The International Journal of Life Cycle Assessment (2023) 28:367–379
https://doi.org/10.1007/s11367-023-02136-2

A new impact pathway towards ecosystem quality in life cycle assessment: characterisation factors for fisheries

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Received: 31 May 2022 / Accepted: 17 January 2023 / Published online: 17 February 2023
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Abstract

Purpose Although life cycle impact assessment methods exist for quantifying land use and its impact on the environment in the "ecosystem quality" area of protection, the impact of sea use on ecosystems has been poorly assessed so far. This paper aims to propose operational characterisation factors for all global fisheries.

Methods For a given intervention, the characterisation factor is defined as the product of the fate factor (inverse of the fish stock growth rate) and the effect factor (depleted fraction of the stock). Characterisation factors are provided for 5000 fish stocks identified by the Food and Agriculture Organization. Both the marginal and average approaches are used, and characterisation factors compatible with the ReCiPe method and the international guidelines of the Life Cycle Initiative hosted by the UN Environment Programme are proposed.

Results and discussion Characterisation factors for regional and global assessments can be employed to address the endemic nature of a species. As an illustration, four contrasting fisheries are presented and compared with land animal production systems. Impacts varied between stocks and between regional and global assessment, particularly with highly endemic species exhibiting impacts comparable to or exceeding land-based animal products.

Conclusions Although in some cases associated uncertainty is large, the proposed method allows endpoint characterisation, in line with the ReCiPe methodology and Life Cycle Initiative, contributing the assessment of fishing impacts on ecosystem quality and a more holistic representation in food impact assessment.

Keywords Stock dynamic model · Overfishing · Impact pathways · LCIA · Biodiversity · Catch

1 Introduction

Humans have always used the land and the sea as food sources. Terrestrial ecosystems have been intensely modified for agricultural purposes since the Neolithic period (Ellis et al. 2013). Although fishing activities date from even further back, impacts on marine communities remained localised and rarely overexploitative until the industrialisation of fishing activities enabled global expansion at an unprecedented scale (Jackson et al. 2001). Regrettably, this intensification of fishing over the past century has rapidly altered the situation. The impact of fisheries on the marine ecosystem has been only relatively recently assessed, but it has undoubtedly been massive for several decades. Fishing has thus modified all marine ecosystems (Pauly 1998), through habitat modification, top-down restructuring of the trophic web (Steneck et al. 2002), reduction of functional redundancy and the ability to provide critical ecosystem functions (Bellwood et al. 2004), ultimately affecting the resilience of the ecosystem to withstand disturbances. Every 2 years, the Food and Agriculture Organization (FAO) provides a detailed report on the state of the world fisheries and aquaculture, addressing all these issues (FAO 2022).

Life cycle assessment (LCA) is the reference approach when addressing the global impacts of products and services. With LCA, practitioners commonly quantify the environmental impacts on three areas of protection (AoP): human health, natural resources and ecosystem quality.

Communicated by Ian Vázquez-Rowe.

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On one hand, LCA has been applied to quantify land use by human activities and its consequences on ecosystems (third AoP). Several approaches have been proposed for this purpose, for example, based on the soil organic carbon content (Milà i Canals et al. 2007) or on biodiversity (de Baan et al. 2013b; Chaudhary et al. 2015; Winter et al. 2018; Chaudhary and Brooks 2018). On the other hand, the LCA community has not yet adequately assessed the impact of sea use on ecosystems.

Sea use influences the two AoPs: natural (biotic) resources and ecosystem quality (Langlois et al. 2014b). The resource AoP is mainly assessed through the human appropriation of the net primary production (Cashion et al. 2016). The same descriptor was used by Langlois et al. (2015) for defining a pathway towards ecosystem quality. Other approaches dedicated to the resource AoP have been proposed, with characterisation factors (CFs) (Langlois et al. 2014a) or indicators (Emanuelsson et al. 2014) based on fishery management parameters, distance-to-target approach (Bach et al. 2022) or characterisation factors (CFs) defined from stock dynamic models (Emanuelsson et al. 2014; Hélias et al. 2018).

Recently, CFs were proposed to assess the impact of seabed destruction on ecosystem quality (Woods and Verones 2019). This promising approach incorporates seafloor destruction as a form of habitat modification as an additive impact of sea use due to trawling in the current life cycle impact assessment (LCIA) framework. However, to the authors' knowledge, no existing approach assesses the impact of biomass removal by fishing on an ecosystem, in compliance with current LCIA guidelines (Verones et al. 2017), despite (over)exploitation representing one of the major causes for the decrease in biodiversity in the oceans (Woods et al. 2016; IPBES 2019). This lack of indicators has been highlighted when comparisons were made between marine- and agricultural-based products: the impacts are not expressed in the same units and are not comparable, which, undoubtedly, represents an important issue for food impact assessment. The present work aims to solve this issue of inconsistency by proposing operational global fishery CFs for characterising ecosystem quality and allowing sea use and land use to be addressed within a single AoP. These CFs comply with international guidelines (Verones et al. 2017) and units, where the inventoried catch (weight of fish) is converted into an ecosystem quality impact. Furthermore, they result from the extension of a recent study on biotic resource depletion (BRD) (Hélias et al. 2018) where the depleted stock fraction (DSF) can provide a tenable link to the ecosystem quality AoP in a manner comparable to potentially affected fraction (PAF) already employed in other impact pathways.

2 Methods

2.1 Ecosystem quality: units and land use models

The Life Cycle Initiative (LC-Initiative) which is hosted by the UN Environment Programme recommends CFs for ecosystem quality AoP in its guidelines (Verones et al. 2017). This results in a potentially disappeared fraction of species over a given time (PDF.year). For the impact of land use, an approach (Chaudhary et al. 2015) has been selected according to the countryside species-area relationship (SAR) model (Pereira et al. 2014). The CFs proposed by Chaudhary et al. (2015) have been updated (Chaudhary and Brooks 2018). The LC Impact method (Verones et al. 2020) was developed using the Chaudhary et al. (2015) framework but with a final metric conversion to PDF at the endpoint. Note that the ReCiPe method (Huijbregts et al. 2016, 2017) uses a former model (de Baan et al. 2013a; Curran et al. 2014) and defines a different (but related) unit based on the number of species that have disappeared over a given time (species.year).

These selected land use CFs are defined by marginal and average approaches to represent the occupation and transformation of a parcel of land (4 sets of CFs are obtained, expressed in lost species/m^2 for occupation and lost species.year/m^2 for transformation). They address the potential species loss resulting from human use of an area per ecoregion. The data obtained for five taxa (mammals, reptiles, birds, amphibians and vascular plants) are then aggregated. This leads to the establishment of regional CFs (expressed in PDF/m^2 for occupation and PDF.year/m^2 for transformation) which assess the loss of the intrinsic function of ecosystems at a regional scale (Frischknecht and Jolliet 2016). Global CFs (expressed in global PDF/m^2 for occupation and global PDF.year/m^2 for transformation) are also provided. They assess the global (and irreversible) loss of the proportion of species in the ecosystem through a vulnerability score (Verones et al. 2015). An improvement of this score has recently been proposed (Kuipers et al. 2019; Verones et al. 2022), defining the "global extinction probability". This latter estimation is more accurate for converting regional CFs into global CFs, by ensuring the regional summation of the conversion factors equals one. The regional fraction of lost species can therefore be translated to a global-scale species extinction potential; aquatic species are also included where previously not. However, the purpose and the main outlines of the design remain the same. See supplementary information for a brief description.

The rationale here is to use a similar approach for fisheries: indeed, the lost species, the related regional PDF and finally the global PDF can be determined from fish stock depletion. Both PDFs can result from marginal and average approaches.

2.2 Marginal CFs for biotic resource depletion

In LCA, abiotic natural resources can be assessed in different ways (Berger et al. 2020; Sonderegger et al. 2020), although depletion is the criterion that is most often investigated. For biotic resources, the depletion of the stock (a species in a habitat) is intrinsically based on its renewability, which depends on the replenishment capacity of living organisms relative to their withdrawal due to human activities. An approach was proposed in a recent study by Hélias et al. (2018) addressing global fisheries as resource depletion. The CFs are based on a marginal approach involving a population model dynamic in order to link the inventory (fish withdrawal) with the impact (stock depletion), which is briefly reported here. The frequently used fish stock dynamics Schaefer model shape (Schaefer 1954) is the foundation of this study:

$$\frac{dB}{dt} = -C + rB \times DSF \tag{1}$$

where B is the fish biomass (tonne), C the annual catch (tonne.year^{-1}), r the growth rate (year^{-1}) and DSF the depleted stock fraction. The latter varies from 0 for a plentiful stock to 1 when it is exhausted. This model illustrates the growth where exponential expansion (rB) is limited by available habitat represented by DSF. The Schaefer model is based on the well-known logistic law of growth and in this case:

$$DSF = 1 - \frac{B}{K} \tag{2}$$

$$DSF = \frac{C}{rB} \tag{3}$$

Two approaches are generally used and recommended in LCA to define CFs, representing a marginal or average change (Frischknecht and Jolliet 2016). A third approach, the linear one, is sometimes used when the current state (as the background concentration for a pollutant) is unknown (Hauschild and Huijbregts 2015), but this is not the case in the present study. Hélias et al. (2018) provide CFs for biotic resource depletion through a marginal approach only ($CF_{BRD,M}$). This is defined as the partial derivative of the impact (∂DSF) according to the inventory (mass of fish removed from the biomass stock, $-\partial B$):

$$CF_{BRD,M} = -\frac{\partial DSF}{\partial B} = \frac{C}{rB^2} \tag{4}$$

See Hélias et al. (2018) for additional details. CF values are provided for all fisheries described in FAO data (global scale). Recently (Hélias and Heijungs 2019), model consistency has been observed between this approach and the abiotic depletion potential (Guinée and Heijungs 1995) (the most commonly used approach to assess abiotic resource depletion in LCA).

2.3 Average CFs for biotic resource depletion

A marginal CF allows for a small change to be assessed from the current situation. However, an average approach is better adapted to address greater changes (often defined as > 5% of the issue as a whole), and both sets of CFs should be provided for an LCIA method (Frischknecht and Jolliet 2016). The CF for average (A) biotic resource depletion ($CF_{BRD,A}$) is defined in LCIA as the average slope of the causal relationship between the inventory E, which is the quantity of fish removed, and the impact, i.e. DSF in this study. This is equivalent to the division of the impact by the overall human intervention (E) (Curran 2017):

$$CF_{BRD,A} = \frac{DSF}{E} \tag{5}$$

To define E as a biotic resource, the timeframe where the catch (extraction rate) occurs needs to be defined. When the system reaches a steady state, the quantity of fish removed is represented by the catch during a timeframe τ:

$$E = C \times \tau \tag{6}$$

On one hand, a too long timeframe does not make sense, as the dynamics of the stock counteract with former withdrawals, which do not affect the current state anymore. On the other hand, a too short timeframe could overlook a part of the human interventions leading to the current state. This timeframe cannot be identical for all stocks and needs to be determined according to the current population resilience, based on its replenishment rate. In dynamical system theory, the responsiveness of a linear-time invariant system is given by its time constant. By analogy, in this study,τ is the time constant of the stock:

$$\tau = \frac{1}{rDSF} \tag{7}$$

E is therefore the quantity of fish removed, corresponding to the current pressure delivered over a given period by the capacity of the stock to counteract changes. When Eqs. (3) and (5–7) are combined, E is equal to B, and the characterisation factor for biotic resource depletion is as follows:

$$CF_{BRD,A} = CF_{BRD,M} = \frac{C}{rB^2} \tag{8}$$

By defining human intervention as the catch over the time constant of the stock, both marginal and average approaches have the same value and a unique set of CFs is provided.

2.4 From depleted stock fraction to ecosystem quality

The impacts affecting ecosystem quality are generally addressed with CF = FF × EF. For a given intervention, the impact is characterised by the product of the fate factor (FF) and the effect factor (EF). The first represents the time period during which the effect occurs, while the second characterises the associated effect. The more detailed relationship CF = FF × XF × EF is often used, although the exposure factor (XF), relating to a toxicity impact, is not relevant for fisheries.

2.4.1 Depleted stock fraction as effect factor

For a biotic resource, an analogy can be observed between the depletion of the resource and the biodiversity impact. Hence, fishing leads to a loss in biodiversity, due to the withdrawal of a part of the living biomass. The *DSF* represents the disappeared fraction of the target stock (a given commercially fished species in its habitat), and from this, the unit for CF_{BRD} can then be defined as the amount of lost target species/tonne.

It is noteworthy that the shape of the equation of the *DSF*, as defined by the Schaefer model, resembles a modelled effect factor for terrestrial acidification (Azevedo et al. 2012; Crespo-Mendes et al. 2019) where the potentially non-occurring fraction is only defined at a biotic community level and not for a specific species of a habitat.

2.4.2 Fate factor

Most of the impacts traditionally quantified in LCA studies that affect ecosystem quality (e.g. ecotoxicity, acidification, eutrophication), result from substance emissions. In this context, the fate factor represents the persistence of a given substance in the media (Cosme et al. 2018). It is usually expressed in years or days. The fate factor is thus driven by transfers between compartments and by substance degradation. For a given compartment, it can be expressed as the inverse of the sum of the removal rates (Cosme et al. 2018) or as a residence time (Rosenbaum et al. 2007).

Since it results from a resource withdrawal rather than an emission, the fate factor of fisheries proposed in this paper is inverted. The principle components of the characterisation factor, however, remain the same where the effect factor represents the impact and the fate factor, its duration. In USEtox®, fate factors are expressed as the inverse of exchange and removal rate constants (Bijster et al. 2018), which is known as the mean lifetime for an exponential law. In the present instance, the model is more complex. The carrying capacity in the model introduces a non-linearity, and the mean lifetime of the model is consequently a function of the magnitude of the elemental flow. In order to avoid this incompatibility with the principles of LCA, where the CF is constant whatever the inventory value, the model can be linearised at the steady state. The fate factor is then defined as

$$FF = \frac{1}{r} \times \frac{K}{B} \tag{9}$$

i.e. the inverse of the growth rate constant tempered by the inverse of the relative biomass. See supplementary materials for details.

2.5 Characterisation factors

The regional CF for the impact of fish catches on ecosystem quality ($CF_{EQ,reg}$, expressed in species.year/kg of fish) is therefore expressed as follows:

$$CF_{EQ,reg} = \frac{K}{rB} \times \frac{C}{rB^2} = \frac{CK}{r^2B^3} \tag{10}$$

This CF is both marginal and average as previously discussed. The species.year unit is used with the ecosystem AoP in the ReCiPe endpoint method (Huijbregts et al. 2016, 2017), and therefore, the impacts on fisheries ecosystems can be directly added to this method. Note that similar to the approach for the land use impact category, this study does not differentiate between the three perspectives of the ReCiPe endpoint method (individualist, hierachist and egalitarian).

The conversion from species.year/kg to regional PDF.year/kg can be easily made by dividing CF_{EQ} by the total number of species in marine regions (233 302) (Horton et al. 2019). The reverse approach is used in the ReCiPe endpoint method to convert PDF.year into species.year. Global CFs ($CF_{EQ,glo}$) should also be provided, as stated by the LC-Initiative guidelines (Verones et al. 2017). From a modelling point of view, the main difference between land use and fisheries lies at the level of intervention. The land use impact is related to a spatial change and affects all species in the corresponding area. For fisheries, the CF is defined for specific, targeted species in a given ecosystem (i.e. the population). In contrast to land use, when using a stock-based modelling approach, the scope of the human intervention through fishing does not include indirect effects on the ecosystem and all of its communities; it solely affects one species, i.e. the caught species. If various species can be caught simultaneously within an ecosystem, the corresponding impacts are additive and assessed separately through inventory flows and associated CFs in the LCA framework.

By considering a PDF linked to the midpoint through the DSF (analogous to PAF) rather than a change in absolute species richness, it is possible to have a representation of species level abundance impacts with are critical to fisheries quantified within the CF.

At population (fish stock) levels, the conversion factor to obtain global PDF from regional PDF only quantifies the endemic character of a given species in given region. This approach is simpler than for the ecosystem level applied to land use. With a reasoning similar to that used for calculating the vulnerability score (Verones et al. 2015) or the global extinction probability (Kuipers et al. 2019; Verones et al. 2022) (except that it takes place at the species level), the endemic conversion factor for obtaining the global PDF from the regional PDF is $B_j/\sum_k B_k$, i.e. the proportion of global biomass in an ecoregion j. The impact can thus be expressed, using all units recommended by LC-Initiative guidelines (Verones et al. 2017), with

$$CF_{EQ,glo,j} = \frac{B_j}{\sum_k B_k} \times CF_{EQ,reg,j} \tag{11}$$

2.6 Operationalisation

Most fish stocks have been poorly described, and the quantification of stock descriptors required to compute CFs in Eq. (10) remains a challenge. To address this issue, the CMSY algorithm (Froese et al. 2017) was chosen, following the methodology described in Hélias et al. (2018) and Hélias (2019). This allows for a global-scale estimation of stock descriptors from catch time series provided by the FAO (2017) and resilience available in FishBase (Froese and Pauly 2016). Estimations of C, r, K and B values are thus provided for all fisheries reported by FAO, considering a stock as a species in an FAO area. The complete description of the approach, its relevance, the management of multi-stock datasets (stocks merging more than one species or more than one habitat) and poor-data stocks (when the available FAO data do not allow the use of CMSY) have been previously discussed in Hélias et al. (2018). The reader can refer to this latter article for more details concerning the validity of biotic resource depletion for ecosystem quality impact. It is also noteworthy that due to this operationalisation and the availability of the data, the term ecoregion refers to FAO major fishing regions. Although these are arbitrary delineations rather than strictly ecological, they serve the same purpose of regionalisation of the approach within current fisheries data constraints and have therefore been considered as a proxy for ecoregions. The occurrence of multiple observed habitats in an FAO area has already been discussed in Hélias et al. (2018).

The relevance of the assessment is determined qualitatively following the approach of Hélias et al. (2018) and Hélias (2019). Results are briefly presented here, ranging from the most reliable to the least trusted:

- Class I corresponds to marine fish stocks with only one species, which have been fully assessed with the CMSY algorithm.
- Class II brings several groups together, also assessed with CMSY. Class II.a is composed of multispecies marine fish stocks with not more than five species. Class II.b lists non-fish mono-species stocks (crustacean, mollusc…). Class II.c encompasses mono-species inland stocks.
- Class III is similar to class II but with multispecies marine fish stocks with more than five species (III.a), non-fish multispecies stocks (III.b) and multispecies inland stocks (III.c).
- Class IV stocks are not directly assessed due to poor data quality. Global aggregated values are used, at species level (IV.a) or group level when values for species are not available (IV.b),

2.7 Case study

As an illustration, four fisheries products have been presented and compared to livestock products. The purpose is not to provide an extensive and accurate LCA but rather to demonstrate how this work can be used by practitioners and to highlight a few results. For this purpose, a simple functional unit has been used without taking the protein content or other nutritional aspects into account. All systems have been assessed for one metric ton of fresh products. The ecoinvent database (Wernet et al. 2016) has been used (v3.5 "allocation at point of substitution" system model implemented in Simapro® v9 software).

Tuna species are fished intensely and are easily identified by consumers. Bluefin tuna species have even been classified as endangered or critically endangered by the International Union for Conservation of Nature (IUCN). The Atlantic bluefin tuna (*Thunnus thynnus*, Scombridae) in the Eastern Atlantic was therefore selected to be assessed. For comparison, yellowfin tuna (*Thunnus albacares*, Scombridae) stocks in the Atlantic Ocean were chosen, since they seem to be surviving in better conditions (near threatened status by IUCN). The ecoinvent process "landed tuna to generic market for marine fish {global}" has been used as an inventory for both species, and only the target species and associated CF differ.

Additionally, two demersal species were also assessed, both being represented by the ecoinvent process "demersal fish to generic market for marine fish {global}". The Alaska pollock (*Theragra chalcogramma*, Dadidae) is one of the most heavily caught and consumed fish in the world. This involves the Northwest Pacific FAO stock. On the contrary, the European seabass (*Dicentrarchus labrax*, Moronidae) catches remain small, and their heavily depleted stocks are becoming an issue as the fisheries are increasingly regulated by emergency measures. The European seabass is included in the Northeast Atlantic FAO stock.

The four fisheries are compared to the terrestrial meat production systems of chicken, pork and beef ("market for chicken/swine/cattle for slaughtering, live weight, {global}" in the ecoinvent database). Impacts are derived for the ReCiPe ecosystem quality endpoint (hierarchist perspective) incorporating the computed regional fishery CFs, expressed in species.years/t. The regional and global fishery CFs also provided by this work (expressed in PDF.year/t) are then used to obtain the impact in a LC-Initiative compatible unit. The results are compared with the impacts of land use (regional and global occupation and transformation) associated with the terrestrial meat products computed with CFs provided by the LC-Initiative guideline report (Frischknecht and Jolliet 2016).

3 Results and discussion

3.1 Overview

The CFs with associated uncertainties for more than the 5 000 stocks listed in FAO data, both regional and global, are available for download at an online deposit (https://doi.org/10.5281/zenodo.3954209). The CFs are expressed in species.years/t and PDF.year/t for use and comparison with ReCiPe endpoint method and LC-Initiative guidelines, respectively.

The regional CFs span over ten orders of magnitude, but the interquartile range is less than two orders of magnitude. The median value is 2.2×10^{-4} species.year/t (9.4×10^{-10} PDF_{reg}.year/t), while the interquartile range varies between 1.85×10^{-4} and 7.2×10^{-3} species.year/t (7.96×10^{-10} and 3.8×10^{-8} PDF_{reg}.year/t). The global CFs span over 13 orders of magnitude, but here again, the interquartile range is more restrained, also covering two orders of magnitude. The global CF median is 4.4×10^{-5} species.year/t (1.9×10^{-10} PDF_{glo}.year/t), and the interquartile range is from 4.8×10^{-7} to 8.6×10^{-4} species.year/t (2.1×10^{-12} to 3.7×10^{-9} PDF_{glo}.year/t). It is noteworthy that $B_j/\sum_k B_k \leq 1$, thus implying that $CF_{EQ,glo}$ is either always less than $CF_{EQ,reg}$ or equal to it if the species is endemic.

3.2 Spatial variation

Fishing pressure does not affect all marine regions equally. Figure 1 addresses this fact by illustrating the CFs per FAO area for class I only, which includes the most reliable categories. This represents a large part of the catch for almost all of the areas. As the values cover several orders of magnitude, the weighted geometrical mean (with catch values) is used.

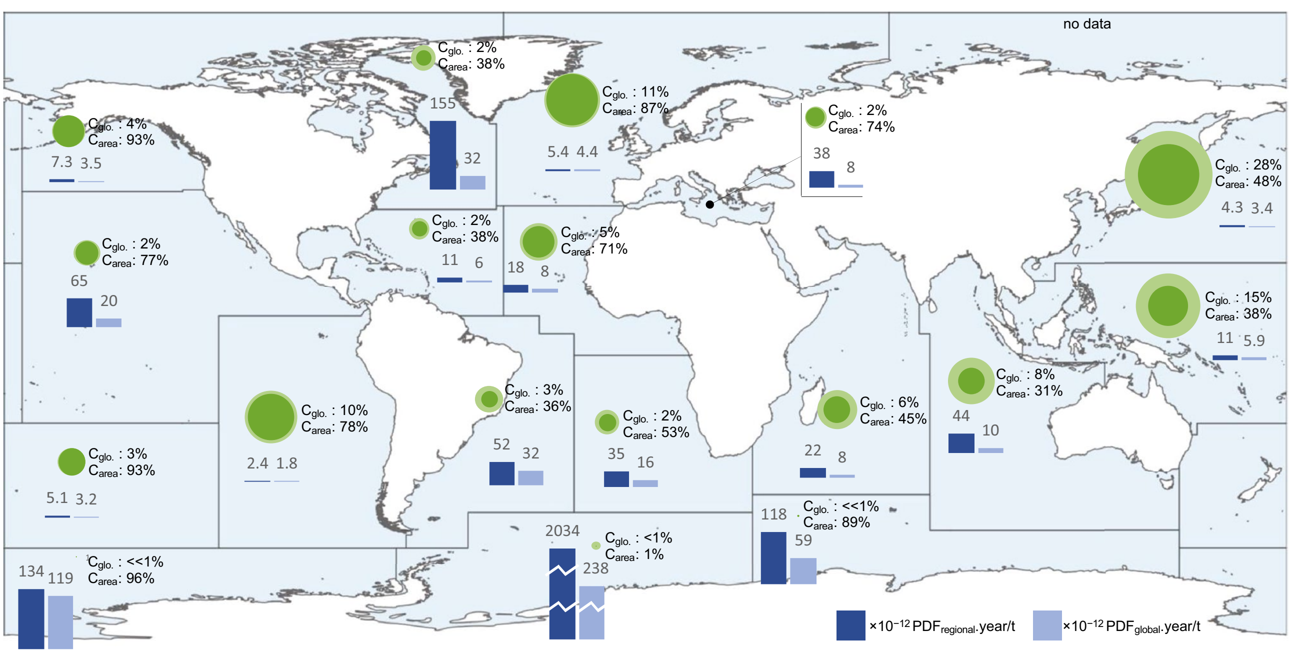


Fig. 1 Regional (dark blue) and global (light blue) weighted geometrical mean of characterisation factors per the Food and Agricultural Organization (FAO) area. Weights are defined according to catches in the area, and only mono-species fish stock directly assessed (class I) is considered. The proportion of the global catch captured in each area (C_{glo}, light green circle) and the proportion of class I stock in the area (C_{reg}, dark green circle) are provided

3.2.1 Catch–impact relationship

As previously observed (Hélias et al. 2018), the most exploited areas present lower impacts per mass of fish than the less exploited areas. For example, the Northeast Atlantic and Northwest Pacific represent 11% and 28% of global catches, respectively, but the average regional impacts per ton of fish are only 0.5×10^{-11} PDF_{reg}.year and 0.4×10^{-11} PDF_{reg}.year, respectively. On the contrary, although the Northwest Atlantic or Mediterranean Sea each encompasses 2% of the whole catch, the average impacts are 15×10^{-11} PDF_{reg}.year and 3.8×10^{-11} PDF_{reg}.year, respectively.

This result is seemingly counter-intuitive (high catch means low impact). However, heavily caught species are fished because their stocks are large and the associated fishing effort is low. For example, Peruvian anchovy (*Engraulis ringens*, Engraulidae) is the most exploited species in the world. Peruvian anchovy catch represents 67% of the Southeast Pacific and 5% of the global catches because it is relatively effortless to fish them. Except during *El Niño* events, this species thrives on an abundance of food related to the Humboldt Current. In addition, its resilience is high. This entails a very high biomass for Peruvian anchovy, and consequently, the most fished species in the world has only been classified as a species of “least concern” by the IUCN. Moreover, corresponding CFs remain relatively low in the Southeast Pacific area, with 6.6×10^{-13} PDF.year/t (this species is only found in this area, which means that global and regional CFs are identical). The average impact in this area is thus very low, so it does not affect the main fisheries. Obviously, this does not indicate that there are no overexploited stocks encountered in this area.

3.2.2 Southern ocean

The Southern Ocean (Antarctic Atlantic, Antarctic Pacific and Antarctic and Southern Indian Ocean FAO areas) present higher average impacts per mass of fish with values ranging between 2×10^{-9} and 1.3×10^{-10} PDF_{reg}.year. Catches are very low, only representing 0.3% of the global catch, and few stocks are exploited. These areas should therefore be evaluated with caution. It is also noteworthy that no class I stocks have been observed in the North polar zone and that the average value for the Arctic sea cannot be determined.

Using available data, only 11 stocks can be categorised in class I in the Southern Ocean, but the results are determined predominantly by three species. The Patagonian toothfish (*Dissostichus eleginoides*, Nototheniidae) represents 93.4% of the assessed catch (class I) in the Southern Indian Ocean and 79.5% in the Southern Atlantic. The status of this species has not been evaluated by the IUCN, but the regional CF is high, mainly in the Southern Atlantic (2.4×10^{-9} PDF_{reg}.year). The global CF in this area is significantly lower, 1.76×10^{-10} PDF_{glo}.year, because the biomass in this area only represents 7% of the global biomass. This species is essentially found in the Southern Indian Ocean (58% of the biomass) and in the Southwest Atlantic (28%). The Antarctic toothfish (*Dissostichus mawsoni*, Nototheniidae) is the only stock that has been assessed in the Southern Pacific, but it represents 96% of catches in this area. This species has also not been evaluated by the IUCN. The third species is the mackerel icefish (*Champsocephalus gunnari*, Channichthyidae), representing 20.2% of catch class I in the Southern Atlantic and 6.4% in the Southern Indian Ocean. This species was considered to have been overfished in these areas by FAO (FAO 2011). The CFs are relatively high in the Southern Atlantic 1.1×10^{-9} PDF_{reg}.year and 8.1×10^{-10} PDF_{glo}.year and more so in the Southern Indian Ocean with 1.3×10^{-8} PDF_{reg}.year and 3.4×10^{-9} PDF_{glo}.year.

The main catch in Antarctic waters is Antarctic krill (*Euphausia superba*, Euphausiidae). It represents 93% of the catch in the Southern Ocean and is only fished in the Southern Atlantic. This stock is part of class II.b with quite low CFs of 1.6×10^{-11} PDF_{reg}.year and 1.4×10^{-11} PDF_{glo}.year. By considering non-fish species in the average determination, lower values are thus found in this area, covering the same order of magnitude as for areas in more temperate latitudes. However, as the CMSY algorithm was not designed to assess non-fish stocks, this result is obviously less reliable and cannot be considered at the same level as fish stocks.

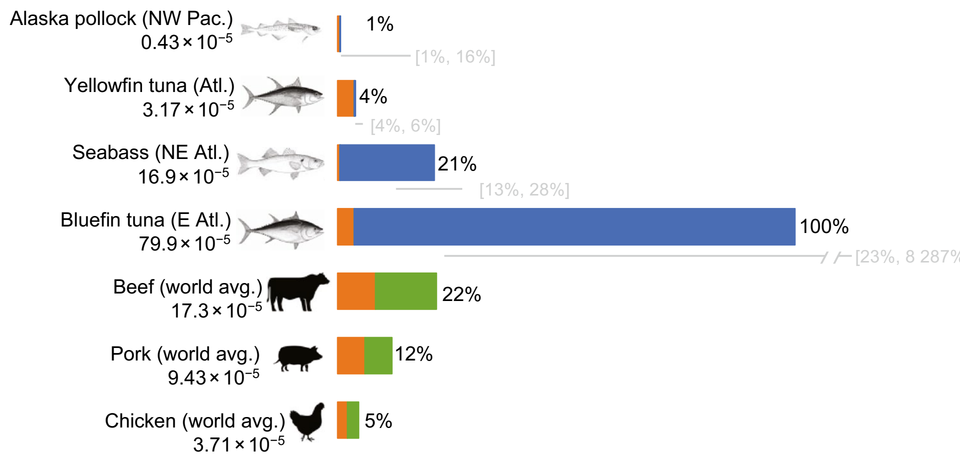
3.3 Case study

A comparison between impacts for four fish stocks and for three land-based meats is provided in Fig. 2 and Fig. 3.

3.3.1 ReCiPe and species years results

The worst system is bluefin tuna (Eastern Atlantic), when assessed in species.years and with the ReCiPe Hierarchist method (Fig. 2). It has a significantly greater impact than the other systems assessed whether terrestrial or marine, as described in the ecoinvent database. Overall fisheries display varied results. The impact on ecosystem quality of Alaska pollock from the Northwest Pacific is very low (1% of the bluefin tuna impact), whereas the result for seabass (Northeast Atlantic) is higher (21%). The uncertainty is represented by the grey line with upper and lower bound values (e.g. 13–28% for the seabass, which corresponds to 9.98×10^{-5} and 21.7×10^{-5} species.year, respectively). Impacts for yellowfin tuna are relatively low, akin to those of chicken farming (world average process) and of the same order of magnitude as pork, whereas seabass has impacts comparable to beef farming. It is interesting to note that when based on ecoinvent data, the ReCiPe endpoint impact associated

Fig. 2 Ecosystem impact of four fisheries and three terrestrial meat production systems. Results are expressed in percentages relative to the worst system: bluefin tuna (100%). The impacts of each of them are given below the names (in species.year). Orange: sum of all ReCiPe (Hierarchist) ecosystem impact except for land use. Green: ReCiPe land use impact. Blue: fishery impact on fish stocks. Grey line: uncertainty range associated with the fishery impact on stocks



with tuna fishery (bluefin and yellowfin tuna) is significantly higher than the impact of demersal fishery (Alaska pollock and Northeast Atlantic seabass). This essentially results from the amount of diesel burned by fishing vessels, which is considerably more significant for tuna fishing. Consequently, yellowfin tuna fishing is almost ten times more impactful than Alaska pollock, despite both yellowfin tuna and Alaska pollock having a similarly low fishery impact. The impact on fish stocks is even more pronounced for seabass but is far exceeded by bluefin tuna.

Uncertainties are determined from CMSY algorithm outputs and highlight the capacity of the calculated stock parameters for fitting the available data. Uncertainty ranges are relatively limited for yellowfin tuna and seabass and consequently do not modify the comparisons between results. They are considerably larger for Alaska pollock and bluefin tuna. When considering uncertainties, it is not possible to conclude that the impact of Alaska pollock would have lower impacts than yellowfin tuna, chicken or pork systems, due to the wide range associated with the pollock and lack of intervals for the land-based systems. For bluefin tuna, the range appears significantly greater because of the very high upper boundary.

3.3.2 Regional and global PDF

Figure 3 focuses on land use (by transformation and occupation) and sea use (by fishing) of the different systems. Considering regional PDF (Fig. 3a), the results are similar to those obtained from ReCiPe, excluding the other impacts. Bluefin tuna thus remains the worst scenario. Land use associated with tuna fisheries is relatively high. This result is surprising for an inventory that does not involve agricultural activities but can be explained by the high diesel consumption of fishing vessels. The land transformation impact of tuna fisheries is presently governed by the transformation of forests into mineral extraction sites, which is associated with the infrastructure for oil extraction to obtain diesel to fuel fishing vessels.

LC-Initiative guidelines provide confidence intervals for CFs. The whole range of uncertainties of the impact can therefore be addressed and not only for fisheries, as is done with ReCiPe. With the confidence intervals, Alaska pollock and yellowfin tuna present a significantly lower impact than seabass, but no other results can be highlighted. This is due to the high uncertainties in the bluefin tuna assessment (see above) and from the very large confidence intervals of land-based productions, where the lower boundary of the interval is negative (i.e. positive effect of land use on biodiversity).

The impacts assessed with global PDF (Fig. 3b) provide some different results. The impacts in global PDF are about tenfold lower for land-based systems (beef, pork and chicken) and Alaska pollock. Both yellowfin tuna and Alaska pollock produce substantially lower impacts than all land-based systems as well as seabass and bluefin stocks, although Alaska pollock exhibits a large uncertainty range, which makes drawing conclusions against other systems difficult. On the contrary, impacts only decrease slightly for seabass (from 71×10^{-11} PDF_{reg}.year to 57×10^{-11} PDF_{glo}.year) and for bluefin tuna (from 330×10^{-11} PDF_{reg}.year to 299×10^{-11} PDF_{glo}.year). The impacts for these two fish stocks are therefore greater than for the other systems, with bluefin tuna retaining the greatest impact two orders of magnitude larger than terrestrial systems. The most noticeable difference at the global scale is that seabass no longer produces results similar to the land-based systems, exhibiting a much higher level of impact. Depending on the confidence intervals, the difference is significant for seabass with respect to yellowfin tuna and land-based productions (i.e. no overlapping of confidence intervals). It is also significant for bluefin tuna with respect to yellowfin tuna and terrestrial-based systems.

Comparison between Fig. 3a and b highlights the importance of including assessments using global PDF. According

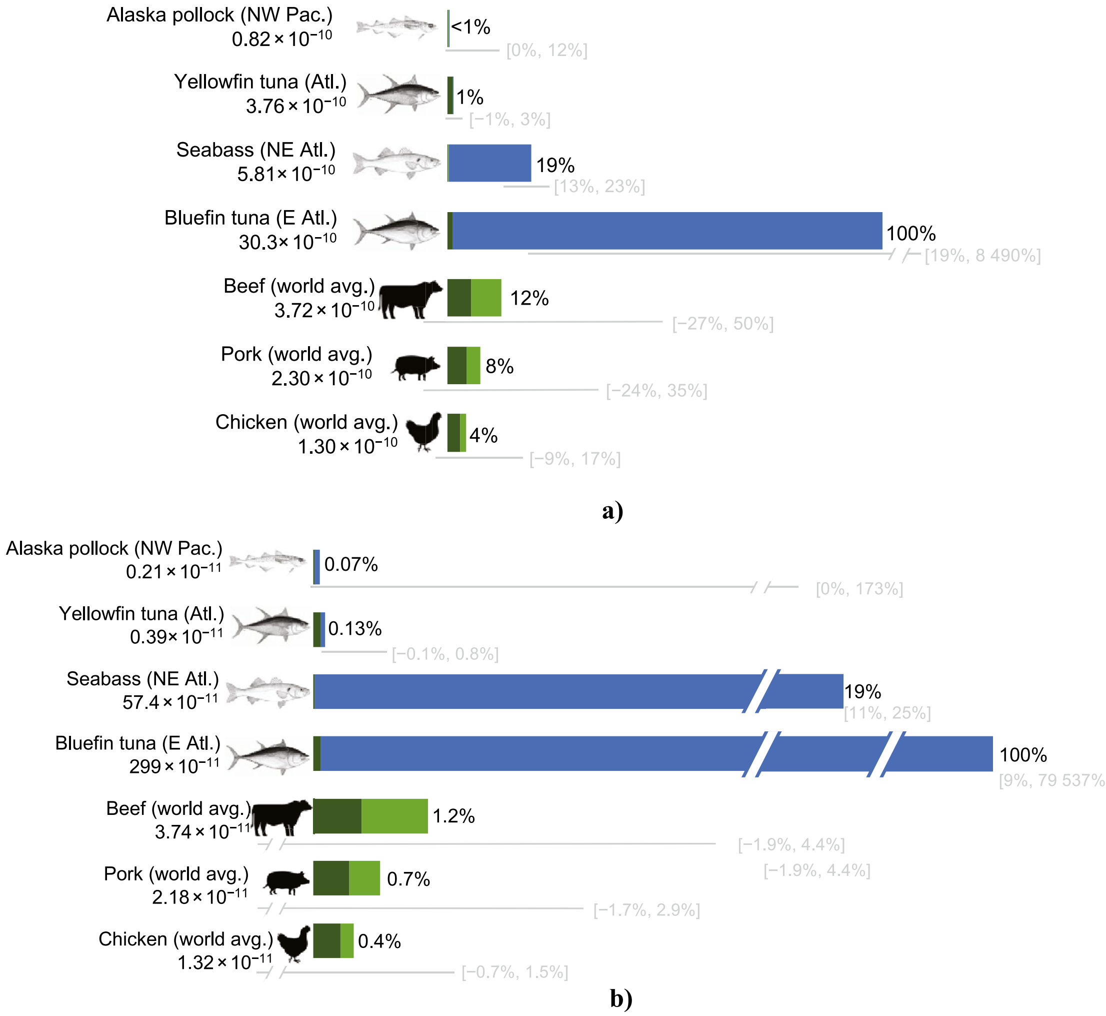


Fig. 3 **a** Regional and **b** global impacts on biodiversity related to land use and fishing for four fisheries and three terrestrial meat production systems. Results are expressed in percentages of the worst system — bluefin tuna (100%) — and the impact of each of them is given below the names (in PDF_{reg}.year or PDF_{glo}.year). Dark green: land transformation. Light green: land occupation. Blue: fishery impact on fish stocks. Grey line: uncertainty range associated with the result

to the data, Atlantic bluefin tuna is strongly endemic to the Eastern Atlantic where 91% of the global biomass is located, the remaining 9% being found in the western Atlantic. The status of European seabass is similar, with 81% of the biomass in the northeastern Atlantic Ocean (seabass can also be found in the Mediterranean Sea and in rare cases in the Central-East Atlantic). Since these species cannot easily be encountered elsewhere, their CFs, when expressed in global PDF, are closer to CFs in regional PDF. The yellowfin tuna is a cosmopolitan species, distributed across all temperate oceans. The Atlantic population only represents 11% of the global stock, and its global PDF value is thus ten times smaller than for the regional PDF. Alaska pollock represents the main population in the Northwest Pacific. It covers 66% of the global biomass (remaining part in the Northeast Pacific) and the difference between regional and global CFs is therefore not significant. However, as the CFs are very low, the results are mainly affected by the extent of land use. Hence, the overall global PDF result is one order of magnitude less than the overall regional PDF.

The inventories involved in this case study do not result from detailed descriptions of systems but only come from available generic datasets. It is important to note that the conclusions derived from the comparisons made cannot be extrapolated. However, marine productions are found to vary within a similar order of magnitude to land-based productions, and the large impact of variations between fish stocks is highlighted. This case study illustrates how the impact due to fishing on an ecosystem can be combined with results from ReCiPe endpoint method and with land use from LC-Initiative guidelines. This exemplifies the introduction of the impact of fisheries into current LCIA methods.

3.4 Relevance of the approach and perspectives

Several aspects concerning the structure of this approach are highlighted and discussed.

Due to its structure

$$\frac{CK}{r^2B^3} = \left(\frac{B}{K}\right)^{-1} \times \frac{1}{r} \times \frac{C}{rB} \times \frac{1}{B} \tag{12}$$

The CF allows us to consider several aspects that are decisive in determining the extent to which a species is endangered or close to extinction. Thus, the ratio of the current biomass to the pristine condition (B/K) accounts for the state of the population, the ratio to its intrinsic growth rate ($1/r$), the restoration dynamics, the ratio of catch to replenishment (C/rB) and the state of anthropogenic pressure. The ratio to current biomass ($1/B$) informs us of the proportion of the stock that is extracted. Furthermore, when global CFs are used, the endemicity of the stock is also introduced. CFs thus aggregate many of the stock descriptors used in fisheries management or that are determinant in defining the status of a species.

The CFs are both marginal and average. The current state of a fish stock results from the intrinsic dynamics counterbalanced by the withdrawal rate (i.e. the elementary inventory flow). The intrinsic dynamics are mainly driven by the state of the stock itself. Due to the model structure, the result is identical whether a marginal variation or all interventions on a time-scale representative of the dynamics of the system are investigated. This undoubtedly represents an advantage, since the threshold of 5% of the impact proposed in the guidelines (Verones et al. 2017) does not have to be applied.

The CFs are expressed in species.year and PDF.year. The unit (species) used in the ReCiPe endpoint method relates to the number of species, while the LC-Initiative guidelines are based on the ecosystem level (PDF). The authors have followed the approach proposed by ReCiPe to convert species to PDF, by dividing the number of species lost by the number of species in marine environments. This differs from the approach of Chaudhary et al. (2015) where the number of extinct species for each taxon is determined directly. The aggregation of taxa, weighted according to the number of species of each, then allows the conversion of a species unit into a PDF unit at the ecosystem level. This second approach is worthwhile when the impact concerns several taxa at the same time. However, as this is not the case in the present study, the ReCiPe approach has been selected.

PDF is the most commonly applied endpoint metric in LCIA methodologies to quantify damage on ecosystem quality (AoP). Recommended for used by the GLAM Lifecycle Initiative (Verones et al. 2017), it represents the loss of biodiversity from an ecosystem as a result of distinct anthropogenic pressures. It is most often calculated using model-derived species richness values (Chaudhary et al. 2015; Dorber et al. 2020). As a biodiversity measurement, species richness is strongly linked to spatial alterations resulting from land use occupation and transformation, and this is reflected in the function of the PDF metric. It is however considered limited in its depiction of the multifaceted nature of biodiversity and changes in environmental quality both by ecological and LCA literature (Curran et al. 2011; Hillebrand et al. 2018; Woods et al. 2018; Lindner et al. 2019). Intra-species abundance data or other indicators (de Baan et al. 2013b) are identified as providing additional, important information on ecosystem structure and function lacking from species richness.

LCIA currently lacks a clear consensus over the definition and structure of PDF. This stems from the various levels of biodiversity that can be assessed and the multitude of metrics available (McGill et al. 2015) and results in a variety of approaches to its calculation. Müller-Wenk (1998) proposes PDF as an indicator measuring change in species diversity, integrated over a certain time and area presented by the life cycle inventory, and it is described by Goedkoop and Spriensma (2001) as the fraction of species which has a high probability of no occurrence in a region due to unfavourable conditions. The superficial nature of these definitions allows for interpretation, and the GLAM Initiative (Verones et al. 2017) recommends PDF should be adapted in order to be able to reflect spatial and inter-species variations. This is currently under discussion within the GLAM working group dedicated to new impact categories, and here, the authors have proposed an adaptation which fulfils both the need to arrive at the recommended harmonised endpoint metric and the inclusion of species level detail necessary to assess impacts in fisheries stocks.

In fisheries, abundance data is crucial for understanding stock status, more than the total number of species found in an ecosystem or fishing area. Therefore, an endpoint metric based on species richness alone does not portray well the changes caused by overexploitation of fisheries on single stocks or within the ecosystem. The inventory flow for the fisheries impact pathway is the direct removal of a portion of each target species, reported as tonnes of biomass in catch data, rather than linked to a change in suitable area available. This renders total species richness unrepresentative of all but the most extreme changes initiated by fisheries, where risk of extinction begins and increases with the decline in species abundance. In order to integrate useful information on the impacts occurring in fish stocks into LCA, this approach proposes a weighted representation of the fractional depletion in individual stocks at the ecosystem scale. Consideration is given to the structural similarities between this approach and that of Potentially Affected Fraction (PAF), the midpoint indicator associated with ecotoxicity impacts and USEtox© which quantifies the fraction of a species exhibiting a change in abundance with exposure to a known level of pressure (Posthuma and de Zwart 2012).

The transition from regional CFs to global CFs is made at the species level. This aspect is crucial for the transition from a regional to a global assessment. Work on the vulnerability score (Chaudhary et al. 2015) or on global extinction probabilities (Kuipers et al. 2019) focuses on determining a conversion factor for a whole ecosystem, thus requiring the collection of information about all species. In the absence of quantitative data on populations, this work relies on data from the IUCN red list (IUCN 2017). This is even more complex for the marine environment, where data are scarce and do not allow for the percentage of threatened species to be estimated (IUCN red list). The work presented here has the advantage that modelling provides an estimate of population size for all regions and that this is carried out at species level. Hence, it is possible to directly fix a regional to global conversion factor at species level.

The CFs are based on the modelling of stock dynamics. This corresponds to the level of fisheries management, since the majority of rules and regulations on fisheries are defined at stock levels. The mechanisms of evolution, adaptation or collapse of ecosystems are obviously more complex and cannot be simply summed up as the addition of direct stock depletions. Any change in the abundance of a population has consequences for the entire food web, which may entail new balances for other species. Although this work is a novel approach addressing the impacts of fisheries on ecosystems, it should be relevant to extend the concept beyond the stock model and towards an ecosystem model, i.e. assessing the extent of the impact of human intervention on ecosystem dynamics.

Human activities have impacts on all ecosystems, whether terrestrial, freshwater or marine. Even though these three categories can be separated (Verones et al. 2020), they can also be grouped together in the ecosystem quality AoP. Expressing impacts for all ecosystems using the same unit makes comparison easier. From a methodological perspective, this approach has relevance to the work currently under discussion in GLAM phase 3 and the development of an impact pathway relating to the impact of biomass removal by fisheries. By providing CFs for 5000 fish stocks, the present work allows for the consequences of fisheries to be taken into account in LCA, which in turn would be useful for food system assessments.

Supplementary Information The online version contains supplementary material available at https://doi.org/10.1007/s11367-023-02136-2.

Author contribution AH and VB conceived the idea. AH and CSC designed the study, carried out all calculations and wrote the manuscript. VB contributed with critical feedback and manuscript revision.

Funding The work of C. Stanford-Clark is funded by the Cost to Coast project of the French National Research Agency and the Quebec Research Fund ANR-19-FQSM-0004.

Data availability The datasets generated during and/or analysed during the current study are available in the Zenodo repository https://doi.org/10.5281/zenodo.3954209 and https://doi.org/10.5281/zenodo.2669064.

Declarations

Conflict of interest The authors declare no competing interests.

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