Biotic resources extraction impact assessment in LCA of fisheries

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ABSTRACT

Because direct environmental impacts of fisheries can hardly be assessed using conventional methods of Life Cycle Assessment (LCA), we suggest building a new methodological framework to account for most of them. We propose a regionalized method of calculation for characterisation factors dedicated to an uptake of biomass through fishing activities (biotic resources extraction impact assessment). These characterisation factors are proposed for the assessment of impacts on biotic resources depletion and on life support functions of marine ecosystems. The method is applied on two examples of fisheries, to demonstrate that it is relevant for comparisons between different fisheries, exploiting different fish species. A discussion on the compatibility of this method with other frameworks is then performed.

Keywords: biotic resources extraction, fisheries, net primary production, maximum sustainable yield

1. Introduction

Life Cycle Assessment (LCA) tends to be exhaustive for the impacts it assesses, but as identified by Pelletier et al., (2007), there is a need of improvement to assess impacts of seafood products. In seafood LCA case studies, most authors deemed necessary to add non-conventional indicators (1) to take into account fish removal from their ecosystem and allow comparisons between terrestrial and aquatic food products, (2) to assess depletion of fish stocks and perturbation of the ecosystem by imbalanced exploitation between trophic levels, (3) to assess seafloor damage. To this aim, they used respectively (1) indicators of net primary production use, (2) small-size ratio of target catch, discard ratio, by-catch ratio and fishing-in-balance index, (3) area of seafloor trawled. In order to harmonize these different proposals, Langlois et al., (2011) suggested the creation of a new impact category, called "sea use" by analogy with "land use", which could allow the assessment of marine ecosystems transformation and occupation impacts. They suggested keeping the most consensual framework of terrestrial land use (Mila i Canals et al., 2007), i.e. defining a quality index whose values could be compared from a use to another and varying according to time to reach a new steady state after a certain time of restoration. They quoted the possibility to use an indicator expressing the life support capability of marine ecosystems.

In the case of biomass removal through fishing activities, impacts are especially strong. First, one or more specific stocks of wild species can be depleted by direct biomass removal and their future use by human as a natural resource can be altered (impacts on Biotic Natural Resources (BNR)). Secondly, the total biomass available for the ecosystem functioning is also decreased by this removal as well as the functioning of the whole ecosystem (impacts on Life Support Functions (LSF)). The biodiversity loss due to fishing is also severe, especially the alpha biodiversity for benthic species due to trawls dredging the seabed, with about 75% of the shelf areas trawled worldwide every year (Kaiser et al., 2002), as well as for commercial species and by-catches, due to a high intensity of direct capture (FAO, 2010).

In marine ecosystems, ecosystem production and biodiversity tend to display correlations (Libralato et al., 2008) and assessing LSF constitute a challenging issue in the present context of worldwide overfishing. Thus, the present study focuses only on the impact assessment of BNR extractions and ecosystem LSF alteration due to fishing activity; the impacts of fishing on biodiversity loss were not considered here. As underlined by Udo de Haes et al., (2002), both BNR and LSF have to be assessed. These authors explain in detail that it does not consist in double counting because two different areas of protection are considered (natural resources and ecosystem quality respectively). This work details and discusses methods for characterisation factors calculations for these two impact pathways. The method is presented in the section 2 and illustrated with an example of fishery in the section 3. Section 4 opens the way to a discussion on the relevance of the proposed methods and on their compatibility with other existing assessment methods.

2. Methods

Two methods of impact assessment are proposed and detailed for BNR and LSF in part 2.1 and 2.2 respectively. One of the constraints considered in this study was to provide some results in comparable units.

2.1. Fishing activities and biotic resources extraction impact assessment

The goal of biotic resources extraction impact assessment is to characterise to what extent the current biotic extractions worsen the possibilities for human society to cover future needs, due to stock reductions as stated by Udo de Haes et al., (2002). One commonly used reference for fish stock status assessment is the maximum sustainable yield (MSY). This is the highest yield in fish production that can be sustained in the long term (Graham, 1935; Schaefer, 1954). It results from the assumptions that current catches at time t (C_t) can be increased up to a certain level by increasing the fishing effort (E) because they are compensated by an equivalent fish production. Above the MSY level and its corresponding E_{MSY} level, the renewal of the resource (reproduction and body growth) cannot keep pace with the removal caused by fishing. In this case further increases in exploitation leads to a reduction in yields (Fig. 1). The MSY can either be calculated through different stock assessment methods or can be estimated empirically (Hilborn and Walters, 1992). Rough stock assessments are performed by FAO but the most interesting database is the RAM Legacy Stock Assessment Database, including biological reference points for over 200 stocks (Ricard et al., 2011).

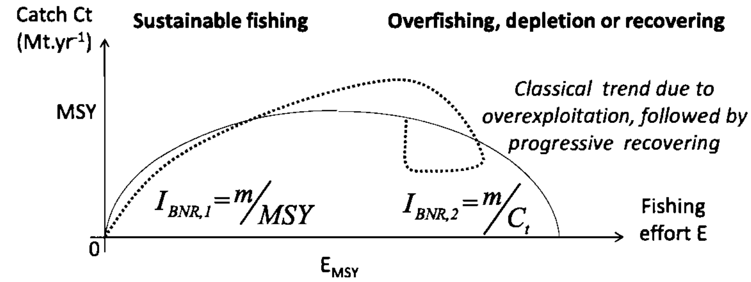

Figure 1. Trends in catches evolution according to fishing effort (in cases of equilibrium states).

We suggest an impact assessment of BNR depletion corresponding to the uptake of a mass (m) of a given marine species using the MSY related points. This allows a differentiation between different fish species, in relation with the size of their stock and the proportion that can be sustainably removed. The environmental impact on biotic natural resources ($I_{BNR,1}$) is thus calculated using the following formula:

$$I_{BNR,1} = \frac{m}{MSY} \quad \text{Eq. 1}$$

Thus, impacts of biotic extraction resources are here expressed in potential time of regeneration, i.e. in time required to restore an uptake of a particular species assuming equilibrium conditions. This equation is valid to assess impacts of biotic extractions as long as stocks are not overexploited (underexploited, moderately exploited or fully exploited, following the typology provided by FAO), i.e. that their catch never exceeded the MSY value. These cases appear on the left side of the graph in Figure 1. Nevertheless, FAO estimates that 32% of the stocks are not in this case, being either (1) overexploited, (2) depleted or (3) recovering from depletion (FAO, 2010). This corresponds to cases where C_t is respectively (1) higher than the MSY value, (2) smaller but decreasing because of previous overexploitation or (3) smaller and increasing. These cases appear on the right side of Fig. 1.

I_{BNR} should express that the uptake of one functional unit from an overexploited stock is worse than the uptake of the same unit from a stock species having the same MSY value and being sustainably exploited. Thus it appears important to multiply $I_{BNR,1}$ by a factor depending on the gap between current catches and MSY in the case of overexploited or recovering stocks. This factor should vary from 1 to infinite for values of C_t varying from MSY to zero (when the stock is severely depleted). One of the easiest possibilities for this factor is the ratio MSY over C_t. Thus $I_{BNR,2}$ would become:

$$I_{BNR,2} = \frac{m}{MSY} \times \frac{MSY}{C_t} = \frac{m}{C_t} \quad \text{Eq. 2}$$

In the particular case of a recent and unsteady overexploitation, where C_t is higher than MSY (Fig. 1; dashed line), we estimated that the impacts should be kept at $I_{BNR,1}$, to avoid minimizing I_{BNR} and to avoid the assessment of a transient state.

2.2. Fishing activities and life support functions assessment

The consensual framework of land use (Mila i Canals et al., 2007) has been developed in a context of intense agricultural and urban occupation as well as habitat transformation. Thus, parameters as time occupation or restoration and area used or transformed were particularly important for this impact assessment. In the case of marine activities were there is seldom continuous occupation and often slow habitat transformation, one of the major issues is to assess the quantity of biomass the ecosystem is deprived of (for fishing activities as well as for other uses, see in the discussion section). A quality index related to the alteration of biomass production capability of the ecosystem could be expressed in free Net Production (fNP). The fNP is the amount of biomass produced remaining in the ecosystem and usable for its own functioning after humans have removed a part of it from the ocean. To account for the trophic level of the biomass removed, we can use equivalence with the corresponding quantity of primary production that was necessary to produce it. Thus the quality index could be expressed in free Net Primary Production equivalent ($fNPP_{eq}$), being the Net Primary Production equivalent (NPP_{eq}) produced by the ecosystem minus the Human Appropriation of Net Primary Production equivalent ($HANPP_{eq}$). Both of them are expressed in kilogram of organic carbon per m² and per year. To fit the framework of (Mila i Canals et al., 2007), the impacts on LSF in marine ecosystems (I_{LSF}) would be the volume defined on Fig.2, expressed in kg of carbon (equivalent to primary carbon which was necessary for its production). For fishing activities, this quantity of carbon the ecosystem is deprived of, directly corresponds to the NPP_{use}, indicator (in kg C_{eq}) used in some LCA studies to quantify the impacts of seafood products, as described by Papatryphon et al., (2004).

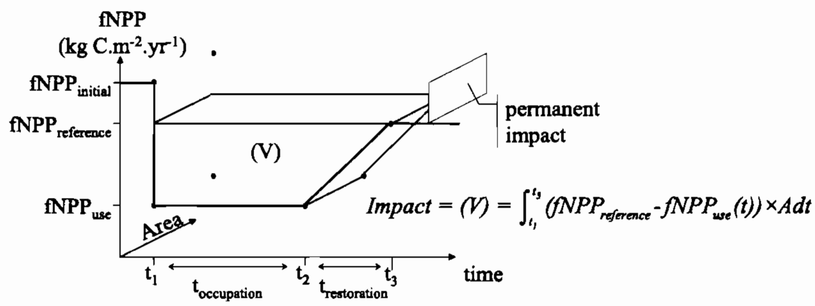

Figure 2. Graphical presentation of sea use impacts on LSF, inspired from (Mila i Canals et al., 2007).

The equivalences between fish masses and primary carbon required to sustain its production can be calculated, considering trophic levels (TL) of the uptake and the transfer efficiency between two trophic levels (TE). Updated values of TL are available per species in the fishbase database (Froese and Pauly, 2012) and updated TE values provided by Libralato et al., (2008) according to the types of ecosystems (i.e. oceanic systems, upwelling systems, tropical shelves, non-tropical shelves, coastal and coral systems). Based on these two parameters and a conservative 1:9 ratio of carbon to wet weight, NPP_{use} for a biomass uptake (m) in kg of wet weight can be calculated in kg of carbon as proposed by Pauly and Christensen (1995):

$$NPP_{use} = \frac{m}{9} \times TE^{TL-1} \quad \text{Eq. 3}$$

This assessment has to be regionalised beyond the regionalisation of TE because the impacts are highly depending on the area where it takes place. Moreover, the value of NPP_{use} allows quantifying how much carbon the ecosystem is deprived of, but it does not provide any information about the relative importance of this uptake relative to the total value of free biomass remaining within the ecosystem. Thus, this "classical" way to assess occupation and transformation impacts can be improved, by adding a factor expressing the scarcity of the biotic resource in the ecosystem. This was suggested by Weidema and Lindeijer (2001) and used by Michelsen (2007) for land use impact assessment. The goal of the factor is to express that for the same amount of biomass removed from the sea, if it is fished in an ecosystem where biomass is scarce, the impacts on ecosystem are worse than if biomass is fished in a fertile one. Two parameters play a role to determine the scarcity of the resource: the ecosystem size ($A_{ecozone}$) and its productivity ($NPP_{mean,ecozone}$). We defined $NPP_{ecozone}$ as the total amount of NPP produced in a given ecozone for a year:

$$NPP_{ecozone} = A_{ecozone} \times NPP_{mean,ecozone} \quad \text{Eq. 4}$$

Apart from LCA, this parameter was also introduced for fishing activities impact assessment by Halpern et al., (2008) and Libralato et al., (2008). For the calculation of the impacts due to sea use on life support functions (I_{LSF}), we suggest the introduction of this factor.

$$I_{LSF} = \frac{NPP_{use}}{NPP_{ecozone}} \quad \text{Eq. 5}$$

Thus I_{LSF} expressed the time required to regenerate the amount of biomass removed from the sea. The classification of the zones is based on the Marine Ecoregions Of the World, developed by Spalding et al., (2007) and recommended for land use impact assessment by Koellner et al., (2012). World maps of NPP values for year 2010 are also available (Oregon State University, 2010). These two types of data were merged in a Geographical Information System software to compute $NPP_{ecozone}$, also using the 200m-isobath (British Oceanographic Data Centre, 2003) and the coastlines (Wessel, 2012).

3. Results

The methods developed in the previous section were applied to two simple case-studies of fisheries. The first one is the fishing of 1 kg of Atlantic cod and the second one of 1 kg of herring. They are both fished along the coastal area of the USA (Gulf of Maine). Data used for this assessment as well as the resulting Characterisation Factors (CF) and impacts are detailed in Table 1.

Table 1. Data used for characterisation factors calculation and results obtained.

	Type of data *[unit]*	Fishery 1	Fishery 2
Inventory data	m *[kg ww]* [a]	1	1
	Ecozone	Gulf of Maine	Gulf of Maine
	Species	Atlantic herring	Atlantic cod
Data used for Characterisation Factors (CF) calculation	Stock status (2004)	Recovering from depletion	Depleted
	Catch *[kg ww.yr⁻¹]* [a]	114 090 [b]	4 950 [c]
	MSY[d] *[kg ww.yr⁻¹]* [a]	194 000	31 159
	TL	3	3.8
	TE (%)	14	14
	NPP_{use} *[kg C_{eq}]*	22	180
	$A_{ecozone}$ *[m^2]*	136 E9	136 E9
	$NPP_{ecozone}$ *[kg $C.yr^{-1}$]*	6.8 E10	6.8 E10
CF	CF_{BNR} *[$yr.kg\ ww^{-1}$]* [a]	8.8 E-15	2.0 E-13
	CF_{LSF} *[$yr.kg\ C^{-1}$]*	3.2 E-10	2.6 E-9
Impact	I_{BNR} *[yr]*	8.8 E-15	2.0 E-13
	I_{LSF} *[yr]*	3.2 E-10	2.6 E-9

[a] ww: wet weight
[b] Average values from 2001 to 2005
[c] Average values from 2003 to 2007
[d] Informative data (not used for these particular assessments).

Both in the case of biotic natural resources extraction impacts and of life support functions, impacts of Atlantic cod fishing are higher than for Atlantic herring. This is due to a previous severe depletion of the cod stock, a relatively small value of its MSY and its higher trophic compared to herring.

4. Discussion

The MSY-related biological reference points have been widely debated, first because they are based on equilibrium conditions or steady states periods not always observed and on the assumption that production in the ecosystem can reach a stable and unique maximum (Larkin, 1977). Furthermore, single species stock assessment methods do not seem accurate for a sustainable management of marine resources and an ecosystem-based management is preferred (Botsford et al., 1997). However, these reference points are still the most commonly used to compare multiple stocks, even if not used by all management agencies (Ricard et al., 2011). The biomass reference point B_{MSY} is the internationally agreed and legally binding reference point for managed fisheries in the United Nations Convention on the Law of the Sea and the United Nations Fish Stock Agreement and provides a useful basis for comparing stocks (Ricard et al., 2011). The expression of $I_{BNR,2}$ as the inverse of current yearly catches can appear as a loss of the information due to the exclusion of MSY. Nonetheless, since this assessment is applied for the interval MSY-extinction of the stock, and C_t is bounded to MSY, MSY is still indirectly taken into account in this assessment. Furthermore it would be difficult to provide a more precise and simple assessment because the impacts induced by fishing on overexploited stocks are hardly predictable. Thus, it is hard to assess these impacts using any simple indicator, except for stocks where information about the current stock biomass (B_t) and the stock biomass at MSY (B_{MSY}) would be available. In these cases, the gap between B_t and B_{MSY} could provide a relevant information on the severity of the impact.

NPP_{use} allows the assessment of impacts due to biomass removal for the biomass landed as well as for the discards, within the same impact category. It should be noted that the calculation of oceanic NPP at a global scale using remote sensing and global models is not very accurate: a factor two exists for resulting NPP values, depending on the methods used for the calculation (Carr et al., 2006). It is mainly due to the integration of the vertical dimension of the sea. This assessment is especially uncertain in coastal areas, due to a high level of sediments in the water column, and in some deep oceanic waters where a chlorophyll deep maximum layer is observed. Moreover, the indicator NPP_{use} also presents some limits: it does not allow the recognition of an imbalance induced by fishing activities. The new impact category we propose encourages the catches in lower trophic level. This could be detrimental if this practice would become excessive.

To allow a good consistency between the different impact categories, BNR and LSF impact assessment must fit existing frameworks. For BNR, the framework is neither well defined nor consensual yet, as no operational methods has been developed in LCA. Udo de Haes et al., (2002) reviewed some suggestions for the operationalisation of BNR assessment, using the balance of exploited biomass for every species, according to its worldwide use and natural replenishment (in kg per year). This balance of overexploited biomass is bounded on zero if the use is smaller than the replenishment. It is then divided by the worldwide stock of this species or its squared value according to the authors. The resulting ratio (Q) is the inverse of the time required to destroy the stock for this species. Udo de Haes et al., (2002) suggested the use of the Red list database edited by the International Union for Conservation of Nature. It provides a level of endangerment of the species, which can be converted to coarse values of (Q), but this method of calculation does not allow a precise differentiation between species (especially for those used below their rate of replenishment). One of the major advantages of our method is that it sidesteps these limitations.

Regarding the applicability of the framework developed for LSF impact assessment to other marine activities, fNPP appears particularly relevant as quality index: in the case of shading impacts due to constructions, or in the case of seafloor destruction due to constructions or destructive fishing, fNPP is also decreased. Thus, both indicator and methodology would be relevant (Langlois et al., in preparation). Moreover, I_{LSF} is compatible with terrestrial land use impact assessment, as the same types of data are also available for terrestrial ecosystems (availability for values of NPP_{use} or $\Delta fNPP_{eq}$ by type of use, biogeographical classifications and maps of NPP).

5. Conclusion

Thanks to these two new impact categories, both impacts on production capability (I_{LSF}) and stock status (I_{BNR}) can be assessed using the same unit (time), which could quite easily be extended to other impact pathways linked with land or sea use. Data required for the I_{BNR} calculation were easily available, and this would be the case for most exploited stocks. The same advantage can be underlined for I_{LSF}. Thus, the methodology proposed for biomass removal from the ocean seems promissing.

Alterations of habitat by biodiversity damage have been excluded, as well as damage of benthic habitats due to trawls. This should constitute the next step of methodological improvement for this impact assessment.

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7. References

Botsford, L. W., Castilla, J. C. and Peterson, C. H., 1997. The Management of Fisheries and Marine Ecosystems. Science 277 (5325), 509–515.

British Oceanographic Data Centre, 2003. GEBCO Digital Atlas, (consulted May 2012 http://www.bodc.ac.uk/projects/international/gebco/)

Carr, M.-E., Friedrichs, M. A. M., Schmeltz, M., Noguchi Aita, M., Antoine, D., Arrigo, K. R., Asanuma, I., et al., 2006. A Comparison of Global Estimates of Marine Primary Production from Ocean Color. Deep-Sea Res Pt II 53 (5-7), 741–770.

FAO, 2010. Part 1: World review of fisheries and aquaculture , in: FAO (Eds), State of World Fisheries and Aquaculture, 2010. Rome, pp. 3–89.

Froese, R. and Pauly, D., 2012. Fishbase. World Wide Web electronic publication

Graham, M., 1935. Modern Theory of Exploiting a Fishery, and Application to North Sea Trawling. ICES Journal of Marine Science 10 (3), 264–274.

Halpern, B. S., Walbridge, S., Selkoe, K. A., Kappel, C. V., Micheli, F., D'Agrosa, C., Bruno, J. F., et al., 2008. A Global Map of Human Impact on Marine Ecosystems. Science 319 (5865), 948–952.

Hilborn, R. and Walters, C. J., 1992. Quantitative Fisheries Stock Assessment: Choice, Dynamics and Uncertainty. Rev Fish Biol Fisher 2 (2), 177–178.

Kaiser, M. J., Collie, J. S., Hall, S. J., Jennings, S. and Poiner, I. R., 2002. Modification of Marine Habitats by Trawling Activities: Prognosis and Solutions. Fish and Fisheries 3 (2), 114–136.

Koellner, T., de Baan, L., Beck, T., Brandao, M., Civit, B., Goedkoop, M., Margni, M., i Canals, L., Müller-Wenk, R., Weidema, B. and Wittstock, B., 2012. Principles for Life Cycle Inventories of Land Use on a Global Scale. Int J Life Cycle Assess 1–13.

Langlois, J., Hélias, A., Delgenes, J. P. and Steyer, J. P., 2011. Review on Land Use Considerations in Life Cycle Assessment: Methodological Perspectives for Marine Ecosystems, in: Finkbeiner, M. (Eds), Towards Life Cycle Sustainability Management. Springer, Berlin, pp. 85–96.

Larkin, P. A., 1977. An Epitaph for the Concept of Maximum Sustained Yield. T Am Fish Soc 106 (1), 1–11.

Libralato, Coll, M., Tudela, S., Palomera, I. and Pranovi, F., 2008. Novel Index for Quantification of Ecosystem Effects of Fishing as Removal of Secondary Production. Marine Ecology Progress Series 355, 107–129.

Michelsen, O., 2007. Assessment of Land Use Impact on Biodiversity. Int J Life Cycle Assess 13 (1), 22–31.

Mila i Canals, L., Bauer, C., Depestele, J., Dubreuil, A., Knuchel, R., Gaillard, G., Michelsen, O., Müller-Wenk, R. and Rydgren, B., 2007. Key Elements in a Framework for Land Use Impact Assessment Within LCA. Int J Life Cycle Assess 12, (1) 5–15.

Oregon State University, 2010. Ocean Productivity, (consulted February 2012 http://www.science.oregonstate.edu/ocean.productivity/custom.php)

Papatryphon, E., Petit, J., Kaushik, S. J. and van der Werf, H. M. G., 2004. Environmental Impact Assessment of Salmonid Feeds Using Life Cycle Assessment (LCA). AMBIO: A Journal of the Human Environment 33 (6), 316–323.

Pauly, D. and Christensen, V., 1995. Primary Production Required to Sustain Global Fisheries. Nature 374 (6519), 255–257.

Pelletier, N. L., Ayer, N. W., Tyedmers, P. H., Kruse, S. A., Flysjo, A., Robillard, G., Ziegler, F., Scholz, A. J. and Sonesson, U., 2007. Impact Categories for Life Cycle Assessment Research of Seafood Production Systems: Review and Prospectus. Int J Life Cycle Assess 12, 414–421.

Ricard, D., Minto, C., Jensen, O. P. and Baum, J. K., 2011. Examining the Knowledge Base and Status of Commercially Exploited Marine Species with the RAM Legacy Stock Assessment Database. Fish and Fisheries (in press).

Schaefer, M. B., 1954. Some Aspects of the Dynamics of Populations Important to the Management of the Commercial Marine Fisheries. Bull. IATTC 1 (2), 27–56.

Spalding, M. D., Fox, H. E., Allen, G. R., Davidson, N., Ferdana, Z. A., Finlayson, M., Halpern, B. S., et al., 2007. Marine Ecoregions of the World: a Bioregionalization of Coastal and Shelf Areas. BioScience 57 (7), 573–583.

Udo de Haes, H. A., Finnveden, G., Goedkoop, M., Hauschild, M., Hertwich, E., Hofstetter, P., Jolliet, O., et al., 2002. Life cycle impact assessment: Striving towards best practice, SETAC Press, Pensacola.

Weidema, B. P. and Lindeijer, E., 2001. Physical Impacts of Land Use in Product Life Cycle Assessment, Technical University of Denmark.

Wessel, P., 2012. GSHHS Coastlines, SOEST Univesity of Hawai'i, Honolulu (consulted February 2012, http://www.soest.hawaii.edu/pwessel/gshhs/index.html).

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