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

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1   **A New Impact Pathway towards Ecosystem Quality in Life Cycle Assessment: Characterisation**

2   **Factors for Fisheries**

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9   ABSTRACT.

10   **Purpose**. Although life cycle impact assessment methods exist for quantifying land use and its impact on the

11   environment in the "ecosystem quality" area of protection, the impact of sea use on ecosystems has been poorly

12   assessed so far. This paper aims to propose operational characterisation factors for all global fisheries.

13   **Methods.** For a given intervention, the characterisation factor is defined as the product of the fate factor (inverse

14   of the fish stock growth rate) and the effect factor (depleted fraction of the stock). Characterisation factors are

15   provided for 5000 fish stocks identified by the Food and Agriculture Organisation. Both the marginal and average

16   approaches are used and characterisation factors compatible with the ReCiPe method and the international

17   guidelines of the Life Cycle Initiative hosted by the UN environment program are proposed.

18   **Results and discussion.** Characterisation factors for regional and global assessments can be employed to address

19   the endemic nature of a species. As an illustration, four contrasting fisheries are presented and compared with

20   land animal production systems. Impacts varied between stocks and between regional and global assessment,

21   particularly with highly endemic species exhibiting impacts comparable to or exceeding land-based animal

22   products.

23 **Conclusions.** Although in some cases associated uncertainty is large, the proposed method allows endpoint

24 characterisation, in line with the ReCiPe methodology and Life Cycle Initiative, contributing the assessment of

25 fishing impacts on ecosystem quality and a more holistic representation in food impact assessment.

26

INTRODUCTION

28 Humans have always used the land and the sea as food sources. Terrestrial ecosystems have been intensely

29 modified for agricultural purposes since the Neolithic period (Ellis et al. 2013). Although fishing activities date

30 from even further back, impacts on marine communities remained localised and rarely over-exploitative until

31 the industrialisation of fishing activities enabled global expansion at an unprecedented scale (Jackson et al. 2001).

32 Regrettably, this intensification of fishing over the past century has rapidly altered the situation. The impact of

33 fisheries on the marine ecosystem has been only relatively recently assessed, but it has undoubtedly been massive

34 for several decades. Fishing has thus modified all marine ecosystems (Pauly 1998), through habitat modification,

35 top-down restructuring of the trophic web (Steneck et al. 2002), reduction of functional redundancy and the

36 ability to provide critical ecosystem functions (Bellwood et al. 2004), ultimately affecting the resilience of the

37 ecosystem to withstand disturbances. Every two years, Food and Agriculture Organisation (FAO) provides a

38 detailed report on the state of the world fisheries and aquaculture, addressing all these issues (FAO 2022).

39 Life Cycle Assessment (LCA) is the reference approach when addressing the global impacts of products and

40 services. With LCA, practitioners commonly quantify the environmental impacts on three areas of protection

41 (AoP): human health, natural resources and ecosystem quality. On one hand, LCA has been applied to quantify

42 land use by human activities and its consequences on ecosystems (third AoP). Several approaches have been

43 proposed for this purpose, for example based on the soil organic carbon content (Milà i Canals et al. 2007) or on

44 biodiversity (de Baan et al. 2013b; Chaudhary et al. 2015; Winter et al. 2018; Chaudhary and Brooks 2018). On

45 the other hand, the LCA community has not yet adequately assessed the impact of sea use on ecosystems.

46   Sea use influences the two AoPs: natural (biotic) resources and ecosystem quality (Langlois et al. 2014b). The

47   resource AoP is mainly assessed through the human appropriation of the net primary production (Cashion et al.

48   2016). The same descriptor was used by Langlois et al (2015) for defining a pathway towards ecosystem quality.

49   Other approaches dedicated to the resource AoP have been proposed, with characterisation factors (CFs) (Langlois

50   et al. 2014a) or indicators (Emanuelsson et al. 2014) based on fishery management parameters, distance-to-target

51   approach (Bach et al. 2022) or characterization factors (CFs) defined from stock dynamic models (Emanuelsson

52   et al. 2014; Hélias et al. 2018).

53   Recently, CFs were proposed to assess the impact of seabed destruction on ecosystem quality (Woods and Verones

54   2019). This promising approach incorporates seafloor destruction as a form of habitat modification as an additive

55   impact of sea use due to trawling in the current life cycle impact assessment (LCIA) framework. However, to the

56   authors' knowledge, no existing approach assesses the impact of biomass removal by fishing on an ecosystem, in

57   compliance with current LCIA guidelines (Verones et al. 2017), despite (over)exploitation representing one of

58   the major causes for the decrease in biodiversity in the oceans (Woods et al. 2016; IPBES 2019). This lack of

59   indicators has been highlighted when comparisons were made between marine- and agricultural-based products:

60   the impacts are not expressed in the same units and are not comparable, which, undoubtedly, represents an

61   important issue for food impact assessment. The present work aims to solve this issue of inconsistency by

62   proposing operational global fishery CFs for characterising ecosystem quality, and allowing sea-use and land-use

63   to be addressed within a single AoP. These CFs comply with international guidelines (Verones et al. 2017) and

64   units, where the inventoried catch (weight of fish) is converted into an ecosystem quality impact. Furthermore

65   they result from the extension of a recent study on biotic resource depletion (BRD) (Hélias et al. 2018) where the

66   depleted stock fraction (DSF) can provide a tenable link to the ecosystem quality AoP in a manner comparable to

67   Potentially Affected Fraction (PAF) already employed in other impact pathways.

68   **METHODS**

69 **Ecosystem quality: units and land use models**. The Life Cycle Initiative (LC-Initiative) which is hosted by UN

70 environment recommends CFs for ecosystem quality AoP in its guidelines (Verones et al. 2017). This results in a

71 Potentially Disappeared Fraction of species over a given time (PDF.year). For the impact of land use, an

72 approach (Chaudhary et al. 2015) has been selected according to the countryside species-area relationship

73 (SAR) model (Pereira et al. 2014). The CFs proposed by Chaudhary et al (2015) have been updated (Chaudhary

74 and Brooks 2018). The LC Impact method (Verones et al. 2020) was developed using the Chaudhary et al (2015)

75 framework, but with a final metric conversion to PDF at the endpoint. Note that the ReCiPe method

76 (Huijbregts et al. 2016, 2017) uses a former model (de Baan et al. 2013a; Curran et al. 2014) and defines a

77 different (but related) unit based on the number of species that have disappeared over a given time

78 (species.year).

79 These selected land use CFs are defined by marginal and average approaches to represent the occupation and

80 transformation of a parcel of land (4 sets of CFs are obtained, expressed in lost species/m² for occupation and lost

81 species.year/m² for transformation). They address the potential species loss resulting from human use of an area

82 per ecoregion. The data obtained for five taxa (mammals, reptiles, birds, amphibians and vascular plants) are then

83 aggregated. This leads to the establishment of regional CFs (expressed in PDF/m² for occupation and PDF.year/m²

84 for transformation) which assess the loss of the intrinsic function of ecosystems at a regional scale (Frischknecht

85 and Jolliet 2016). Global CFs (expressed in global PDF/m² for occupation and global PDF.year/m² for

86 transformation) are also provided. They assess the global (and irreversible) loss of the proportion of species in the

87 ecosystem through a vulnerability score (Verones et al. 2015). An improvement of this score has recently been

88 proposed (Kuipers et al. 2019; Verones et al. 2022) , defining the "global extinction probability". This latter

89 estimation is more accurate for converting regional CFs into global CFs, by ensuring the regional summation of

90 the conversion factors equals one. The regional fraction of lost species can therefore be translated to a global scale

91 species extinction potential, aquatic species are also included where previously not. . However, the purpose and

92 the main outlines of the design remain the same. See supplementary information for a brief description.

93   The rationale here is to use a similar approach for fisheries: indeed, the lost species, the related regional PDF and

94   finally the global PDF can be determined from fish stock depletion. Both PDFs can result from marginal and

95   average approaches.

96   **Marginal CFs for biotic resource depletion.** In LCA, abiotic natural resources can be assessed in different ways

97   (Berger et al. 2020; Sonderegger et al. 2020), although depletion is the criterion that is most often investigated.

98   For biotic resources, the depletion of the stock (a species in a habitat) is intrinsically based on its renewability,

99   which depends on the replenishment capacity of living organisms relative to their withdrawal due to human

100 activities. An approach was proposed in a recent study by Hélias et al (2018) addressing global fisheries as resource

101 depletion. The CFs are based on a marginal approach involving a population model dynamic in order to link the

102 inventory (fish withdrawal) with the impact (stock depletion), which is briefly reported here. The frequently-

103 used fish stock dynamics Schaefer model shape (Schaefer 1954) is the foundation of this study.

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

104 where B is the fish biomass (tonne), C the annual catch (tonne.year^{-1}), r the growth rate (year^{-1}), and DSF the

105 depleted stock fraction. The latter varies from 0 for a plentiful stock to 1 when it is exhausted. This model

106 illustrates the growth where exponential expansion (rB) is limited by available habitat represented by DSF. The

107 Schaefer model is based on the well-known logistic law of growth and in this case,

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

108

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

109 Two approaches are generally used and recommended in LCA to define CFs, representing a marginal or average

110 change (Frischknecht and Jolliet 2016). A third approach, the linear one, is sometimes used when the current

111   state (as the background concentration for a pollutant) is unknown (Hauschild and Huijbregts 2015), but this is

112   not the case in the present study. Hélias et al. (2018) provide CFs for biotic resource depletion through a marginal

113   approach only ($CF_{BRD,M}$). This is defined as the partial derivative of the impact (∂DSF) according to the inventory

114   (mass of fish removed from the biomass stock, $-\partial B$).

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

115   See Hélias et al. (2018) for additional details. CF values are provided for all fisheries described in FAO data (global

116   scale). Recently (Hélias and Heijungs 2019), model consistency has been observed between this approach and the

117   abiotic depletion potential (Guinée and Heijungs 1995) (the most commonly used approach to assess abiotic

118   resource depletion in LCA).

119   **Average CFs for biotic resource depletion.** A marginal CF allows for a small change to be assessed from the

120   current situation. However, an average approach is better adapted to address greater changes (often defined as

121   >5% of the issue as a whole) and both sets of CFs should be provided for an LCIA method (Frischknecht and

122   Jolliet 2016). The CF for average (A) biotic resource depletion ($CF_{BRD,A}$) is defined in LCIA as the average slope of

123   the causal relationship between the inventory E, which is the quantity of fish removed, and the impact, i.e. DSF

124   in this study. This is equivalent to the division of the impact by the overall human intervention (E) (Curran 2017).

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

125   To define E as a biotic resource, the timeframe where the catch (extraction rate) occurs needs to be defined.

126   When the system reaches a steady state, the quantity of fish removed is represented by the catch during a

127   timeframe τ.

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

128 On one hand, a too long timeframe does not make sense, as the dynamics of the stock counteract with former

129 withdrawals, which do not affect the current state anymore. On the other hand, a too short timeframe could

130 overlook a part of the human interventions leading to the current state. This timeframe cannot be identical for

131 all stocks and needs to be determined according to the current population resilience, based on its replenishment

132 rate. In dynamical system theory, the responsiveness of a linear-time invariant system is given by its time

133 constant. By analogy, in this study, τ is the time constant of the stock.

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

134 E is therefore the quantity of fish removed, corresponding to the current pressure delivered over a given period

135 by the capacity of the stock to counteract changes. When eqs (3) and (5)–(7) are combined, E is equal to B and

136 the characterisation factor for biotic resource depletion is as follows:

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

137 By defining human intervention as the catch over the time constant of the stock, both marginal and average

138 approaches have the same value and a unique set of CFs is provided.

139 **From depleted stock fraction to ecosystem quality.** The impacts affecting ecosystem quality are generally

140 addressed with $CF = FF \times EF$. For a given intervention, the impact is characterized by the product of the fate

141 factor (FF) and the effect factor (EF). The first represents the time period during which the effect occurs while

142 the second characterises the associated effect. The more detailed relationship $CF = FF \times XF \times EF$ is often used,

143 although the exposure factor (XF), relating to a toxicity impact, is not relevant for fisheries.

144 *Depleted stock fraction as effect factor.* For a biotic resource, an analogy can be observed between the depletion

145 of the resource and the biodiversity impact. Hence, fishing leads to a loss in biodiversity, due to the withdrawal

146 of a part of the living biomass. The *DSF* represents the disappeared fraction of the target stock (a given

147 commercially fished species in its habitat) and from this, the unit for CF_BRD can then be defined as the amount of

148 lost target species/tonne.

149 It is noteworthy that the shape of the equation of the *DSF,* as defined by the Schaefer model, resembles a modelled

150 effect factor for terrestrial acidification (Azevedo et al. 2012; Crespo-Mendes et al. 2019) where the potentially

151 non-occurring fraction is only defined at a biotic community level and not for a specific species of a habitat.

152 *Fate factor.* Most of the impacts traditionally quantified in LCA studies that affect ecosystem quality (e.g.

153 ecotoxicity, acidification, eutrophication, etc.), result from substance emissions. In this context the fate factor

154 represents the persistence of a given substance in the media (Cosme et al. 2018). It is usually expressed in years

155 or days. The fate factor is thus driven by transfers between compartments and by substance degradation. For a

156 given compartment, it can be expressed as the inverse of the sum of the removal rates (Cosme et al. 2018) or as a

157 residence time (Rosenbaum et al. 2007).

158 Since it results from a resource withdrawal rather than an emission, the fate factor of fisheries proposed in this

159 paper is inverted. The principle components of the characterisation factor however, remain the same where the

160 effect factor represents the impact and the fate factor, its duration. In USEtox®, fate factors are expressed as the

161 inverse of exchange- and removal-rate constants (Bijster et al. 2018), which is known as the mean lifetime for an

162 exponential law. In the present instance, the model is more complex. The carrying capacity in the model

163 introduces a non-linearity and the mean lifetime of the model is consequently a function of the magnitude of the

164 elemental flow. In order to avoid this incompatibility with the principles of LCA, where the CF is constant

165 whatever the inventory value, the model can be linearised at the steady state. The fate factor is then defined as

166

$$FF = \frac{1}{r} \times \frac{K}{B} \qquad\qquad (9)$$

167 i.e., the inverse of the growth rate constant tempered by the inverse of the relative biomass. See supplementary

168 materials for details.

169 **Characterisation factors.** The regional CF for the impact of fish catches on ecosystem quality ($CF_{EQ,reg}$, expressed

170 in species.year/kg of fish) is therefore expressed as follows

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

171 This CF is both marginal and average as previously discussed. The species.year unit is used with the ecosystem

172 AoP in the ReCiPe endpoint method (Huijbregts et al. 2016, 2017) and therefore the impacts on fisheries

173 ecosystems can be directly added to this method. Note that similar to the approach for the land use impact

174 category, this study does not differentiate between the three perspectives of the ReCiPe endpoint method

175 (individualist, hierachist and egalitarian).

176 The conversion from species.year/kg to regional PDF.year/kg can be easily made by dividing CF_{EQ} by the total

177 number of species in marine regions (233 302) (Horton et al. 2019). The reverse approach is used in the ReCiPe

178 endpoint method to convert PDF.year into species.year. Global CFs ($CF_{EQ,glo}$) should also be provided, as stated

179 by LC-Initiative guidelines (Verones et al 2017). From a modelling point of view, the main difference between

180 land use and fisheries lies at the level of intervention: The land use impact is related to a spatial change and

181 affects all species in the corresponding area. For fisheries, the CF is defined for specific, targeted species in a

182 given ecosystem (i.e. the population). In contrast to land use, when using a stock-based modelling approach, the

183 scope of the human intervention through fishing does not include indirect effects on the ecosystem and all of its

184 communities; it solely affects one species, i.e. the caught species. If various species can be caught simultaneously

185 within an ecosystem, the corresponding impacts are additive and assessed separately through inventory flows

186 and associated CFs in the LCA framework.

187   By considering a PDF linked to the midpoint through the DSF (analogous to PAF) rather than a change in

188   absolute species richness, it is possible to have a representation of species level abundance impacts with are

189   critical to fisheries quantified within the CF.

190   At population (fish-stock) levels, the conversion factor to obtain global-PDF from regional-PDF only quantifies

191   the endemic character of a given species in given region. This approach is simpler than for the ecosystem level

192   applied to land use. With a reasoning similar to that used for calculating the vulnerability score (Verones et al.

193   2015) or the global extinction probability (Kuipers et al. 2019; Verones et al. 2022) (except that it takes place at

194   the species level), the endemic conversion factor for obtaining the global-PDF from the regional-PDF is

195    $B_j/\sum_k B_k$, i.e. the proportion of global biomass in an ecoregion j. The impact can thus be expressed, using all

196   units recommended by LC-Initiative guidelines (Verones et al. 2017), with

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

197   **Operationalisation.** Most fish stocks have been poorly described and the quantification of stock descriptors

198   required to compute CFs in equation (10) remains a challenge. To address this issue, the CMSY algorithm

199   (Froese et al. 2017) was chosen, following the methodology described in Hélias et al. (2018) and; Hélias (2019).

200   This allows for a global scale estimation of stock descriptors from catch time-series provided by FAO (2017) and

201   resilience available in FishBase (Froese and Pauly 2016). Estimations of C, r and B values are thus provided for

202   all fisheries reported by FAO, considering a stock as a species in an FAO area. The complete description of the

203   approach, its relevance, the management of multi-stock datasets (stocks merging more than one species or more

204   than one habitat) and poor-data stocks (when the available FAO data do not allow the use of CMSY) have been

205   previously discussed in Hélias et al. (2018). The reader can refer to this latter article for more details concerning

206   the validity of biotic resource depletion for ecosystem quality impact. It is also noteworthy that due to this

207   operationalisation and the availability of the data, the term ecoregion refers to FAO major fishing regions.

208   Although these are arbitrary delineations rather than strictly ecological, they serve the same purpose of

209 regionalisation of the approach within current fisheries data constraints, and have therefore been considered as

210 a proxy for ecoregions. The occurrence of multiple observed habitats in an FAO area has already been discussed

211 in Hélias et al. (2018).

212

213 The relevance of the assessment is determined qualitatively following the approach of Hélias et al. (2018) and

214 Hélias (2019). Results are briefly presented here, ranging from the most reliable to the least trusted.

215 • Class I corresponds to marine fish stocks with only one species, which have been fully assessed with

216   the CMSY algorithm.

217 • Class II brings several groups together, also assessed with CMSY. Class II.a is composed of multispecies

218   marine fish stocks with not more than five species. Class II.b lists non-fish mono-species stocks

219   (crustacean, mollusc…). Class II.c encompasses mono-species inland stocks.

220 • Class III is similar to class II but with multispecies marine fish stocks with more than five species

221   (III.a), non-fish multi-species stocks (III.b) and multi-species inland stocks (III.c).

222 • Class IV stocks are not directly assessed due to poor data quality. Global aggregated values are used, at

223   species level (IV.a) or group level when values for species are not available (IV.b),

224 **Case study.** As an illustration, four fisheries products have been presented and compared to livestock products.

225 The purpose is not to provide an extensive and accurate LCA, but rather to demonstrate how this work can be

226 used by practitioners and to highlight a few results. For this purpose, a simple functional unit has been used

227 without taking the protein content or other nutritional aspects into account. All systems have been assessed for

228 one metric ton of fresh products. The ecoinvent database (Wernet et al. 2016) has been used (v3.5 "allocation at

229 point of substitution" system model implemented in Simapro® v9 software).

230   Tuna species are fished intensely and are easily identified by consumers. Bluefin tuna species have even been

231   classified as endangered or critically endangered by the International Union for Conservation of Nature (IUCN).

232   The Atlantic bluefin tuna (*Thunnus thynnus*, Scombridae) in the Eastern Atlantic was therefore selected to be

233   assessed. For comparison, Yellowfin tuna (*Thunnus albacares*, Scombridae) stocks in the Atlantic Ocean were

234   chosen, since they seem to be surviving in better conditions (near threatened status by IUCN). The ecoinvent

235   process "landed tuna to generic market for marine fish {global}" has been used as an inventory for both species,

236   and only the target species and associated CF differ.

237   Additionally, two demersal species were also assessed, both being represented by the ecoinvent process

238   "demersal fish to generic market for marine fish {global}". The Alaska pollock (*Theragra chalcogramma*,

239   Dadidae) is one of the most heavily caught and consumed fish in the world. This involves the Northwest

240   Pacific FAO stock. On the contrary, the European seabass (*Dicentrarchus labrax*, Moronidae) catches remain

241   small and their heavily depleted stocks are becoming an issue as the fisheries are increasingly regulated by

242   emergency measures. European seabass is included in the Northeast Atlantic FAO stock.

243   The four fisheries are compared to the terrestrial meat production systems of chicken, pork and beef ("market

244   for chicken/swine/cattle for slaughtering, live weight, {global}" in the ecoinvent database). Impacts are derived

245   for the ReCiPe ecosystem quality endpoint (hierarchist perspective) incorporating the computed regional

246   fishery CFs, expressed in species.years/t. The regional and global fishery CFs also provided by this

247   work(expressed in PDF.year/t) are then used to obtain the impact in a LC-Initiative compatible unit. The results

248   are compared with the impacts of land use (regional and global occupation and transformation) associated with

249   the terrestrial meat products computed with CFs provided by the LC-Initative guideline report (Frischknecht

250   and Jolliet 2016).

251   **RESULTS AND DISCUSSION**

252 **Overview**. The CFs with associated uncertainties for more than the 5 000 stocks listed in FAO data, both

253 regional and global, are available for download at an online deposit DOI: 10.5281/zenodo.3954209 (##Note to

254 reviewers: the link will be accessible after acceptance of the article, for the review process the CFs are available

255 here https://www.dropbox.com/s/10fnovqnke4vmg4/CF-EcoQual-Fisheries.xlsx?dl=0 ##). The CFs are expressed

256 in species.years/t and PDF.year/t for use and comparison with ReCiPe endpoint method and LC Initiative

257 guidelines respectively.

258 The regional CFs span over ten orders of magnitude but the interquartile range is less than two orders of

259 magnitude. The median value is 2.2×10^{-4} species.year/t (9.4×10^{-10} PDF$_{reg}$.year/t), while the interquartile range

260 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

261 span over 13 orders of magnitude but here again the interquartile range is more restrained, also covering two

262 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

263 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

264 noteworthy that $B_j/\sum_j B_j \leq 1$, thus implying that $CF_{EQ,glo}$ is either always less than $CF_{EQ,reg}$, or equal to it if

265 the species is endemic.

266 **Spatial variation.** Fishing pressure does not affect all marine regions equally. Fig. 1 addresses this fact by

267 illustrating the CFs per FAO area for class I only, which includes the most reliable categories. This represents a

268 large part of the catch for almost all of the areas. As the values cover several orders of magnitude, the weighted

269 geometrical mean (with catch values) is used.

270 *Catch-impact relationship.* As previously observed (Hélias et al. 2018), the most exploited areas present lower

271 impacts per mass of fish than the less exploited areas. For example, the Northeast Atlantic and Northwest

272 Pacific represent 11% and 28% of global catches respectively, but the average regional impacts per ton of fish

273 are only 0.5×10^{-11} PDF$_{reg}$.year and 0.4×10^{-11} PDF$_{reg}$.year respectively. On the contrary, although the Northwest

274 Atlantic or Mediterranean Sea each encompass 2% of the whole catch, the average impacts are 15×10^{-11}

275 PDF_{reg}.year and 3.8×10^{-11} PDF_{reg}.year respectively.

276 This result is seemingly counter-intuitive (high catch means low impact). However, heavily caught species are

277 fished because their stocks are large and the associated fishing effort is low. For example, Peruvian anchovy

278 (*Engraulis ringens*, Engraulidae) is the most exploited species in the world. Peruvian anchovy catch represents

279 67% of the Southeast Pacific and 5% of the global catches because it is relatively effortless to fish them. Except

280 during *El Niño* events, this species thrives on an abundance of food related to the Humboldt Current. In

281 addition, its resilience is high. This entails a very high biomass for Peruvian anchovy and consequently, the

282 most fished species in the world has only been classified as a species of "least concern" by the IUCN. Moreover,

283 corresponding CFs remain relatively low in the Southeast Pacific area, with 6.6×10^{-13} PDF.year/t (this species is

284 only found in this area, which means that global and regional CFs are identical). The average impact in this area

285 is thus very low, so it does not affect the main fisheries. Obviously, this does not indicate that there are no

286 overexploited stocks encountered in this area.

287 *Southern Ocean.* The Southern Ocean (Antarctic Atlantic, Antarctic Pacific, and Antarctic and Southern Indian

288 Ocean FAO areas) present higher average impacts per mass of fish with values ranging between 2×10^{-9} and 1.3

289 $\times 10^{-10}$ PDF_{reg}.year. Catches are very low, only representing 0.3% of the global catch, and few stocks are

290 exploited. These areas should therefore be evaluated with caution. It is also noteworthy that no class I stocks

291 have been observed in the North polar zone and that the average value for the Arctic sea cannot be determined.

292 Using available data, only 11 stocks can be categorised in class I in the Southern Ocean, but the results are

293 determined predominantly by three species. The Patagonian toothfish (*Dissostichus eleginoides*, Nototheniidae)

294 represents 93.4% of the assessed catch (Class I) in the Southern Indian Ocean, and 79.5% in the Southern

295 Atlantic. The status of this species has not been evaluated by IUCN but the regional CF is high, mainly in the

296 Southern Atlantic (2.4×10^{-9} PDF_{reg}.year). The global CF in this area is significantly lower, 1.76×10^{-10}

297   PDF_{glo}.year, because the biomass in this area only represents 7% of the global biomass. This species is essentially

298   found in the Southern Indian Ocean (58% of the biomass) and in the South-West Atlantic (28%). The Antarctic

299   toothfish (*Dissostichus mawsoni*, Nototheniidae) is the only stock that has been assessed in the Southern Pacific

300   but it represents 96% of catches in this area. This species has also not been evaluated by IUCN. The third

301   species is the mackerel icefish (*Champsocephalus gunnari*, Channichthyidae), representing 20.2% of catch class

302   I in the Southern Atlantic and 6.4% in the Southern Indian Ocean. This species was considered to have been

303   overfished in these areas by FAO (FAO 2011). The CFs are relatively high in the Southern Atlantic 1.1×10^{-9}

304   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

305   3.4×10^{-9} PDF_{glo}.year.

306   The main catch in Antarctic waters is Antarctic krill (*Euphausia superba*, Euphausiidae). It represents 93% of

307   the catch in the Southern Ocean, and is only fished in the Southern Atlantic. This stock is part of class II.b with

308   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

309   determination, lower values are thus found in this area, covering the same order of magnitude as for areas in

310   more temperate latitudes. However, as the CMSY algorithm was not designed to assess non-fish stocks, this

311   result is obviously less reliable and cannot be considered at the same level as fish stocks.

312   [Insert Fig.1 about here]

313   **Case study**. A comparison between impacts for four fish stocks and for three land-based meats are provided in

314   Fig. 2 and Fig 3.

315   *ReCiPe and species.years results.* The worst system is bluefin tuna (Eastern Atlantic), when assessed in

316   species.years and with the ReCiPe Hierarchist method (Fig. 2). It has a significantly greater impact than the

317   other systems assessed whether terrestrial or marine, as described in the ecoinvent database. Overall fisheries

318   display varied results. The impact on ecosystem quality of Alaska pollock from the Northwest Pacific is very

319   low (1% of the bluefin tuna impact), whereas the result for Seabass (Northeast Atlantic) is higher (21%). The

15

320 uncertainty is represented by the grey-line with upper and lower bound values (e.g. 13%–28% for the seabass,

321 which corresponds to 9.98×10^{-5} and 21.7×10^{-5} species.year respectively). Impacts for yellowfin tuna are

322 relatively low, akin to those of chicken farming (world average process) and of the same order of magnitude as

323 pork, whereas seabass has impacts comparable to beef farming. It is interesting to note that when based on

324 ecoinvent data, the ReCiPe endpoint impact associated with tuna fishery (bluefin and yellowfin tuna) is

325 significantly higher than the impact of demersal fishery (Alaska pollock and Northeast Atlantic seabass). This

326 essentially results from the amount of diesel burned by fishing vessels, which is considerably more significant

327 for tuna fishing. Consequently, yellowfin tuna fishing is almost ten times more impactful than Alaska pollock,

328 despite both yellowfin tuna and Alaska pollock having a similarly low fishery impact. The impact on fish stocks

329 is even more pronounced for seabass, but is far exceeded by bluefin tuna.

330 Uncertainties are determined from CMSY algorithm outputs and highlight the capacity of the calculated stock

331 parameters for fitting the available data. Uncertainty ranges are relatively limited for yellowfin tuna and seabass

332 and consequently do not modify the comparisons between results. They are considerably larger for Alaska

333 pollock and bluefin tuna. When considering uncertainties, it is not possible to conclude that the impact of

334 Alaska pollock would have lower impacts than yellowfin tuna, chicken or pork systems, due to the wide range

335 associated with the pollock and lack of intervals for the land-based systems. For bluefin tuna, the range appears

336 significantly greater because of the very high upper boundary.

337 [Insert Fig.2 about here]

338 *Regional and global PDF.* Fig. 3 focuses on land use (by transformation and occupation) and sea use (by fishing)

339 of the different systems. Considering regional PDF (Fig. 3.a), the results are similar to those obtained from

340 ReCiPe, excluding the other impacts. Bluefin tuna thus remains the worst scenario. Land use associated with

341 tuna fisheries is relatively high. This result is surprising for an inventory that does not involve agricultural

342 activities, but can be explained by the high diesel consumption of fishing vessels. The land transformation

343 impact of tuna fisheries is presently governed by the transformation of forests into mineral extraction sites,

344 which is associated with the infrastructure for oil extraction to obtain diesel to fuel fishing vessels.

345 LC-Initiative guidelines provide confidence intervals for CFs. The whole range of uncertainties of the impact

346 can therefore be addressed and not only for fisheries, as is done with ReCiPe. With the confidence intervals,

347 Alaska pollock and yellowfin tuna present a significantly lower impact than seabass, but no other results can be

348 highlighted. This is due to the high uncertainties in the bluefin tuna assessment (see above) and from the very

349 large confidence intervals of land-based productions, where the lower boundary of the interval is negative (i.e.

350 positive effect of land use on biodiversity).

351 The impacts assessed with global PDF (Fig3.b) provide some different results. The impacts in global PDF are

352 about ten-fold lower for land-based systems (beef, pork and chicken), and Alaska pollock. Both yellowfin tuna

353 and Alaska pollock produce substantially lower impacts than all land-based systems as well as seabass and

354 bluefin stocks, although Alaska pollock exhibits a large uncertainty range, which makes drawing conclusions

355 against other systems difficult. On the contrary, impacts only decrease slightly for seabass (from 71×10^{-11}

356 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).

357 The impacts for these two fish stocks are therefore greater than for the other systems, with bluefin tuna

358 retaining the greatest impact two orders of magnitude larger than terrestrial systems. The most noticeable

359 difference at the global scale is that seabass no longer produces results similar to the land-based systems,

360 exhibiting a much higher level of impact. Depending on the confidence intervals, the difference is significant

361 for seabass with respect to yellowfin tuna and land-based productions (i.e. no overlapping of confidence

362 intervals). It is also significant for bluefin tuna with respect to yellowfin tuna and terrestrial based systems.

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

364 According to the data, Atlantic bluefin tuna is strongly endemic to the Eastern Atlantic where 91% of the

365 global biomass is located, the remaining 9% being found in the western Atlantic. The status of European seabass

366   is similar, with 81% of the biomass in the North-eastern Atlantic Ocean (seabass can also be found in the

367   Mediterranean Sea, and in rare cases in the Central-East Atlantic). Since these species cannot easily be

368   encountered elsewhere, their CFs, when expressed in global PDF, are closer to CFs in regional PDF. The

369   yellowfin tuna is a cosmopolitan species, distributed across all temperate oceans. The Atlantic population only

370   represents 11% of the global stock, and its global PDF value is thus ten times smaller than for the regional PDF.

371   Alaska pollock represents the main population in the Northwest Pacific. It covers 66% of the global biomass

372   (remaining part in the Northeast Pacific) and the difference between regional and global CFs is therefore not

373   significant. However, as the CFs are very low, the results are mainly affected by the extent of land use. Hence,

374   the overall global PDF result is one order of magnitude less than the overall regional PDF.

375   [Insert Fig. 3 about here]

376   The inventories involved in this case study do not result from detailed descriptions of systems but only come

377   from available generic datasets. It is important to note that the conclusions derived from the comparisons made

378   cannot be extrapolated. However, marine productions are found to vary within a similar order of magnitude to

379   land-based productions and the large impact of variations between fish stocks is highlighted. This case study

380   illustrates how the impact due to fishing on an ecosystem can be combined with results from ReCiPe endpoint

381   method and with land use from LC-Initiative guidelines. This exemplifies the introduction of the impact of

382   fisheries into current LCIA methods.

383   **Relevance of the approach and perspectives.** Several aspects concerning the structure of this approach are

384   highlighted and discussed.

385   Due to its structure:

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

386

387   The CF allows us to consider several aspects that are decisive in determining the extent to which a species is

388   endangered or close to extinction. Thus, the ratio of the current biomass to the pristine condition (B/K)

389   accounts for the state of the population, the ratio to its intrinsic growth rate ($1/r$) the restoration dynamics, the

390   ratio of catch to replenishment (C/rB) the state of anthropogenic pressure. The ratio to current biomass ($1/B$)

391   informs us of the proportion of the stock that is extracted. Furthermore, when global CFs are used, the

392   endemicity of the stock is also introduced. CFs thus aggregate many of the stock descriptors used in fisheries

393   management or that are determinant in defining the status of a species.

394   The CFs are both marginal and average. The current state of a fish stock results from the intrinsic dynamics

395   counterbalanced by the withdrawal rate (i.e. the elementary inventory flow). The intrinsic dynamics are mainly

396   driven by the state of the stock itself. Due to the model structure, the result is identical whether a marginal

397   variation or all interventions on a time scale representative of the dynamics of the system are investigated. This

398   undoubtedly represents an advantage, since the threshold of 5% of the impact proposed in the guidelines

399   (Verones et al. 2017) does not have to be applied.

400   The CFs are expressed in species/year and PDF.year. The unit (species) used in the ReCiPe endpoint method

401   relates to the number of species while the LC-Initiative guidelines are based on the ecosystem level (PDF). The

402   authors have followed the approach proposed by ReCiPe to convert species to PDF, by dividing the number of

403   species lost by the number of species in marine environments. This differs from the approach of Chaudhary et

404   al. (2015) where the number of extinct species for each taxon is determined directly. The aggregation of taxa,

405   weighted according to the number of species of each, then allows the conversion of a species-unit into a PDF-

406   unit at the ecosystem level. This second approach is worthwhile when the impact concerns several taxa at the

407   same time. However as this is not the case in the present study, the ReCiPe approach has been selected.

408   PDF is the most commonly applied endpoint metric in LCIA methodologies to quantify damage on ecosystem

409   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 allow for

interpretation, and the GLAM Initiative (Fu et al. 2020) 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 over-exploitation 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

434 renders total species richness unrepresentative of all but the most extreme changes initiated by fisheries, where

435 risk of extinction begins and increases with the decline in species abundance. In order to integrate useful

436 information on the impacts occurring in fish stocks into LCA this approach proposes a weighted representation

437 of the fractional depletion in individual stocks at the ecosystem scale. Consideration is given to the structural

438 similarities between this approach and that of PAF (Potentially Affected Fraction), the midpoint indicator

439 associated with ecotoxicity impacts and USEtox© which quantifies the fraction of a species exhibiting a change

440 in abundance with exposure to a known level of pressure (Posthuma and de Zwart 2012).

441 The transition from regional CFs to global CFs is made at the species level. This aspect is crucial for the

442 transition from a regional to a global assessment. Work on the vulnerability score (Chaudhary et al. 2015) or on

443 global extinction probabilities (Kuipers et al. 2019) focuses on determining a conversion factor for a whole

444 ecosystem, thus requiring the collection of information about all species. In the absence of quantitative data on

445 populations, this work relies on data from the IUCN red list (IUCN 2017). This is even more complex for the

446 marine environment, where data are scarce and do not allow for the percentage of threatened species to be

447 estimated (IUCN red list). The work presented here has the advantage that modelling provides an estimate of

448 population size for all regions and that this is carried out at species level. Hence it is possible to directly fix a

449 regional to global conversion factor at species level.

450 The CFs are based on the modelling of stock dynamics. This corresponds to the level of fisheries management,

451 since the majority of rules and regulations on fisheries are defined at stock levels. The mechanisms of evolution,

452 adaptation or collapse of ecosystems are obviously more complex and cannot be simply summed up as the

453 addition of direct stock depletions. Any change in the abundance of a population has consequences for the

454 entire food web, which may entail new balances for other species. Although this work is a novel approach

455 addressing the impacts of fisheries on ecosystems, it should be relevant to extend the concept beyond the stock

456 model and towards an ecosystem model, *i.e.* assessing the extent of the impact of human intervention on

457 ecosystem dynamics.

458   Human activities have impacts on all ecosystems, whether terrestrial, freshwater or marine. Even though these

459   three categories can be separated (Verones et al. 2020), they can also be grouped together in the ecosystem

460   quality AoP. Expressing impacts for all ecosystems using the same unit makes comparison easier. From a

461   methodological perspective, this approach has relevance to the work currently under discussion in GLAM

462   Phase 3 and the development of an impact pathway relating to the impact of biomass removal by fisheries. By

463   providing CFs for 5000 fish stocks, the present work allows for the consequences of fisheries to be taken into

464   account in LCA, which in turn would be useful for food system assessments.

465   **Supporting Information**.

466   Additional information concerning the transformation from regional to global CFs are provided (PDF file)

467   CFs are available for download in a data repository, DOI: 10.5281/zenodo.3954209

468

469   **Author Contributions**

470   AH and VB conceived the idea. AH and CSC designed the study, carried out all calculations, and wrote the manuscript.

471   VB contributed with critical feedback and manuscript revision.

472   **Conflict of interest**

473   The authors have no relevant financial or non-financial interests to disclose. The authors have no competing

474   interests to declare that are relevant to the content of this article.

475   **Data Availability Statements**

476   The datasets generated during and/or analysed during the current study are available in the Zenodo repository, DOI:

477   10.5281/zenodo.3954209 and 10.5281/zenodo.2669064

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614

616   Figure captions

617

618   Figure 1. Regional (dark blue) and global (light blue) weighted geometrical mean of characterisation factors per FAO

619   (Food and Agricultural Organisation) area. Weights are defined according to catches in the area and only mono-species

620   fish stock directly assessed (Class I) are considered. The proportion of the global catch captured in each area (C_{glo},

621   light green circle) and the proportion of class I stock in the area (C_{reg}, dark green circle) are provided.

622

623   Figure 2. Ecosystem impact of four fisheries and three terrestrial meat production systems. Results are expressed in

624   percentages relative to the worst system: Bluefin tuna (100%).The impacts of each of them are given below the names

625   (in species.year). Orange: sum of all ReCiPe (Hierarchist) ecosystem impact except for land use. Green: ReCiPe Land

626   use impact. Blue: Fishery impact on fish stocks. Grey line: uncertainty range associated with the fishery impact on

627   stocks.

628

629   Figure 3. a) Regional and b) global impacts on biodiversity related to land use and fishing for four fisheries and three

630   terrestrial meat production systems. Results are expressed in percentages of the worst system- Bluefin tuna (100%)-

631   and the impact of each of them are given below the names (in PDF_{reg}.year or PDF_{glo}.year). Dark green: Land

632   transformation. Light green: Land occupation. Blue: Fishery impact on fish stocks. Grey line: uncertainty range

633   associated with the result.

634

635   Figure 1

$C_{glo.}$: 2%
C_{area}: 38%

155

$C_{glo.}$: 11%
C_{area}: 87%

32

5.4 4.4

$C_{glo.}$: 4%
C_{area}: 93%

7.3 3.5

$C_{glo.}$: 2%
C_{area}: 74%

38

8

$C_{glo.}$: 28%
C_{area}: 48%

4.3 3.4

$C_{glo.}$: 2%
C_{area}: 38%

11 6

$C_{glo.}$: 5%
C_{area}: 71%

18 8

$C_{glo.}$: 2%
C_{area}: 77%

65

20

$C_{glo.}$: 15%
C_{area}: 38%

11 5.9

$C_{glo.}$: 10%
C_{area}: 78%

2.4 1.8

$C_{glo.}$: 3%
C_{area}: 36%

52 32

$C_{glo.}$: 2%
C_{area}: 53%

35 16

$C_{glo.}$: 6%
C_{area}: 45%

22 8

$C_{glo.}$: 8%
C_{area}: 31%

44 10

$C_{glo.}$: 3%
C_{area}: 93%

5.1 3.2

$C_{glo.}$: <<1%
C_{area}: 89%

118 59

$C_{glo.}$: <<1%
C_{area}: 96%

134 119

2034 238

$C_{glo.}$: <1%
C_{area}: 1%

636

$\times 10^{-12}$ PDF$_{regional}$·year/t $\times 10^{-12}$ PDF$_{global}$·year/t

Figure 2

Alaska pollock (NW Pac.)
0.43×10^{-5} — 1%
[1%, 16%]

Yellowfin tuna (Atl.)
3.17×10^{-5} — 4%
[4%, 6%]

Seabass (NE Atl.)
16.9×10^{-5} — 21%
[13%, 28%]

Bluefin tuna (E Atl.)
79.9×10^{-5} — 100%
[23%, 8 287%]

Beef (world avg.)
17.3×10^{-5} — 22%

Pork (world avg.)
9.43×10^{-5} — 12%

Chicken (world avg.)
3.71×10^{-5} — 5%

641

Figure 3

Alaska pollock (NW Pac.)
0.82 × 10⁻¹⁰

<1%
[0%, 12%]

Yellowfin tuna (Atl.)
3.76 × 10⁻¹⁰

1%
[−1%, 3%]

Seabass (NE Atl.)
5.81× 10⁻¹⁰

19%
[13%, 23%]

Bluefin tuna (E Atl.)
30.3× 10⁻¹⁰

100%
[19%, 8 490%]

Beef (world avg.)
3.72 × 10⁻¹⁰

12%
[−27%, 50%]

Pork (world avg.)
2.30 × 10⁻¹⁰

8%
[−24%, 35%]

Chicken (world avg.)
1.30 × 10⁻¹⁰

4%
[−9%, 17%]

642

643

a)

Alaska pollock (NW Pac.)
0.21 × 10⁻¹¹

0.07%
[0%, 173%]

Yellowfin tuna (Atl.)
0.39× 10⁻¹¹

0.13%
[−0.1%, 0.8%]

Seabass (NE Atl.)
57.4 × 10⁻¹¹

19%
[11%, 25%]

Bluefin tuna (E Atl.)
299 × 10⁻¹¹

100%
[9%, 79 537%]

Beef (world avg.)
3.74 × 10⁻¹¹

1.2%
[−1.9%, 4.4%]
[−1.9%, 4.4%]

Pork (world avg.)
2.18 × 10⁻¹¹

0.7%
[−1.7%, 2.9%]

Chicken (world avg.)
1.32 × 10⁻¹¹

0.4%
[−0.7%, 1.5%]

644

645

b)

646

33