4 phases, namely (i) the definition of the goal and scope, (ii) the life cycle inventory (LCI), (iii) the life cycle impact assessment (LCIA), and (iv) the interpretation of results (ISO 2006). Within the system boundary, LCA distinguishes between foreground processes, which are directly influenced or controlled by the decision-maker, and background processes, which are more generic and support activities beyond direct control (Frischknecht, 1998).

During the LCI phase, the exchanges of material or energy flows between the pro-

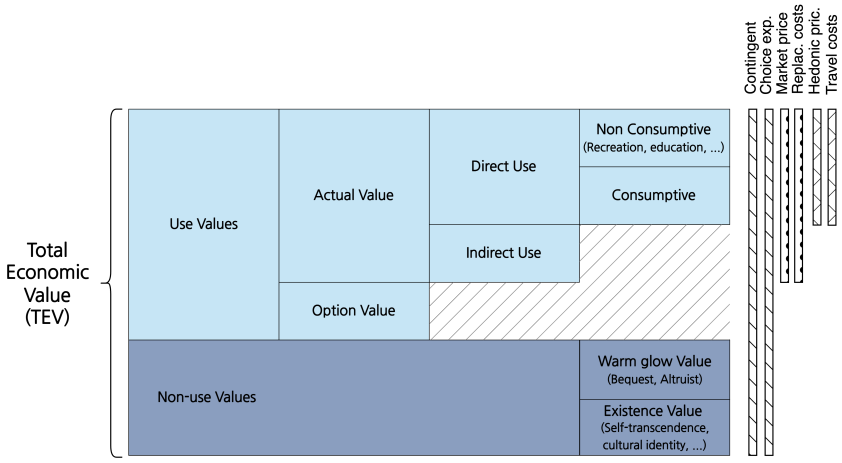


FIGURE A.3 Total Economic Value (TEV) framework and examples of valuation methods that could be used to assess it (partially or totally), adapted from Davidson (2013). Valuation methods are grouped by families. hatched bars : stated preferences methods, doted bars : revealed preferences methods, cross hatched bars : market-based methods.

duct system and the ecosphere, called elementary flows, are compiled in the inventory (ISO, 2006). The LCIA phase aims at translating the inventory flows into potential environmental impacts. In this respect, characterization factors (CFs) are multiplied with the different elementary flows to calculate impact scores. The latter are combined to provide an environmental impact profile. Impact indicators are distinguished at the "problem" level (with an indicator chosen anywhere along the cause effect chain and typically expressed in equivalent of a reference elementary flow such as global warming, in $kgCO_2e$) and at the "damage" level (with an indicator chosen at the end of the cause effect chain measuring the damage on an AoP and expressed in absolute values, e.g., climate change-driven potential impact on human health, in DALYs). Notwithstanding impact scores at damage level encompass a higher level of model uncertainty (Bare et al., 2000), they might provide more relevant and tangible information to decision-makers (Bare et al., 2000 ; Torkayesh et al., 2022).

Damage-oriented LCIA methodologies (compiling several characterization models) such as IMPACT World+ (Bulle et al., 2019), ReCiPe (Huijbregts et al., 2017) or LC-Impact (Verones et al., 2020) enable aggregating different damage-oriented impact categories to different AoP. These AoP represent entities that are important for society and that LCA aims to safeguard (Schaubroeck & Rugani, 2017 ; Verones et al., 2017).

Historically, three AoP have been considered by the LCIA community : human health, ecosystem quality and natural resources. These AoP are central to well-known impact assessment methodologies such as LC-Impact (Verones et al., 2020), IMPACT2002+ (Jolliet et al., 2003) or ReCiPe (Huijbregts et al., 2017). However, different ethical perspectives on what should be sustained can lead to the development of different AoP (Schaubroeck & Rugani, 2017). Starting from the premise - derived from the authors‘ perspective - that sustainability is an anthropocentric concept, Schaubroeck et Rugani (2017) proposes a single AoP : human well-being.
However, the choice of what an AoP entails - hence what society deems worth protecting - is based on value assumptions, ethical perspectives, and epistemology of the research community and thus open to debate (Freidberg, 2018 ; Pascual et al., 2021).

Integrating ES into Life Cycle Assessment

Despite shared sustainability ambitions, the LCA and ES communities have grown separately, and Vanderwilde et Newell (2021) noted little interaction between them. However, in the past decade, efforts have been made to integrate ES into LCA (De Luca Peña et al., 2022, 2024 ; Heitbrink, 2024 ; Maia de Souza et al., 2018 ; Rugani et al., 2022 ; Vanderwilde & Newell, 2021 ; Y. I. Zhang, Anil & Bakshi, 2010 ; Y. I. Zhang, Singh & Bakshi, 2010).

The review article from De Luca Peña et al. (2022) defines three *categories* of research approaches for integrating ES and LCA methodologies :

— the "post-analysis" category involves conducting ES and LCA analyses independently, with results interpreted in parallel.

— the "integration through combination of results" group entails conducting the analyses separately but combining their results prior to interpretation.

— the "integration through the complementation of a driving method" approach designates one methodology -either LCA or ES- as the primary framework, enhanced by tools or insights from the other. In this latter category, LCA is the most frequent driving method (in 17 studies out of 20 according to De Luca Peña et al. (2022)).

Within this last category (integration through the complementation of LCA as the driving method), the classification of Rugani et al. (2023) can be used to further distinguish research efforts into four *paradigms* : 1- the inclusion of ES as part of the functional unit of a system, 2- the inclusion of human activities' use of ES through the life cycle inventory, 3- the inclusion of regulation and maintenance services as a means of mitigating environmental impacts, and 4- the potential impacts of human activities on ecosystems' capacity to provide ES.

Rugani et al. (2023)'s paradigms are not mutually exclusive and can be combined. They suggest that integrating ES into LCA can be relevant at every phase of the assessment. Moreover, Rugani et al. (2019, 2023) suggest that the best approach to integrate ES flows might depend on their type ; some ES flows should serve as func-

tional units (e.g., provisioning services like crops), while others should be considered elementary flows in the LCI stage (e.g., provisioning services such as freshwater or biomass). Certain ES flows should be assessed in LCIA, such as the impact of human activities on carbon sequestration. Finally, some ES flows such as aesthetic services, ecological habitats that support species should be considered separately, as "natural capital components to be protected".

Building on the classifications of De Luca Peña et al. (2022) and Rugani et al. (2022), a third exhaustive classification of approaches for ES-LCA integration is formulated in table A.1. The latter further details some of the categories of Rugani et al. (2022). The "integration through the functional unit" approach is further divided into two parts : the formulation of a functional unit which can be provided by an ES (e.g. FU="Capture 1 ton of $CO_{2_{eq}}$"), and the treatment of an ES via the issue of multi-functionality (e.g. in the assessment of an agro-ecosystem providing food among other ES (Boone et al., 2019)). Similarly, building on Hardaker et al. (2022), the "potential impacts on ecosystems' capacity to provide services" approach is divided into two parts in table A.1 : the impacts due (1) to stress to and (2) to transformation of ecosystems. The reader is invited to refer to the supplementary information for additional details on the classification.

This third classification, developed in this paper, was used to position each reviewed study proposing to integrate ES in LCA. The detailed result can be found in the SI. The right-hand column of table A.1 summarizes the findings of this review by enumerating the number of times an approach was mentioned or used in a reviewed publication.

In response to the methodological challenges involved in the "LCA as driving method" approach, Rugani and colleagues (2022) suggested "intertwining" LCA and ES tools in a framework where inventories and midpoint indicators are connected, while the goal and scope phase, the endpoint-oriented results and the interpretation phase are shared. This perspective aligns with the "integration through combination of results"

Tableau A.1 Classification of approaches to integrate ES and LCA derived from the combination and enhancement of De Luca Peña et al. (2022)'s and Rugani et al. (2023)'s classifications. n is the number of propositions using or mentioning the approach for an ES-LCA integration among the 43 reviewed studies. One study can use or mention several approaches. See ESM1 for additional information.

Classification by De Luca Peña et al. (2022)	**Classification by Rugani et al. (2023)**	**Classification in this work**	n
LCA as "driving method"	In the Functional Unit	In the Functional Unit	2
		Considering multi-functionality	3
	In the inventory	In the inventory	19
	Regulation and maintenance services as a means of mitigating impacts	Impact mitigation/avoidance by regulating and maintenance ES	2
	Potential impacts on ecosystem's capacity to provide services	Impacts on ecosystems' capacity to provide ES : stress	16
		Impacts on ecosystems' capacity to provide ES : transformation	17
"Post-analysis" : LCA and ES performed independently		"Post-analysis" : LCA and ES performed independently	1
"Combination" of : damage-oriented results		"Combination" of damage-oriented results	6

approach proposed by De Luca Peña et al. (2022), which advocates for moving beyond the exclusive reliance on LCA to deliver comprehensive assessments that encompass ecological dynamics, ES benefits, and environmental impacts.

The intertwining of ES and LCA tools and the application of ES assessment at the foreground level is particularly interesting for product systems such as Nature-based solutions (Alshehri et al., 2023 ; Babí Almenar et al., 2023 ; Larrey-lassalle, Perfido & Naneci, 2022 ; Rugani et al., 2022) such as revegetation. In this context, the joint use of tools allows underlining both the degree to which the system benefits from (or depend on) nature's services and the extent of its environmental impact throughout its life cycle.

Rugani et al. (2022)'s suggestion to interweave ES and LCA tools neither eliminate nor are incompatible with the intention to further develop life cycle characterization models and factors assessing potential impacts of various drivers on ecosystems' capacity to provide services.

Challenges in defining the interface between the ecosphere and the technosphere for the integration of ES in LCA

Following the ISO14044 standard (ISO, 2006), the LCA methodology distinguishes between two spheres : the technosphere (human-made system) and the ecosphere (natural environment). Elementary flows represent exchanges between these two spheres, which are compiled into the life cycle inventory and subsequently used to calculate the potential environmental impacts of a product system. Clearly defining the boundary between the ecosphere and the technosphere is therefore fundamental for conducting an LCA.

However, establishing a strict demarcation between human and natural systems is challenging, particularly given the extent of human-induced environmental transformations (Jones et al., 2018 ; J. Liu et al., 2021 ; Plumptre et al., 2021). This interconnection has led to the concept of social-ecological systems (SES) (Berkes, Folke & Colding, 2000), which recognizes that human and natural systems are deeply intertwined. SES reflect a shift in sustainability debates from a dichotomous understanding of human-nature relationships (humans and nature as opposed entities) to an integrated perspective (humans and nature as two entities that form one whole system) (West et al., 2020). Although this latter perspective still recognizes a conceptual distinction between human and natural components, it highlights their strong interdependencies (West et al., 2020).

Within SES, ES emerge at the interface of human and natural systems. In ES research, services provided by social-ecological systems may be classified as ES or excluded from the analysis based on the degree of anthropization, depending on the scope of the study (Barot et al., 2017 ; Martínez-García, Martínez-Paz & Alcon, 2025).

Agriculture provides a clear example of this complexity (Barot et al., 2017). Agricultural systems, which provide food for human societies (a provisioning ES), depend on natural processes such as water availability and soil fertility. However, these processes are also heavily influenced by human interventions, including irrigation, fertilization,

and land management. As a result, agricultural ES are co-produced through the interaction of natural and human systems. The degree of human influence varies along a continuum : from wild blueberry harvesting to hydroponic greenhouse production, passing through organic and industrial farming, human involvement in generating the same food provisioning service differs substantially.

In this context, the complex nature of ES and the lack of a clear boundary between the ecosphere and the technosphere present significant challenges for their integration into LCA. This ambiguity complicates the distinction between what qualifies as an ES and what does not, as well as determining whether impacts on these services should be taken into account in LCA, in which the characterization of impacts applies, in theory, only to exchanges between the ecosphere and the technosphere (Rosenbaum et al., 2015b).

Moreover, in the context of integrating ES into the functional unit of an LCA, it is essential to recognize that while an ES can fulfill a specific function (e.g., capturing $CO_{2_{eq}}$), highly anthropized technological systems, which the LCA community would classify as part of the technosphere, can also serve the same purpose.

Rationale for integrating ES into Life Cycle Impact Assessment

As a widely used and increasingly popular environmental assessment tool, LCA has the potential to inform and raise awareness among decision makers about the potential impacts of their systems (Hellweg et al., 2023). The integration of ES concepts within LCIA can promote the attention of decision makers toward potential impacts on ES, and reveal the "hidden costs or benefits to society of the LCI flows degrading or supporting functional ecosystems" (Othoniel et al., 2019). In this sense, accounting for ES in LCA can serve as a means of promoting the conservation of functioning ecosystems.

Moreover, the current exclusion of ES from LCA impact profiles constitutes a significant blind spot in LCA impact results (Rugani et al., 2022). Since LCA aims to provide a comprehensive assessment of environmental impacts, this omission can dis-

tort sustainability-driven decision-making. By not explicitly addressing ES, impact profiles risk encouraging decisions that inadvertently compromise ecosystems' ability to provide essential services (X. Liu, Ziv & Bakshi, 2018b).

A critical review on the proposals to integrate ES into LCIA

Proposals to integrate ES into LCIA as an Area of Protection

Proposals have been made to integrate ES as LCIA impact indicators both at the midpoint (e.g., Saad et al. (2011)) and at the damage levels (e.g., Cao et al. (2015)). Due to the relevance of impact scores at the damage level for decision-making processes (Bare et al., 2000), various authors recommend developing damage-oriented characterization models and factors for ES (Bulle et al., 2019 ; Hardaker et al., 2022 ; Heitbrink, 2024 ; Verones et al., 2017).

Methodological advances and innovations in LCIA models have led the UNEP-SETAC Life Cycle Initiative to review the LCIA framework and its AoP (Verones et al., 2017). The working group identified two categories of entities deserving protection that need to be distinguished in LCIA : entities pertaining to (i) human and human-made systems and to (ii) natural systems. Within these categories, the authors consider that entities can have "intrinsic", "instrumental", and "cultural" values (Table A.2).

Tableau A.2 Classification of AoP depending on the type of object deserving protection and the social value assigned to it, as per Verones et al. (2017). Verones et al. (2017) distinguish intrinsic, instrumental and cultural values, which can emerge from human and human-made systems or from natural systems. In this work, we suggest to replace cultural values* by relational values.

Type of value / **Type of objects**	**Intrinsic value**	**Instrumental values**	**Cultural values***
Human and human-made systems	Human health	Socio-economic assets	Cultural Heritage
Natural systems	Ecosystem quality	Natural resources & ecosystem services	Natural heritage

Current ES classifications categorize cultural ES under the instrumental value of natural systems (Haines-young, Paddocks & Lane, 2023). Consequently, using the term "cultural" to denote a distinct value domain, separate from instrumental values, may create confusion. The term "cultural values" in Verones et al. (2017)'s classification therefore diverges from the value classifications introduced in Section A. The emerging concept of "relational values" may provide a more comprehensive framework for capturing the full spectrum of value types.

Verones and colleagues' double-entry table suggests creating an AoP for each category of entity and corresponding type of value. In this way, "human health" and the "ecosystem quality" are described as entities with intrinsic value that belong to the "human/human-made" and "natural" systems, respectively. Similarly, Verones et al. (2017) consider "cultural heritage" and "natural heritage" as AoP representing entities with cultural (relational) values, pertaining to "human/human-made" and "natural systems", respectively. There are various interpretations of heritage values (Azzopardi et al., 2023), and we call for further research to discuss the category of values heritage values fall into. Finally, Verones et al. (2017) define "Socio-economic assets" and "Natural resources and ES" as entities with instrumental values pertaining to the "human/human-made" and "natural" systems, respectively.

As discussed in section A, the elaboration of AoP reflects specific ethical perspectives. Verones and colleagues' classification, which organizes AoP by type of value and system, exemplifies a common approach in mainstream scientific frameworks that emphasize a distinction between human and natural systems (Levis et al., 2024). The challenges associated with this division echo those discussed in section A. This dichotomy, characteristic of many Western scientific traditions, conceptually separates human from nature. In this context, "separation" refers to an ontological understanding of humans and nature as distinct entities ; within Western philosophy and science, nature is often conceptualized in opposition to humans and human-made objects (Ducarme & Couvet, 2020). However, this separation may be less relevant in alternative knowledge systems, such as Indigenous frameworks, where human and natural systems are often viewed as inseparable (Díaz et al., 2015, 2018 ; Pascual

et al., 2023). Section A of the Supplementary Information further explores this classification of values, particularly with regard to the boundary between natural and human systems and the concept of intrinsic values. Despite these limitations, we adopt the framework of Verones et al. (2017) for the remainder of our analysis.

Verones et al. (2017)'s proposal to develop a "Natural resources & Ecosystem services" AoP reflects the growing interest among LCA practitioners and LCIA model developers to include information on ES and instrumental values in LCA endpoint-oriented impact profiles. This increasing interest is also illustrated by the creation of a specific task force group within the 3rd phase of the Global Life Cycle Impact Assessment Method (GLAM) initiative (Life Cycle Initiative, 2021). However, only a few studies have provided operational ES characterization models and factors at the damage level (e.g., Arbault et al. (2014), Cao et al. (2015) et Othoniel et al. (2019)).

This shortcoming could be due to the absence of methodological clarity and guidelines regarding optimal approaches for the development of an ES AoP and associated characterization models. The following paragraphs explore the lack of structure for the development of the latter.

The LCIA community treats Natural Resources and provisioning Ecosystem Services separately despite their interconnectedness

The LCIA community has largely overlooked the inherent connection between natural resources and provisioning ES. This disconnect may stem, in part, from historical ES classifications that focused primarily on services derived from biotic processes and excluded geo-resources. The adoption of a more holistic classification in CICES v5.2 provides an opportunity to bridge this gap and facilitates a more coherent integration of natural resources and provisioning ES within LCIA.

Historically, the integration of market-valued natural resources into LCIA frameworks preceded the inclusion of broader ES categories (UNEP/SETAC 2019).

Research studies have been carried out on abiotic resources such as water (Pradinaud et al., 2019a), metal or mineral resources (Ardente, Beylot & Zampori, 2023 ; Beylot

et al., 2024 ; Charpentier Poncelet et al., 2021 ; Greffe, Margni & Bulle, 2022 ; Pradel, Garcia & Vaija, 2021 ; Sonderegger, Pfister & Hellweg, 2020), soil (Sonderegger et al., 2020), marine surfaces (Taelman et al., 2014), or biotic resources (Crenna, Sozzo & Sala, 2018) such as wood (Odppes, Bulle & Ugaya, 2021) or halieutic resources (Hélias, Stanford-Clark & Bach, 2023 ; Langlois et al., 2014 ; Luong, Dewulf & Laender, 2020).

Although these natural resources are provided by biophysical (e.g. timber or fish provisioning) or geophysical provisioning ES (e.g. mineral resources), most studies do not explicitly refer to ES concepts. The absence of efforts to clarify the links between resources and ES within the LCIA community has led to the parallel development of models, tailored for natural resources (Sonderegger et al., 2017) on one hand and for other ES (Rugani et al., 2019) on the other.

These siloed development efforts have resulted in conceptual and terminological inconsistencies.

Within the resources-oriented LCIA community, the term "functionality of natural resources" is used to refer to the instrumental value that a resource provides to society, whereas in the ES community, "ecological functions" describe the biophysical processes that sustain ecosystems.

Moreover, both ES and resources-oriented LCIA communities use the term "service" to denote the benefits that ecosystems or resources provide to society, referred to as "ES" and "resources services", respectively. However, in practice, the level of detail in these services differs. CICES classifications delineate resources at the class level (e.g., CICES v5.2 class 4.2.1.2 : *mineral substances used for material purposes*), while resources-oriented characterization models in LCIA often extend beyond resource supply to specify the societal service provided by the resource (e.g., *materials used for mobility*, such as aluminum used for vehicles). Harmonization and clarification of the level of services to be considered in the characterization models could help clear up this ambiguity. Research efforts reconciling CICES with resources' services such as Whiting, Carmona et Carrasco (2022)'s classification could be explored to address

these limitations.

Several proposals for integrating ES into AoP have been published but many lack clarity and consistency

Although Verones et al. (2017)'s social value classification suggests a common AoP for natural resources and ES (see Table A.2), their conceptual LCIA framework does not clearly differentiate between the two. In Figure 1 of their article, resources and ES are depicted separately, reinforcing the distinction discussed in section A. Following Verones et al. (2017)'s recommendation, Bulle et al. (2019) implemented a "Resources and Ecosystem Services" AoP in their LCIA methodology. However, the terminology used for this AoP has caused confusion : while it was intended to represent "the services provided by both resources and ecosystems", it has often been interpreted as "resources" (on the one hand) and "the services provided by ecosystems" (on the other).

Other scholars offered different approaches for integrating ES into AoP. In their theoretical LCIA framework, Hardaker et al. (2022) proposed separating ES completely from natural resources, recommending a standalone "ecosystem services" AoP to accompany existing AoP for human health, ecosystem quality, and natural resources. Their ES-specific AoP combines two aspects : (1) the use of ES supply (to what extent do ES mitigate midpoint impacts generated by the studied system ?) and (2) the change of ES supply (to what extent does the studied system change ecosystems' capacity to provide ES ?) (see paradigm 3 in section A). These proposals represent innovative contributions to LCA by emphasizing human *dependence* on natural capital and ES, an aspect typically overlooked in traditional LCA impact profiles and crucial for the ES community (IPBES, 2019 ; Natural Capital Coalition, 2016). However, one should keep in mind that the notion of human reliance on natural systems does not correspond to an assessment of the impact of the product system on the environment, but rather to an assessment of the dependence of the product system on the environment. Hence, it is debatable to position this as an impact pathway in the impact assessment framework.

Taelman et al. (2023) offered a different perspective by proposing to integrate ES into existing AoP ; the "Natural Resources" AoP encompasses provisioning ES, the "Ecosystem Quality" AoP is broadened to include Regulating and Maintenance ES, and the "Human health and well-being" AoP "covers both human health impacts and cultural ES". Similarly, Rugani et al. (2019) establish links between *Provisioning ES* and the Natural Resources and Ecosystem Quality AoP, and between both *Regulation and maintenance*, and *Cultural ES* with Human well-being and Ecosystem Quality AoP. These works are among the few that highlight the similarities between provisioning ES and natural resources (see section A).

However, by separating ES between AoP, Taelman et al. (2023)'s proposal risks overlooking the intricate links between ES flows and the broader AoP of human health and ecosystem quality (see section A), a limitation identified by the authors. Moreover, this proposal diverges from Verones et al.'s approach, which advocates for separate AoP for each system type, based on its specific value type (Table A.2).

Modeling purposes for Natural resources (which are provisioning ES) lack transparency and coherence

Similarly to AoP creation, the modeling "purposes" behind LCIA characterization models are shaped by what developers believe should be protected (Freidberg, 2018). For natural resources - which are here but have been barely classified as provisioning ES (5.2) - these purposes vary widely (Dewulf et al., 2015).

The diversity of natural resources, from minerals, metals, fossil fuels and air components to land and water, combined with their typological differences (stocks, funds, and flows), poses a significant challenge to harmonize impact modeling across different approaches (Pradinaud et al., 2019a ; Sonderegger et al., 2017). However, beyond the characteristics of the resources, the aspect of the resource that is intended to be protected plays a critical role in shaping the characterization models.

The various modeling methods that have been proposed (Sonderegger et al., 2017) often address different research questions, which in turn have led to the development

of different impact assessment approaches. For example, some LCIA models aim to assess the potential impacts on the (reduced) availability of natural resources for future generations (e.g. Pradinaud et al. (2019a)), their stock in the earth's crust (Oers & Guinée, 2016 ; Oers, Guinée & Heijungs, 2020), the efforts required to make this stock accessible (Vieira et al., 2017), their exergy content (Dewulf et al., 2007) or cumulative energy demand (Frischknecht et al., 2007), among other indicators.

Dewulf et al. (2015) explored the diverse interpretations of the AoP specific to natural resources, identifying three key safeguard subjects : (1) the Asset of Natural Resources, (2) the Provisioning Capacity of Natural Resources, and (3) the Role of Natural Resources in Global Functions. Similarly, Berger et al. (2020) noted that the selection of a characterization model often depends on its intended application, underscoring the variety of perspectives within the field.

However, the recent consensus of Verones et al. (2017) advocates that characterization models should focus on the *instrumental value* of natural resources, particularly in terms of the services they provide to humans (Verones et al., 2017). This shift is reflected in the AoP "Resources and Ecosystem Services" of the IMPACTWorld+ methodology, where models are increasingly expected to capture the functionality of resources for human society (Bulle et al., 2019).

However, earlier models often failed to incorporate this functional perspective. For example, early mineral resource-oriented methods focus on resource extraction (Berger et al., 2020). However, the extraction process fails to address a loss of resources' service per se : a resource extracted from the lithosphere and used in the technosphere can still provide instrumental value for human society if recycled (Greffe, Margni & Bulle, 2022). In contrast, the degradation and dissipation of materials threaten the functionality of natural resources (Charpentier Poncelet et al., 2021 ; Graedel & Miatto, 2022 ; Greffe, Margni & Bulle, 2022). In response, newer characterization models are evolving to assess the potential loss of services due to resource dissipation (Beylot et al., 2024 ; De Bruille, 2014 ; Greffe, Margni & Bulle, 2022). The operational damage-oriented LCIA methods on resource dissipation assess the impact in terms of loss of exchange value (Ardente, Beylot & Zampori, 2023 ; Charpentier Poncelet

et al., 2021 ; Saldivar et al., 2023), quantified with the market price of the dissipated resource. Efforts to clarify the extent of instrumental value (use, non-use, see section A) that these evaluation methods can cover are needed to promote harmonization between characterization models (Greffe, Margni & Bulle, 2022).

Assessment of ES (beyond natural resources) lacks harmonization, completeness and guidelines

In this section, the term "ES" is used as adopted by the LCIA community, which distinguishes between ES and services provided by natural resources, although the latter derive from provisioning ES.

Despite more than a decade of attempts to integrate ES concepts into LCA, their integration at the LCIA level remains in its infancy (Hardaker et al., 2022 ; Rugani et al., 2022).

Our literature review identifies two schools of thought regarding the assessment of ES in LCA : i) researchers focusing on grasping the ES supply-demand relationship introduced in section A, and ii) researchers focusing on modeling the impacts of human activities on ecosystems' capacity to provide services. Although these approaches differ in perspective, their proposals overlap in part.

i) Academics interested in the ES "supply-demand" relationship

Although conventional LCA primarily focuses on assessing potential environmental impacts, the ES community places a significant emphasis on the dependence of human activities on natural capital and on the ES supply-demand relationship (Burkhard et al., 2012 ; Xue & Bakshi, 2022). Some LCA researchers have proposed theoretical frameworks and models that align with this philosophy (B. Bakshi & Small, 2011 ; Hardaker et al., 2022 ; X. Liu et al., 2020 ; Rugani et al., 2019 ; Xue & Bakshi, 2022).

The approaches proposed by X. Liu, Ziv et Bakshi (2018b) and X. Liu et al. (2020)

aim to extend the classic steps of LCA to incorporate the notions of *demand* and *supply* of ES at different spatial scales. Their proposals build on B. Bakshi et Small (2011)'s Techno-Ecological Synergies (TES) framework aiming at "understanding and designing synergies between technological and ecological systems". Through this framework, the authors reinterpret the environmental intervention matrix (used in conventional LCA to compile exchanges between the ecosphere and the technosphere) into a *demand* matrix. In the latter, flows of resources are referred to as demand for provisioning ES, and flows of substances emissions are referred to as demand for regulation and maintenance services. By comparing demand and supply flows for each ES, X. Liu, Ziv et Bakshi (2018b) and X. Liu et al. (2020) derive sustainability scores : if demand induced by the activity exceeds the supply potential, the activity is deemed unsustainable.

The demand for regulation and maintenance services reflects the third paradigm of Rugani et al. (2022) (see section A), which posits that by exerting pressures on ecosystems, human activities generate a form of *demand* for such ES to mitigate the damage they induce. Similarly, Hardaker et al. (2022) emphasize the demand for regulating and maintenance services by introducing an impact indicator that quantifies "the portion of global ES used to mitigate impacts at the problem level".

Demand for ES : straddling between the inventory and impact assessment The interpretation of Hardaker et al. (2022)'s indicator as an "impact" is debatable. This indicator differs from the impact indicators conventionally used in LCIA that reflect a change in the state of the system as a result of a cause-effect chain (Hauschild, Michael Z and Huijbregts, 2015). Rather, one could interpret it as a descriptor or the state of a system that accounts for the quantity of services it provides. In this context, proposals to quantify the demand for ES could be regarded as part of the LCI phase rather than of the LCIA phase.

This ambiguity on the positioning of demand-supply approaches within the LCA

framework can also be found in X. Liu, Ziv et Bakshi (2018b) and X. Liu et al. (2020). The authors' reinterpretation of the environmental matrix into a "demand matrix" reinforces the link between their approach and the LCI phase. However, quantifying the demand for regulation and maintenance services requires an upstream impact assessment step. It is therefore difficult to classify this approach as part of the LCI or the LCIA phases.

From a LCIA perspective, their conceptualization of demand can be interpreted in two different ways depending on the type of ES ; the demand for provisioning ES reflects the consumption of natural resources, which is captured at the LCI level, while the demand for regulation and maintenance services can be viewed as a positive impact associated with the mitigation of the impacts of the product systems and embedded into the impact assessment phase.

The ES demand-supply approach to assess "absolute environmental sustainability" Some LCA scholars have emphasized the connections between ES concepts, including the relationship between ES supply and demand, and absolute environmental sustainability (AES) metrics (Hardaker et al., 2022 ; X. Liu & Bakshi, 2018 ; X. Liu, Ziv & Bakshi, 2018b ; Xue & Bakshi, 2022).

Several LCA-based AES assessment methods have been proposed to "evaluate whether an anthropogenic system can be considered environmentally sustainable in an absolute sense"(Bjorn et al., 2020). In these methods, the impact of the anthropogenic system is compared to its "assigned carrying capacity". These estimates of carrying capacity are often derived from the planetary boundary framework (Bjørn et al., 2016), which defines a "safe operating space" constrained by nine ecological boundaries (Steffen et al., 2015).

Some scholars have emphasized the relevance of ES concepts in promoting AES. Hardaker et al. (2022) argue that the integration of ES in LCA methodologies enables the evaluation of how the environmental burdens of product systems contribute to the encroaching on planetary boundaries. In the same perspective, other researchers propose to revise AES metrics based on ES. Through their TES framework, B. R.

Bakshi, Ziv et Lepech (2015) et X. Liu et Bakshi (2018) quantify AES by comparing "human demand for ES (emissions, resource use) with natural supply (carrying capacity of ecosystems)" (Xue & Bakshi, 2022). Unlike planetary boundary-based AES methods, which use downscaling to estimate local carrying capacity, the TES-LCA framework relies on biophysical ES models to estimate ES supply, "analogous to safe operating space", at different scales (Xue & Bakshi, 2022).

Challenges on the application of the supply-demand approaches As mentioned above, quantification of the demand for regulation and maintenance services to mitigate potential impact presupposes an upstream impact assessment step, which in turn implies that a relevant impact category has already been selected during the goal and scope definition phase of LCA. Such an approach presents a challenge as at that early stage there is often no prior knowledge of which ES will be involved. The implementation of demand-supply approaches could eventually build on the iterative nature of LCA ; Initial conclusions drawn from the interpretation phase could inform refinements to the goal and scope definition, leading to subsequent refinements in ES integration in both the LCI and LCIA phases.

The methodologies proposed by X. Liu, Ziv et Bakshi (2018b),X. Liu et al. (2020) and Rugani et al. (2022) involve expanding LCI flows to include the ES demand. Despite the conceptual advances of their proposals, the latter remain difficult to implement across an entire product life cycle ; the current structure and scope of LCA, which requires analyzing the environmental performance of background systems with unknown or globally distributed locations, do not allow for an explicit representation of the demand for ES mitigating these systems' impacts. Achieving this would require high-resolution regionalization. In practice, such flows can only be assessed at the foreground level, and X. Liu, Ziv et Bakshi (2018b)'s approach remains constrained by the current capabilities of LCA software and LCI databases.

To answer similar challenges and account for both foreground and background processes within their "mitigation indicator", Hardaker et al. (2022) propose developing

a database of regionalized characterization factors at various spatial scales, while acknowledging the challenges this proposal involves.

ii) Impacts on ecosystems' capacity to provide ES.

A second school of thought for the assessment of ES in LCA follows a more classic LCIA perspective and aims at modeling potential impacts on ecosystems' capacity to provide services.

In parallel with their "mitigation" indicator introduced earlier, Hardaker et al. (2022) proposed a second indicator to capture the impact of a product system in terms of "changes in ES supply". This indicator reflects two impact pathways identified by Hardaker et al. (2022).

The first pathway highlights how ecosystem transformation, primarily driven by land use changes, alters its structure and functions and therefore threatens its capacity to provide services (Hardaker et al., 2022). This impact pathway aligns closely with common ES LCIA models, which are primarily designed to assess the potential impacts of land use.

The second pathway captures changes in an ecosystem's structure and functions caused by other environmental pressures -such as the emission of substances into soil, air, or water- "exceeding the carrying capacity of ecosystems to assimilate the emissions that cause these problems" (Hardaker et al., 2022).

The different proposals to integrate ES in LCIA have been developed on the basis of one or both of these pathways. Vanderwilde et Newell (2021) classified these research efforts into three clusters : a) the land use cluster, b) the biodiversity and ES cluster and c) the dynamic modeling. The following paragraphs discuss the latter.

The *land use cluster* gathers characterization models and factors developed to assess

the potential impacts on ES due to Land Use and Land Use Change (LULUC), linking land transformation to changes in ecosystem functions (Koellner & Geyer, 2013). The UNEP/SETAC LULCIA taskforce has laid the foundations for these approaches by developing modeling guidelines for ES and biodiversity impacts (De Baan et al., 2013 ; Koellner & Geyer, 2013 ; Milà i Canals et al., 2007). Subsequently, other models have been developed on the basis of these models and guidelines or to improve the characterization of land use (Bos et al., 2016 ; Cao et al., 2015). These land use impact-oriented models have the advantage of being operational in relation to existing inventory data (Rugani et al., 2022).

However, current land use characterization practices have significant limitations.

First, they may not adequately account for the multitude of pressures that influence the capacity of the ecosystem to provide ES (Alejandre, Bodegom & Guinée, 2019). Since land use is commonly employed as a proxy for ecosystem transformations, there is also a risk of double-counting if future models incorporate additional pressure pathways (e.g. emission of substances into soil) that are not currently considered.

Second, the representativeness of land use impacts is constrained by the scale and resolution of available inventory data (Othoniel et al., 2016 ; Rugani et al., 2022). Most ES depend on regional and local ecosystem conditions, which are often poorly captured by generalized land use categories in current LCI databases. Research initiatives aimed at refining inventory data resolution, such as Regioinvent (Agez, in review), offer promising pathways for enhancing the accuracy of land use inventory flows and, consequently, their relevance for ES assessments.

Furthermore, ES supply is time dependent (Othoniel et al., 2016). The ability to capture ES dynamics through land use flows is likely hindered by the seasonality of certain regulation and maintenance ES, which do not necessarily follow the same temporal patterns as technosphere flows modeled within the same dataset.

Finally, ES are often provided through bundles (Saidi & Spray, 2018), and existing models for land use impact assessment are not adequate to analyze human influence on the capacity of ecosystems to deliver multiple ES (Hardaker et al., 2022 ; Othoniel

et al., 2016).

The *biodiversity and ES cluster* brings together methods establishing a relationship between biodiversity, ecosystem functions and ES (see section 5.7). Among these proposals, the framework proposed by Oginah et al. (2023) links the impacts on structural biodiversity to ES through changes in functional biodiversity. Following this framework, T. Li et al. (2023) proposed a characterization framework linking microplastic and nanoplastic emissions to potential impacts on soil ES. These literature studies are promising attempts to capture ES concepts. However, the limited number of even very recent studies belonging to this cluster reflects a high degree of complexity and sophistication potentially associated with the approach.

The *dynamic modeling group* encompasses proposals to adopt dynamic integrated models to quantify ES impacts by assessing the interactions between ecological and human systems. In this sense, Arbault et al. (2014) used the GUMBO model (Boumans et al., 2002) which "simulates context-dependent cause-effect chains between the natural sphere and the human sphere" to retrieve time- and scenario-dependent CFs, while Othoniel et al. (2019) "developed an integrated system dynamics model of land cover and ecosystem services" building on the MIMES framework (Boumans et al., 2015) to develop CFs at the scale of Luxembourg and its municipalities. Both studies show the benefits of using system dynamics to quantify CFs to assess impacts on ecosystem capacity to supply services. Nevertheless, the generated CFs are spatial and/or temporal context-dependent and, therefore, their use cannot be easily generalized. Only the approach can be replicated, ideally, on a per-country or even regional basis.

Among these different clusters, the cascade model has been endorsed by several studies to support the development of ES-related CFs (Maia de Souza et al., 2018 ; Pavan & Ometto, 2018 ; Rugani et al., 2019). Although Rugani et al. (2019) suggested guidelines for the computational structure, first studies that build on the cascade

model are emerging (Alshehri et al., 2023 ; Lago-olveira, Moreira & Gonz, 2025 ; X. Liu et al., 2020 ; Pavan & Ometto, 2018).

Despite all these advances, the integration of ES into LCIA remains limited. Firstly, the overall coverage of ES impacts in LCIA is low (Alejandre, Bodegom & Guinée, 2019 ; De Luca Peña et al., 2022 ; Hardaker et al., 2022 ; Othoniel et al., 2016 ; Rugani et al., 2019 ; Vanderwilde & Newell, 2021). Although ES classifications such as CICES intend to be exhaustive, only a limited number (mainly terrestrial) of ES have been addressed in LCIA (Alejandre, Bodegom & Guinée, 2019 ; Vanderwilde & Newell, 2021). Cultural ES, in particular, have been overlooked or completely excluded (Alejandre, Bodegom & Guinée, 2019 ; Soldati et al., 2023 ; Vanderwilde & Newell, 2021). Secondly, the focus has predominantly been on land use as a key pressure on ES, while other critical environmental pressures are underrepresented (Hardaker et al., 2022 ; Rugani et al., 2022).

Finally, analysis of the various proposals to integrate ES into LCIA models through the lens of ES knowledge reveals a lack of terminological clarity regarding ES concepts. The reviewed LCA literature often refers to ES without specifying whether they consider ecosystem functions, ES flows, or benefits (Hardaker et al., 2022 ; Othoniel et al., 2016 ; Pavan & Ometto, 2018). Given the conceptual difference between these concepts, underlined in the cascade model, the absence of a clear terminology hampers positioning the different proposals for ES integration in LCIA, and the understanding of the entity modelers intend to assess.

Metrics for assessing potential impacts on ES at the damage level are often unclear, conductive to potential inconsistencies.

Authors have suggested assessing ES at the damage level using various metrics such as emergy (Odum, 1996), human welfare (Arbault et al., 2014) or monetary valuation (Alejandre, Bodegom & Guinée, 2019 ; Bruel et al., 2016 ; Cao et al., 2015 ; Othoniel et al., 2019). Monetary valuation is often proposed to express the costs of losing ES for society (Bulle et al., 2019). Nevertheless, there is no recommendation on the

monetary valuation method to use while "different valuation methods may not be measuring the same economic construct and therefore values from different methods may not be directly comparable" (R. S. De Groot, Wilson & Boumans, 2002). ES-oriented characterization models at the damage level (e.g. Ardente, Beylot et Zampori (2023), Cao et al. (2015), Charpentier Poncelet et al. (2021), Othoniel et al. (2016) et Saldivar et al. (2023)) frequently employ, and sometimes combine, different monetary valuation methods to account for both damage and adaptation costs. However, as per section A, different valuation methods may capture distinct aspects of the use and non-use values of a system. Although economists generally accept aggregating results from multiple methods, ensuring transparency in method selection is crucial for maintaining consistency across models used in comparative analyses.

How ES interact with existing AoP lacks clarity.

The impacts mechanisms on human health, ecosystem quality, and ES are interrelated and should not be considered (and therefore modeled) in silos (Schaubroeck & Rugani, 2017).

Important research efforts have been conducted to assess the role of biodiversity in the supply of ES, leading to the development of the ecological subfield of biodiversity-ecosystem functioning (BEF) (Balvanera et al., 2006 ; Cardinale et al., 2012 ; Isbell et al., 2017 ; Tilman, Isbell & Cowles, 2014). BEF researchers have evidenced the influence of biodiversity on ecosystem functioning, showing that loss of species is likely to disrupt patterns of material and energy flows, underpinning ecosystem functioning (Hong et al., 2022 ; Isbell et al., 2017 ; Jochum et al., 2020). Callesen (2016) notes that the loss of some species that play an essential and quantitatively important role in the structure and processes in ecosystems, called *keystones* species, "may trigger a cascade of effects on ecosystem structures and functions", resulting in a loss of ES. As a result, the ability of ecosystems to provide services might be altered. Thus, biodiversity has been shown to play a key role in maintaining human well-being (Pohjanmies et al., 2017 ; Sandifer, Sutton-Grier & Ward, 2015). The link between

the ecosystem quality AoP and ES has already been discussed by LCIA scholars (De Souza et al., 2013 ; Koellner & Geyer, 2013 ; T. Li et al., 2023 ; Oginah et al., 2023 ; Woods et al., 2018). However, outside of the recent approach suggested by Oginah et al. (2023) (see A), current LCIA practices and characterization models hardly express this interconnectivity. Clarifying these links in LCIA could shed light on current gaps and offer interesting research avenues for LCIA method developers. Since characterization models pointing to the AoP Ecosystem Quality result from a transformation of ecosystem functions, it is likely that these cause-effect chains can be extended towards the ES AoP.

Ecosystems are also closely related to human health. By definition, ES play a crucial role in supporting human health, directly and indirectly (Costanza et al., 1997 ; Sandifer & Sutton-Grier, 2014). The consistency of such support is clear for ES such as food provisioning or pest control. However, less direct contributions can be acknowledged ; human contact with biodiversity and nature can result in psychological and physiological health benefits (J. C. Fisher et al., 2023 ; Sandifer, Sutton-Grier & Ward, 2015 ; Zhou et al., 2024).

Thus, by influencing the supply of ES, a change in ecosystem functions can affect human health. However, one of the main challenges is to determine how to quantitatively account for the effect of such a change and integrate it into characterization models in a manner that ensures both reproducibility and practical applicability.

Some characterization models already feature links between the AoP. Cao et al. (2015) assess the potential impact of land use on the capacity of the ecosystem to filtrate water as the cost to society to adapt to this loss of service. The authors distinguish countries that are economically able to adapt (having to assume economic costs to avoid losing service) from others. When not able to adapt, countries face potential impacts on Human Health due to a loss of water provisioning.

Although this complementary impact pathway has not yet been operationalized, this proposal highlights that impact scores and affected AoP can vary depending on

whether adaptive or nonadaptive behaviors are considered. It is therefore essential for LCIA method developers to explicitly state whether adaptation measures are accounted for and to ensure consistency in their modeling assumptions and choices.

Conclusion

Despite growing interest in integrating ES into LCIA models and factors, progress remains hindered by conceptual, methodological, and terminological inconsistencies. Although natural resources and provisioning ES are inherently connected, they are often treated separately, resulting in fragmented approaches within LCIA models.

Various proposals have been made to integrate ES as a new AoP or within existing AoP, but the absence of consensus is conducive to the development of inconsistent models which follow distinct modeling objectives. Despite several proposals to incorporate ES into characterization models, these efforts are frequently undertaken following different schools of thought, without harmonized guidelines to unify these approaches. In addition, different valuation methods used to assess the potential impacts on ES at the damage level address varying aspects of ES, but the rationale behind their selection is rarely discussed or explicitly justified. Finally, critical interactions between ES concepts and existing AoP remain insufficiently addressed in current LCIA practices, despite representing promising avenues for further research and development. As a result, the integration of ES in LCIA remains underdeveloped and inconsistent, with limited coverage across different categories of ES and types of environmental pressure.

This critical review attempts to establish a foundational understanding of ES concepts for the progress of LCIA, identifying the most urgent challenges to account for the ES value in impact assessment models. Its conclusions call for the development of a comprehensive, consistent, and globally accepted framework to facilitate the integration of potential impacts on ecosystems' capacity to provide services at the damage level in LCIA. By establishing a structured approach to modeling potential impacts on ES, the LCIA community can better capture the complexity of ES, ultimately

supporting more informed decision making.

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Data Availability

Data sharing is not applicable to this article.

Declarations

The authors declare no competing interests.

SUB-APPENDIX A - Discussion on Verones et al. (2017)'s values classification

The classification proposed by Verones et al. (2017) has yielded a number of observations, which are further detailed in this section.

As detailed in the body of the manuscript, any classification of AoP is the result of choices (conscious or unconscious), influenced by the perspectives and backgrounds of the researchers (Schaubroeck & Rugani, 2017). These choices are value-laden and therefore can never be neutral (Freidberg, 2018). Here, we align with Freidberg (2018) who states that discussing the positioning of researchers is crucial to minimize biases, and it should be kept in mind that methodological choices are not free from the influence of personal values.

In this context, we understand the current definition of AoP as a reflection of what the LCA and LCIA communities currently consider the most important to protect. The table of Verones and colleagues is the result of a consensus ; it reflects an agreement that has been reached between LCA experts operating within a western scientific framework. Like any consensus, it can change over time in light of new inputs and discussions. These ongoing developments and adaptations are processes that are inherent in scientific research.

As a reminder, the classification defines AoP dividing into three types of values (intrinsic, instrumental, and relational) and two types of systems (human/human-made and natural).

On the types of values

In Verones et al. (2017)'s original work, the term "cultural" is used instead of "relational". We suggested replacing this term by "relational", which seems more aligned with the current state of the art in the research community working on values (Himes et al., 2024).

The partition of value types in Verones and colleagues' framework aligns with the concept of plurality of values, which receives more and more attention in academic works and in the science-policy interface (Pratson, Adams & Gould, 2023). For example, it is included in the notion of contributions of nature to people (NCP) (IPBES, 2022). Verones et al. (2017)' framework allows to acknowledge this plurality of values while conceptually separating the different types of values.

To develop AoP accounting for different types of value, it is important to keep this plurality of values in mind when developing characterization models. As highlighted by Azzopardi et al. (2023) "Intrinsic, instrumental and relational values are not mutually exclusive ; something can be important for its own sake, for its benefits and for the relationships people have with it that are more than merely instrumental". For example, a lake has intrinsic value, independent of human use or perception. At the same time, it can provide instrumental value, such as serving as a recreational site for swimming (a cultural ES, having instrumental value). In addition, individuals may develop a relational value with the lake, such as a deep, non-substitutable emotional connection to it from having grown up nearby.

In this context, when an entity has intrinsic, instrumental, and/or relational values and is subject to environmental pressures, parallel cause-effect chains should be considered to assess potential impacts on each type of value. However, the modeling and evaluation of these impacts must remain independent for each value type. It is therefore essential to ensure that the valuation method used for one category of value (e.g., monetary valuation for instrumental value) does not inadvertently encompass other types (e.g., relational value) to avoid double counting issues.

On the types of systems

Verones and colleagues' proposal conceptually separates natural and human (made) systems.

This separation seems quite straightforward when it comes to associating the AoP 'human health' with a human-made and human-made system and 'ecosystem quality'

with a natural system. However, the relevance of this separation could be discussed.

As described in 4.3, defining a clear boundary between natural and human systems is not straightforward given the extent of nature's anthropization (Jones et al., 2018 ; J. Liu et al., 2021 ; Plumptre et al., 2021). The close interconnection between these systems has led to the concept of social-ecological systems (SES) (Berkes, Folke & Colding, 2000). The idea of SES reflects the change in sustainability debates from a dichotomic understanding of human-nature relationships (humans and nature as opposed entities) to a dualistic view (humans and nature as two entities that form one whole system) (West et al., 2020). This vision still draws a conceptual dividing line between humans and nature, but recognizes their close interactions (West et al., 2020).

Agriculture can illustrate this complexity (Barot et al., 2017). Agricultural systems, which provide food for human societies, depend on natural processes such as water availability and soil fertility. However, these processes are also heavily influenced by human interventions, including irrigation, fertilization, and land management. As a result, agricultural systems are co-produced through the interaction of natural and human systems. The degree of human influence varies along a continuum : from wild blueberry harvesting to hydroponic greenhouse production, passing through organic and industrial farming, human involvement in supporting an agricultural system and its production differs substantially. This raises the question of the nature of the agricultural system : Should it be considered as a human or a natural system ? Can there be a unique answer to this question, given the diversity of agricultural systems and their varying levels of anthropization ? The absence of categorical answers to these questions reveals the shortcomings of Verones and colleagues' human/natural system categorization.

Instrumental values, which Verones et al. (2017) propose to separate into social-economic assets on one hand, and natural resources and SE on the other, are arguably also the result of interactions between human and natural systems. ES, resulting from the combination of ecological functions and human inputs (Burkhard et al., 2014), also exist at the interface between human and natural systems.

Similarly, for relational values, a separation between cultural and natural heritage is proposed. However, since relational values inherently reflect the connections between humans and nature, a strict dichotomy between the relational values of natural and human systems appears questionable.

On intrinsic values

On the quantifiability of intrinsic values In Verones et al. (2017)'s framework, the AoPs 'Human Health' and 'Ecosystem Quality' cover the intrinsic value of human systems and natural systems, respectively. Since AoP aggregate impact results, Verones et al. (2017)' proposal presupposes that it is possible to quantify impacts on intrinsic values. However, the concept of intrinsic value and its measurement by humans is a topic of ongoing debate among experts (Batavia & Nelson, 2017 ; Hargrove, 1992). The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) defines 'intrinsic values' as 'the value inherent to nature, independent of human experience and evaluation, and thus beyond the scope of anthropocentric valuation approaches' (Díaz et al., 2015, 2018). The latter part of the definition (i.e., 'beyond the scope of anthropocentric valuation approaches') suggests that intrinsic values cannot be quantified. In this respect, the development of impact scores to assess quantified impacts on intrinsic values is subject to debate.

On the premise that the human health AoP covers an intrinsic value. To our knowledge, there is no scientific consensus that human health represents the intrinsic value of human and human-made systems.

If we can admit that human health has an intrinsic value, we can also recognize that it has an instrumental value, as it enables individuals to contribute to society, participate in economic activities, and pursue personal and collective well-being. Furthermore, human health has a relational value, as it influences social cohesion, cultural practices, and intergenerational well-being.

In this context, the intrinsic dimension of human health can be seen as one of the components of the intrinsic value of human systems, for which a more in-depth

definition could be provided.

The present work does not aim to propose an alternative to Verones' classification. However, based on the limitations of the framework we have identified here, we suggest further reflection in future research.

SUB-APPENDIX B - Classification of ES-LCA integration efforts

Building on the integration classifications of ES-LCA proposed by De Luca Peña et al. (2022) and Rugani et al. (2023), we developed a third holistic classification system. We invite researchers to further discuss and enhance this classification. We identified 4 main approaches to integrate ES in LCA, where, different "LCA stakeholders" can be involved :

- **the LCA practioner** is responsible for conducting the entire LCA study, following the guidelines established by the ISO 14044 standard. Their tasks include defining the goal and scope of the study, building the LCI by collecting both foreground and background data, selecting and applying the LCIA characterization methods, and interpreting the results. In the process of building the inventory, they may need to integrate ES models to capture specific impacts in the foreground system.
- **the LCI database developer :** focuses on the development and maintenance of LCI databases, which provide the foundational data for LCA studies (e.g., the ecoinvent database).
- **the LCIA method developer :** is responsible for developing and refining LCIA methods, used to translate inventory data into potential environmental impacts across different impact categories. Their work involves designing impact pathways and developing characterization models and factors. In relation to ES, their tasks might include developing characterization models and factors to be included in LCIA methodologies and assessing potential impacts on ecosystems' capacity to provide services.

In our revised classification, ES concepts can be apprehended in LCA through :

— **the functional unit.**

The function of the system studied can be fulfilled through an ES. This could be a provisioning ES (e.g., *"providing 1 m^3 of water"*), regulating and maintenance ES (e.g., *"protecting 10 km of coastline"*), or cultural ES (e.g., *"swimming in a lake"*). In this approach, it is the LCA practitioner, conducting the study, who is responsible for identifying the relevant ES for their product system. Life cycle models of Nature-based solutions (NbS), such as green roofs for climate adaptation in cities, are also a prominent example of this category. By definition, NbS are systems that produce ES that can be apprehended through the functional unit of a LCA (Larrey-lassalle, Perfido & Naneci, 2022).

ES can lead to challenges of multifunctionality, as illustrated in Boone et al. (2019) et Bragaglio et al. (2018), who compare organic and conventional farming systems. In this study, the authors perform an economic allocation to address the multi-functionality issue presented by agricultural production, which simultaneously produces food (provisioning ES) and delivers regulating and maintenance ES, such as soil quality regulation, disease control, pollination, or erosion control.

— **the inventory**. Links can be established between life cycle inventory (LCI) construction and ES. These links may involve the LCA practitioner and/or the life cycle inventory database developer.
  — Certain flows usually considered in the life cycle inventory can be explicitly understood as ES. For example, a water input can be considered as a provisioning ES flow. The quantity of such LCI flows (elementary flows) is estimated by the LCA practitioner. To quantify these flows, the LCA practitioner can turn to knowledge on ES, for example, to consider the influence of regulation and maintenance ES.
  — Certain ES are already taken into account by LCI database developers when developing datasets. For example, land use change (e.g. the ecoinvent dataset *"land use change, annual crop"*) includes impacts related to soil

management practices in terms of soil organic carbon changes. This modeling implicitly reflects a change in the provision of regulating and maintenance ES between different types of land use. Similarly, LCI datasets such as those for "crop systems" or "forest wood extraction" account for CO_2 captured from the atmosphere and stored as biogenic carbon input. Although these carbon flows represent ES, they are not explicitly identified as such in the datasets.

— **the impact assessment.** The concept of ES can be understood through the lens of the impact assessment. However, in this regard, two schools of thought can be considered. The reader is invited to refer to the body of the manuscript for further discussion on this topic.

— **a combination of LCA and ES-specific tools used in parallel.** Finally, to leverage the advantages provided by both LCA and ES methodologies, some authors suggest performing both types of analyses in parallel and interpreting the results by combining or comparing them.

This new classification system was used to review the 43 studies we identified in the literature which propose to integrate ES into LCA (see ESM1).

Electronic Supplementary Material

The following supplementary information file presents a mapping of propositions for the integration or use of ES concepts with(in) LCA at different LCA steps (as per ISO 14040-44) i.e. goal and scope, life cycle inventory, and life cycle impact assessment (LCIA).

A "Y" in the table indicates that the corresponding approach (in rows) is employed for a given proposition (in columns). This table helps understand the state of the art of propositions of frameworks or methods to integrate or combine ES indicators and methods with LCI and LCIA.

Classification by De Luca Pena (2022)	Classification in this work	Description	LCA stakeholder	Singh and Bakshi (2009)	Müller-Wenk & Brandão (2010)	Zhang et al. (2010a)	Zhang et al. (2010b)	Viglia et al. (2013)	Rugani et al. (2013)	Schaubroeck et al. (2013)	Koellner et al. (2013)	Brandao and Mila i Canals (2013)	Saad et al. (2013)	Núñez et al. (2013)
				Describe the eco-LCA tool based on IO	CRP CF ; change in soil carbon sequestration	Review for ES-LCA: existing methods are inadequate to emncompass all ES so thermodynamics should be explored + encourage developing integrated models (TES)	Eco-LCA : ecologically-based LCA : integrates LCA with other methods to include ES	Integrate LCA and ES conducted independently	Emergy for ES-LCIA	Framework to perform an LCA on an integrated T-ES	Guidelines for land use LCIA models (SETAC\UNEP)	BPP : impact on ecosystems" capacity to produce biomass	FRP and WPP : impact on water quantity and quality ? ERP	Global spatially resolved method to assess land occupations impacts on the erosion regulation.
LCA as "driving method"	In the Functional Unit	The functional unit is an ES	LCA practitioner											
		ES engender multi-functionality issues	LCA practitioner											
	In the inventory	An ES concept is included in the inventory	LCA practitioner/LCI database developer	Y		Y	Y			Y				Y
	In the assessment phase	ES mitigate environmental impacts	LCA practitioner											
			LCIA method developer											
		Impacts on the capacity of ecosystems to provide ES are adressed: stress	LCA practitioner											
			LCIA method developer						Y					
		Impacts on the capacity of ecosystems to provide ES are addressed: transformation	LCA practitioner											
			LCIA method developer		Y						Y	Y	Y	Y
"post-analysis" : LCA and ES performed in parrallel	ES and LCA are both used	ES and LCA results are interpreted independently	LCA practitioner					Y						
"combination of results" : damage-oriented results are aggregated		ES and LCA damage-oriented results are interpreted conjointly	LCA practitioner											

Classification by De Luca Pena (2022)	Classification in this work	Description	LCA stakeholder	Muñoz et al. (2014) LCA of bio-based ethanol	Xue et al. (2014) LCA of carbon footprint of tillage systems (converted into a "cost")	Arbault et al. (2014) Integrated earth system dynamic modeling (GUMBO) to derive CFs by applying resource extraction diracs	Cao et al. (2015) ECF to convert midpoint to endpoint indicators	Bakshi et al. (2015) TES-LCA	Bruel et al. (2016) Measures envionrmental externalities	Bos et al. (2016) LANCA model	Chaplin-Kramer et al. (2017) use predictive land-change modelling and inVEST to calculate key elements of LCI	van Zelm et al. (2018) Development of CF for soil erosion due to crop production using the EPIC biophysical crop model	Jeswani et al. (2018) LCA of breakfast cereals (land use impacts using LULCIA CFs)	Liu et al. (2018a) Part I
LCA as "driving method"	In the Functional Unit	The functional unit is an ES	LCA practitioner											
		ES engender multi-functionality issues	LCA practitioner											
	In the inventory	An ES concept is included in the inventory	LCA practitioner/LCI database developer	Y				Y			Y			Y
	In the assessment phase	ES mitigate environmental impacts	LCA practitioner											
			LCIA method developer											
		Impacts on the capacity of ecosystems to provide ES are adressed: stress	LCA practitioner					Y	Y					Y
			LCIA method developer			Y						Y		Y
		Impacts on the capacity of ecosystems to provide ES are addressed: transformation	LCA practitioner											
			LCIA method developer	Y			Y			Y			Y	
"post-analysis" : LCA and ES performed in parrallel	ES and LCA are both used	ES and LCA results are interpreted independently	LCA practitioner											
"combination of results" : damage-oriented results are aggregated		ES and LCA damage-oriented results are interpreted conjointly	LCA practitioner		Y									

Classification by De Luca Pena (2022)	Classification in this work	Description	LCA stakeholder	Liu et al. (2018b) Part II	Liu et al. (2018c) ES encouraging TES Synergies	Pavan and Ometto (2018) conceptual framework for land use impacts on soil ES building on the cascade model (ES are linked to soil functions)	Blanco et al. (2018) Framework to include ES as a midpoint category	Rugani et al (2019) Cascade model to include ES in LCA	Boone et al. (2019) LCA of conventional vs. organic farming. Using allocation procedure to divide potential impacts between crop and ES production	Othoniel et al. (2019) Use of spatial and dynamic integrated model (combining CLUE, inVEST, BIOME-BGC, MAES) of land use to derive CFs	Liu et al. (2019) TES-LCA framework on urban farms	Liu et al (2020) Application of the cascade framework to rice farming systems	Briones-Hidrovo et al. (2020) methodological approach combining ES and LCA to assess hydropower schemes	Bragaglio et al. (2020) compare 4 beefs production systems using economic allocation for the potential ES supply
LCA as "driving method"	In the Functional Unit	The functional unit is an ES	LCA practitioner					Y						
		ES engender multi-functionality issues	LCA practitioner						Y					Y
	In the inventory	An ES concept is included in the inventory	LCA practitioner/LCI database developer	Y	Y		Y	Y			Y	Y		
	In the assessment phase	ES mitigate environmental impacts	LCA practitioner											
			LCIA method developer											
		Impacts on the capacity of ecosystems to provide ES are adressed: stress	LCA practitioner	Y	Y						Y	Y		
			LCIA method developer			Y		Y			Y			
		Impacts on the capacity of ecosystems to provide ES are addressed: transformation	LCA practitioner											
			LCIA method developer				Y	Y		Y				
"post-analysis" : LCA and ES performed in parrallel	ES and LCA are both used	ES and LCA results are interpreted independently	LCA practitioner											
"combination of results" : damage-oriented results are aggregated		ES and LCA damage-oriented results are interpreted conjointly	LCA practitioner					Y					Y	

Classification by De Luca Pena (2022)	Classification in this work	Description	LCA stakeholder	Larrey-Lassalle et al. (2022)	Hardaker et al. (2022) framework for endpoint ES indicators combining indicator of ES use for mitigation and potential impact on ES supply	Rugani et al. (2022) (section 4) Proposition to intertwine LCA and ES for assessing NBS	Taelman et al. (2023) ESA-LCA	Taelman et al. (2023) LCA_+ES	Babí Almenar et al. (2023) develop a modelling framework for nature-based solutions into two parts : foreground (dynamic modelling) and background (static, relies on LCA)	Alsheri et al. 2023 modify Rugani et al. 2023 to the soil remediation and brownfield redevelopment context	Lago-olveira et al. (2024) propose a methodological framework for developing spatially explicit midpoint CFS to assess the impact and benefits of diffferent agriculturals practices on the supply of ES. Build on the cascade model, define new land use inventory flows and use inVEST
LCA as "driving method"	In the Functional Unit	The functional unit is an ES	LCA practitioner	Y							
		ES engender multi-functionality issues	LCA practitioner	Y							
	In the inventory	An ES concept is included in the inventory	LCA practitioner/LCI database developer	Y		Y				Y	Y
	In the assessment phase	ES mitigate environmental impacts	LCA practitioner	Y							
			LCIA method developer		Y						
		Impacts on the capacity of ecosystems to provide ES are adressed: stress	LCA practitioner			Y					
			LCIA method developer		Y						
		Impacts on the capacity of ecosystems to provide ES are addressed: transformation	LCA practitioner			Y				Y	
			LCIA method developer		Y			Y			Y
"post-analysis" : LCA and ES performed in parrallel	ES and LCA are both used	ES and LCA results are interpreted independently	LCA practitioner								
"combination of results" : damage-oriented results are aggregated		ES and LCA damage-oriented results are interpreted conjointly	LCA practitioner				Y		Y		

Classification by De Luca Pena (2022)	Classification in this work	Description	LCA stakeholder	Oginah et al. (2023) LCIA framework linking toxic substances emissions to the environment to damages on AoP "Ecosystem services"	Li et al. (2023) LCIA framework linking micro- and nano-plastics emissions to the environment to damages on AoP "Ecosystem services"	Martínez-García et al. (2025) Using ES and LCA to assess ES and EDS on agroecosystems
LCA as "driving method"	**In the Functional Unit**	The functional unit is an ES	LCA practitioner			
		ES engender multi-functionality issues	LCA practitioner			
	In the inventory	An ES concept is included in the inventory	LCA practitioner/LCI database developer			
	In the assessment phase	ES mitigate environmental impacts	LCA practitioner			
			LCIA method developer			
		Impacts on the capacity of ecosystems to provide ES are adressed: stress	LCA practitioner			
			LCIA method developer	Y	Y	
		Impacts on the capacity of ecosystems to provide ES are addressed: transformation	LCA practitioner			
			LCIA method developer			
"post-analysis" : LCA and ES performed in parrallel	**ES and LCA are both used**	ES and LCA results are interpreted independently	LCA practitioner			
"combination of results" : damage-oriented results are aggregated		ES and LCA damage-oriented results are interpreted conjointly	LCA practitioner			Y

Short version	Year of Publication	Citation
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Müller-Wenk & Brandão (2010)	2010	Müller-Wenk R, Brandão M (2010) Climatic impact of land use in LCA-carbon transfers between vegetation/soil and air. Int J Life Cycle Assess 15:172–182. https://doi.org/10.1007/s11367-009-0144-y
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ANNEXE B – SUPPLEMENTARY INFORMATION : REVAMPING THE LCIA FRAMEWORK THROUGH THE LENS OF ECOSYSTEM SERVICES

Ce matériel supplémentaire se réfère à l'article présenté au chapitre 5.

Revision of the computational structure of LCA to characterize some intermediate flows

As mentioned in the manuscript, some potential impacts are arguably omitted by restricting the characterization phase to elementary flows (at the boundary between the ecosphere and the technosphere) only. A prominent example are irrecoverable alloying elements, such as copper in aluminum 1000 series alloy used in vehicles. Copper cannot be recovered to be used in another product and is therefore inaccessible to other users while remaining in the technosphere. Since it represents a form of loss of functionality, this impact could be aggregated within the AoP Ecosystem and resources Service (currently 'Resources and ecosystem services'). To do so, the flow of copper in aluminum should be characterized. Therefore, we argue for accounting for the potential impacts of some intermediate flows.

With the current LCA computation structure, characterization of intermediate flows is currently not possible. In a process-based LCA, the current calculation of the impact score s of a product system assessment that answers a final demand f is calculated as per equation 1, proposed by Heijungs et Sun (2002) :

$$s = Q \cdot B \cdot A^{-1} \cdot f \tag{1}$$

where : Q is the characterization matrix of size [t, m] with t the number of impact categories and m the number of elementary flows, B is the environmental matrix (also called the intervention matrix) of size [m, n], n the number of intermediate flows, A^{-1} is the technology matrix of size [n, n] and f is the final demand vector of size [n, 1].

Multiplication of matrix B by A^{-1} and the vector of final demand f results in a vector of elementary flows $\dot{g}$ of size [m,1] and scaled to the functional unit (equation 2).

$$\dot{g} = B \cdot A^{-1} \cdot f \tag{2}$$

Therefore the impact score s can be rewritten as per equation 3 :

$$s = Q \cdot \dot{g} \tag{3}$$

The Q characterization matrix has t rows and m columns, to characterize the m elementary flows for each impact category. It is inadequate to characterize additional flows.

Heijungs (2024) highlighted this computational limit, which does not allow to capture the potential impact of so-called “activity variables”, defined as “a measure of process activity which can be related to process output” (UNEP/SETAC, 2020) such as kilowatt-hour of electricity which might induce potential bird mortality due to collisions on power lines. To solve this shortcoming, the author suggested building a new characterization matrix (or “intensity matrix”) for impacts categories that work on activity variables. At this stage, it is not clear whether this approach could be used for other types of entities than activity variables, such as intermediate flows. As per activity variables, the current computational structure is inadequate to take into account the potential impacts due to intermediate flows. We also propose a possible way to solve this issue, which would also require developing an additional characterization matrix. This new matrix would have t rows and n columns, n being the number of intermediate flows in the A matrix. To enable this calculation, it is key that (i) A is a square matrix where (ii) rows and columns of matrix A match (e.g., the unit process ’production of hydroelectricity’ is in column 1 and the intermediate flow ’1 kWh of hydroelectricity’ is in row 1) and (iii) the intermediate flows are scaled at the unit process (e.g., 1 kWh).

Providing these conditions, we suggest to build a new characterization matrix designed for intermediate flows of size [t, n] named Q^*. This matrix contains characterization (or other term to define) factors of the n intermediate flows for each impact category. Then, multiplying Q^* by the inverse of A and by the vector of final demand f, we obtain the potential impact of intermediate flows for the system of the product under study.

The new way of calculating LCIA impact could be following equation 4 :

$$s = Q \cdot \dot{g} + Q^* \cdot A^{-1} \cdot f \tag{4}$$

The following paragraph illustrates the proposition with the example of electricity generation from offshore wind turbine. To distribute the produced electricity, high-voltage grid lines are used.

The functional unit is 1 kWh of electricity produced from offshore wind turbines and transported through the grid. The technology matrix A of the system is defined as per equation 5 :

$$\mathbf{A} = \begin{bmatrix} 1 & 0 \\ -1 & 1 \end{bmatrix} \tag{5}$$

Its inverse A^{-1} is as per equation 6 :

$$A^{-1} = \begin{bmatrix} 1 & 0 \\ 1 & 1 \end{bmatrix} \tag{6}$$

The vector of final demand f of the system is as per equation 7 :

$$f = \begin{bmatrix} 1 \\ 0 \end{bmatrix} \tag{7}$$

We here suggest considering the characterization factor proposed by Gilad et al.

(2024) to assess the potential damage-oriented impact of the transmission line activity variable due to bird death by collision. Latter factor equals 3.9E-15 $PDF \cdot yr/kWh$. Assuming that this characterization factor is the only characterization factor that assesses activity variables or dissipation processes in the system, Q^* is defined as per equation 8 :

$$Q^* = \begin{bmatrix} 0 & 0 \\ 3.9E-15 & 0 \end{bmatrix} \tag{8}$$

Regarding the potential impact of elementary flows, we need to define the matrices B and Q.

We consider a simplified environmental matrix B is as per equation 9 :

$$B = \begin{bmatrix} 0 & 0 \\ 2 & 0 \\ 0.1 & 0.4 \end{bmatrix} \tag{9}$$

And we have the characterization matrix Q as per equation 10 :

$$Q = \begin{bmatrix} 0 & 0 & 0 \\ 1E-15 & 2E-15 & 3E-15 \end{bmatrix} \tag{10}$$

Then the impact score s can be calculated as follows :

$$s = Q \cdot B \cdot A^{-1} \cdot f + Q^* \cdot A^{-1} \cdot f \tag{11}$$

which gives under a matrix formulation :

$$s = \begin{bmatrix} 0 & 0 & 0 \\ 1E-15 & 2E-15 & 3E-15 \end{bmatrix} \cdot \begin{bmatrix} 0 & 0 \\ 2 & 0 \\ 0.1 & 0.4 \end{bmatrix} \cdot \begin{bmatrix} 1 & 0 \\ 1 & 1 \end{bmatrix} \cdot \begin{bmatrix} 1 \\ 0 \end{bmatrix} + \begin{bmatrix} 0 & 0 \\ 3.9E-15 & 0 \end{bmatrix} \cdot \begin{bmatrix} 1 & 0 \\ 1 & 1 \end{bmatrix} \cdot \begin{bmatrix} 1 \\ 0 \end{bmatrix} \tag{12}$$

which is equivalent to :

$$s = \begin{bmatrix} 0 & 0 \\ 4.3E-15 & 1.2E-16 \end{bmatrix} \cdot \begin{bmatrix} 1 & 0 \\ 1 & 1 \end{bmatrix} \cdot \begin{bmatrix} 1 \\ 0 \end{bmatrix} + \begin{bmatrix} 0 & 0 \\ 3.9E-15 & 0 \end{bmatrix} \cdot \begin{bmatrix} 1 & 0 \\ 1 & 1 \end{bmatrix} \cdot \begin{bmatrix} 1 \\ 0 \end{bmatrix} \tag{13}$$

which is equivalent to :

$$s = 4.4E-15 \ + \ 3.9E-15 = 8.3E-15 \ PDF \cdot yr \tag{14}$$

Application to ecosystem disservices

The revamped framework proposed in this work can be extended to account for ecosystem disservices (EDS) in parallel to ES. The concept of "ecosystem service supply" is complemented by the notion of "EDS supply" representing the potential for ecosystems to generate disservices. Similarly, the "ES flow" concept is mirrored by the "EDS flow", which refers to the portion of the EDS potential to which populations are effectively exposed to. Unlike ES flows, which meet a desired demand, EDS flows represent detrimental exposures. Therefore, the corresponding impact pathways do not involve a "demand" for EDS, but rather an "exposure" to EDS flows. In response to changes in EDS flows, populations can implement adaptation strategies to mitigate negative impacts, similar to the adaptation mechanisms observed for changes in ES flows.

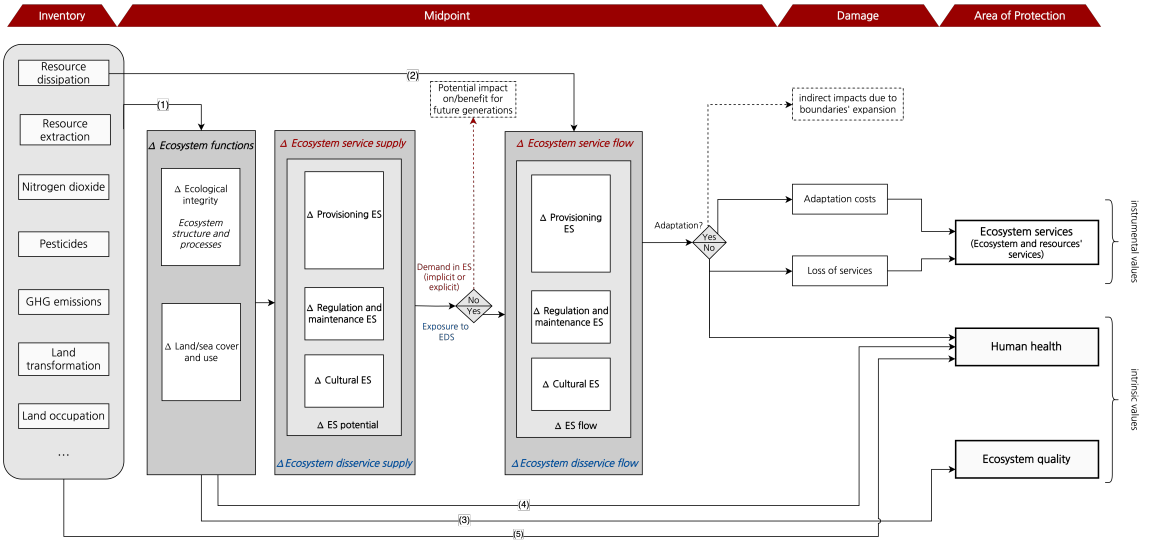


FIGURE B.1 Extended revamped LCIA framework for considering ecosystem disservices (EDS) and related impact pathways.

However, as underlined in the body of the manuscript, the integration of EDS into the proposed framework requires a comprehensive, complementary, and nonoverlapping classification of both ES and EDS. Currently, those criteria are not met by the last version of the Common International Classification of Ecosystem Services (CICES 5.2).

Valuation methods

Section 5.2.2 suggests the use of economic valuation methods to assess potential damages due to a change in ES flow. However, these methods face several theoretical and practical limitations (J.-m. Salles, 2011 ; Tinch et al., 2019).

Economic approaches to ecosystem valuation are grounded in a neoliberal logic, understanding nature as a provider of goods, services, and 'natural capital' in the context of capitalist frameworks (Spash & Aslaksen, 2015). In this context, critics have pointed out the risks of reducing nature to a mere means to an end based on

human preferences (Gould et al., 2024 ; Muradian & Gómez-Baggethun, 2021 ; Spash & Aslaksen, 2015), and the risks of commodification of nature, referring to the transformation of natural resources and ES into marketable goods and services (Kosoy & Corbera, 2010).

As mentioned above, humans form and express values throughout different value dimensions, which go beyond what is generally captured through economic models (IPBES, 2022 ; Spash, 2000). Mainstream economic valuation can only capture a limited range of values, namely those based on human utility (Gould et al., 2024). The use of economic valuation that reduces nature to a mere means to an end therefore undermines the ethical and moral dimensions of conservation (Batavia & Nelson, 2017). It may enhance attitudes of dominance and exploitation (Muraca, 2016 ; Spash & Aslaksen, 2015) rather than respectful human-nature relationships (Comberti et al., 2015). In parallel, it risks marginalizing stakeholders holding noneconomic perspectives, particularly indigenous and local communities (Chan et al., 2012).

Such critics have, however, little applicability to the revised LCIA framework proposed in this study. In the latter, economic valuation is used to assess potential impacts on instrumental values, operating alongside separate assessments for intrinsic—and potentially relational—values. Other value dimensions are explicitly considered within distinct AoP, ensuring a more comprehensive evaluation framework.

In response to concerns about the commodification of nature, researchers have emphasized that, rather than treating ES as market commodities, the economic valuation of their instrumental value aims to highlight and enhance awareness of their critical role in society (L. M. Brander et al., 2024). This study adopts a similar perspective.

Beyond ethical concerns, economic valuation methods face methodological challenges.

The expression of ES value in economic terms is based on the assumption that individual consumer preferences are substitutable and commensurable, allowing them to

be aggregated for comparison (Spash & Hache, 2022). However, these methodological assumptions strongly simplify consumer responses and their context, treating individuals as homogeneous, passive, asocial, and ahistorical agents (Gould et al., 2024 ; Spash, 2011). Moreover, treating individual preferences as commensurable metrics overlooks the complex ways in which values are socially constructed (Spash, 2015). As a result, market-based and contingent valuation methods often fail to account for social dynamics, political contexts, power structures, and systemic inequalities that shape individual preferences (Spash, 2000 ; Spash & Hache, 2022). In practice, this leads to an inappropriate transfer of monetary values between different socio-economic and environmental contexts, compromising their reliability and applicability (Gould et al., 2024).

Moreover, market-based assumptions, such as substitutability and marginality, often fail to apply to ES. Unlike conventional goods, many ES cannot be easily replaced or exchanged ; for example, the loss of a keystone species or the collapse of an ecosystem cannot be offset by technological substitutes, as their ecological functions are often irreplaceable. Gómez-Baggethun et al. (2010) caution that market-based approaches can foster a false sense of security, potentially leading to underinvestment in conservation efforts.

Furthermore, economic valuation tends to favor material ES, as they are more readily quantifiable, while nonmaterial ES, such as spiritual and symbolic interactions with nature, the creation of traditional ecological knowledge, and cultural heritage, are more challenging to assess using monetary metrics. As a result, these 'hard-to-evaluate' services are often excluded from economic evaluations, reducing their visibility and leading to a distorted representation of the importance people place on nature (IPBES, 2019).

Economic valuation practices are often subject to procedural justice concerns, as responses and behaviors that do not align with economic models are often dismissed as 'irrational' or 'inexplicable' and therefore excluded from decision-making processes (Spash, 2000). This exclusion systematically silences diverse perspectives, particularly those of historically marginalized groups, including 'women under patriarchy,

indigenous communities under colonialism, the uneducated in productivist societies, and the financially poor under capitalism' (Spash, 2022).

Beyond these social justice concerns, scholars also highlight practical challenges in the economic valuation of ES, particularly the plurality of valuation methods and the uncertainties they entail. Various methods, such as consumer surplus, avoided costs, and exchange value approaches, are commonly used, but may produce inconsistent results and are often not directly comparable (L. M. Brander et al., 2024 ; L. Brander et al., 2022). Moreover, economic valuation methods are inherently subject to uncertainty, which can arise from the structure of the model ('structural uncertainty') or from the parameters used in the calculations ('parametric uncertainty') (Boithias et al., 2016).

In the light of these critiques, we acknowledge that economic valuation of ES should be applied with a clear awareness of its limitations, and to represent potential impacts on instrumental values complementing indicators on other value dimensions.

Electronic Supplementary Material

The following supplementary information file presents the analysis of the characterisation models used in the IMPACTWorld + LCIA methodology (Bulle et al. 2019 https ://doi.org/10.1007/s11367-019-01583-0) in the light of the revamped LCIA framework proposed in the manuscript.

The first table provides a description of the cause effect chains considered in the IMPACTWorld+ impact categories, and how they can be positioned within the revamped LCIA framework.

The second table provides an analysis of the coverage of impact categories by IMPACTWorld+ characterisation models and identifies research gaps and priorities.

Impact category (from IMPACT World, Bulle et al., 2019)	AoP	Re-commanded R or Interim I	Covered impacts	Description of mechanisms	Pathway in framework	Ecosystem service involved
Climate change	HH	R	Cardiovascular diseases	Climate change, by increasing the frequency and intensity of heatwaves (Δecosystem structure) increases risks of cardiovacular diseases (HH AoP)	0-4	NA
			Malaria	Climate change, by enlarging the distribution areas of mosquitos due to increasing temperatures (Δecosystem structure), increases the transmission of malaria (c) (HH AoP)	0-4	NA
			Diarrhoea	Climate change, by altering precipitations patterns (Δecosystem processes), decreases water availability (Δprovisioning ES). Without adaptation measures (c), this can lead to reduced hygiene practices and to the consumption of inadequate water sources, and result in diarrheoal diseases (HH AoP)	0-1-c	Provisioning of water
			Floods	Climate change, by increasing the frequency and intensity of extreme weather events (Δecosystem structure), causes more frequent and severe floods. This can lead to displacement of communities and contamination of water sources, and result in waterborne diseases and injuries (HH AoP)	0-4	NA
			Malnutrition	Climate change, by increasing temperatures and altering precipitation patterns (Δecosystem processes), reduces crop yields (Δprovisioning ES). Without adaptation measures (c), this can lead to food scarcity and poor nutritional quality, and result in malnutrition (HH AoP)	0-1-c	Provisioning of food
Climate change	EQ	R	Species disappearance	Climate change, by altering species distribution areas due to increasing temperatures (Δecosystem structure), can lead to the extinction of species (EQ AoP)	0-3	NA
Marine Acidification	EQ	R	Population affected	Marine acidification, linked to a decrease of marine pH (Δecosystem structure), can lead to the potential disappearance of calcifying species (EQ AoP).	0-3	NA
Ozone Layer Depletion	HH	R	Immune suppression; skin cancer; cataract	Ozone depletion, by decreasing the capacity of the ozone layer to absorb and scatter sun's ultraviolet radiation (Δecosystem structure), increases human exposure to these radiations. Without adaptation measures (c), this can compromise immune system function and increases vulnerability to skin cancer or cataract (HH AoP)	0-1-c	Protection from Sun's ultraviolet radiations
Freshwater acidification	EQ	R	Species affected by decrease of pH	Freshwater acidification, linked to a decrease in freshwater pH (Δecosystem structure), can lead to the potential disappearance of fish species (EQ AoP).	0-3	NA
Terrestrial acidification	EQ	R	Species affected by decrease of pH	Terrestrial acidification results from the atmospheric deposition of emitted pollutants (e.g nitrogen oxides (NOx), ammonia (NH3) and sulfur dioxide (SO2)), and their subsequent deposition, which can induce a decrease the soil's solution pH (Δecosystem structure). This decrease of pH can lead to the disappearance of vascular plant species (EQ AoP).	0-3	NA
Freshwater eutrophication	EQ	R	Species affected by decreased levels of oxygen	Freshwater eutrophication, linked to an increase in limiting nutrient levels, can cause algal blooms, which subsequent mortality can deplete oxygen levels (Δecosystem structure). This can lead to the potential disappearance of aquatic species (EQ AoP).	0-3	NA
Marine eutrophication	EQ	R	Species affected by decreased levels of oxygen	Marine eutrophication, linked to an increase in limiting nutrient levels, can cause algal blooms, which subsequent mortality can deplete oxygen levels (Δecosystem structure). This can lead to the potential disappearance of marine species (EQ AoP).	0-3	NA
Freshwater ecotoxicity	EQ	R	Freshwater species affected by chemical exposure	Freshwater ecotoxicity, linked to the introduction of toxic substances and the increase of their concentration in the environment (Δecosystem structure), can lead to the potential disappearance of sensitive freshwater species (EQ AoP).	0-3	NA
			from outdoor emissions	Outdoor emissions of toxic substances (e.g. factories, car combustion, agriculture) can alter the outdoor air composition by introducing hazardous pollutants into the atmosphere (Δecosystem structure). Pollutants can jeopardize human health, increasing risks of cancers and non-cancer diseases (HH AoP)	0-4	NA
Human toxicity	HH	R	from indoor emissions	Indoor emissions of toxic substances (e.g., cleaning agents, tobacco smoke, building materials) can alter indoor air composition by introducing hazardous pollutants into the living environment. Since this indoor environment is part of the technosphere and not the ecosphere, no ecosystem functions change occurs. These toxic substances can directly jeopardize human health, increasing risks of cancers and non-cancer diseases (HH AoP)	5	NA

			from pesticides in crops	Pesticide residues ingested from crops can introduce hazardous chemicals into the human diet. Since this exposure occurs within the food chain (i.e. in the technosphere) rather than the ecosphere, no changes in ecosystem functions are involved. These pesticide residues can directly jeopardize human health, increasing risks of cancers and non-cancer diseases (HH AoP)	5	NA
Terrestrial ecotoxicity	EQ	I	Terrestrial species affected by chemical exposure	Terrestrial ecotoxicity, through the introduction and accumulation of toxic substances (Δecosystem structure), can lead to the potential disappearance of sensitive terrestrial species (EQ AoP).	0-3	NA
Marine ecotoxicity	EQ	I	Marine species affected by chemical exposure	Marine ecotoxicity, through the introduction and accumulation of toxic substances (Δecosystem structure), can lead to the potential disappearance of sensitive marine species (EQ AoP).	0-3	NA
Particulate matter formation	HH	R	respiratory impacts, cardiovascular morbidity, lung cancer and other morbidity	Emission of particulate matter (PM) can alter air composition (Δecosystem structure). Inhalation of PM can cause health issues including chronic respiratory issues, cardiovascular morbidity, lung cancer and other morbidity (HH AoP)	0-4	NA
Photochemical oxidant formation	HH	R	Respiratory diseases	Photochemical oxidant formation, primarily driven by ground-level ozone (O_3) and other reactive compounds produced by the reaction of sunlight with substances like nitrogen oxides (NO_x) and volatile organic compounds (VOCs) (Δecosystem structure) can lead to respiratory problems and cardiovascular effects (HH AoP)	0-4	NA
Ionizing radiation	HH	R	Cancer and severe hereditary effects	Release of radioactive material to the environment can alter the concentration of soil, water and air in radioactive components (Δecosystem structure). Exposure of humans to radioactive elements by inhalation, consumption of contaminated food or water can increase risks of cancers and heredity effects (HH AoP)	0-4	NA
	EQ	R	Freshwater species affected	Ionizing radiations, induced by radioactive releases in freshwater environments (Δecosystem structure), can lead to the potential disapperance of freshwater species (EQ AoP).	0-3	NA
Water availability	HH	R	Malnutrition (agricultural)	Water use, by decreasing local water availability (Δecosystem structure), can increase water competition between users, and lead to shortages for agricultural and fisheries users' (ΔES flow). If they do not adapt agricultural and fisheries users can face water shortages, which can have repercussions on crops and aquaculture productivity, leading to problems of malnutrition (c) (HH AoP). If they adapt (a), the population must bear the economic costs of adaptation (ES AoP, identified but not modeled) and potential impacts due to the system boundaries expansion (e, identified but not modeled).	0-1-c	Provisioning of water
			Water-related diseases (domestic)	Water use, by decreasing local water availability (Δecosystem structure), can increase water competition between users, and lead to shortages for domestic users' (ΔES flow). If they do not adapt, domestic users can face water shortages, which can lead them to have inadequate hygiene pratices or to consume unsafe water sources (c). These behaviors can lead to water-related diseases including diarrheoal diseases (HH AoP). If they adapt (a), the population must bear the economic costs of adaptation (ES AoP, identified but not modeled) and potential impacts due to the system boundaries expansion (a, identified but not modeled).	0-1-c	Provisioning of water
Water availability (terrestrial)	EQ	R	Terrestrial plants affected	Excessive groundwater withdrawal, by lowering of the groundwater level (Δecosystem structure), can cause hydric stress for terrestrial plants. This can lead to the disappearance of latter species (EQ AoP).	0-3	NA
Water availability (freshwater)	EQ	R	Freshwater species disapperance	Water consumption, by changing the water supply to aquatic ecosystems (Δecosystem processes), can intensify pressures on freshwater species and lead to a potential disapperarance of fraction of fish species (EQ AoP).	0-3	NA
Water availability	R&ES	I	Adaptation costs to the loss of water availability	Water use, by decreasing local water availability (Δecosystem structure), can increase water competition between users. Competiting users able to adapt (a) will have to pay for adapting to the decreased availability, facing economic costs (AoP ES) and potential impacts due to the system boundaries expansion (e, identified but not modeled). If no adaptation measures are implemented, decreased water availability can result in potential impacts on HH (c, identified but not modeled) and EQ (d, identified but not modeled).	0-1-a	Provisioning of water
Land Transformation	EQ	R	Species disappearance	Land transformation, particularly the artificialisation of natural and semi-natural ecosystems, by altering the ecosystem states (Δland cover), can lead to species disappearance (EQ AoP).	0-3	NA

Land Occupation	EQ	R	Species disappearance	Land occupation, by altering the ecosystem state (Δland occupation), can lead to species disappearance (EQ AoP).	0-3	NA
Land transformation - mechanical filtration	HH	I	Malnutrition & water-related diseases (water availabity) due to land transformation impact on mechanical filtration	Land transformation, particularly the artificialisation of natural and semi-natural ecosystems, by altering soil mechanical filtration properties (Δecosystem processes), can reduce local water availability (ΔES supply). This can increase water competition between users, and lead to impact pathways similar to the water availability impact category.	0-1-c	Provisioning of water
Land transformation - mechanical filtration	R&ES	I	Adaptation costs to the loss of mechanical water filtration	Land transformation, by altering land cover (Δland cover) can modify the ability of soil to filtrate water through mechanical processes (Δecosystem processes). This can lower water availability and increase water competition between users. Competiting users able to adapt (a) will have to pay for adapting to the decreased availability, facing water purification process economic costs (AoP ES) and potential impacts due to the system boundaries expansion (e, identified but not modeled). If no adaptation measures are implemented, decreased water availability can result in potential impacts on HH (c, identified but not modeled) and EQ (d, identified but not modeled).	0-1-a	mechanical filtration
Land transformation - physico-chemical filtration	HH	I	Malnutrition & water-related diseases (water availabity) due to land transformation impact on physico-chemical filtration	Land transformation, by altering soil physico-chemical filtration properties (Δecosystem processes), can reduce local water availability (ΔES supply). This can increase water competition between users, and lead to impacts pathways similar to the water availability impact category.	0-1-c	Provisioning of water
Land transformation - physico-chemical filtration	R&ES	I	Adaptation cost to the loss of physico-chemical water filtration	Land transformation, particularly the artificialisation of natural and semi-natural ecosystems, by altering land cover (Δland cover) can modify the ability of soil to filtrate water through physico-chemical processes (Δecosystem processes). This can lower water availability and increase water competition between users. Competiting users able to adapt (a) will have to pay for adapting to the decreased availability, facing water purification process economic costs (AoP ES) and potential impacts due to the system boundaries expansion (e, identified but not modeled). If no adaptation measures are implemented, decreased water availability can result in potential impacts on HH (c, identified but not modeled) and EQ (d, identified but not modeled).	0-1-a	Physico-chemical filtration
Land transformation - freshwater recharge	HH	I	Malnutrition, water-related diseases (water availabity) due to land transformation impact on freshwater recharge	Land transformation, by altering freshwater recharge properties (Δecosystem processes), can reduce local water availability (ΔES supply). This can increase water competition between users, and lead to impacts pathways similar to the water availability impact category.	0-1-c	Provisioning of water
Land transformation - freshwater recharge	R&ES	I	Adaptation cost to the loss of freshwater recharge	Land transformation, by altering land cover (Δland cover) can modify soil's freshwater recharge capacity (Δecosystem processes). This can lower water availability and increase water competition between users. Competiting users able to adapt (a) will have to pay for adapting to the decreased availability, facing water supply costs (AoP ES) and potential impacts due to the system boundaries expansion (e, identified but not modeled). If no adaptation measures are implemented, decreased water availability can result in potential impacts on HH (c, identified but not modeled) and EQ (d, identified but not modeled).	0-1-a	freshwater recharge
Land transformation - erosion resistance	R&ES	I	Replacement costs for the loss of erosion control	Land transformation, by altering land cover (Δland cover), can impact soil's erosion resistance (Δecosystem processes). This alteration can leads to reduced vegetation cover, which is critical in anchoring soil and reducing surface runoff. The diminished erosion resistance can result in increased soil erosion rates. Users dependent on stable land for agriculture and infrastructure can invest in replacement erosion control measures (a) and face replacement costs (AoP ES) and potential impacts due to the system boundaries expansion (e, identified but not modeled).. If no adaptation measures are implemented, heightened erosion can lead to impacts on HH (c, identified but not modeled) and EQ (d, identified but not modeled).	0-1-a	Erosion resistance

Land transformation - biotic production	R&ES	I	Damage costs due to the lost of biotic production	Land transformation, by altering land cover (Δland cover), can impact biotic production (Δecosystem processes). This alteration often leads to changes in vegetation cover, which is essential for maintaining soil fertility and supporting diverse biological communities. The reduced biotic production can result in decreased crop yields. If no adaptation measures are implemented, reduced biotic production can lead to damages (b) assessed in terms of loss of yield generated (AoP ES) as well as impacts on HH (c, identified but not modeled) and EQ (d, identified but not modeled).	0-1-b	biotic production
Land occupation - mechanical filtration	HH	I	Malnutrition, water-related diseases (water availabity) due to land occupation impact on mechanical filtration	Land occupation, by altering soil mechanical filtration properties (Δecosystem processes), can reduce local water availability (ΔES supply). This can increase water competition between users, and lead to impacts pathways similar to the water availability impact category.	0-1-c	Provisioning of water
Land occupation - mechanical filtration	R&ES	I	Adaptation costs to the loss of mechanical water filtration	Land occupation, by altering land cover (Δland use) can modify the ability of soil to filtrate water through mechanical processes (Δecosystem processes). This can lower water availability and increase water competition between users. Competiting users able to adapt (a) will have to pay for adapting to the decreased availability, facing water purification process economic costs (AoP ES) and potential impacts due to the system boundaries expansion (e, identified but not modeled). If no adaptation measures are implemented, decreased water availability can result in potential impacts on HH (c, identified but not modeled) and EQ (d, identified but not modeled).	0-1-a	mechanical filtration
Land occupation - physico-chemical filtration	HH	I	Malnutrition, water-related diseases (water availabity) due to land occupation impact on physico-chemical filtration	Land occupation, by altering soil physico-chemical filtration properties (Δecosystem processes), can reduce local water availability (ΔES supply). This can increase water competition between users, and lead to impacts pathways similar to the water availability impact category.	0-1-c	Provisioning of water
Land occupation - physico-chemical filtration	R&ES	I	Adaptation costs to the loss of physico-chemical water filtration	Land occupation, by altering land cover (Δland use) can modify the ability of soil to filtrate water through physico-chemical processes (Δecosystem processes). This can lower water availability and increase water competition between users. Competiting users able to adapt (a) will have to pay for adapting to the decreased availability, facing water purification process economic costs (AoP ES) and potential impacts due to the system boundaries expansion (e, identified but not modeled). If no adaptation measures are implemented, decreased water availability can result in potential impacts on HH (c, identified but not modeled) and EQ (d, identified but not modeled).	0-1-a	physico-chemical filtration
Land occupation - freshwater recharge	HH	I	Malnutrition, water-related diseases (water availabity) due to land occupation impact on freshwater recharge	Land occupation, by altering freshwater recharge properties (Δecosystem processes), can reduce local water availability (ΔES supply). This can increase water competition between users, and lead to impacts pathways similar to the water availability impact category.	0-1-c	Provisioning of water
Land occupation - freshwater recharge	R&ES	I	Adaptation costs to the loss of freshwater recharge	Land occupation, by altering land cover (Δland use) can modify soil's freshwater recharge capacity (Δecosystem processes). This can lower water availability and increase water competition between users. ompetiting users able to adapt (a) will have to pay for adapting to the decreased availability, facing water supply costs (AoP ES) and potential impacts due to the system boundaries expansion (e, identified but not modeled). If no adaptation measures are implemented, decreased water availability can result in potential impacts on HH (c, identified but not modeled) and EQ (d, identified but not modeled).	0-1-a	freshwater recharge
Land occupation - erosion resistance	R&ES	I	Replacement costs for the loss of erosion control	Land occupation, by altering land cover (Δland use), can impact soil's erosion resistance (Δecosystem processes). This alteration can leads to reduced vegetation cover, which is critical in anchoring soil and reducing surface runoff. The diminished erosion resistance can result in increased soil erosion rates. Users dependent on stable land for agriculture and infrastructure can invest in replacement erosion control measures (a) and face replacement costs (AoP ES). If no adaptation measures are implemented, heightened erosion can lead to impacts on HH (c, identified but not modeled) and EQ (d, identified but not modeled).	0-1-a	Erosion resistance

Land occupation - biotic production	R&ES	I	Damage costs due to the lost of biotic production	Land occupation, by altering land cover (Δland use), can impact biotic production (Δecosystem processes). This alteration often leads to changes in vegetation cover, which is essential for maintaining soil fertility and supporting diverse biological communities. The reduced biotic production can result in decreased crop yields. If no adaptation measures are implemented, reduced biotic production can lead to damages (b) assessed in terms of loss of yield generated (AoP ES) as well as impacts on HH (c, identified but not modeled) and EQ (d, identified but not modeled).	0-1-b	biotic production
Mineral ressource use	R&ES	I	Adaptation costs to the loss of functionality of mineral resources	Resource dissipation, by dissipating the flow of service offered by a resource in the technosphere (2) through the scattering and dissipation of materials into the environment in forms that are difficult or impossible to reclaim, prevents recycling and circular economy, thus impeding other users from benefiting from it. This dissipation results in a loss of functionality. Users can adapt to this loss of functionality (a) by, for example, making additional procurement efforts or substituting other, functionally equivalent resources.	2-a	provisioning of mineral resource
Fossil energy use	R&ES	I	Adaptation costs to the loss of functionality of fossil energy	Use of fossil energy, due to the consuming nature of the use, can dissipate a flow of service of energie production (2). Users can adapt to the loss of functionality using a backup technology (a)	2-a	Energy production

Impact category	Area of protection (AoP)	Detailled impact categories (if several models for the same AoP)	Recommended in IW+ (R), Interim in IW+ (I), Absent from IW+ (A)	Pathway	Relevance of including the pathways (expert opinion, only for models absent in IW+) : +: potentially relevant, ++: relevant	Comment on relevance
Climate change	HH	Cardiovascular diseases, malaria,	R	0-4		
		Diarrhoea	R	0-1-c		
		Malnutrition	R	0-1-c		
	EQ		R	0-3		
	R&ES		I	0-1-b		
Acidification	HH		A		+	Acidification primarily affects ecosystems and ecosystem services. Human health impacts might occur indirectly through degraded air quality or contaminated water sources.
	EQ	Marine, terrestrial, freshwater	R	0-3		
	R&ES		A		++	Acidification can disrupt ecosystem services such as water purification, soil fertility, and fish and coral reef populations in aquatic ecosystems (Doney et al. 2020; Doo et al. 2020), reducing the benefits humans derive from these services.
Ozone layer depletion	HH		R	0-1-c		
	EQ		A		++	Increased UV radiation due to ozone depletion can harm terrestrial and aquatic ecosystems, especially by affecting phytoplankton, plants, and animals sensitive to UV exposure, potentially causing disruptions in food chains and leading to species disappeance.
	R&ES		A		++	Increased UV radiation due to ozone depletion can impact ecosystem services by affecting the productivity of agricultural systems, fisheries, and water purification processes.
Mineral resources use	HH		A		+	If populations do not adapt to the loss of functionality of mineral resources, they may be faced with impacts on human health: for example, the loss of functionality of phosphorus can lead to a loss of crop productivity and malnutrition. The intensity of the impact can vary from one mineral resource to another
	EQ		A		-	
	R&ES		I	2-a		
Fossil energy use	HH		A		+	If human population does not adapt to the loss of functionality of fossil, they may be faced with indirect impacts on human health, such as energy shortages for healthcare and essential services, or for agriculture and food supply chains.
	EQ		A		-	
	R&ES		I	2-a		
Eutrophication	HH		A		+	Eutrophication can degrade water quality, leading to harmful algal blooms that may produce toxins harmful to humans.
	EQ	Marine, freshwater	R	0-3		
	R&ES		A		++	Eutrophication can degrade water quality, affect fish stocks, and reduce recreational value, thus impairing services like water purification, fisheries, and tourism.
Toxicity	HH	from outdoor emissions	R	0-4		
		from pesticides in crops, from indoor emissions	R	5		
	EQ	Ecotoxicity (freshwater, terrestrial, marine)	R (freshwater), I (terrestrial, marine)	0-3		
	R&ES		A		++	Toxic substances can impair ecosystem services like water purification, soil fertility, and food production.
Particulate matter formation	HH		R	0-4		
	EQ		A		+	Particulate matter can affect plant growth and damage ecosystems by depositing harmful substances on vegetation and in water bodies.
	R&ES		A		+	PM formation can reduce air quality and visibility, affecting services like tourism and outdoor activities.
Photochemical oxidant formation	HH		R	0-4		
	EQ		A		++	Photochemical oxidant formation can damage plants, reduce crop yields, and harm sensitive ecosystems, leading to biodiversity loss and decreased productivity in agricultural and forest systems.
	R&ES		A		++	Photochemical oxidant formation can reduce crop yields, forest health, and carbon sequestration capabilities, thereby impacting ecosystem services like food production and climate regulation.
Ionizing radiations	HH		R	0-4		
	EQ		R	0-3		

Ionizing radiations	R&ES		A		++	Ionizing radiation can impair ecosystem services by affecting food production (e.g., agriculture, fisheries), water quality, and natural habitat functioning. Contamination of ecosystems can reduce their capacity to support human livelihoods and disrupt services like food provisioning and water purification.
Water availability	HH		R	0-1-c		
	R&ES		I	0-1-a		
	EQ	Freshwater, terrestrial, Thermally polluted water	R	0-3		
Land transformation	HH	Mechanical filtration, Physico-chemical filtration, Freshwater recharge	I	0-1-c		
	EQ	Biodiversity	R	0-3		
	R&ES	Mechanical filtration, Physico-chemical filtration, Freshwater recharge, Erosion resistance	I	0-1-a		
		Biotic production	I	0-1-b		
Land occupation	HH	Mechanical filtration, Physico-chemical filtration, Freshwater recharge	I	0-1-c		
	EQ	Biodiversity	R	0-3		
	R&ES	Mechanical filtration, Physico-chemical filtration, Freshwater recharge, Erosion resistance	I	0-1-a		
		Biotic production	I	0-1-b		

Legend

currently in IMPACTWorld+ (IW+)

not included in IW+, relevance of integration is limited

not included in IW+, relevance of integration is important

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ANNEXE C – SUPPLEMENTARY INFORMATION : CHARACTERAZING SEA-LEVEL RISE POTENTIAL IMPACTS ON COASTAL ECOSYSTEM SERVICES IN LIFE CYCLE IMPACT ASSESSMENT

Ce matériel supplémentaire se réfère au chapitre 6.

Method for calculating the cumulative GHG emissions

This section details the calculation of cumulative greenhouse gas (GHG) emissions for each RCP scenario.

The RCPDatabase (accessible here) provides historical data and future projections of GHG emissions under different RCP scenarios. It includes emissions of various GHGs (CO_2, CH_4, N_2O, HFCs, PFCs, etc.) across multiple sectors such as surface transportation, international shipping, and aviation.

Conversion of CO2 emissions CO_2 emissions are reported in PgC/year (Petagrams of carbon). To express them in kilograms of CO_2, values are multiplied by $3.67 * 10^{12}$.

Carbon dioxide (CO_2) consists of one carbon (C) atom and two oxygen (O) atoms. Given the atomic masses of carbon (12 g/mol) and oxygen (16 g/mol), the molecular weight of CO_2 is :

$$12 + 2 \times 16 = 44 \text{ g/mol.} \tag{15}$$

Building on the assumption that 1 GtC corresponds to 3.67 $GtCO_2$ when carbon is fully oxidized to CO_2, the mass ratio of CO2 to carbon is calculated as per :

$$\frac{44}{12} = 3.67 \tag{16}$$

The additional factor of 10^{12} is applied to convert gigtaonnes (Gt) of CO_2 into kilograms (kg).

Conversion of GHG emissions to $CO_{2,eq}$ Emissions of different GHG are converted to a common unit, $kgCO_{2,eq}$, using their Global Warming Potential (GWP) values from the IPCC AR6.

Tableau C.1 Global Warming Potential of greenhouse gases (Smith et al., 2021) used to convert emissions into $kg_{CO2,eq}$. Note that some values are missing , leading to zero values in the table C.2

Substance	**GWP**	**Substance**	**GWP**
Black carbon		Halon 1202	216
C_2F_6	12400	Halon 1211	1930
C_6F_{14}	8620	Halon 1301	7200
CCl_4	2200	Halon 2402	2170
CF_4	7380	HCFC-141b	860
CFC-11	5560	HCFC-142b	2300
CFC-114	9430	HCFC-22	1960
CFC-115	9600	HFC-125	3740
CFC-12	11200	HFC-134a	1530
CFC-13	12400	HFC-143a	5810
CH_3Br	2.43	HFC-227ea	3600
CH_3CCl_3	161	HFC-23	14600
CH_3Cl	5.54	HFC-245fa	8690
CH_4	27.9	HFC-32	771
CO		N_2O	273
CO_2	1	NH_3	
Organic carbon		NOx	
SF_6	25200	VOC	

Cumulative emissions over the period 2015-2100 The *RCPDatabase* provides emission data at 5- or 10-year intervals rather than annually. To estimate cumulative GHG emissions, we assume a linear trend between consecutive data points and integrate emissions over time accordingly.

Tableau C.2 Cumulative GHG emissions over the period 2015–2100 ($kgCO_{2,eq}$)

Substance | Scenario	**RCP2.6**	**RCP4.5**	**RCP8.5**
CO_2	1.08E+15	2.65E+15	6.90E+15
CH_4	4.56E+14	7.41E+14	1.65E+15
N_2O	2.32E+14	3.09E+14	4.78E+14
SF_6	2.17E+12	9.44E+12	2.76E+13
HFC23	5.93E+11	5.93E+11	5.93E+11
HFC32	5.36E+12	5.36E+12	5.36E+12
HFC125	4.28E+13	8.77E+12	3.75E+13
HFC134a	2.94E+13	3.46E+13	6.39E+13
HFC143a	4.11E+13	1.40E+13	2.36E+13
HFC227ea	1.92E+11	7.81E+10	1.80E+11
HFC245fa	3.47E+12	6.53E+13	1.12E+14
CF_4	2.60E+12	5.88E+12	7.49E+12
C_6F_{14}	8.90E+10	8.90E+10	8.90E+10
C_2F_6	1.23E+12	2.07E+12	2.90E+12
Black carbon	0.00E+00	0.00E+00	0.00E+00
Organic carbon	0.00E+00	0.00E+00	0.00E+00
CO	0.00E+00	0.00E+00	0.00E+00
NOx	0.00E+00	0.00E+00	0.00E+00
VOC	0.00E+00	0.00E+00	0.00E+00
NH_3	0.00E+00	0.00E+00	0.00E+00
CFC-11	5.84E+12	5.84E+12	5.84E+12
CFC-12	3.34E+12	3.34E+12	3.34E+12
CFC-13	7.01E+09	7.01E+09	7.01E+09
CFC-114	2.55E+08	2.55E+08	2.55E+08
CFC-115	3.64E+09	3.64E+09	3.64E+09
CCl_4	1.07E+11	1.07E+11	1.07E+11
CH_3CCl_3	1.75E+09	1.75E+09	1.75E+09
HCFC-22	1.14E+13	2.44E+13	2.44E+13
HCFC-141b	1.54E+12	1.93E+12	1.93E+12
HCFC-142b	9.19E+11	9.19E+11	9.19E+11
Halon 1211	1.05E+11	1.05E+11	1.05E+11
Halon 1202	0.00E+00	0.00E+00	0.00E+00
Halon 1301	1.71E+11	1.71E+11	1.71E+11
Halon 2402	2.19E+09	2.19E+09	2.19E+09
CH_3Br	3.26E+10	3.26E+10	3.26E+10
CH_3Cl	1.67E+12	1.67E+12	1.67E+12
Total	1.92E+15	3.88E+15	9.35E+15

Adaptation of ES-matrix to estimate value of IUCN maps

The ESVD (L. M. Brander et al., 2024) provides average values for ecosystem service (ES) supply by biome, expressed in int$2020 per hectare per year (see Table 1 in L. M. Brander et al. (2024)). However, more detailed data disaggregated by habitat type (or EFGs) is not available. To estimate the value of habitats threatened by ESL events, biome-level proxies were therefore assigned to each EFG.

This process is detailed in the electronic supplementary material (ESM), which includes the matrix used to allocate ES values (per unit area) to each habitat as mapped by the IUCN.

Additional analysis on ES valuation

The following subsections present additional results from the valuation of coastal zones at risk of inundation.

Evolution of GESV under threat of inundation

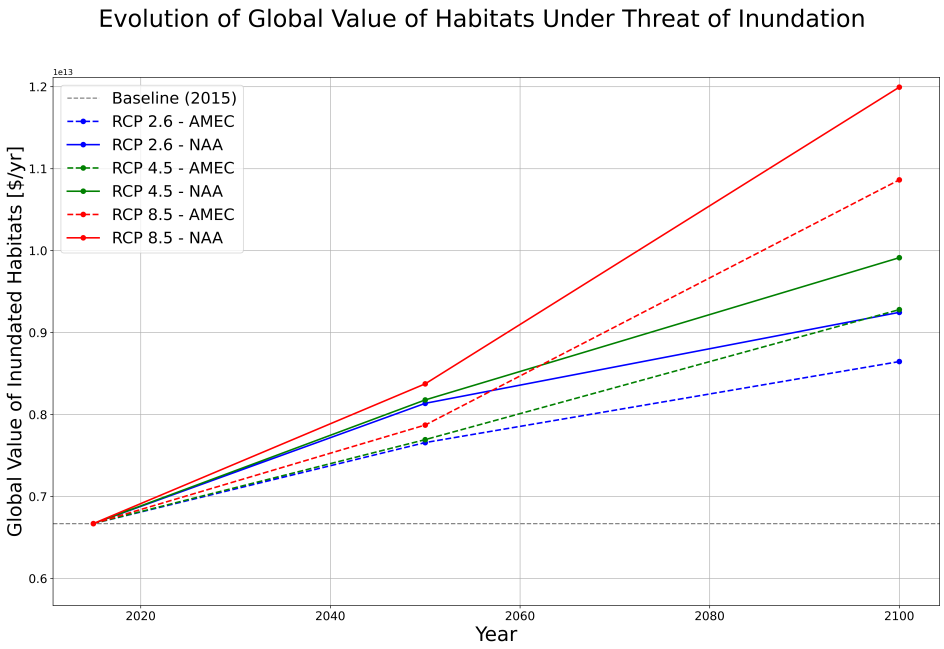
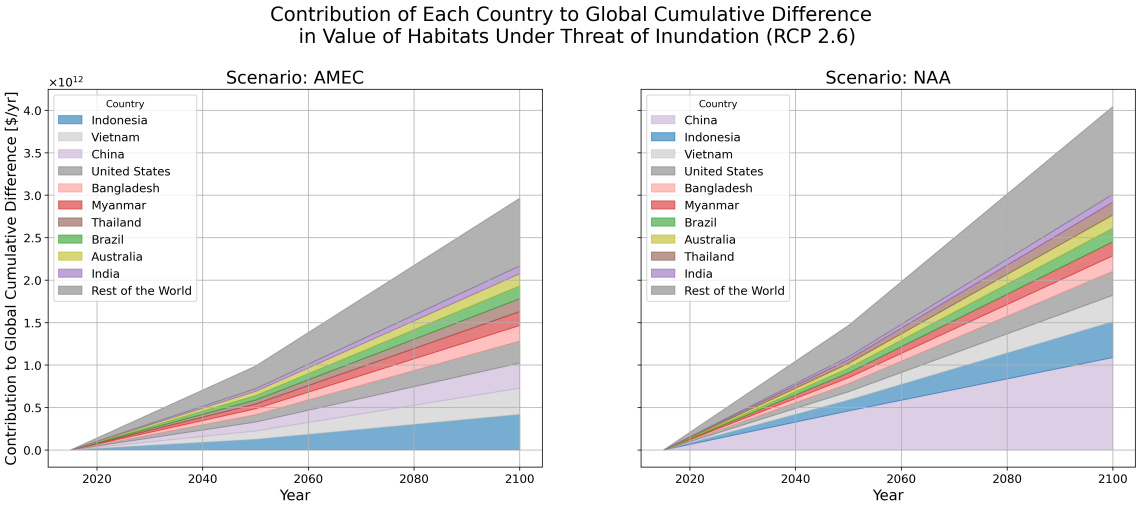
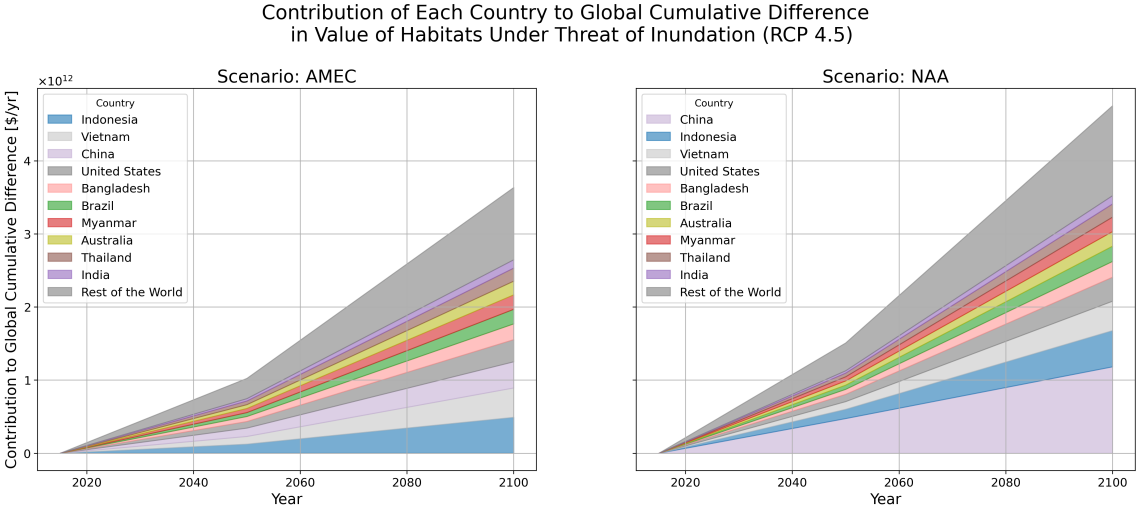
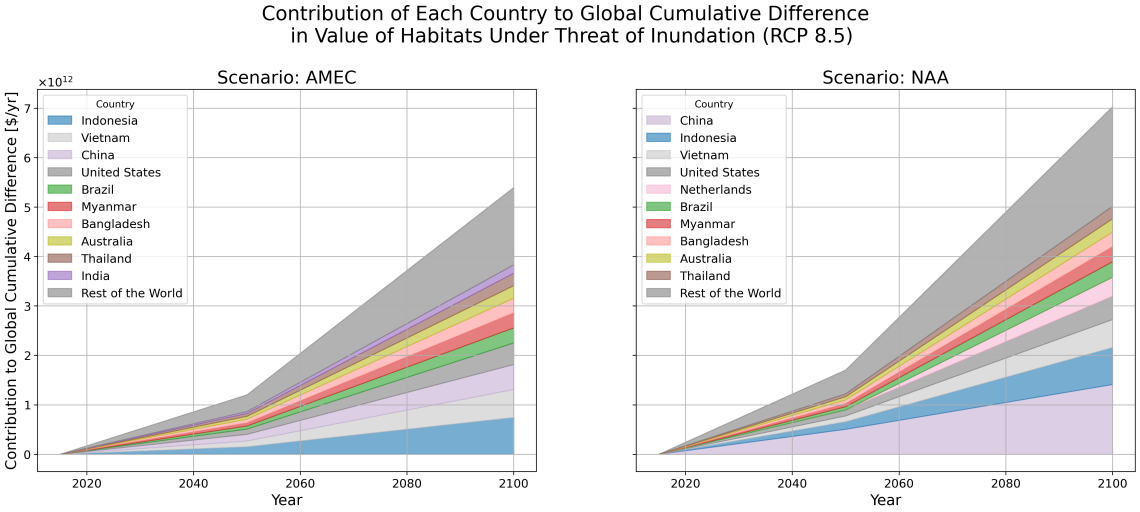


FIGURE C.1 Evolution of global value under threat of flooding, for all scenarios.

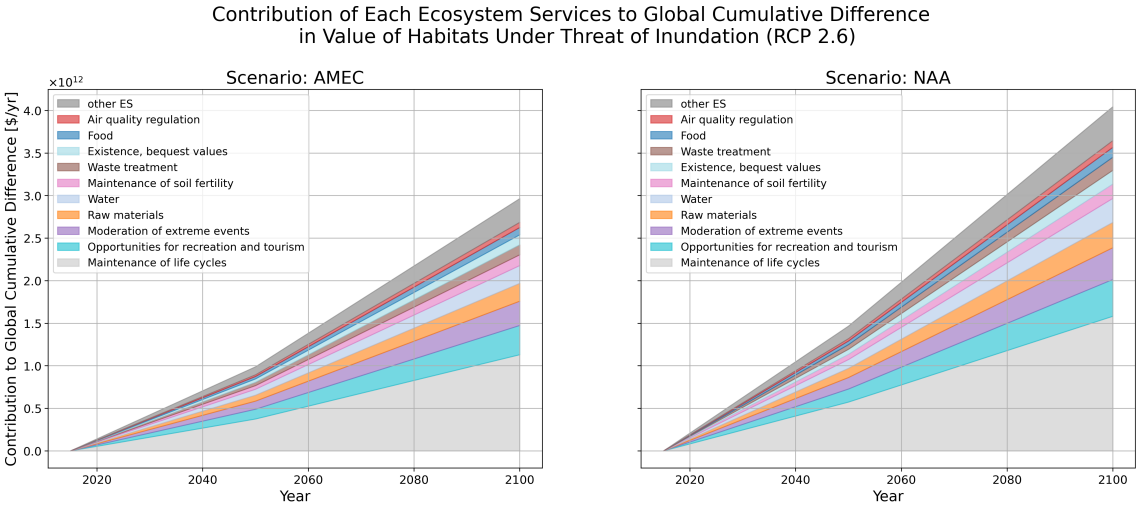
Contribution analysis

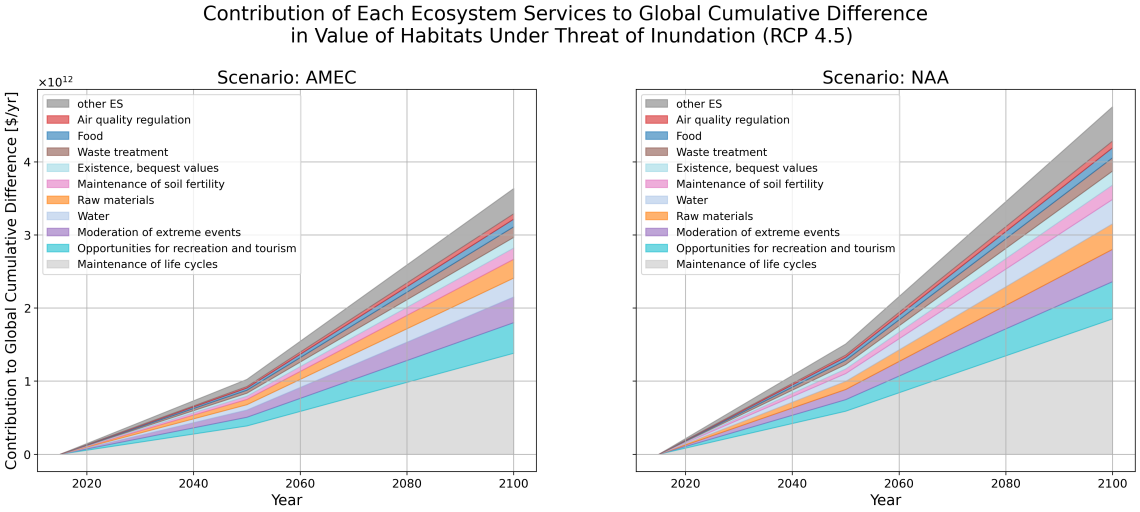


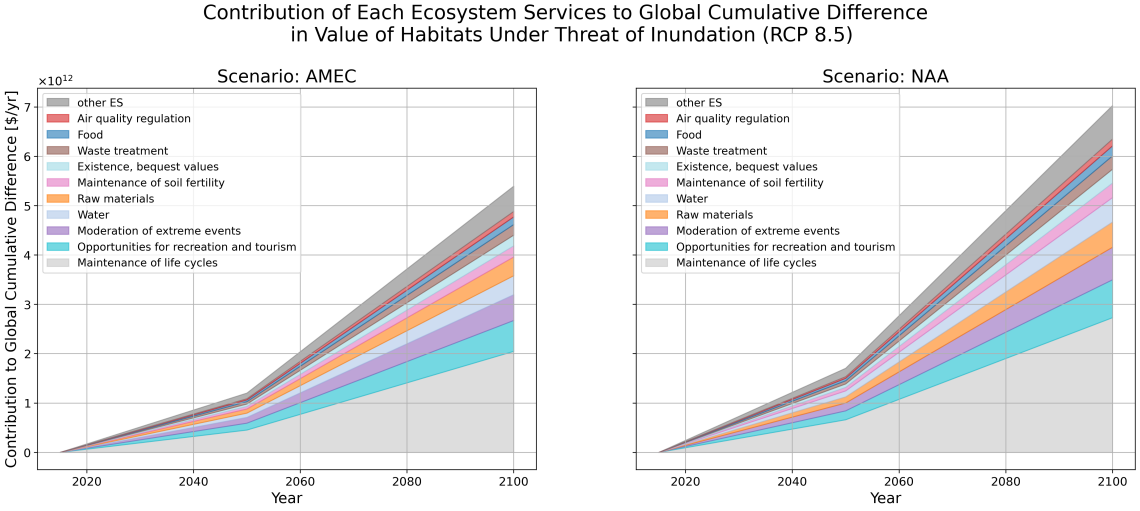




Contribution analysis per country







Contribution analysis per ecosystem services

The social cost of carbon

The damage function on SLR

To calculate the economic costs due to SLR, Diaz (2016) minimizes the sum of adaptation costs and damages :

$$\min \sum_{t} \left[\frac{1}{(1+r)^t} \left(\text{ProtectionCost}_{s,t} + \text{RetreatCosts}_{s,t} + \text{InundationCosts}_{s,t} + \text{WetlandCost}_{s,t} + \text{FloodCost}_{s,t} \right) \right] \tag{17}$$

where t : annual time-step, r : discount rate of 4%, s :adaptation strategy.

In this equation, the damage costs due to inundation are calculated based on the value of the land, estimated by its average rents for agricultural land. The damage costs due to wetland loss are based on the willingness to pay for ES. The approach here is close to the approach followed in our study, while using fewer types of habitats and more outdated ES valuation. The damage costs due to flooding focus on damages on human assets.

ANNEXE D - SUPPLEMENTARY INFORMATION - FRESHWATER CONSUMPTION AND DOMESTIC WATER DEPRIVATION IN LCIA : REVISITING THE CHARACTERIZATION OF HUMAN HEALTH IMPACTS

Ce matériel supplémentaire se réfère au chapitre 7.

Sanity check - Comparison of WASH DALYs values

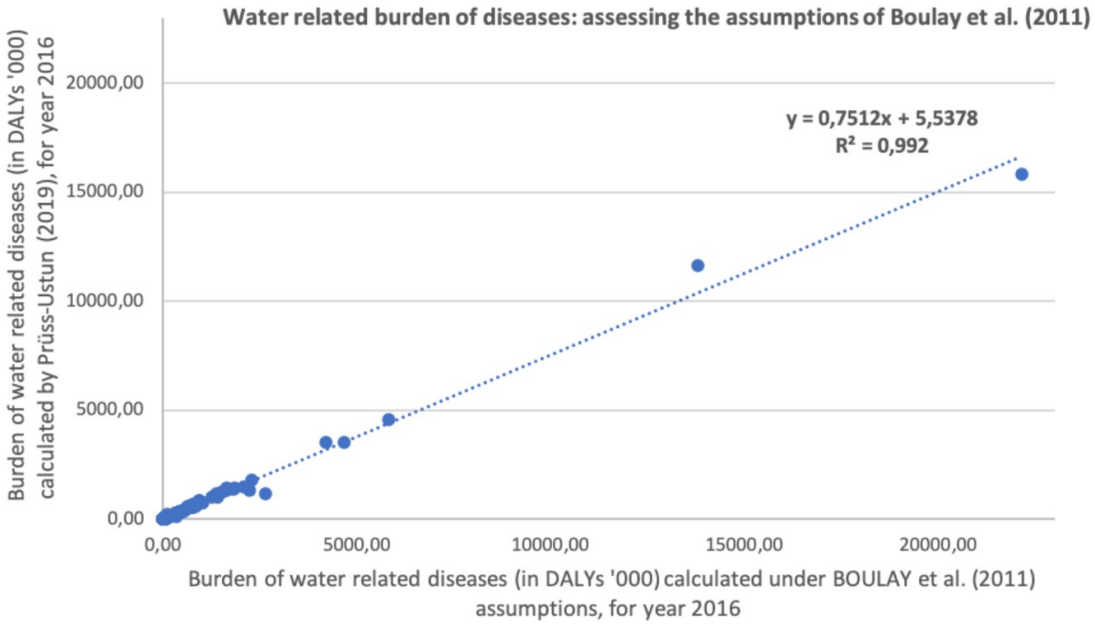


Figure 1 : Sanity check comparing Pruss-Ustun et al. (2019) results with Boulay et al. (2011)'s assumptions regarding water-related DALYs

Although the values proposed by Boulay et al. (2011) are 25% higher, the correlation is strong between the latest model's assumptions and data regarding the water related burden of diseases. Considering its expertise and accuracy, we chose the data proposed by Prüss-Ustün et al. (2019) to be integrated in this work.

Insights on the fate factor

The fate factor developed in this work results from the multiplication of two ratios: HWCtotal/AMC (where AMC accounts for *Availability Minus Water Consumption*)

which can be linked to *water scarcity*, and HWCdom/HWCtotal, accounting for *water distribution among users*.

$$FF_{dom} = \frac{HWC_{total}[m^3]}{AMC[m^3]} \times \frac{HWC_{dom}[m^3]}{HWC_{total}[m^3]} \quad \text{Eq1}$$

Their multiplication (and the simplification of the term HWCtotal) allows the calculation of the fate factor as the ratio of HWCdom to AMC, as presented in the main text.

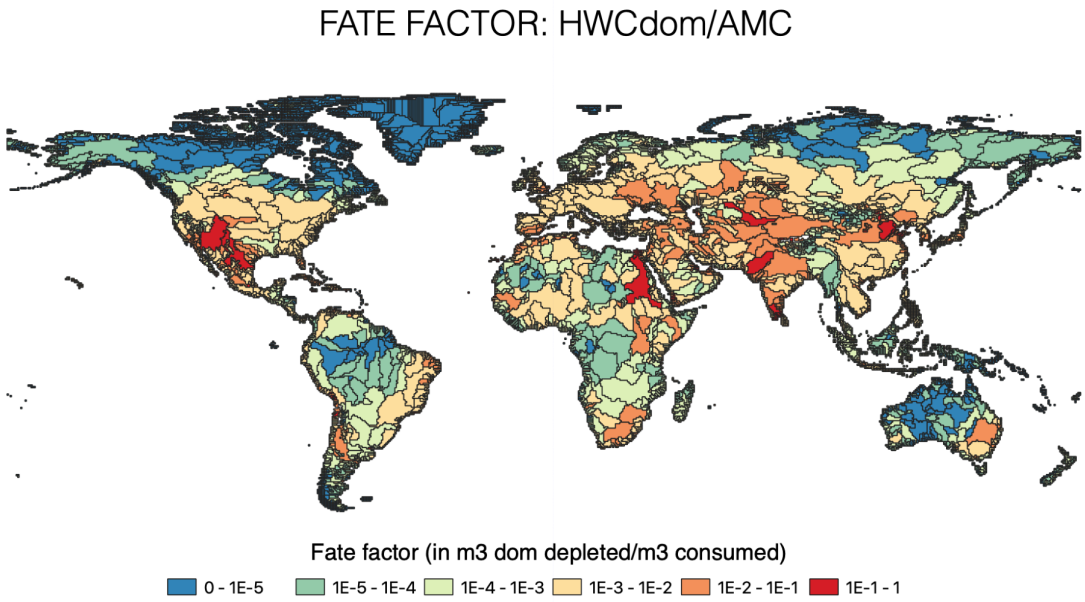
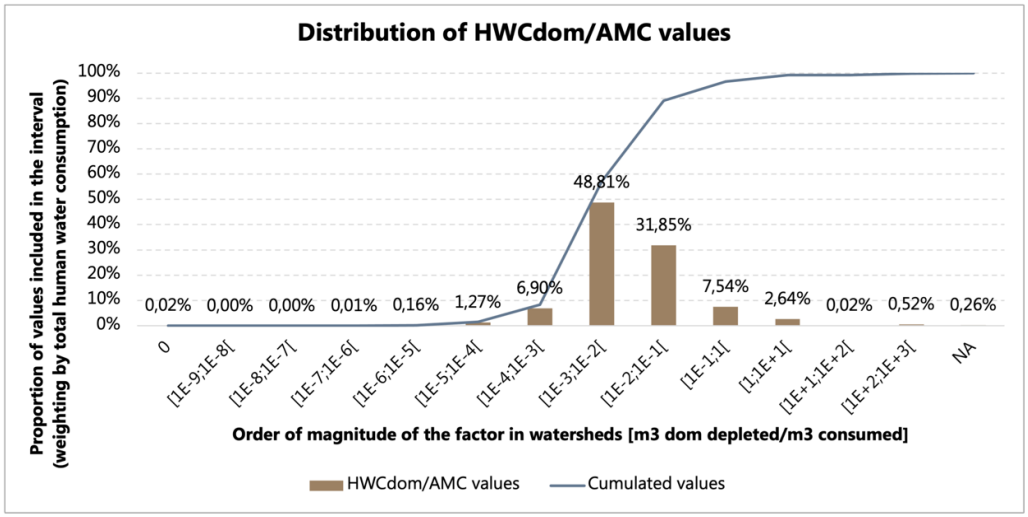


Figure 2: Map of fate factors

A



B

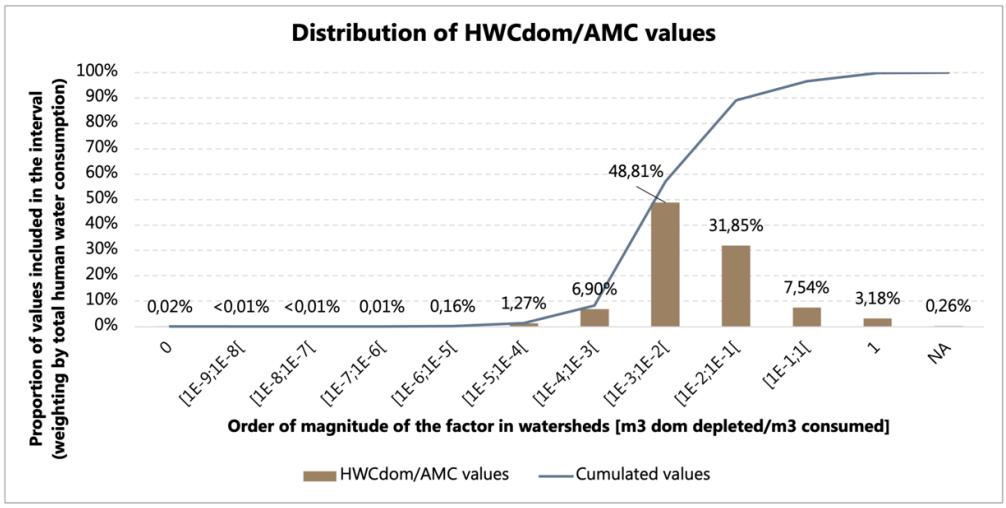


Figure 3: Distribution of fate factors values according to their order of magnitude. Fate factors are expressed in m^3 dom depleted/m^3 consumed. Proportions are calculated based on the total human water use intensity in each watershed. Values above 1 are not limited to 1 in A but are in B

Fate factors as defined in the model (i.e. limited to 1, see figure B) range from 0 to 1 m3 dom depleted/m3 consumed. Those results are aggregated by percentage of water consumption amount in watersheds to assess the distribution of the values in line with LCA inventories (Figure 3). More than 99% of the values (99.55%) are superior to 1E-5, and almost 81% (80.66%) are included between 1E-3 and 1E-1 m^3 dom depleted/m^3 consumed.

In order to provide better insights on the modeling and to understand the characteristics of watersheds, the following paragraphs offer discussion on the differences of both *water*

scarcity (HWCtotal/AMC) and *users distribution (HWCdom/HWCtotal)* within watersheds.

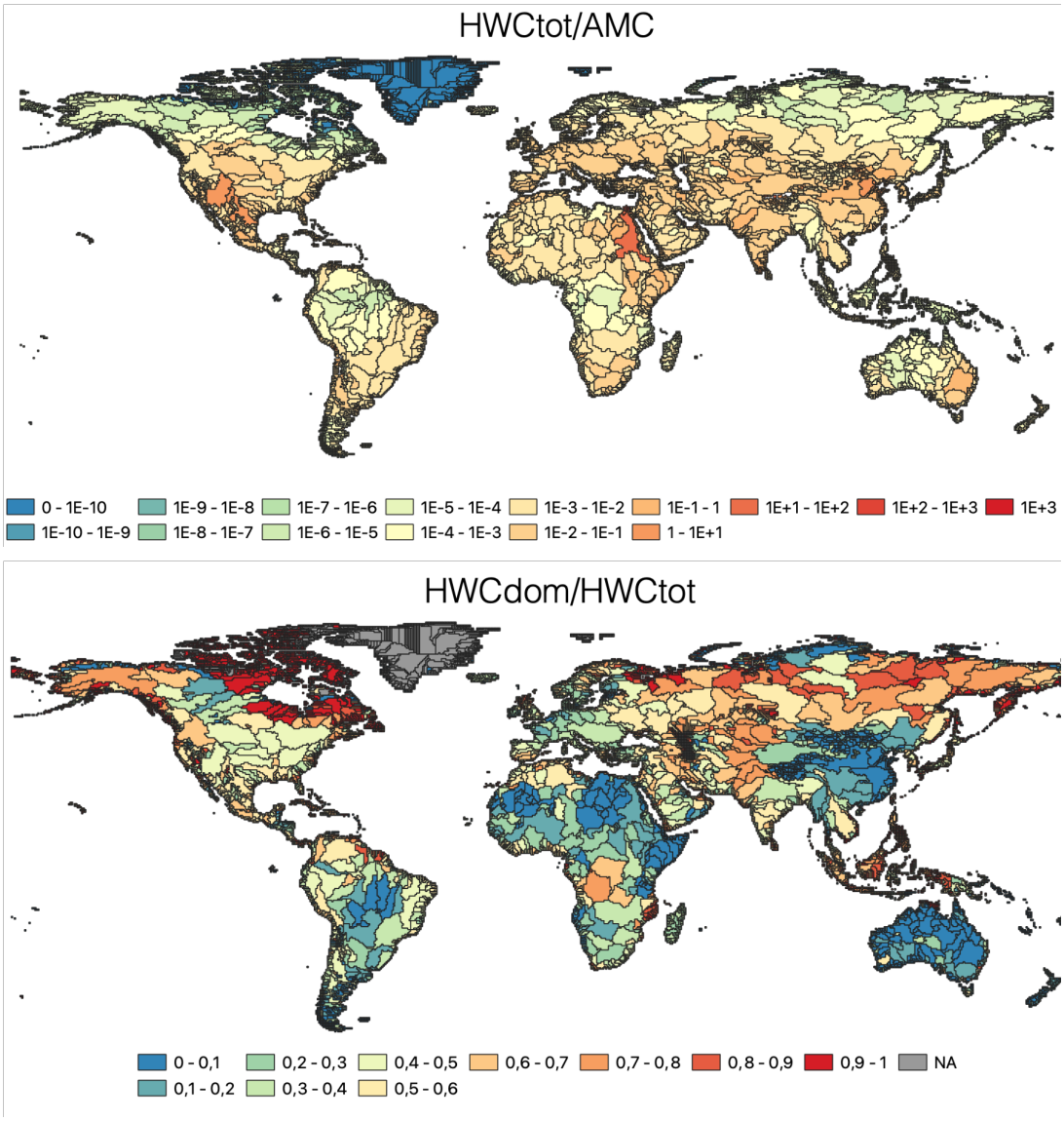
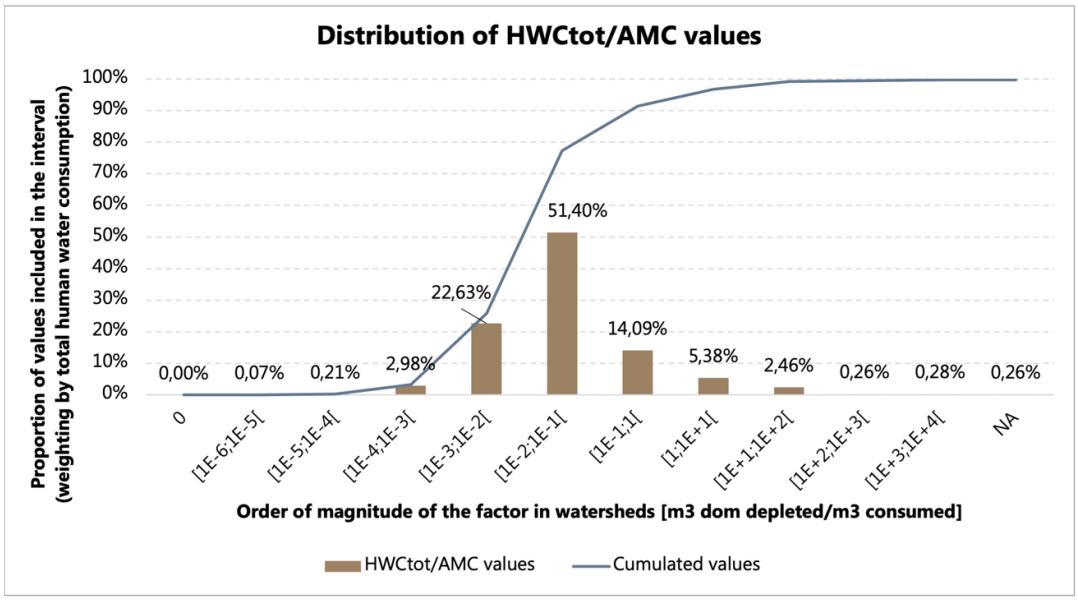


Figure 4: Geographic representation of A- water scarcity (unitless) and B-water distribution among users in watersheds (in % water consumption). Obtained by manipulating WaterGAP data (Flörke et al. 2013) through QGIS. Caution is required in the interpretation of the maps giving the non-uniformity of the legends with the Fate factor map's one (Figure 1).

Water scarcity (HWCtotal/AMC) echoes the physical water depletion potential we define within the text. Figure 4.a evidences the regions where water scarcity is high, such as the Nile valley, California, Nevada or Arizona. In low population areas such as central Australia, northern Canada, the Amazon rainforest, northern Russia or Saharan Africa, low values of Hwtotal/AMC can be explained by low water demand. Water distribution among users (Figure 4.b) allows to see water competition between users through the lens of domestic water. Domestic water use intensity is particularly high in northern Canada and in some Eastearn Europe regions.

A)



B)

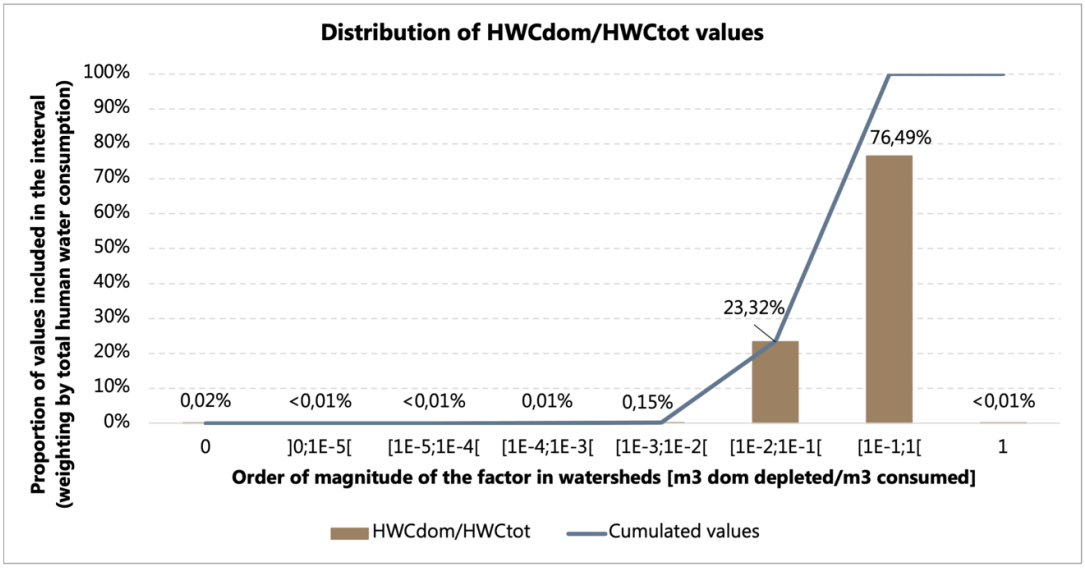


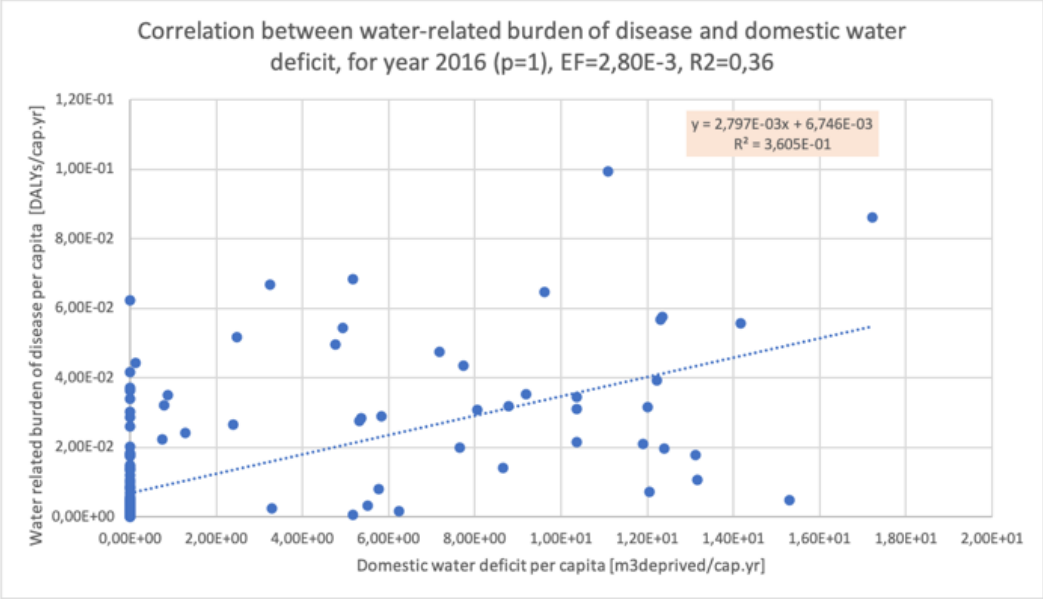
Figure 5: Distribution of values A- for HWCtot/AMC, assessing water scarcity, and B- for HWCdom/tot assessing domestic water share in total water use

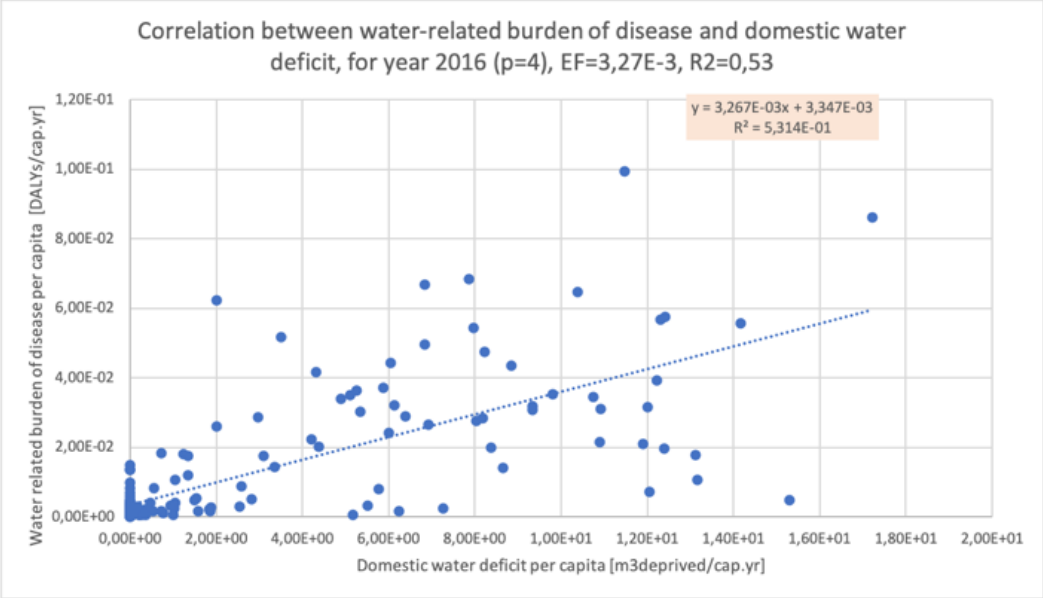
Figure 4 and Figure 5 show the wider distribution of HWCtot/AMC values compared to HWCdom/HWCtot. While nearly ninety nine (98.93) percent of the values of the ratio HWCtot/AMC are distributed between 1E4 and 1E+2 (6 orders of magnitude), 99.81% of the values of the ratio HWCdom/HWCtot are included between 1E-2 and 1 (2 orders of magnitude). That highlights the important sensitivity of the fate factor to water scarcity (HWCtot/AMC).

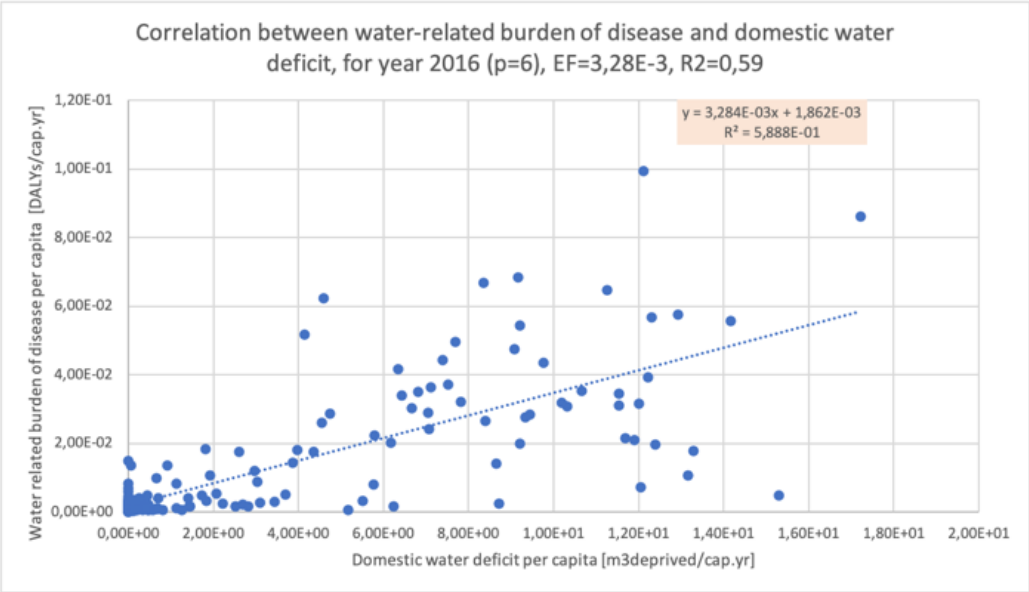
Sensitivity analysis of p

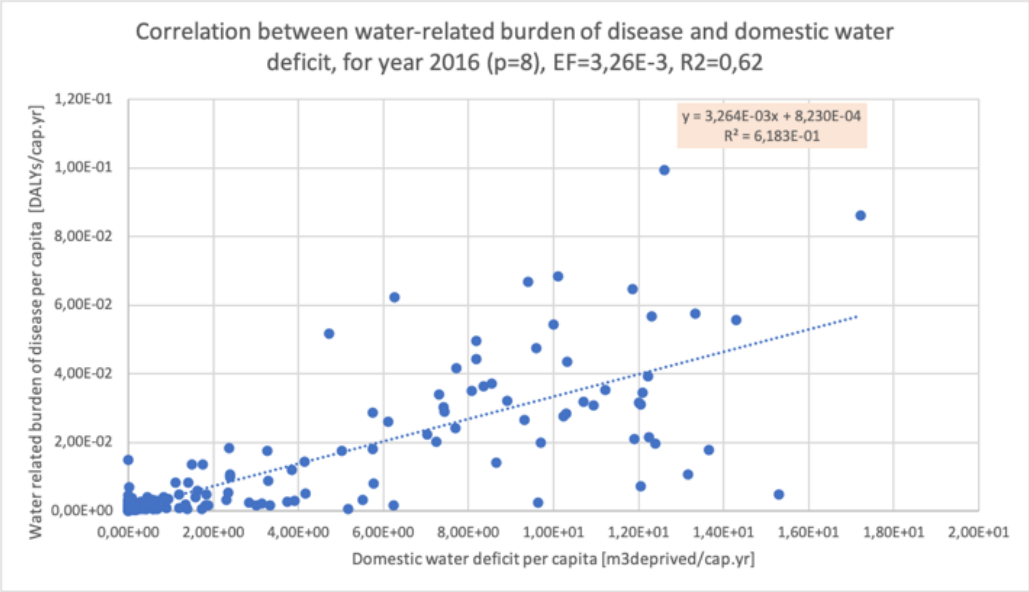
A sensitivity analysis is carried out on the value p to assess its influence on the effect factor value. The relevance of the correlation between the water-related burden of disease and the domestic water deficit is considered as a basis to this analysis. The inconsistency of the results when p=1 confirms the relevance of the integration of a factor p to discriminate domestic water use between households with and without water access on premise: the high number of countries suffering from water-related diseases with no domestic water deficit is considered inadequate. The correlation between water related burden of diseases and domestic water deficit shows great results for values of p superior to 4. The value used in this work is set to 6,3 in respect to these results and to the work of Larson, Minten, and Razafindralambo (2006). Nonetheless, the sensitivity

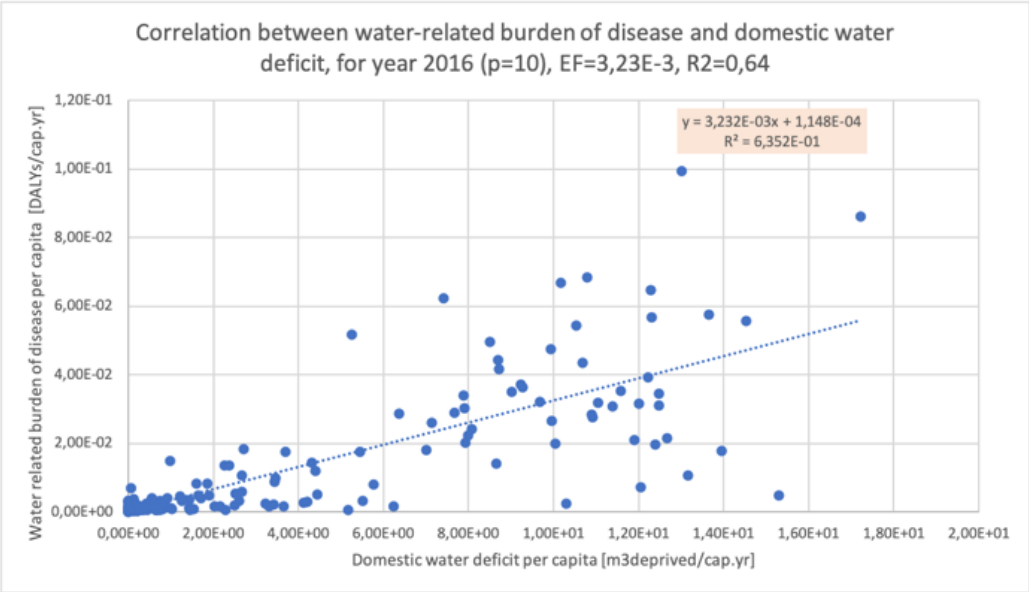
analysis shows that a change of the value has little influence on the value of the effect factor and its order of magnitude [2.80E-3 DALY/m^3 – 3.07E-3 DALY/m^3].






Correlation between water-related burden of disease and domestic water deficit, for year 2016 (p=6), EF=3,28E-3, R2=0,59
y = 3,284E-03x + 1,862E-03
R² = 5,888E-01
Water related burden of disease per capita [DALYs/cap.yr]
1,20E-01
1,00E-01
8,00E-02
6,00E-02
4,00E-02
2,00E-02
0,00E+00
0,00E+00 2,00E+00 4,00E+00 6,00E+00 8,00E+00 1,00E+01 1,20E+01 1,40E+01 1,60E+01 1,80E+01 2,00E+01
Domestic water deficit per capita [m3deprived/cap.yr]


Correlation between water-related burden of disease and domestic water deficit, for year 2016 (p=8), EF=3,26E-3, R2=0,62
y = 3,264E-03x + 8,230E-04
R² = 6,183E-01
Water related burden of disease per capita [DALYs/cap.yr]
1,20E-01
1,00E-01
8,00E-02
6,00E-02
4,00E-02
2,00E-02
0,00E+00
0,00E+00 2,00E+00 4,00E+00 6,00E+00 8,00E+00 1,00E+01 1,20E+01 1,40E+01 1,60E+01 1,80E+01 2,00E+01
Domestic water deficit per capita [m3deprived/cap.yr]



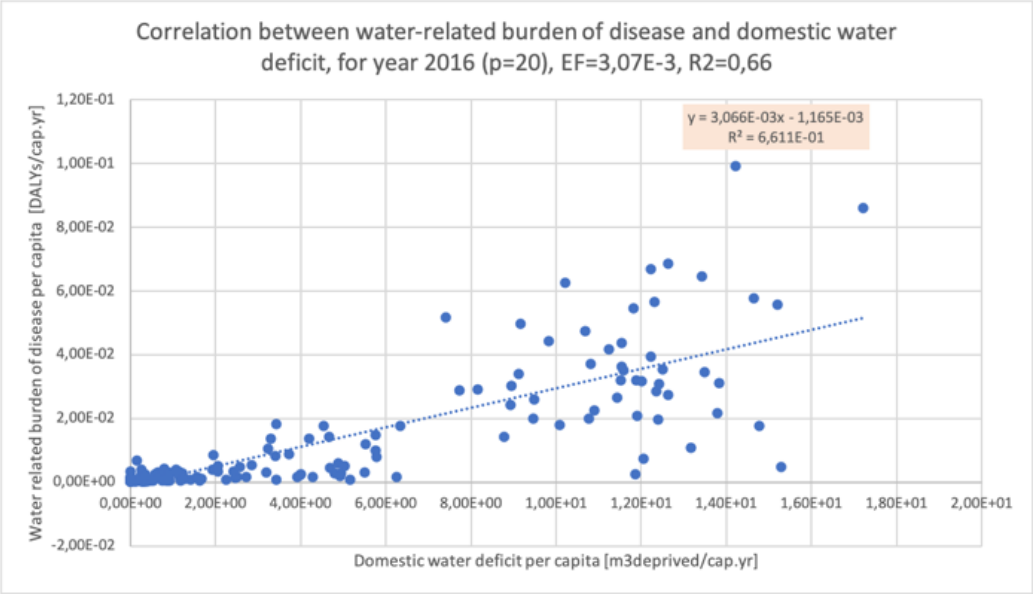


Figure 6: Correlation between water-related burden of disease and domestic water deficit, for year 2016

Correlation between water related burden of diseases and private access to water supplies

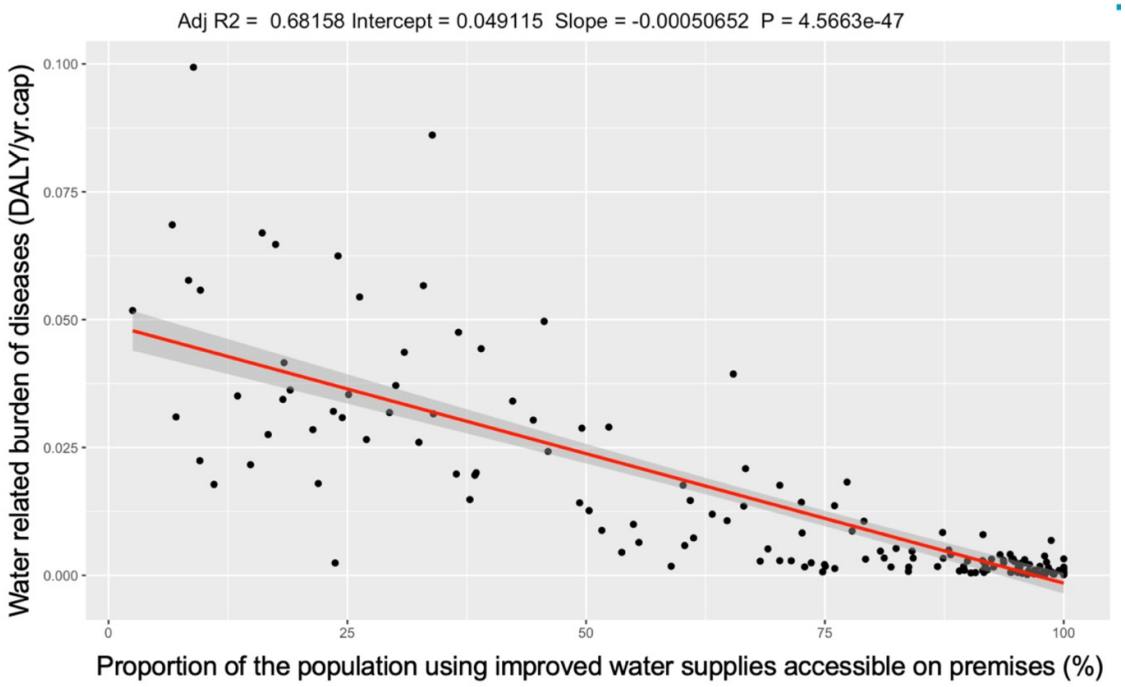


Figure 7: Correlation between water related burden of diseases and private access to water supplies

A strong correlation is demonstrated between the coverage of private access to domestic water supply and the water-related burden of disease per capita (R2=0,68, P=4,57e-47).

Boulay et al. (2011) « general » and "unknown" models

Several sub-models were developed building on Boulay et al. (2011) in order to be implemented in life cycle assessment applications. They differ mainly by the specificity of the data used to calculate the fate factor. The "Unknown" model was developed to be integrated into the IMPACT World+ (Bulle et al. 2019) life cycle assessment methodology, while the "General" model data is used in this work to compare the results with the revised model. The following section summarizes how they differ.

The fate factor used in the **Unknown model** discriminates between groundwater and surface water scarcity.
It defines a final scarcity indicator as the weighted mean of the scarcity factors: $\alpha_{surface}$ and $\alpha_{groundwater}$. They are each the result of the application of a logarithmic function to their corresponding "raw" scarcity indices
($\alpha^*_{surface}$ and $\alpha^*_{groundwater}$). The surface $\alpha^*_{surface}$ "raw" scarcity indicator is calculated as a consumption to availability (CTA) ratio, where consumption refers to surface water consumption and availability as the statistical low flow Q90. The groundwater $\alpha^*_{groundwater}$ "raw" scarcity indicator is also calculated as a consumption to availability (CTA) ratio, where consumption refers to ground water consumption and availability as the renewable groundwater resource available.
In Boulay et al. (2011), a logarithmic function is applied to these "raw" factors (CTA ratios) to obtain the fate (named scarcity) factors $\alpha_{surface}$ and $\alpha_{groundwater}$. It is defined by thresholds based on exisiting international recommendations on the definition of stress at the time developed for WTA and converted for CTA use: ratios are set equal to 0 if inferior to 0,0022 and to 1 if superior to 0,196. Then, $\alpha_{unknown}$ is calculated as the consumption-weighted sum of $\alpha_{surface}$ and $\alpha_{groundwater,}$ as per:

$$\alpha_{unknown} = \alpha_{surface} \times (1 - fg) + \alpha_{groundwater} \times fg$$

where fg is the fraction of usage dependent on groundwater.

The **General** model calculates $\alpha_{general}$ by applying the same logarithmic function to a "general" CTA ratio that does not discriminate between water source (surface or ground), but is defined as the ratio between total water consumption and total water availability ($\alpha_{general^*}$= CU/(Q90+GWR)). The logarithmic function thresholds are the same (0,0022 and 0,196).

Since the thresholds remain the same and that $\alpha^*_{surface}$ values are globally higher than $\alpha^*_{groundwater}$ and $\alpha^*_{general,}$ the application of the logarithmic function on these "raw" scarcity indices result in a higher proportion of zero values for $\alpha_{general}$ values (compared to $\alpha_{unknown}$ calculated as the weighted sum of $\alpha_{surface}$ and $\alpha_{groundwater}$). This translates into a "general" water stress being generally lower (because of more resources available for all uses) than a surface-specific or groundwater-specific water stress where the water use may not necessarily be optimized with the most abundant source

Fate factor comparison

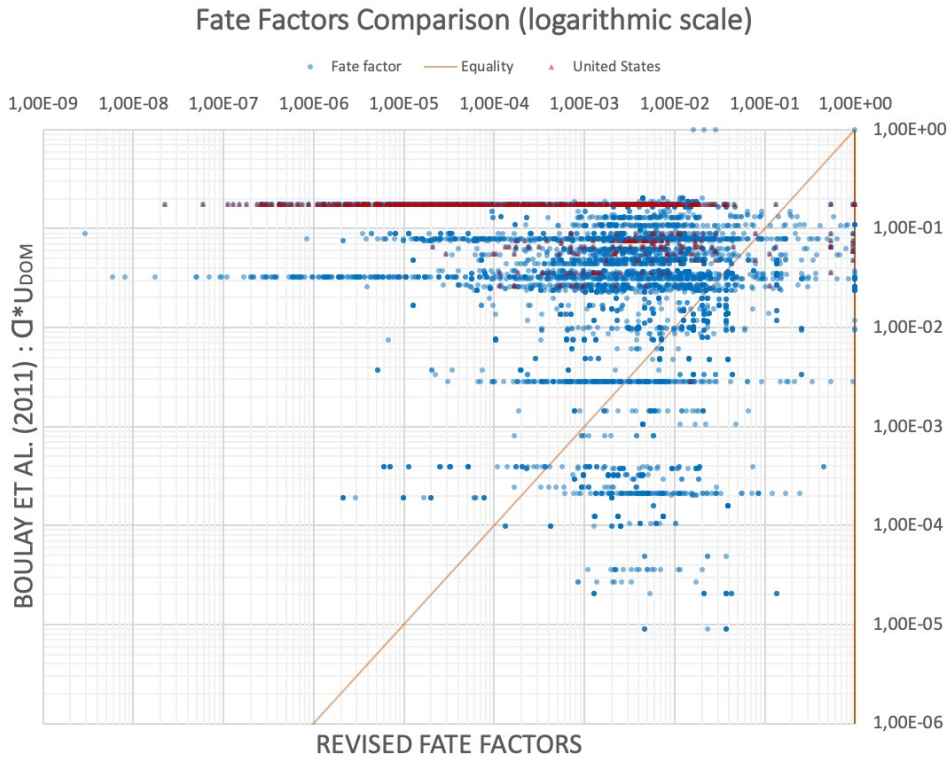


Figure 8: Comparison of revised (this study) vs Boulay et al. (2011)'s non zero fate factors [in m3 dom depleted/m3 dom used]. Blue points abscissa (ordinate respectively) coordinates account for sub river basin fate factors values in the revised model (the combination of fate factors α and the proportion of use by domestic users U_{dom} in Boulay et
al. (2011)). Due to the use of a logarithmic scale, null values are not represented; they are the only truncated values of the graph. The orange equality line represents coordinates for which revised and Boulay et al (2011)'s fate factors are equal for a sub river basin cell. Red points account for United States fate factors.

Figure 8 represents the revised fate factors in comparison with the combination of non null Boulay et al. (2011)'s fate factors α and the proportion of use by domestic users. Dots aligned on a horizontal line illustrate the variability of fate factors due to a higher spatial resolution of this research work versus the single value of a lower resolution of Boulay et al. (2011). For instance, Boulay et al. (2011) provides 37 different fate factors in the United States, compared to 721 in the revised model. 69.7% of the points are located at the left of the equality line, showing an overall decrease of non null fate factors values in the revised model. For instance, in the United States, 79.9%, (0.92% and 19.1%) of the revised fate factors present lower (equal and higher respectively) values compared to Boulay et al. (2011).

Revised exposure factors

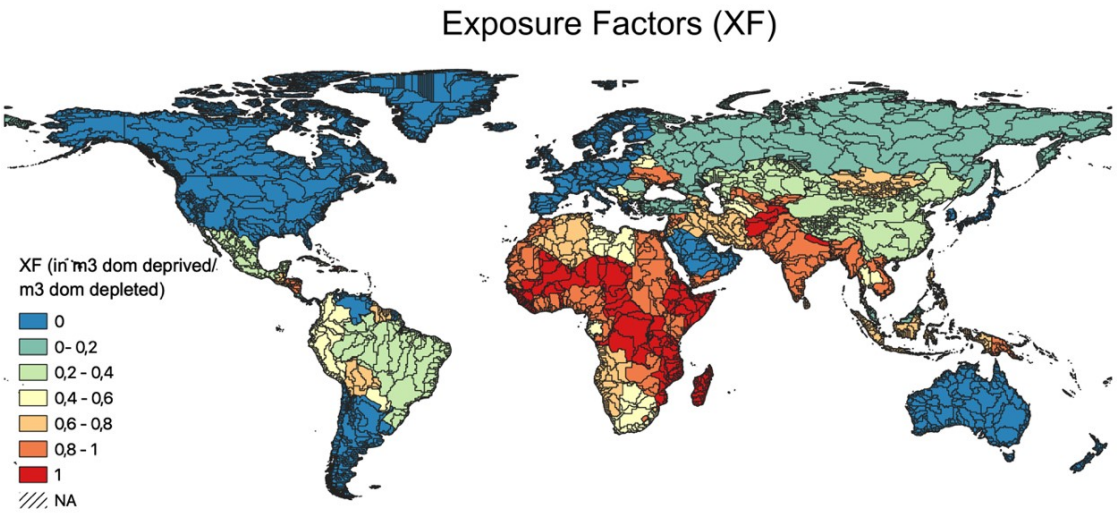


Figure 9: Exposure factors

Country scale CF

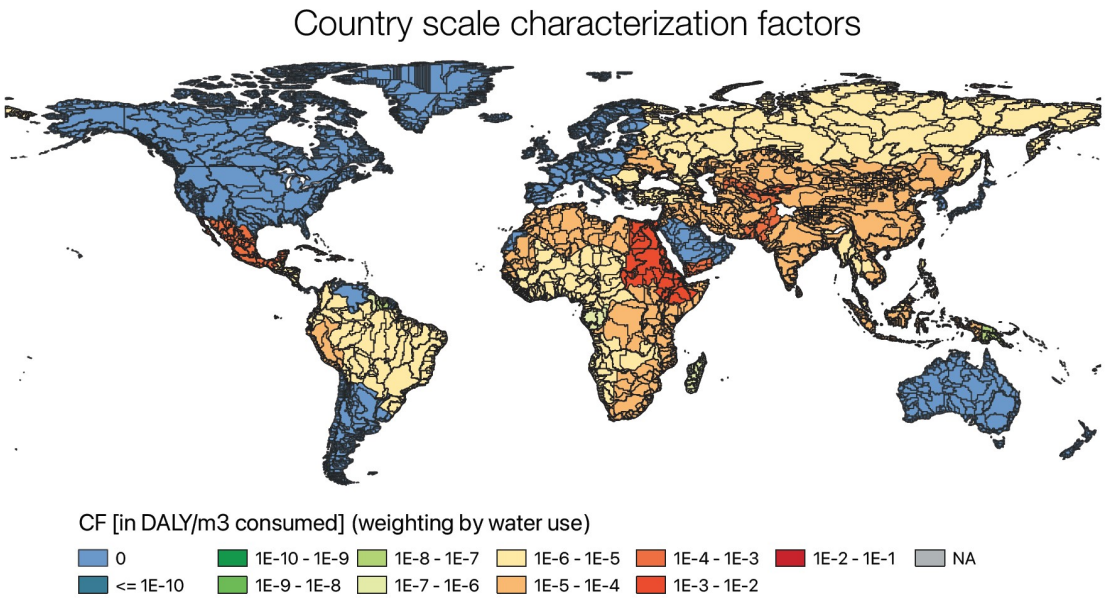


Figure 10: Country scale CF

Country scale CFs (annual) are calculated considering a domestic water use weightening.

Sensibility analysis: Variation of minimum water requirements

To assess the sensibility of the results to the value of minimum human water requirements (MWR), we calculated the effect factors by adjusting the value of the MWR from 10 to 500 liters per day per capita.

Minimum water requirements			
In liters per day per capita (lpdc)	In m3	EF (global, as the slope of the linear regression)	R2
10	0,01	2,50E-02	0,15
20	0,02	9,92E-03	0,34
30	0,03	5,97E-03	0,46

40	0,04	4,15E-03	0,51
50	0,05	3,13E-03	0,54
60	0,06	2,51E-03	0,56
70	0,07	2,07E-03	0,57
80	0,08	1,73E-03	0,57
90	0,09	1,49E-03	0,57
100	0,1	1,30E-03	0,57
110	0,11	1,15E-03	0,57
120	0,12	1,01E-03	0,56
130	0,13	9,30E-04	0,6
140	0,14	8,48E-04	0,55
150	0,15	7,77E-04	0,55
160	0,16	7,17E-04	0,54
170	0,17	6,65E-04	0,54
180	0,18	6,19E-04	5,28
190	0,19	5,79E-04	0,52
200	0,2	5,43E-04	0,51
500	0,5	3,37E-04	0,43

A rise of the minimum water requirements results in the increase of domestic water deficit. Since the effect factor is calculated as the ratio of water-related burden of diseases to the domestic water deficit, this trend leads to a decrease in the effect factor by an order of magnitude similar to the evolution of water requirements (EF=3.13E-3 DALY/m3 deprived for water requirements= 50lpcd, EF=3.37E-4 DALY/m3 deprived for water requirements= 500lpcd).

Exposure factors

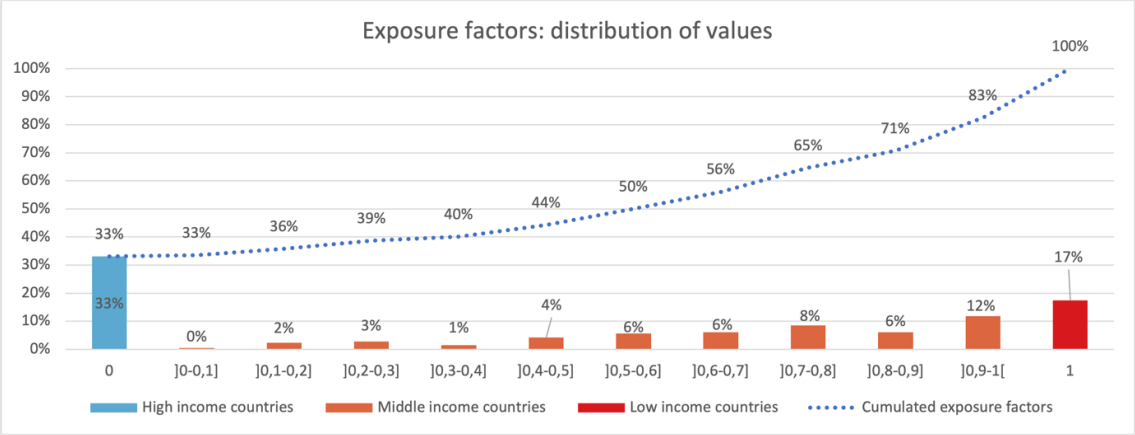


Figure 11: Distribution of exposure factor values. Each column represents the percentage of total countries whose exposure factors are included in the corresponding range. Blue dotted line represents the cumulative percentage of total countries whose exposure factors are equal or inferior to the corresponding range

Among the 212 updated factors, 70 countries are capable to fully adapt to water depletion (XF=0 for 33%, Figure 6's blue column) while 37 don't (XF = 1 for 17%) and exposure factor is included between 0 and 1 for 105 middle income countries (50%).

Comparison with Boulay et al. (2011) CFs

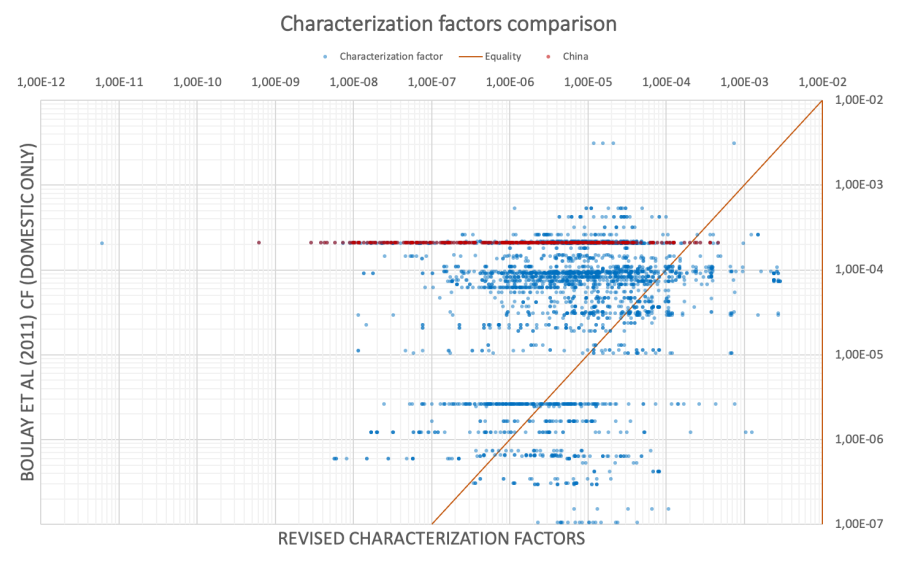


Figure 12: Comparison of revised (this study) vs Boulay et al. (2011)'s non null characterization factors [in DALY/m3 consumed]. Due to the use of a logarithmic scale, null values are not shown. Red points account for one CF calculated in China by Boulay et al. (2011)

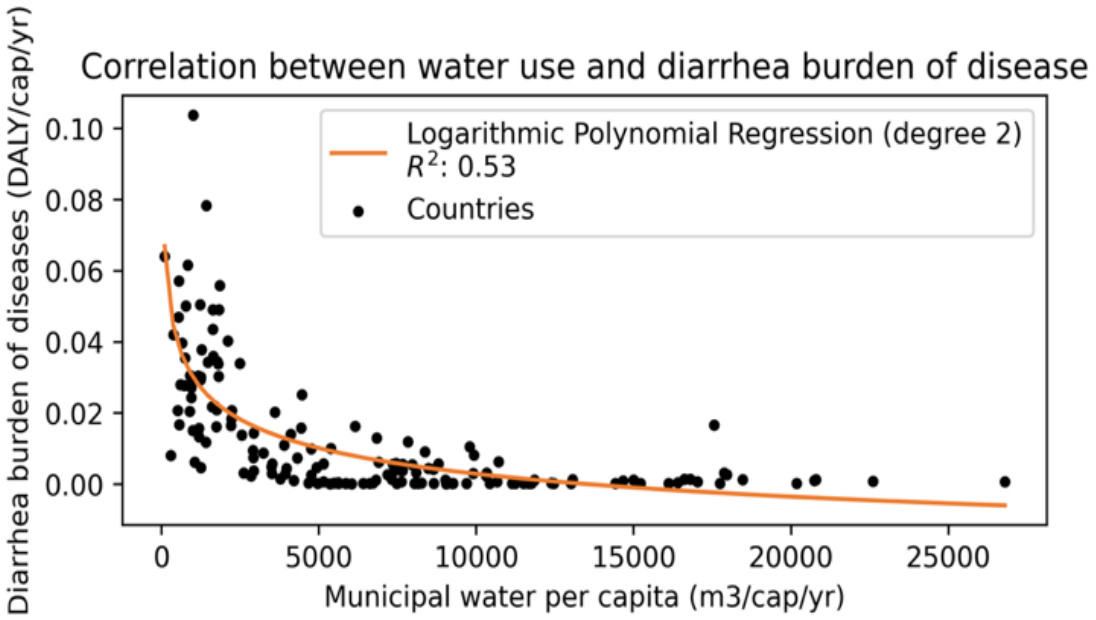
Figure 12 shows the comparison between the non null revised CFs of this study and the ones developed by Boulay et al. (2011). These last CFs account for the domestic impact pathway only (excluding the agricultural and fisheries impact pathways) and consider a generic water quality level and without any specification related to the source of water. Overall, with points located above the line of equality, the non zero revisited CFs show lower values from the original CFs, directly linked with the changes in the fate factor, as described above. The lines of horizontal points show the relevance of the increased geographical resolution of the fate factor following the update of data from WaterGAP (Flörke et al. 2013; Müller Schmied et al. 2014). That allows a higher discrimination within a region, and a better representation of the local realities (current CFs) than single values obtained by Boulay et al (2011).

ANNEXE E - SUPPLEMENTARY INFORMATION: ASSESSING THE POTENTIAL HUMAN HEALTH IMPACTS OF FRESHWATER CONSUMPTION: CONSIDERING INEQUALITIES IN WATER AVAILABILITY TO ASSESS THE CONSEQUENCES OF DOMESTIC WATER DEPRIVATION

Ce matériel supplémentaire se réfère au chapitre 8.

1. Correlation between water use per capita and diarrhea burden of disease.

A



B

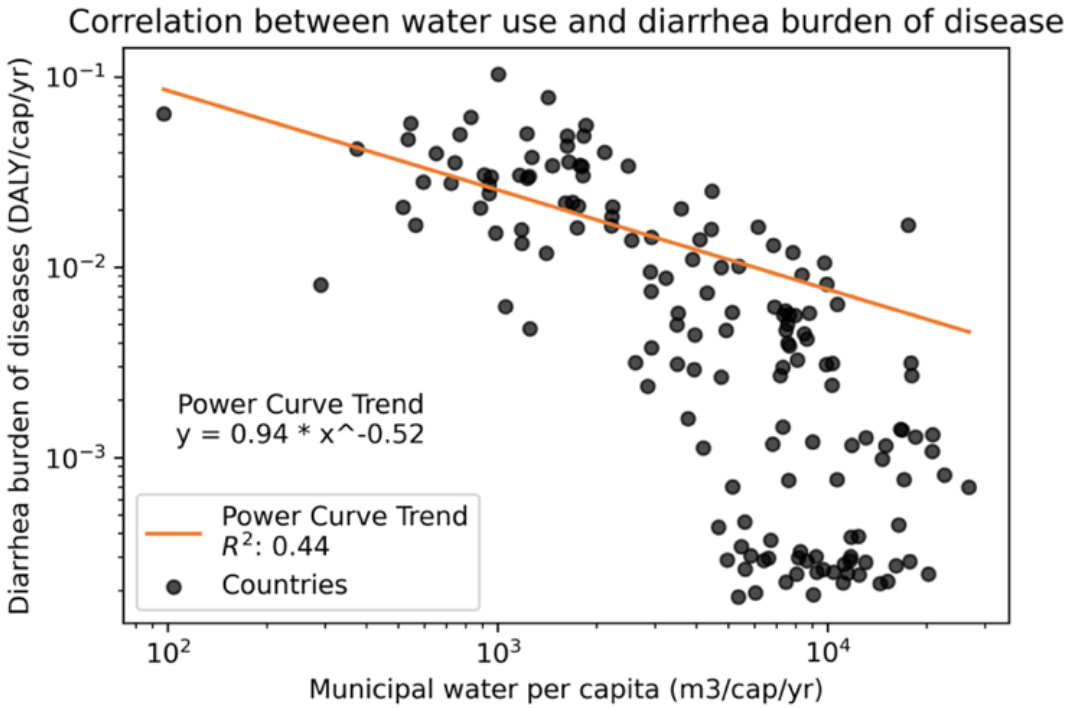


Figure 1: Link between municipal water use per capita and diarrhea burden of diseases (2019).

Figure 1 shows the logarithmic (above) and power (below) relationships between municipal water use per capita and diarrhea burden of diseases per capita. The relationship shows that the larger municipal water use per capita (used as a proxy for domestic water use), the lower the corresponding diarrhea burden of disease. s

2. Proportion of population in each domestic water use class.

The Joint Monitoring Program (WHO/UNICEF, n.d.) provides global data of basic and safely managed drinking water coverage for 2020. It distinguishes five levels of water service: 1- surface water, 2- unimproved, 3- limited, 4- Basic, 5- safely managed. To be consistent with the three classes of domestic water users we define in this work, we merge our classification with the one from the JMP as per:

Class of domestic users (as defined in this work)	Service level defined in the JMP data reporting

1- users using less than 20 liters per day per capita (lpcd) (identified by the index [<20])	"Surface water" and " unimproved"
2- users using between 20 and 50 lpdc [20-50]	"Limited" ,"basic" , "at least basic"*
3- users using more than 50 lpdc [<50]	"at least basic"* ,"Safely managed"

* We distinguish two situations for the "at least basic" indicator. In high income countries, "at least basic" is assumed to be equivalent to the class 3 (users using more than 50 lpdc). In other countries, "at least basic" is assumed to be equivalent to the class 2 (users using between 20 and 50 lpdc [20-50]).

3. Original exposure levels from Fewtrell et al. (2007) and Fry et al. (2011)

Fewtrell et al (2007) defined 7 levels of exposure. Fry et al. (2010) associated quantities of water consumption to each one of those 7 exposure levels.

Table 1: Exposure levels and associated risk ratios and daily water consumption estimates, adapted from Fry et al. (2010) based on Fewtrell et al. (2007). Lines with a green background are the exposure levels we used in this work.

Exposure scenario	Description of water and sanitation access	RR of diarrheal diseases	Daily water use per capita (L)
I	ideal situation, corresponding to the absence of transmission of diarrheal diseases through poor water, sanitation, and hygiene. This situation does not exist in reality.	1	
II	population having access to improved water supply and sanitation services in countries where more than 98% of the population is served by those services; generally corresponds to regulated water supply and full sanitation coverage, with partial	2.5	110

	treatment of sewage, and is typical in developed countries.		
III	(c) IV plus continuous piped water supply (b) IV and improved personal hygiene (a) IV plus improved water quality (point of use treatment)	no estimate 3.75 4.5	110 70 50
IV	population having access to improved water supply and improved sanitation in countries where less than 98% of the population is served by water supply and sanitation services, and where water supply is likely not to be routinely controlled.	6.9	50
Va	population having access to improved sanitation but no improved water supply in countries where less than 98% of the population is served by water supply and sanitation services, and where water supply is likely not to be routinely controlled.	6.9	35
Vb	population having access to improved water supply, but not served with improved sanitation in countries which are not extensively covered by those services, and where water supply is not likely to be routinely controlled (less than 98% coverage).	8.7	20
VI	no improved water supply and no basic sanitation in a 11 10 country where less than 98% of the population is served by those services and where water supply is not routinely controlled	11	10

To calculate the cEF in our water classes ([<20],[20-50],[>50]), which represent ranges of water consumption per day, we need to "pair" exposure levels whose associated water consumption represents a boundary of the water classes (e.g 20 Lpcd or 50 Lpcd for class [20-50]). For the class [<20] ([>50] respectively) we needed not only the exposure level corresponding to 20 Lpcd (50 Lpcd) but also the exposure level linked to the lowest (highest) water consumption estimates. Therefore we kept only the exposure levels that could be associated to create the 3 classes as per Table 2.

Table 2: Levels of exposure from Table 1 that are used and paired in this work, to calculate the cEF per class of users.

Class of users	Levels of exposure paired to calculate the cEF [daily water consumption in the levels of exposure]
[<20]	Level Vb [20 lpcd] and below (VI [10 lpcd])
[20-50]	Level Vb [20 lpcd] and IV [50 lpcd]
[>50]	Level IV [50 lpcd] and above (II [110 lpcd])

4. Class specific effect factors

We defined and used three classes of users to represent the whole population (<20, 20-50, >50). We also calculated values of cEF for other classes [10-35], [10-50] (cf. Figure 2).

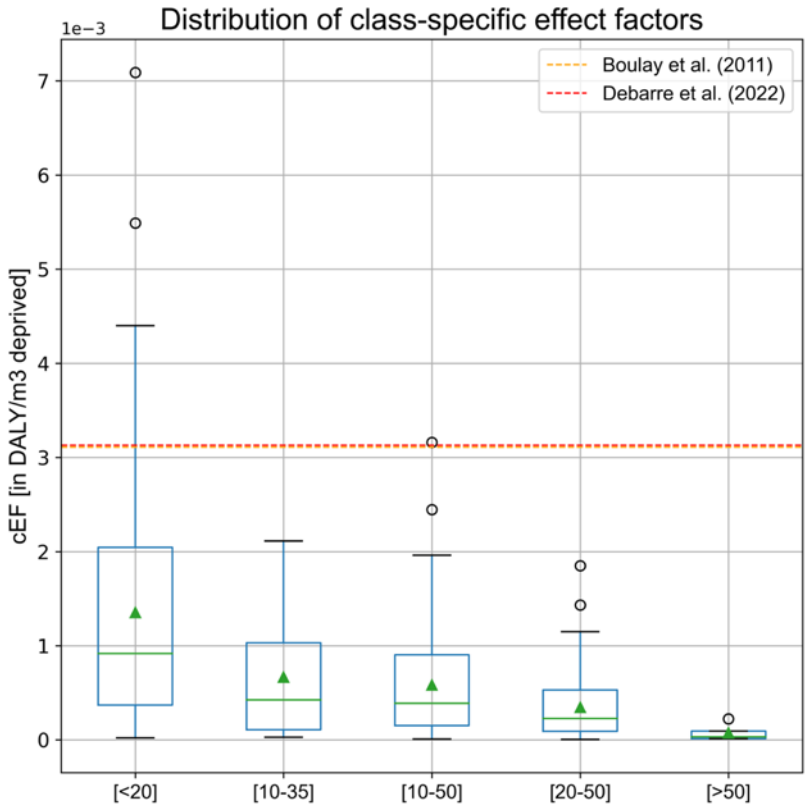


Figure 2: Distribution of class-specific EF (cEFi,j) per country, shown per class of users. Green triangles indicate the arithmetic mean ($cEF_{i,global}$) and green lines indicate the median value. Classes of users sometimes overlap and can not be used together to represent the total population (e.g. [10-50] and [20-50]).

Class of population	<20	10-35	10-50	20-50	>50
count	107	10	118	110	4
mean	1.35e-03	6.67e-04	5.82e-04	3.44e-04	7.53e-05
std	1.33e-03	7.17e-04	5.90e-04	3.45e-04	1.00e-04
min	2.28e-05	2.78e-05	1.02e-05	5.95e-06	1.36e-05
max	7.09e-03	2.11e-03	3.16e-03	1.85e-03	2.23e-04

5. Effect factor: statistics per sub-region

The body of manuscript include a table of effect factor per regions. The following Table 3 provides a similar table including sub-regions.

Table 3: Effect factor statistics per sub-regions

	mean	count	std	min	max
Australia and New Zealand	7.55e-05	2	2.72e-07	7.53e-05	7.57e-05
Central Asia	2.00e-04	5	1.01e-04	8.92e-05	3.52e-04
Eastern Asia	1.88e-04	8	1.37e-04	7.53e-05	3.94e-04
Eastern Europe	1.14e-04	10	4.68e-05	7.73e-05	2.25e-04
Latin America and the Caribbean	2.17e-04	46	1.18e-04	7.53e-05	5.80e-04
Melanesia	4.80e-04	5	2.88e-04	9.13e-05	8.72e-04
Micronesia	1.98e-04	6	1.74e-04	7.53e-05	4.85e-04
Northern Africa	2.72e-04	7	1.24e-04	1.39e-04	4.74e-04
Northern America	1.07e-04	5	5.59e-05	7.65e-05	2.07e-04
Northern Europe	8.62e-05	15	1.09e-05	7.53e-05	1.09e-04
Polynesia	1.96e-04	9	8.15e-05	8.20e-05	3.44e-04
South-eastern Asia	2.98e-04	11	1.46e-04	7.53e-05	4.71e-04
Southern Asia	3.19e-04	9	1.23e-04	9.73e-05	5.06e-04
Southern Europe	1.03e-04	16	3.18e-05	7.53e-05	1.85e-04
Sub-Saharan Africa	4.85e-04	50	1.45e-04	9.06e-05	7.30e-04
Western Asia	1.62e-04	18	1.12e-04	7.53e-05	4.48e-04
Western Europe	7.67e-05	9	1.77e-06	7.53e-05	8.01e-05

6. Effect factor: proxies of country scale EF for countries where pi values are not available

Values of the proportion of population in each class of domestic users (pi), that are estimated via the "service level" as defined in the JMP data, were not available for 26 countries (see table below). We assigned the sub-regional value of EFs as a proxy for these missing values. Because of its singularity, no proxy has been assigned to Antarctica.

ISO3 missing ES values	Name	Region	Sub-region	Proxy (by sub-region)
ABW	Aruba	Americas	Latin America and the Caribbean	2.17E-04

AIA	Anguilla	Americas	Latin America and the Caribbean	2.17E-04
ALA	Åland Islands	Europe	Northern Europe	8.62E-05
ANT	#N/A	#N/A	#N/A	#N/A
ARG	Argentina	Americas	Latin America and the Caribbean	2.17E-04
ATG	Antigua and Barbuda	Americas	Latin America and the Caribbean	2.17E-04
BHS	Bahamas	Americas	Latin America and the Caribbean	2.17E-04
COK	Cook Islands	Oceania	Polynesia	1.97E-04
COM	Comoros	Africa	Sub-Saharan Africa	4.85E-04
DMA	Dominica	Americas	Latin America and the Caribbean	2.17E-04
ERI	Eritrea	Africa	Sub-Saharan Africa	4.85E-04
ESH	Western Sahara	Africa	Northern Africa	2.72E-04
FLK	Falkland Islands (Malvinas)	Americas	Latin America and the Caribbean	2.17E-04
GNQ	Equatorial Guinea	Africa	Sub-Saharan Africa	4.85E-04
GRD	Grenada	Americas	Latin America and the Caribbean	2.17E-04
HRV	Croatia	Europe	Southern Europe	1.03E-04
JEY	Jersey	Europe	Northern Europe	8.62E-05

KNA	Saint Kitts and Nevis	Americas	Latin America and the Caribbean	2.17E-04
MSR	Montserrat	Americas	Latin America and the Caribbean	2.17E-04
SJM	Svalbard and Jan Mayen	Europe	Northern Europe	8.62E-05
TCA	Turks and Caicos Islands	Americas	Latin America and the Caribbean	2.17E-04
TKL	Tokelau	Oceania	Polynesia	1.97E-04
TWN	Taiwan, Province of China	Asia	Eastern Asia	1.88E-04
VAT	Holy See	Europe	Southern Europe	1.03E-04
VCT	Saint Vincent and the Grenadines	Americas	Latin America and the Caribbean	2.17E-04
ZWE	Zimbabwe	Africa	Sub-Saharan Africa	4.85E-04

7. Region-wide characterization factors

- **Computation at the subriver basin level**

CFs are calculated at the river basin (assuming constant national XF and EF). To do so, CF are calculated combining maps of FF (defined at the sub river basin level), XF (defined at the country level) and EF (defined at the country level). A python script has been created for this computation task and can be shared on request along with shapefiles of the CFs.

- **Aggregation method**

While we provide CF at the subriver basin level, current life cycle inventory databases limit their use due to coarse regionalization of water inventory flows. For instance, in the ecoinvent database (Wernet et al. 2016), different geographies of

inventory flows are developed (‘countries’, 'aluminium', 'legacy', 'states', 'electricity','un-subregions', 'special', 'russia', 'un-regions','usa-electricity'). To be operationalized, CF must be aggregated by regions. Regional-scaled CFs are calculated by aggregating CF by water consumption; regional-scaled CF are calculated as the weighted sum of CF (subriver basin) multiplied by the percentage of water use in the subriver basin compared to water consumption in the region.

Water consumption data is provided by the WaterGAP model (Müller Schmied et al. 2020) at the subriver basin level. Boundaries of subriver basins do not match with those of ecoinvent geographies.
To calculate regions-scaled CFs, shapefiles of water consumption and subriver basin CFs are intersected, leading to smaller "sub-subriver basins". Water consumption in sub-subriver basins is estimated based on water use in the subriver basin, and on the surface of the “sub-subriver basin” compared to consumption subriver basin (Water consumption in sub-subriver basin = area sub-subriver basin/area subriver basin * water consumption subriver basin). We assume here that water consumption is proportional to surface.

Then, regional-scaled CF are calculated as per : $\sum_i CF_i * \frac{Water\ consumption_i}{Water\ consumption\ in\ region}$.
CFs at the subriver, global, country and ecoinvent regions levels are provided in the ESM1.

- **Country-specific CFs**

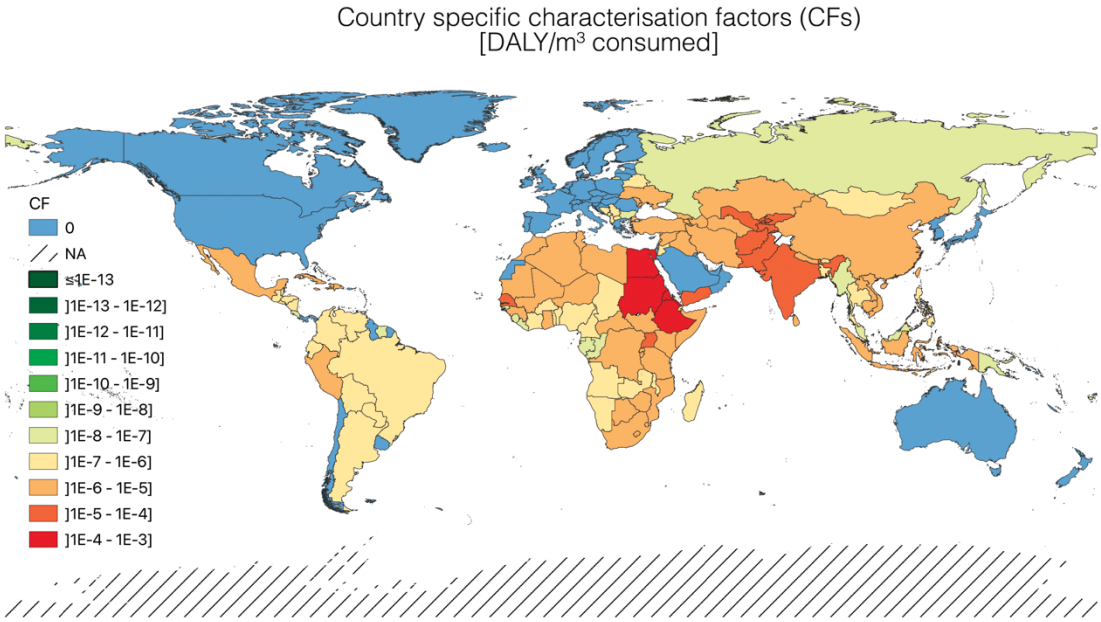


Figure 3: Country-wide characterization factors

Figure 3 shows the map of country-specific CFs calculated as the weighted sum of CF at the sub-watershed level, weighted by water consumption. Table of values is provided is the ESM2.

- **Global CF**

Global CF is equal to 2.00e-5 DALY/m^3 consumed.

8. Sanity check : normalization

To assess our results, we calculated the total global potential impact using the CFs we developed and tested the order of magnitude of the result.

We used data of water consumption from the WaterGAP 2.2d model (Müller Schmied et al. 2020). The latter model provided data on water consumption for different sectors and different months, per river basin. We compiled the total water consumption over one year per river basin.

The total water consumption (summing all river basins) reaches 1.38e+12 m^3 globally.

As per the body manuscript, total potential impact reaches **2.77e+7** DALYs for year 2019.

9. Comparison of CF results

- **Debarre et al. (2022) vs. revised CFs.**

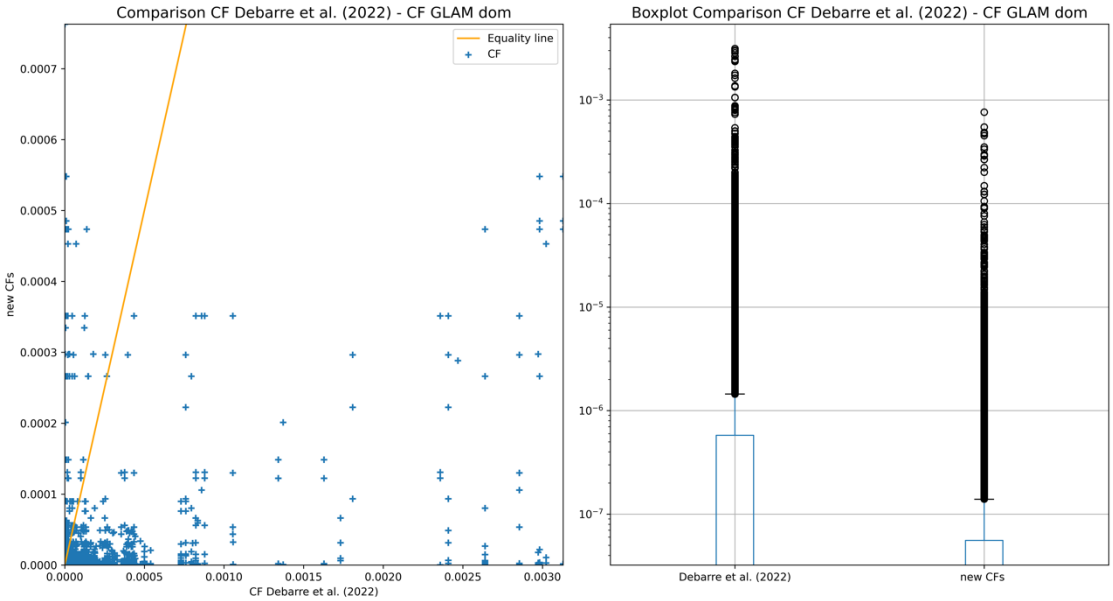


Figure 4:Comparison of characterization factors at the subriver basin scale: Debarre et al. (2022) vs this paper.

Figure 4 shows the comparison of the CF with the CF developed in Debarre et al. (2022) The yellow line shows the equality line ; points at the right of the equality line indicate that new CFs are lower than Debarre et al. (2022)'s CFs. the concentration of points to the right of the equality line shows a downward trend in factors. Boxplots on the right show the distribution of CFs in both papers that show the same trend.

- **Agricultural CF (GLAM1) vs revised CFs.**

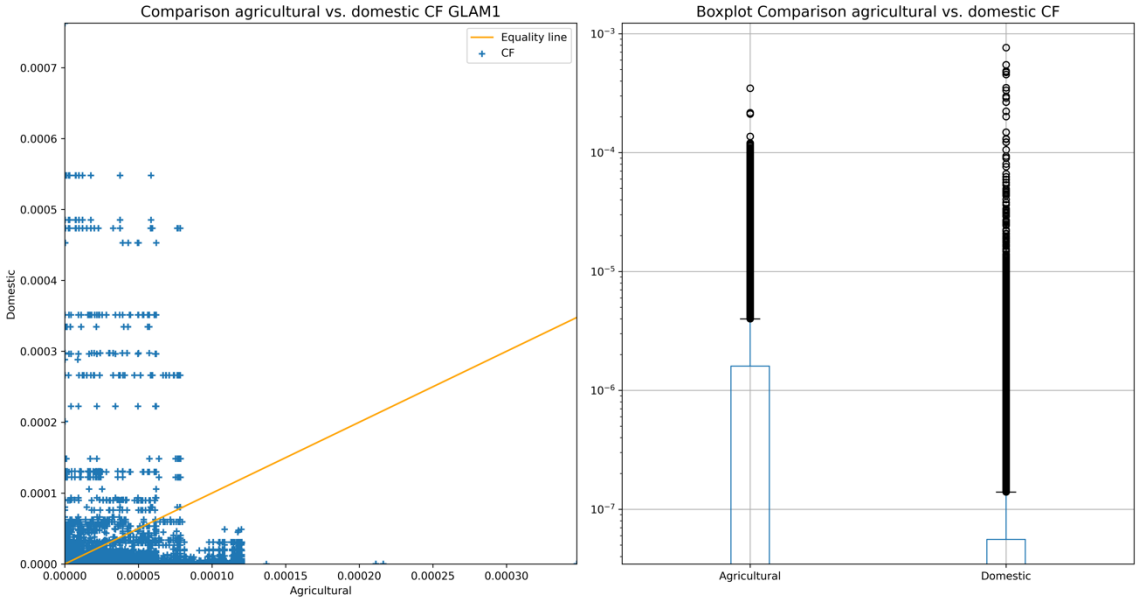


Figure 5:Comparison of characterization factors at the subriver basin scale: Motoshita et al. (2016) vs this paper. Motoshita et al. (2018) provide CF assessing the potential impacts due to agricultural water deprivation. Yellow line show the equality line. Boxplots on the right show the distribution of CFs in both papers.

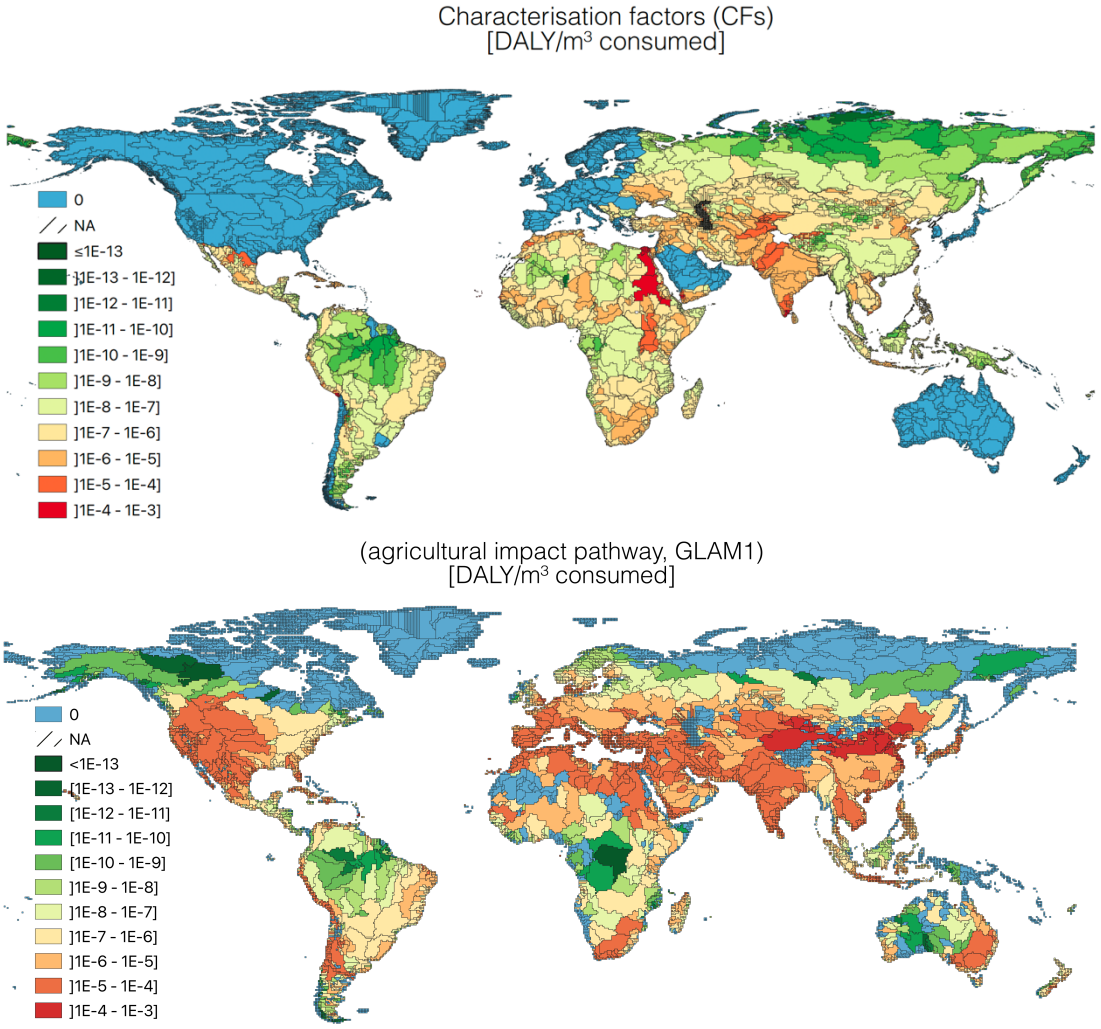


Figure 6: Comparison of water deprivation characterization factors : domestic (this work) vs. agricultural (Motoshita et al. (2018)) water deficit.

Figure 6 shows the comparison of the computed domestic water deficit characterization factors with the agricultural deficit characterization factors calculated by (Motoshita et al. 2016). The trends are not fully identical. Domestic water deprivation is a local issue; the water is consumed, depleted and deprived in the same region. Considering economic adaptation capacities based on the GNI per capita for the domestic impact pathway leads to high values of the CFs in poor regions where competition for water is important.

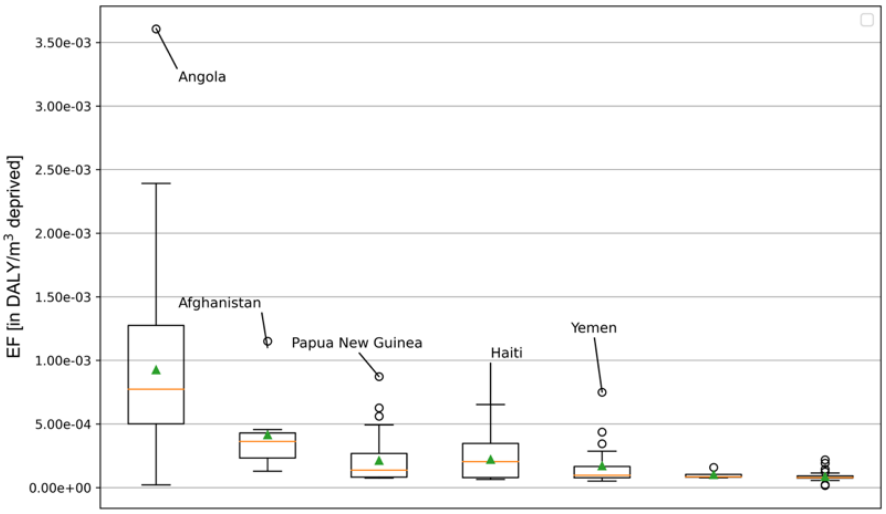
In the agricultural model however, trade dimensions are considered to account for the trade of crops across the world. Therefore, water competition in a high-income

country can lead to reduced crop productions and lead to malnutrition issues in other part of the world (namely in low-income countries).

10. Sensitivity analysis: cEF per country

Due to limited data availability, we used generic, global $cEF_{i,global}$ to be computed in country-specific EFs in our model. Nonetheless, country-specific $cEF_{i,j}$ (see Figure 1 of main manuscript) could be relevant to consider disparities in terms of vulnerability to health impacts due water deprivation. One could consider that, due to different levels of access to healthcare, or different hygiene practices habits, two countries could deal differently with the health consequences of depriving people of the same volume of domestic water. In the following sensitivity analysis, country-specific $cEF_{i,j}$ were used to be computed in country-specific EF and sub-river basin CFs.

- **Effect factor calculated using country specific cEF**



	Sub-Saharan Africa	South Asia	East Asia and Pacific	Latin America and Caribbean	Middle East and North Africa	North America	Europe and Central Asia
mean	9.25e-04	4.14e-04	2.13e-04	2.21e-04	1.70e-04	1.01e-04	8.59e-05
count	50	8	41	45	20	4	59
std	6.66e-04	3.20e-04	1.77e-04	1.42e-04	1.71e-04	3.82e-05	3.07e-05
min	2.08e-05	1.29e-04	7.42e-05	6.35e-05	5.08e-05	7.65e-05	1.61e-05
max	3.61e-03	1.15e-03	8.72e-04	6.53e-04	7.50e-04	1.58e-04	2.19e-04

Figure 7: Distribution of country-specific EFs calculated using country specific class-specific EF cEF. EFs are grouped by regions. Green triangles indicate arithmetic mean. Orange line indicated the median value.

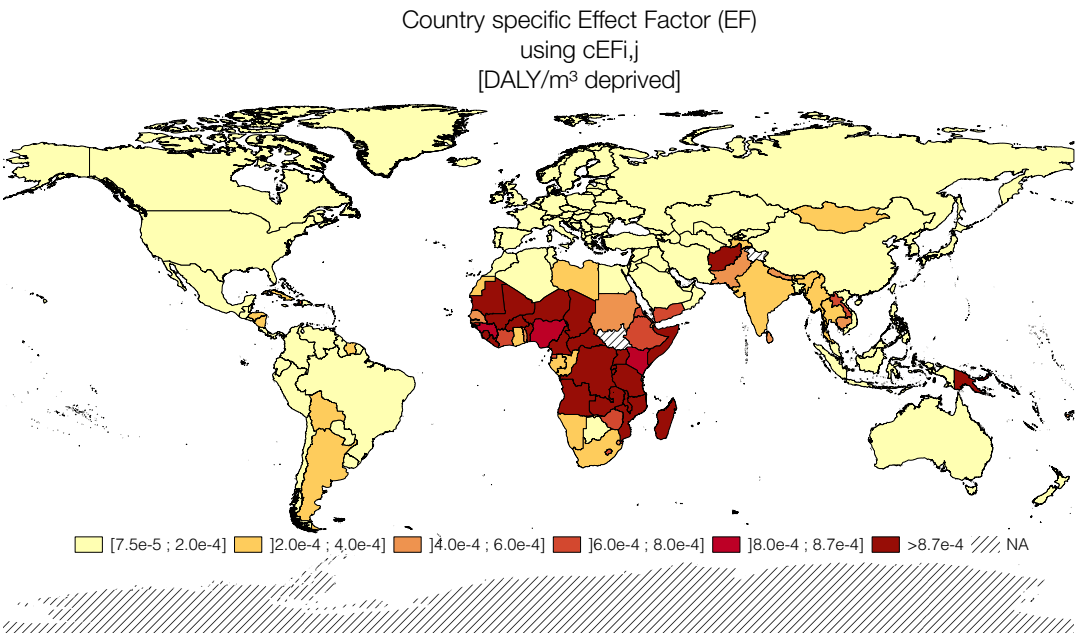


Figure 8: Country-specific EF using country specific class specific EF ($cEF_{i,j}$).

Figure 7 below shows the distribution of country-specific effect factors country, also represented on a global map in Figure 8.

- **Comparison with EF calculated using generic cEF**

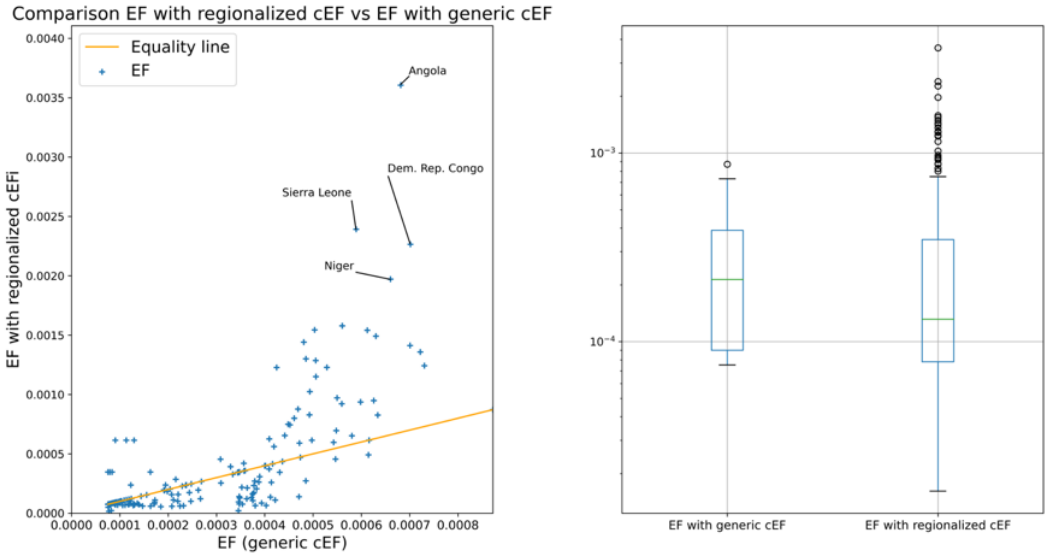


Figure 9: Comparison of effect factors : EF using generic $cEF_{i,global}$ vs EF computed with country-specific $cEF_{i,j}$.

Figure 9 shows the comparison of EFs calculated using generic cEF (x axis) and using country specific cEF (y axis). The equality line allows to visualize the variation of EFs when computed with generic and specific cEFi. this variation is particularly pronounced in the case of Angola, the Democratic Republic of Congo, Sierra Leone and Niger for which the EF values calculated with specific cEFi are higher.

- **Characterization factors**

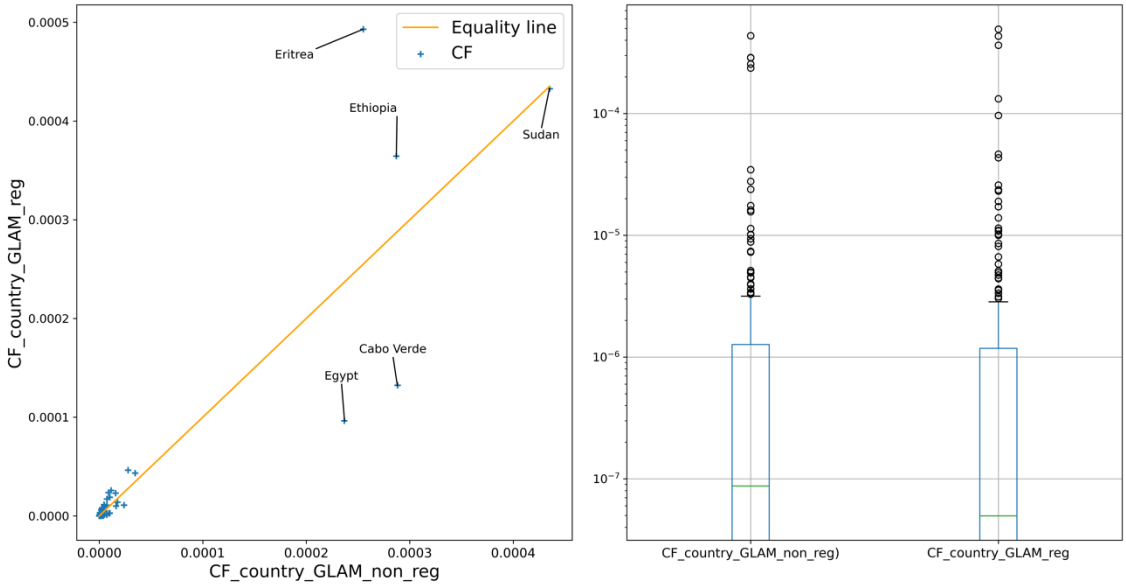


Figure 10: Comparison of characterisation factors : CF calculated with EF using generic $cEF_{i,global}$ vs CF calculated with EF computed with country-specific $cEF_{i,j}$.

Figure 10 shows the comparison of CF computed with EFs calculated using generic (non-regionalized) cEF (x axis) and using country specific (regionalized) cEF (y axis).

11. Data management and availability

A code is available to recompute CFs and can be shared on request.

Required data can be downloaded on a Borealis repository, available on request.

REFERENCES ASSOCIATED WITH THE ANNEXE E

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