

Sea use impact category in life cycle assessment: characterization factors for life support functions

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Abstract

Purpose The impact of human activities on marine environments is poorly addressed by the scope of life cycle impact assessment (LCIA). The aim of this study is to provide characterization factors to assess impacts of sea use such as fishing activities or seafloor destruction and transformation on the life support functions of marine ecosystems.

Methods The consensual framework of land use for ecosystem services damage potential assessment was applied, according to the recent United Nations Environment Programme-Society for Environmental Toxicology and Chemistry (UNEP-SETAC) guidelines, using the free net primary production as a quality index of life support functions.

Results and discussion The impact of shading, biomass removal, seafloor destruction, and artificial habitat creation on the available quantity of organic biomass for the ecosystem functioning was quantified at the midpoint level with a

common unit (kg of organic carbon equivalent). It included effects of human interventions on both the ecosystem production potential and the stock of biomass present within the ecosystem. Characterization factors (CF) for biomass removal vary from 0.1 kg_{Ceq}kg⁻¹ for seaweed to 111.1 kg_{Ceq}kg⁻¹ for tunas, bonitos, and billfishes. CF for seafloor destruction range from 0.164 kg_{Ceq}m⁻² for a temperate seagrass ecosystem to 0.342 kg_{Ceq}m⁻² for an intertidal tropical rocky habitat. **Conclusions** This study provides an operational method in order to compute sea use impact assessment.

Keywords Ecosystem services · Free net primary production · Life cycle impact assessment · Life support functions · Marine ecosystem · Sea use

1 Introduction

Human activities lead to particularly strong environmental changes (Agardy et al. 2005; Pauly et al. 2005; Halpern et al. 2008) within marine ecosystems. The recent development of marine bioenergy (Inger et al. 2009) and the present-day high fishing rates point to the necessity of assessing the corresponding environmental impacts. As life cycle impact assessment (LCIA) methods try to represent global environmental damage due to human activities, they should allow for the evaluation of all these changes. Several studies underlined the need for a sea use impact category focusing on fishing or aquaculture activities (Pelletier et al. 2007; review by Vázquez-Rowe et al. 2012 and Avadi and Fréon 2013). Langlois et al. (2014a) also underlined this necessity and suggested extending the scope of this new kind of category from fisheries to marine construction, aquaculture, and navigation. They summarized the different pathways that should be included within the sea use impact category, including cause-effect chains to quantify the impacts

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from 1) biotic natural resource depletion, 2) climate change, 3) ecosystem services damage potential, and 4) biodiversity damage potential (Fig. 1). They suggested focusing on characterization factors related to biotic primary production, as an easily computed and relevant proxy for one of the ecosystem services: life support functions (LSF), sensu the Millennium Ecosystem Assessment of the marine ecosystems (Diaz et al. 2005). This approach is particularly suitable in a context of severe overfishing (Coll et al. 2009; FAO 2010) that diminishes the quantity of biomass available for ecosystem functioning. The present study aims at performing this assessment, by providing characterization factors (CF) for the impacts of sea use on LSF. The method of assessment may be applicable to every type of marine ecosystem and its related activities, including, as aforementioned, fishing, marine construction, navigation, and aquaculture. It may also point to the impact associated to these activities such as shading, biomass removal, benthic destruction, and artificial habitat creation as identified by Langlois et al. (2014a) in the case of LSF damage potential assessment.

In the present study, the methodology used for the CF calculation corresponding to LSF damage potential assessment is first explained. The complete list of CF is then presented. The consistency of the results is finally discussed, as well as limitations and perspectives resulting from this methodological framework.

2 Methods

In this section, when an impact is the result of an equation, in the right-hand side, the first part of the product is the inventory data and the second part is the CF; each part is in square brackets. In Fig. 1, the red pathway illustrates the outline of this study, highlighting the impact pathways for LSF involved

here. In this pathway, coming from the human intervention level to the areas of protection, impacts are assessed at the midpoint level (rounded rectangle in Fig. 1). Indirect pathways linking other human activities (e.g., toxic component emissions, nitrate release) to the LSF of the sea are not represented because they are already represented through other midpoint impacts in LCIA (e.g., ecotoxicity, eutrophication).

2.1 Framework for life support functions impact assessment

Some recent developments in land use impact assessment sealed the framework of Mila i Canals et al. (2007a), as a result of a large cooperation of authors through the United Nations Environment Programme-Society for Environmental Toxicology and Chemistry (UNEP-SETAC) Life Cycle Initiative (Koellner et al. 2013). Its graphical representation is provided in Fig. 2. This framework has been applied using the free net primary production in primary carbon equivalent ($fNPP_{eq}$) as a quality index, as recommended by Langlois et al. (2014a) for the impact assessment of LSF damage potential at the midpoint level. $fNPP_{eq}$ is the flow of biotic production available for the ecosystem functioning, expressed in kilograms of organic carbon per square meter per year ($kg C_{eq} m^{-2} year^{-1}$). If a part of the potentially produced biomass should be removed by human activity, then this amount would not be present anymore in the ecosystem during a certain time, thus decreasing the LSF performed by the ecosystem. An exception to this decreasing in performance could be cases of over-availability of biomass. This is particularly the case for harmful algal blooms, but as far as we know, there is no substantial in situ exploitation of microalgae to date which solves the issue of a beneficial removal by humans of primary production.

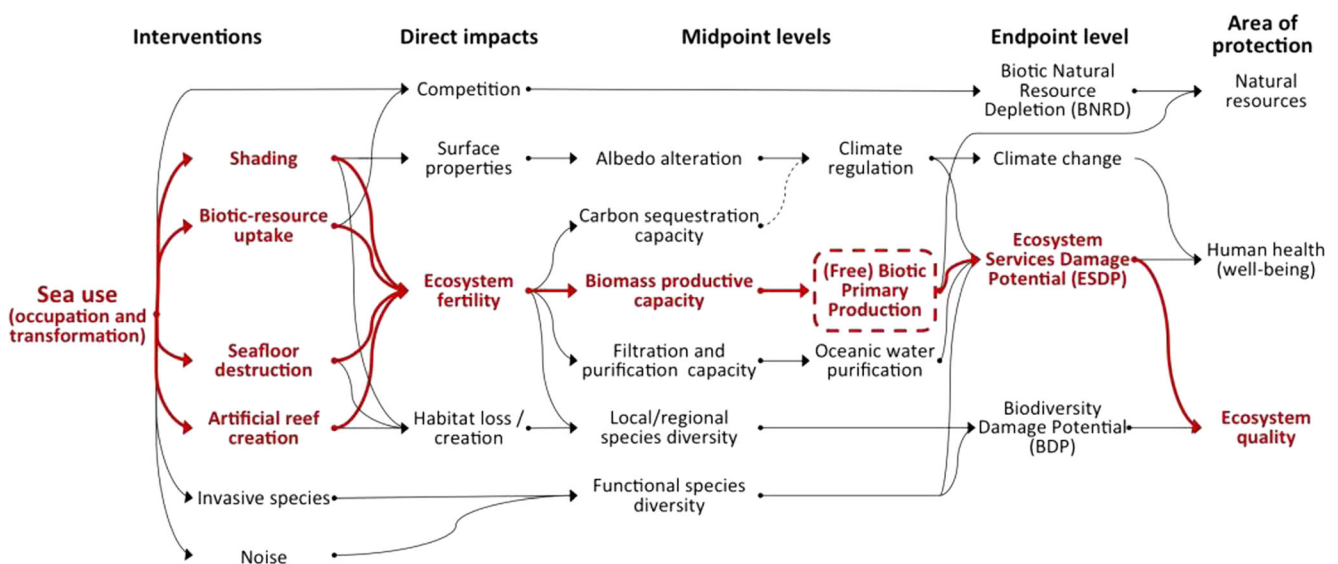


Fig. 1 Impact pathways for life support functions and their location in the global cause-effect chain of sea use. Impact pathways for life support functions are in *bold* and *red* (adapted from Langlois et al. 2014a)

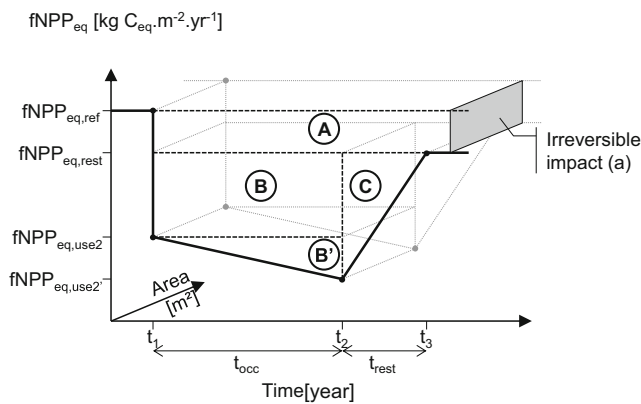


Fig. 2 Graphic representation of sea use impacts. Adapted from the representation of the land use impact proposed by Mila i Canals et al. (2007a, b) (for abbreviations, see text)

In this study, an area has been transformed from a sea use type (1) to a sea use type (2) at t_1 and has then been occupied during a certain time of occupation (t_{occ}). This second sea use type creates an amount of impacts on LSF due to sea transformation (TI_{LSF}) and sea occupation (OI_{LSF}). Its current quality level before transformation ($fNPP_{eq,ref}$) is degraded (or enhanced) to a worsened (or better) state ($fNPP_{eq,use2}$). When sea occupation (2) stops at t_2 , the area regains a reference quality level ($fNPP_{eq,rest}$) after a period of restoration (t_{rest}). Irreversible impacts of transformation on LSF ($TI_{LSF,irrev}$) are represented by the section (a) of volume A or by volume A if a time horizon is considered. Reversible impacts of transformation on LSF ($TI_{LSF,rev}$) are represented by volume C , and reversible impacts of occupation on LSF ($OI_{LSF,rev}$) are represented by volumes B (according to quality-level changes of transformation) and B' (according to quality changes occurring during occupation). The area $B+B'$ represents the total quality change during occupation. To simplify this explanation, a case without any initial activity was chosen as sea use type (1). For terrestrial ecosystems, it was recommended by the UNEP-SETAC guidelines (Koellner et al. 2013) to neglect the time of transformation before the occupation and restoration times, as well as the quality changes occurring during occupation (i.e., $fNPP_{eq}(t_1)$ equals $fNPP_{eq}(t_2)$ and therefore $B'=0$).

Considering reversible changes and applying these hypotheses (transformation time and quality changes during occupation neglected), the corresponding equations for impact assessment are as follows:

$$OI_{LSF,rev} = [A \times t_{occ}] \times [fNPP_{eq,rest} - fNPP_{eq,use2}] \quad (1)$$

$$TI_{LSF,rev} = [A] \times \left[\frac{(fNPP_{eq,rest} - fNPP_{eq,use2}) \times A \times t_{rest}}{2} \right] \quad (2)$$

These two equations can be used when there is no previous activity, or if there are no transformation impacts related to this

previous activity. In those situations, $fNPP_{eq,use2}$ is simply replaced by $fNPP_{eq,use1}$. Further details on how to account for transformation impacts of previous activities are provided in the UNEP-SETAC guidelines (Koellner et al. 2013).

The application of this framework to the particular cases of biomass removal (fishing, extensive aquaculture, and most shellfish aquacultures), shading, seafloor destruction, and artificial habitat creation is illustrated in Fig. 3 and detailed in the following section. $fNPP_{eq}$ can be modified by two types of effects: changes in the available stock of exploited biomass (light grey on the graph) or changes in the production potential in the ecosystem (dark grey on the graph).

2.2 Impacts of biomass removal on LSF

In the case of fishing activities, the equivalence between a given weight of fish and the primary carbon required to sustain its production can be easily calculated. This quantity is called net primary production use (NPP_{use}), considering trophic levels (TL) of the catch and the transfer efficiency (TE) of the production between two adjacent trophic levels. NPP_{use} for a biomass uptake (m) in kilograms of wet weight can be calculated in kilograms of primary carbon equivalent (Pauly and Christensen 1995). It has already been used as an indicator for LCIA of fisheries (reviews by Avadi and Fréon 2013) and aquaculture products (review by Henriksson et al. 2012; Jerbi et al. 2011; Efole Ewoukem et al. 2012). The NPP_{use} is the impact value ($SI_{LSF,remov}$, in Eq. (3)) and corresponds to the quantity of carbon that does not benefit the ecosystem, due to the uptake of biomass within the available stock.

Therefore, there is no need to estimate the duration of destruction t_{destr} , although for the sake of clarity it is represented in Fig. 3a (which allows the representation of NPP_{use} as a volume). Transformation impacts occur when the biomass exits the ecosystem. The overall ecosystem productivity is not negatively affected and compensation processes take place to replace part of the removed biomass, although the total biomass during exploitation always remains lower than the original one (Graham 1935; Schaefer 1954). This issue of biomass depletion has been accounted for under the area of protection of natural resources, as a biotic natural resource depletion (BNRD) (Emanuelson et al. 2012; Langlois et al. 2014b).

$$SI_{LSF,remov} = NPP_{use} = [m] \times \left[\frac{TE^{1-TL}}{9} \right] \quad (3)$$

In this formula, the mass (m , in kg) is an inventory data, characterizing the functional unit of the assessment, and the second part of the equation is the CF. Here, 1:9 is a conservative value of the ratio of C to wet weight (Pauly and Christensen 1995) and TE is expressed as a

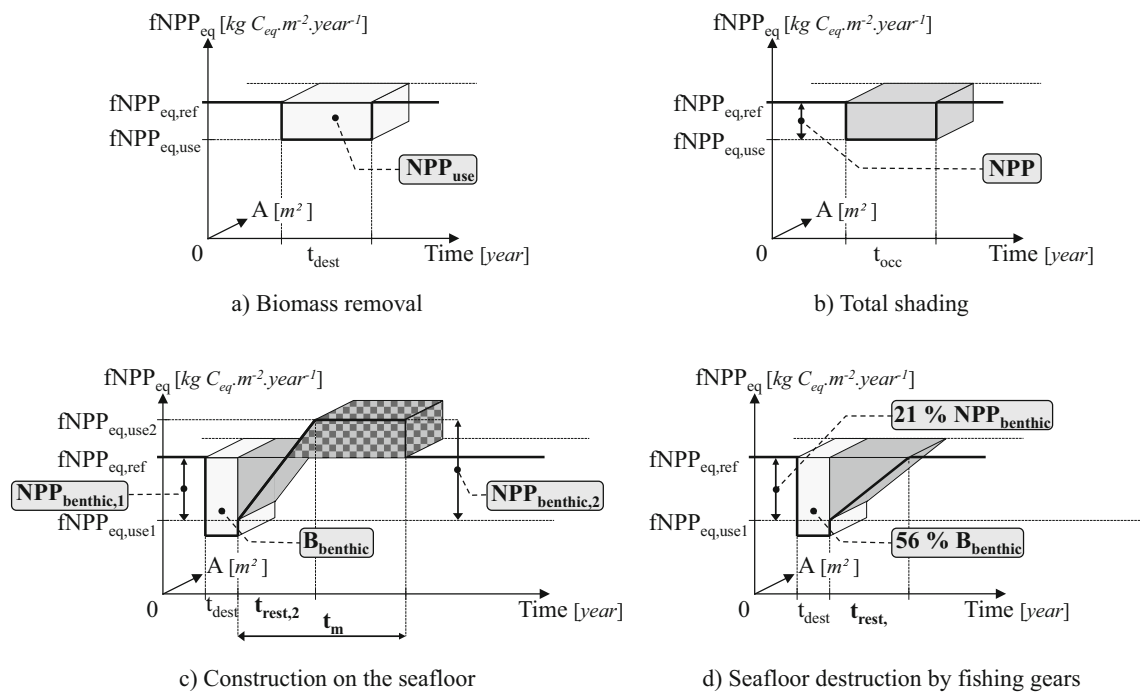


Fig. 3 Representation of sea use on life support functions due to a biomass removal, **b** shading at the sea surface, **c** moored constructions, and **d** seafloor destruction due to fishing gear. Impact due to stock

changes appears in *light grey* and that due to NPP changes appears in *dark grey* (*checkered* for positive impact); mandatory data for the calculations appear in *bold*

ratio.¹ Complementary sources of data used for the CF calculation and discussion on their accuracy are provided in section 1 of the Electronic Supplementary Material. The reason why the removal of extensively cultivated organisms and most shellfish aquacultures should be accounted for in the same manner is that here, part or the whole food uptake comes from the ecosystem where the animals are farmed. This situation contrasts with intensive aquaculture where there is nearly no local uptake of food from the ecosystem because fish are entirely fed by humans, like in the case of aquaculture of salmonids (salmon, trouts). The impact of cultivating these later species, including aquafeed production and eutrophication among other factors, is properly taken into account in conventional life cycle assessments (LCAs) (Henriksson et al. 2012).

Discarded organic material, such as non-commercial fish thrown back into the sea after having been fished out, is not considered as biomass removed from the ocean. Nonetheless, the corresponding mortality must be taken into account, either as a fishery-specific impact as reviewed in Vázquez-Rowe et al. (2012) and Avadi and Fréon (2013), or as a BNRD under the area of protection of natural resources a BNRD as previously indicated. The organic carbon physically contained within a

mass of discarded biomass (D , in kg) should not be accounted for as primary matter the ecosystem is deprived of. The ratio of 1:9 is still used here for the conversion of wet weight to carbon. Thus, the impact of discards on LSF is expressed as

$$SI_{LSF, discard} = D \frac{TE^{1-TL}}{9} - \frac{D}{9} = [D] \times \left[\frac{TE^{1-TL}-1}{9} \right] \quad (4)$$

In cases where part of the discard survives fishing operations, predator mortality during its return to its original habitat, and additional mortality due to injury and stress, a correction factor (equal to the best estimate of the actual mortality rate) must be applied to the discarded biomass in the previous equation (Eq. (4)).

2.3 Impacts of shading on LSF

Shading can lead to a decrease in the capacity of production of phytoplankton biomass, by reducing or preventing photosynthesis (Johnson et al. 2008). In this study, the shading impact of an opaque structure floating at the sea surface ($OI_{LSF, shade}$) was considered. The annual averaged value of NPP characterizing the study site (NPP_{local}) was assumed to be threatened, depending on the elapsed time of occupation, leading to

$$OI_{LSF, shade} = [A \times t_{occ}] \times [NPP_{local}] \quad (5)$$

where A is an inventory data for the section of the actual shading area. This area can be estimated as the projected

¹ In most papers, TL is expressed as a percentage and, following Pauly and Christensen (1995), the numerator of the equation is equal to TE^{TL-1} . But this equation is only valid in the special case where $TE=10\%$.

shadow of the structure on a horizontal plane, integrated during the whole daytime period. Nutrients remain available in the zone, because production in the water column is avoided without destruction or removal of biomass. Thus, primary production can instantaneously resume, as soon as the floating structure has been removed. NPP values can be computed from satellite radiometer reflectance in terms of productivity, allowing for a worldwide coverage (Oregon State University 2010). Ocean color satellite-borne sensors provide an estimate of light penetration (via a relationship between the blue-to-green reflectance ratio) and of attenuation, in the water column (Behrenfeld and Falkowski 1997). This approach is now routinely used for the open ocean where phytoplankton itself is the main contributor to attenuation (Gattuso et al. 2006).

World maps of monthly NPP values for 9 years, from 2003 to 2011, were used to calculate the annual marine productivity of the oceans (Oregon State University 2010). Details on how data were computed are provided in section 2 of the Electronic Supplementary Material.

The above-mentioned impacts apply only to phytoplankton. In cases of large floating infrastructure, the impact on seagrass, macroalgae, microalgae, and/or autotrophic coral can be assimilated to seafloor transformation, as detailed below.

2.4 Impact of seafloor destruction on LSF

In coastal waters, the major sources of seabed disturbance are near-bed currents, wind-induced waves, and bottom trawling and dredging (Charpy-Roubaud and Sournia 1990). The latter type of damage is shared with offshore mining by dredging, sucking, and drilling. Even if less frequent, benthic constructions moored on the seafloor also disturb marine ecosystems (Halpern et al. 2008).

In coastal and shallow environments, benthic primary production represents an important part of the NPP: the seafloor receives a significant amount of sunlight in shallow waters and can therefore sustain benthic primary production with seagrass, macroalgae, microalgae, and/or autotrophic coral (Gattuso et al. 2006). It has been estimated that primary production of microbenthic algae ($50 \text{ g C m}^{-2} \text{ year}^{-1}$) and macrobenthic algae ($375 \text{ g C m}^{-2} \text{ year}^{-1}$) contributes to about 10 % of the total primary production in the oceans (Charpy-Roubaud and Sournia 1990). These two types of benthic producers were dealt with in the assessment, except for particular types of ecosystems with specific values of biomass and production (mangroves and seagrass meadows) (Mateo et al. 2006). For example, in addition to their local importance, seagrass meadows contribute to 1 % of NPP at the global scale (Duarte and Chiscano 1999).

The case of seafloor destruction due to moored constructions (corresponding to a total destruction of the standing biomass and of its production potential) is dealt with first,

followed by the case of fishing gear (corresponding to a less intensive destruction).

Seafloor transformation (moored constructions) Impacts due to anchored constructions are illustrated in Fig. 3c. Two distinct types of impact on the free biotic production can be observed:

- (1) The present biomass is destroyed, covered by the hard structures anchored on the seafloor; it corresponds to an uptake of biomass within the standing stock, in light grey on the graph.
- (2) Benthic production disappears because the biomass is not present anymore, but it regenerates progressively until reaching a new steady state after a certain time of restoration. It corresponds to transformation impacts, due to changes in the seafloor production potential until the ecosystem has recovered, in dark grey. In the case of a hard structure moored on a soft-bottom seafloor characterized by a poor production potential, there can be some positive effects due to the transformation (the positive effects we can see in the second stage of recovering are marked with small white crosses in Fig. 3c). Indeed, a hard structure (such as concrete blocks or rocks) can be comparable to a hard-bottom seafloor, playing the role of an artificial reef (Moura 2010).

Because recolonization occurs as soon as the occupation begins on hard structures in marine ecosystems, occupation impacts (*sensu* land use impact assessment) can be neglected.

The first type of impact (initial destruction) accounts for the destroyed biomass. This impact, due to destruction of the standing biomass on the seafloor ($SI_{LSF,seafloor}$), is the quantity of organic carbon that does not benefit the ecosystem during this phase. Although this biomass still remains within the ecosystem, in most cases, it is not available for the ecosystem functioning anymore, as it is flattened between the seafloor and the anchored construction (except if it is swept away before construction and not removed from the ecosystem). Remineralization can take place but it is slowed down and neglected here because chemical nutrients are usually not limiting factors at depth. The available data are not provided in the units of $\Delta fNPP$ (the available data are in kg C m^{-2} instead of $\text{kg C m}^{-2} \text{ year}^{-1}$; therefore, as for biomass removal, t_{destr} is represented only for the sake of clarity). It can be directly calculated using the following equation instead of the general formula of sea use framework (Eqs. (1) and (2)):

$$SI_{LSF,seafloor} = [A] \times [B_{benthic,1}] \quad (6)$$

where A is the area destroyed (in m^2). It is an inventory data. $B_{benthic}$ (kg C m^{-2}) is the benthic biomass (including microphytobenthos, Cahoon 1999; macrophytobenthos,

Charpy-Roubaud and Sournia 1990; and macrozoobenthos, Ricciardi and Bourget 1999; Cusson and Bourget 2005) of the sediment column, or global primary values for seagrass (Green and Short 2003) and mangrove ecosystems (Mateo et al. 2006), whose values depend on the ecosystem type where the transformation occurs (see [Electronic Supplementary Material](#) for more details).

After destruction, benthic production NPP_{benthic} disappears until the full restoration of the ecosystem. Immediately after construction, recolonization can take place and a new ecosystem can appear. Thus, impacts due to transformation of the seafloor ($TI_{\text{LSF,seafloor}}$) can be expressed as

$$TI_{\text{LSF,seafloor}} = [A] \times \left[\frac{NPP_{\text{benthic}2} \times t_{\text{rest}2}}{2} + (NPP_{\text{benthic}1} - NPP_{\text{benthic}2} \times t_m) \right] \quad (7)$$

where A is the area transformed, being the same inventory data as in Eq. (6). In the case of a construction, the second sea use type is comparable to a rocky habitat; thus $NPP_{\text{benthic}2}$ and $t_{\text{rest}2}$ are values associated to rocky habitats in the studied biome. $NPP_{\text{benthic}1}$ is the value of the benthic NPP in the type of ecosystem where the construction is built. Irreversible impacts are taken into account over the timeframe t_m . In LCA, this time horizon depends on the perspective chosen for the assessment. It is usually chosen as 20, 100, or 500 years for individualist, hierarchist, or egalitarian perspectives respectively (Guinée et al. 2001).

This framework can be extended to any other seafloor transformation, including a transformation from a hard bottom to a soft bottom (for example, if a construction, which was previously built on a soft bottom, is removed from the ocean). The two formulas provided (Eqs. (6) and (7)) can still be used for these cases.

t_{rest} is the time needed for the seafloor to recover a new steady state after the disturbance, i.e., the time needed for the biomass to regenerate in the disturbed area. Average values of restoration time have been estimated according to the type of bottom substrate. The seafloor characteristics strongly influence the capacity for benthos to recover a steady state after disturbance. Initial responses and recovery rates of the seabed can be predicted from the physical stability of the seabed (Dernie et al. 2003). For this reason, Nilsson and Ziegler (2007) built a method of assessment of destructive fishing impacts in LCA, distinguishing sandy, rocky, and muddy floors. As the biomass of secondary benthic organisms also depends on the type of substrate, this classification was also used in this study. The Marine Life Information Network's (MarLIN) recoverability classes (Hiscock et al. 1999) and the work of Nilsson and Ziegler (2007) have been used to calculate t_{rest} (see [Electronic Supplementary Material](#)).

Similarly to B_{benthic} , NPP_{benthic} values included microphytobenthos (Cahoon 1999) and macrophytobenthos

(Charpy-Roubaud and Sournia 1990) of the sediment, or global primary values for seagrass (Green and Short 2003) and mangrove ecosystems (Mateo et al. 2006) (see [Electronic Supplementary Material](#) for more details). **Fishing gear** Seafloor destruction due to fishing activities is caused by towed bottom-fishing gear. Potential impacts of fishing gears are specific with respect to the type of fishing gear, disturbance regime (i.e., frequency), habitat, and environment. A meta-analysis of their impacts has been performed for intertidal dredging, scallop dredging, or trawling (Collie et al. 2000), highlighting a substantial need for more data in this field. Hiddink et al. (2006) estimated that the bottom trawl fleet reduced benthic biomass and production by 56 and 21 % respectively in average, in comparison to a situation without fishing activities. There are no occupation impacts on average, because fishing gear does not remain on the seafloor after its passage. Nevertheless, production is decreased, as the reduced biomass cannot fully assume its role of production. This lack of production takes place while the biomass recuperates, which can take a long time in some ecosystems like the deep coral ones exploited in high latitude of the northern hemisphere. Thus, it is part of the transformation impacts, considering that the nature of the seafloor is identical before disturbance and at the end of the restoration period. Impacts are illustrated in Fig. 3d.

Hence, the impact of seafloor destruction due to fishing gear on benthic stock $SI_{\text{LSF,seafloor_trawl}}$ can be expressed as follows (based on Eq. (6) with the same representation of unused t_{destr} on Fig. 3d):

$$SI_{\text{LSF,seafloor_trawl}} = [A] \times [B_{\text{benthic}} \times 56\%] \quad (8)$$

For the impact of transformation, the equation is based on Eq. (7):

$$TI_{\text{LSF,seafloor_trawl}} = [A] \times \left[NPP_{\text{benthic}} \times 21\% \times \frac{t_{\text{rest}}}{2} \right] \quad (9)$$

All data for biomass, production, or restoration times used for seafloor destruction impact assessment are summarized in the [Electronic Supplementary Material](#), as well as all the impact assessment formulas on LSF.

3 Result and discussion

3.1 Characterization factors for sea use impact assessment on LSF

For biomass removal, characterization factors (CF) have been calculated in terms of groups of most commonly found species associated to different types of ecosystems. Values, as well as the data used for the calculation, are associated with Table 1.

Table 1 Characterization factors for impacts of sea use on LSF (CF_{LSF}) due to biomass removal. Trophic level (TL) and trophic efficiency (TE) values come from Pauly and Christensen (1995)

Ecosystem type	Groups of species	TL	TE	$CF_{LSF,remov}$ ($kg_{C_{eq}}kg^{-1}$)	
Oceanic (gyre) systems	Tunas, bonitos, billfishes	4.2	0.12	98.3	
	Krill	2.2	0.12	1.4	
Upwelling systems	Anchovies, sardines	2.6	0.05	13.4	
	Jacks	3.2	0.05	80.9	
	Mackerels	3.3	0.05	109.2	
	Squids	3.2	0.05	80.9	
	Small pelagics	2.8	0.1	7.0	
Tropical shelves	Misc. teleosteans	3.5	0.1	35.1	
	Jacks, mackerels	3.3	0.1	22.2	
	Tunas, bonitos, billfishes	4	0.1	111.1	
	Squids, cuttlefishes, octopuses	3.2	0.1	17.6	
	Shrimps, prawns	2.7	0.1	5.6	
	Lobster, crabs, and other invertebrates	2.6	0.1	4.4	
	Sharks, rays, chimaeras	3.6	0.1	44.2	
	Cods, hakes, haddockes	4.3 ^a	0.14	73.0	
Non-tropical shelves	Redfishes, basses, congers	3.4	0.14	12.4	
	Miscellaneous marine fishes	3.2	0.14	8.4	
	Jacks, mullets, sauries	3.8	0.14	27.3	
	Herrings, sardines, anchovies	3.0	0.14	5.7	
	Shrimps and other crustaceans	2.3	0.14	1.4	
	Squids, cuttlefishes, octopuses	3.2	0.14	8.4	
	Flounders, halibuts, soles	2.9	0.14	4.7	
	Mackerels, cutlassfishes	3.4	0.14	12.4	
	Diadromous fishes	2.4	0.14	1.7	
	Sharks, rays, chimaeras	3.7	0.14	22.4	
	Coastal and coral reefs	Bivalves and other molluscs	2.1	0.13	1.0
		Miscellaneous marine fishes	2.8	0.13	4.4
Herrings, sardines, anchovies		3.2	0.13	9.9	
Seaweeds		1.0	0.13	0.1	
Jacks, mackerels		3.3	0.13	12.1	
Diadromous fishes		2.8	0.13	4.4	
Shrimps, prawns		2.6	0.13	2.9	
Crustaceans and other invertebrates		2.4	0.13	1.9	
	Turtles	2.4	0.13	1.9	

^a The original value (3.8) was updated from the FishBase database (Froese and Pauly 2012)

CF_{LSF} range from $0.1 kg_{C_{eq}}kg^{-1}$ for seaweed to $111.1 kg_{C_{eq}}kg^{-1}$ for tunas, bonitos, and billfishes, with a non-weighted average value of $24.9 kg_{C_{eq}}kg^{-1}$. Values are mainly driven by TL of the organisms and to a lesser extent by their TE.

CF for shading impacts are local values of NPP. Their distribution is illustrated on the world map in Fig. 4.

Strong geographical variations depend mostly on the latitude, on physical processes such as the presence of

upwelling, and on depth in shallow waters. CF are also provided by marine provinces on a worldwide scale map in the [Electronic Supplementary Material](#). In average, values range from 117 to $307 g C m^{-2} year^{-1}$ for deep-sea and coastal areas respectively.

CF for impacts of the seafloor destruction are summarized in Table 2. Note that as NPP values related to benthic production are often identical for a given climate zone, none irreversible impact is observed for the main part of the ecosystem and thus CFs according to time horizon are identical.

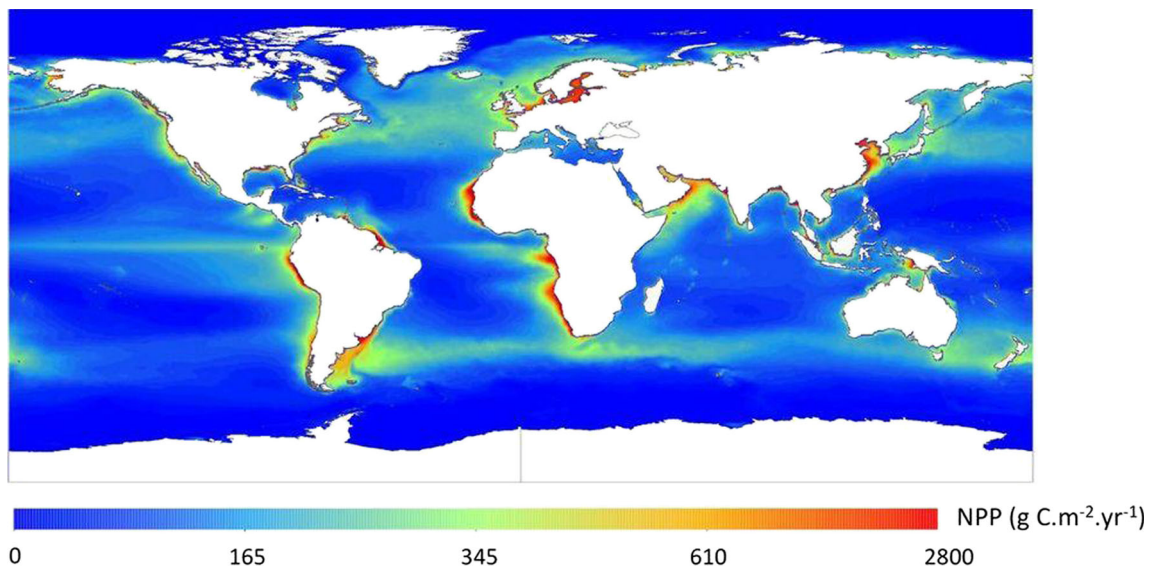


Fig. 4 World map of the characterization factors for impacts of sea use on LSF due to shading (yearly NPP, calculated from the Oregon State University 2010)

3.2 Limits and perspectives

The use of $fNPP_{eq}$ as a quality index for marine ecosystems is of significant interest for LCIA. It allows the expression at the midpoint level and in the same unit, of impacts from different types of disturbance, including biomass removal, shading, seafloor destruction, and artificial habitat creation. This is particularly important for activities such as algaculture, which induces all four of these interventions at the same time. It should be particularly interesting for comparison studies on sea settlement as well. Although certain human activities produce noise that is considered to have a significant impact (McCarthy 2004), it was not considered here due to the lack of relevant impact data. Nonetheless, including noise using the $fNPP_{eq}$ approach should be possible, providing that data describing associated mortalities and disturbance of marine organisms become available.

Concerning human intervention related to biomass removal, the LSF impact assessment proposed here appears particularly relevant, demonstrating a good comparison between different fisheries and/or aquaculture or between aquaculture types. Further precise calculations could even be performed, using data of TL by species, from the FishBase database (Froese and Pauly 2012), instead of average TL by group of species. Moreover, NPP_{use} has already been used by several authors in LCA (review by Henriksson et al. 2012; Jerbi et al. 2011; Efole Ewoukem et al. 2012). Indirect impact of fishing is not accounted for in the present method, as other authors have done using NPP_{use} . It occurs when fishing concerns the low trophic levels, thus leading to a lack of production for the higher levels, due to a lower amount of available feed, with repercussions along the whole food chain. Libralato et al. (2008) proposed to take this phenomenon into account.

Considering these indirect impacts of fishing in the present approach would lead to double counting of the biomass removal impacts, as the method already includes the potential lack of biomass for the ecosystem functioning.

The impacts on LSF due to shading may also be refined by expressing impacts of partial shading due to depth and to light penetration through the structure. For example, partial shading is provided by aquaculture structures (nets, ropes) or organisms (e.g., algae or bivalves).

Data are particularly scarce for production and standing biomass of phytobenthos (see [Electronic Supplementary Material](#)). In particular, data provided at the global scale according to the type of seafloor substrate and to the type of biomes for macrophytobenthos are strongly lacking. For this reason, the beneficial effects of constructions are not fully addressed at present: the impacts of biomass destruction and of time of restoration for the ecosystem can be accounted for, but the level of production for macrophytes on rocky and soft bottom cannot be distinguished with the current state of knowledge.

The effects of artificial reefs on benthic biomass and production have been widely debated during the past decades (Grossman et al. 1997). Pickering and Whitmarsh (1997) reviewed the “attraction versus production” debate. They remind that artificial reefs provide additional habitats which increase the environmental carrying capacity and thereby the abundance and biomass of reef biota. Simultaneously, they underlined that artificial reefs can also serve as purely aggregating devices, whereby the behavioral preferences of fish result in aggregation on and around artificial reefs without any increase in biomass. In their review, Grossman et al. (1997) were even more critical towards the consequences of artificial reefs, underlying potential deleterious effects and

Table 2 Characterization factors for seafloor destruction ($SI, \text{kg}_{\text{C}_{\text{eq}}}\text{m}^{-2}$) and transformation ($TI, \text{kg}_{\text{C}_{\text{eq}}}\text{m}^{-2}\text{year}^{-1}$) due to construction ($CF_{\text{LSF}_{\text{seafloor}}}$) and to fishing gear ($CF_{\text{LSF}_{\text{trawl}}}$) per biome, type of substrate on the seafloor, depth, and time horizon of the modeling). Value are expressed in kilograms of carbon equivalent ($\text{kg}_{\text{C}_{\text{eq}}}\text{m}^{-2}$)

Ecosystem type	$CF_{\text{LSF}_{\text{seafloor}}}$				$CF_{\text{LSF}_{\text{trawl}}}$	
	SI	$TI_{20\text{y}}$	$TI_{100\text{y}}$	$TI_{500\text{y}}$	SI	TI
Intertidal						
Polar, rocky habitat	0.322	1652	1652	1652	0.181	0.347
Polar, sedimentary habitats	0.218	1652	1652	1652	0.122	0.055
Temperate, rocky habitat	0.308	1.910	1.910	1.910	0.173	0.401
Temperate, sedimentary habitats	0.204	1.910	1.910	1.910	0.114	0.064
Tropical, rocky habitat	0.342	2.148	2.148	2.148	0.192	0.451
Tropical, sedimentary habitats	0.207	2.148	2.148	2.148	0.116	0.072
Depth <50–60 m						
Polar, rocky habitat	0.216	1652	1652	1652	0.121	0.347
Polar, muds	0.214	1652	1652	1652	0.120	0.111
Polar, sands	0.214	1652	1652	1652	0.120	0.055
Temperate, rocky habitat	0.202	1.910	1.910	1.910	0.113	0.401
Temperate, muds	0.200	1.910	1.910	1.910	0.112	0.128
Temperate, sands	0.200	1.910	1.910	1.910	0.112	0.064
Tropical, rocky habitat	0.205	2.148	2.148	2.148	0.115	0.451
Tropical, muds	0.202	2.148	2.148	2.148	0.113	0.144
Tropical, sands	0.202	2.148	2.148	2.148	0.113	0.072
Mangroves	0.205	14.648	64.648	314.648	0.115	1.017
Seagrasses (temperate)	0.164	10.988	46.348	223.148	0.092	0.851
Depth >60 m						
Polar, rocky habitat	0.004	0.000	0.000	0.000	0.002	0.000
Polar, muds	0.001	0.000	0.000	0.000	0.001	0.000
Polar, sands	0.001	0.000	0.000	0.000	0.001	0.000
Temperate, rocky habitat	0.004	0.000	0.000	0.000	0.002	0.000
Temperate, muds	0.001	0.000	0.000	0.000	0.001	0.000
Temperate, sands	0.001	0.000	0.000	0.000	0.001	0.000
Tropical, rocky habitat	0.004	0.000	0.000	0.000	0.002	0.000
Tropical, muds	0.001	0.000	0.000	0.000	0.001	0.000
Tropical, sands	0.002	0.000	0.000	0.000	0.001	0.000

mentioning that very few studies unambiguously demonstrate that artificial reefs enhance regional fish production. Since these reviews, the beneficial effects of artificial reefs on production have been quantified in many studies (Bortone et al. 2011).

The values retained for the standing biomass on the seafloor, and for the reduction of benthic biomass and production (56 and 21 % respectively) by bottom trawling, are extremely rough, and the resulting values of SI and TI (Table 2) must be improved when necessary. These values should be adjusted according to the actual fishing intensity, the gear type, the type of seafloor, and a better resolution of the bottom depth effect. A particular attention should be paid to deep water trawling, in particular when performed on hard bottom occupied by long-lived sessile organisms such as coral, gorgonians, etc. Although poorly known, the reduction of biomass and production is certainly high, and the restoration time likely to be

longer than in shallower waters. Because only photosynthetic production was considered in the estimation of the benthic production NPP_{benthic} , null values are considered for bottom depth >60 m. In the real world, the productivity of these areas depends mostly on the “rain” of particulate matter coming from upper layers. These particles result from feces and cadavers of organisms (mostly planktonic) leaving the photic area. Although they are partly degraded, they still contain a substantial part of organic matter that is used by heterotrophic organisms. Their production can be considered, in a sense, as a primary production (Frontier and Pichod-Viale 1991). Based on this consideration, a rough estimate of the heterotrophic primary production in benthic areas >60 m depth could be estimated by a fraction of the autotrophic primary production that occurs in the upper layers. The major limit of this approach would result from the horizontal currents that can generate a mismatch between the photic layer and the benthic

areas, a mismatch that is expected to increase according to bottom depth.

In this study, no interaction was considered to exist between the different interventions. They can lead to negative impacts on the environment which are not well understood yet (Pauly et al. 2005); therefore, it does not seem realistic to aim for a better goal in the current state of knowledge.

3.3 Links and differences with the surrounding impact categories

Even if sea use impact assessment is similar to the land use impact assessment, some methodological differences can be noticed. Firstly, it is recommended to neglect the time of transformation t_{destr} in land use impact assessment (Koellner et al. 2013). However, this time artificially occurs, when expressing impacts due to uptake or destruction of the stock of biomass (Fig. 3a–d). Secondly, the occupation phase also appears differently in these figures: only shading impacts can be assumed as an occupation, such as they are defined for land use impact assessment, because biomass uptake and destruction take place instantaneously, and recolonization occurs as soon as the occupation starts.

Furthermore, $\Delta fNPP_{\text{eq}}$ are not equal in sea use and land use, due to the effects on the stock and to changes in the production potential. In land use impact assessment (when considering variations in organic matter or in biodiversity for example), this type of difference does not appear. This is because the quantity of standing primary biomass present in seawater is not directly correlated with its production rate, hence the difference in the heights of volumes representing stock changes (light grey) and changes in NPP (dark grey) on both panels c and d of Fig. 3. Indeed phytoplankton can be grazed by other producers almost as rapidly as it has been produced (Longhurst 2007). Primary biomass at sea is often as much influenced by the dynamics existing in the upper levels of the food web than it is by the dynamics of primary production itself.

Indicators in LCIA usually express an amount of substance that causes impact. In contrast, $fNPP$ expresses the amount of a beneficial substance that is not present, causing a negative impact, as organic matter does in soil (Mila i Canals 2003; Mila i Canals et al. 2007b). Thus, the parallel with terrestrial land use is well established. Advantages and limits of this midpoint indicator are provided by Langlois et al. (2014a).

In addition to the relevance of this indicator (most of the oceans are experiencing severe overfishing and lack of biotic resources), one of the main reasons why it has been selected is that it could be a means to relate land use with sea use quite easily. Some pathways for land use already express impacts in NPP units (Núñez et al. 2013). This should be the next step in methodological development for this new impact category, in order to compare results of land use with sea use (particularly

relevant for comparisons between fish and meat, between biofuels from terrestrial crops and from seaweed for example). Another perspective is the integration of this sea use in the metric of current LCIA endpoint methods, and it should be addressed in future works.

The framework for the assessment of LSF damage potential has already been applied for the particular case of biomass removal due to fishing activities (Langlois et al. 2014b). Following Johnson et al. (2008), Langlois et al. (2014b) included a scarcity factor, depending on the scarcity of NPP within the considered ecosystem. This allowed for a regionalized assessment of the relative importance of the carbon the ecosystem is deprived of due to human activity, compared to the total value of free carbon available within the ecosystem. This approach including scarcity of NPP was also suggested by Weidema and Lindeijer (2001) and used by Michelsen (2007) for land use impact assessment. Halpern et al. (2008) and Libralato et al. (2008) also suggested it for fishing activities impact assessment, apart from LCA. The goal of this scarcity factor was to express the fact that for the same amount of biomass removed from the sea, if fishing occurs in an ecosystem where biomass is scarce, the corresponding impacts on the ecosystem would be worse than if biomass were fished in a fertile one. Since these studies were performed, UNEP-SETAC guidelines (Koellner et al. 2013) have been published, with recommendations to express absolute values for ecosystem services damage potential (ESDP) assessment, instead of relative values. Therefore, this scarcity factor should not be used for ESDP impact assessment. In contrast, this scarcity factor should be used when computing a characterization factor for biotic-resource depletion impact, as did Langlois et al. (2014b).

Many uncertainties and assumptions have been made at every stage of CF calculation: uncertainties on the trophic level by group of fish species, on transfer efficiency by type of ecosystem, on NPP measured by remote sensing, and on average values of benthic production and standing biomass (see the discussions on data accuracy in the [Electronic Supplementary Material](#)). Moreover, intrinsic deficiencies in accuracy are associated to the use of $fNPP_{\text{eq}}$ as indicator. For instance, the microbial primary production for benthos is not accounted for in NPP measurements, despite its importance within the benthos (MacIntyre et al. 1996).

LCA is still in its infancy and this is even truer for LCAs applied to fisheries and aquaculture. This is a first intent of quantifying sea use, taking into account the state of the art, and the method must be improved along with progress in this research field. Despite the above-mentioned uncertainties, the goal of this study, which is to propose a harmonized and consistent method of LCIA for comparisons between different activities and interventions, has been successfully achieved. Impacts can be expressed using the same metric (kg of organic matter in primary organic carbon equivalent).

This methodological progress is operational and can be directly used by LCA practitioners.

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