

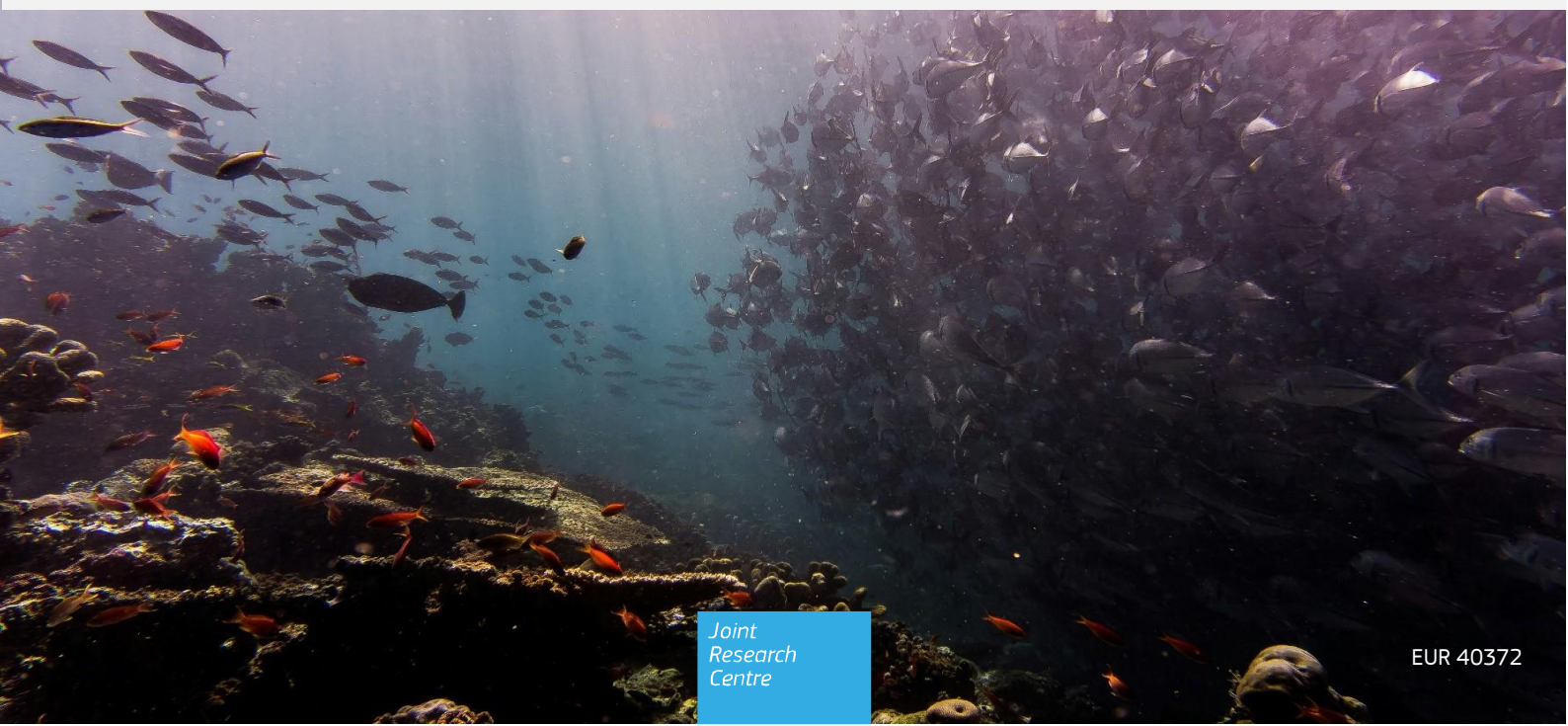


Exploring environmental impact methodologies to quantify fish stock exploitation and seabed impacts of fishing

Evaluating LCA and Non-LCA Approaches

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Abstract

This report evaluates methods from Life Cycle Assessment (LCA) and fisheries science concerning the impacts of fishing on exploited stocks and the seabed, using three case studies to assess feasibility and make recommendations for the European Footprint (EF) development. For fish stock exploitation, both fields offer suitable methods, but challenges remain in integrating them with the 16 EF indicators. The LCA method adopts an "Intrinsic biodiversity" approach, considering each kilogram removed as impactful. In contrast, the fisheries approach uses an "instrumental approach," viewing fishing as sustainable if stocks are not overfished, aligning removals with biomass renewal. The report highlights integration challenges and suggests consulting stakeholders to choose the best-suited approach for EF applications. Regarding seabed impacts, the LCA method lacks maturity and global data support for EF use, but this may improve. Meanwhile, the semi-quantitative approach from fisheries science is recommended for inclusion as a new characterization factor, having been successfully applied to the case studies and suitable for all EU seafood products. Lastly, a qualitative approach combining letter grades for stock depletion and seabed impact is discussed as a contingency, intended as a third and second option, respectively, for resource and seabed evaluations.

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1 Introduction

1.1 Background for the project

To contribute to improving the efficiency of resource use in Europe, Life Cycle Thinking methodologies, such as Life Cycle Assessment (LCA) have been developed to standardised approaches to reporting and communicating the environmental performance of products, services and companies. This standardised approach is intended to bring clarity to consumers by preventing misleading claims and reducing the diversity of various eco-labelling brands or systems.

According to ISO 14040, LCA is the process of compiling and assessing the inputs, outputs, and potential environmental impacts of a product system throughout its entire life cycle. To help LCA practitioners effectively implement LCA, additional guidelines have been developed in alignment with this standard. The European Commission has introduced the Environmental Footprint (EF) methods to improve the comparability of LCA when applied to products and organizations.

The Environmental Footprint (EF) for Product (PEF) and Organisation (OEF) methods have been developed since 2013 and are the main focus for addressing this environmental performance labelling issue within the European Union (EU). The EF methods include the LCA steps: 1) **the goal and scope** phase, the aims of the study are defined, namely the intended application, the reasons for carrying out the study and the intended audience. Main methodological choices are made in this step, in particular the exact definition of the functional unit, the identification of the system boundaries, the identification of the allocation procedures, the studied impact categories and the Life Cycle Impact Assessment (LCIA) models used, and the identification of data quality requirements; 2) **the Life Cycle Inventory** (LCI) phase involves the data collection and the calculation procedure for the quantification of inputs and outputs of the studied system. Inputs and outputs concern energy, raw material and other physical inputs, products and co-products and waste, emissions to air/water/soil, and other environmental aspects. Data collected concern foreground processes (e.g. for a consumer good manufactures, the manufacturing and packaging of a product) and background processes (e.g. for a consumer good manufactures, the production of purchased electricity and materials). Data are validated and put in relationship to the process units and functional unit; 3) the **Life Cycle Impact Assessment** (LCIA) phase, where LCI results are associated to environmental impact categories and indicators. This is done through LCIA methods which firstly classify emissions into impact categories and secondly characterise them to common units so as to allow comparison; 4) the **Life Cycle Interpretation phase**, where results from LCI and LCIA are interpreted in accordance to the stated goal and scope. This step includes completeness, sensitivity, and consistency checks.

DG Environment and the Joint Research Centre (JRC) of the European Commission (EC) are currently leading further development and implementation of this approach, in cooperation with other EC services, LCA experts and stakeholders.

One such product area is the harvest of wild seafood products. Fishing and harvesting of wild products deviate from typical food production, in that it relies on the variable natural production of different stocks, outside the influence of husbandry and agricultural practices. Furthermore, the activity of fishing and harvesting of marine products involves exerting pressures on wild habitats, such as those on the seabed, across shared territories and different jurisdictions.

The current EF methods contain 16 environmental impact categories upon which the value chain of different products can contribute. None of these existing impact categories reflect the impact of wild seafood harvesting. There is an imperative to include such impacts in eco-labels and other

environmental claims for fishery activities and relevant products, as also addressed in the proposed Directive on substantiation and communication of explicit environmental claims (proposed Directive: COM/2023/166 final).

To further develop the EF methods, the EC sought advice on how best to include fisheries environmental impacts into the existing EF workflow. Specifically, a panel of **four LCA and fisheries experts** - indicated in this report as the “experts”, and corresponding to the four leading authors - were contracted to review current best practices for evaluating the impact of marine biological resource exploitation (fisheries removals) and impacts on the seabed derived from marine harvesting activities (fishing) based on LCA and non-LCA approaches.

This report addresses the incorporation of wild marine fisheries into the EF methodology.

1.2 Summary of first report

This work builds from a previous activity detailed in Annex 1 where the four leading authors identified methods relevant to assessing the impact of fishing on the marine environment, categorised by two dimensions. The first dimension addressed the mechanism of impact, or in LCA parlance the potential impact pathways, which were stipulated by the JRC as being the extraction/direct exploitation of natural wild resources and the impact of harvesting activities on the seabed. The second dimension divided methods by LCA and non-LCA approaches (Table 1).

Table 1. Four dimensions of inquiry, covering the different disciplinary approaches and impact indicators.

	Impact of depletion of targeted biotic resources	Impact upon seabed from fishing activities
LCA approaches	LCA methods for quantifying the depletion of targeted biotic resources	LCA methods for quantifying seabed impacts
Non-LCA approaches (originating from fisheries science)	Non-LCA methods for quantifying the depletion of targeted biotic resources	Non-LCA methods for quantifying seabed impacts

Source: Authors’ own elaboration.

Within these four categories, across these two dimensions, all identified methods were evaluated against common criteria and the most appropriate methods were proposed. Due to the disparate approaches and sometimes philosophically incommensurable approaches used across the LCA and non-LCA realms, it was not possible to select and recommend a unique approach that can be considered absolutely better than the others. Therefore, four final methods, one from each category across the two dimensions, were put forward as the most appropriate for inclusion in environmental labelling of seafood products.

Regarding the **depletion of targeted biotic resources (extraction / exploitation of wild marine stocks), the LCA method recommended** was based on that described by Hélias et al. (2023) and started from the point that any and all extractions are an impact on an otherwise unimpacted state. Conversely, the non-LCA method proposed was based on STECF (2023) grading system, which starts from the point that wild stocks are a renewable resource that can, in principle, be sustainably harvested, indefinitely, with stable detrimental environmental impacts (not changing

over time). Aside from these fundamental differences in definition, there are other strengths and weaknesses of each approach both in the form of varying assumptions and in the ability to operationalise the methods for all seafood products in the EU, which are summarised in the table below (Table 2).

Table 2. Summary of characteristics of methods originating from LCA and fisheries science regarding the impact of depletion of targeted biotic resources. Highlighted green and red are considered the key features of discussion

Originating from LCA	Originating from Fisheries Science
<p>Helias et al. (2023)</p> <ul style="list-style-type: none"> • “Intrinsic biodiversity” approach: the first kg removed from the sea has an impact • Completeness • Peer-reviewed publication • Temporal average over 3 years (2014-2016) • Quantified uncertainty of CFs is provided • Full compatible with EF • GLAM recommended unit (PDF.yr/kg fish) • Not well aligned with worldwide fisheries management policies • Default values are based on CMSY computations (controversial in fishery science) and are given for “meta-stocks” • It is recommended to compute values for stocks if data available • Acceptability: concept of reference state as being the “pristine state” is unusual for fishery scientists • Impact pathway: missing link in species interactions 	<p>Normalised STECR (2023 – updated in STECR, 2024)</p> <ul style="list-style-type: none"> • “Instrumental approach”: as long as the stock is not overfished, fishing removals are aligned with biomass renewal, and the impact does not increase over time • Fully aligned with worldwide standards of fisheries management and SDG 14 • Impact per kilo is not influenced by the size of the population but by its productivity / management • Applicable to any seafood product adhering to EU’s CMO regulation (EU reg. 1379/2013) • Not aligned directly with LCA conceptualisation (bell-shaped impact, “intrinsically additivity” through management dynamic) • Adaptation was needed to convert letter-grading to normalised impact per kilo – a new proposal for LCA-compatible normalisation on impact contribution to worldwide overfishing was explored • Operational as it is but would need fine tuning peer’s validation

Source: Authors’ own elaboration.

Regarding the **impacts of harvesting on the seabed**, the LCA method recommended was based on Woods & Verones (2019), which considers both the disturbance impact of fishing activities, but also the ecological resilience (in the form of recovery times) of the habitats being impacted. For the non-LCA method a comprehensive and systematic literature review found no suitable methods for quantifying seabed impact by kilogram of seafood product, rather the entirety of the academic literature explored on this topic in the first report considers seabed impact in the form of per-unit-area impact by all fishing activities, thus not directly translatable to a fishery product sold on the market. The recommended method in this case was again based on a STECF (2023) grading system, specifically focussed on seabed impacts. This method utilises details of which fishing gears are used in harvesting as well as species associations with primary habitat types to provide a generally applicable work-flow capable of operating with diverse seafood products, so long as they comply with existing EU reporting requirements. The main comparison between these two methods for seabed impact involves the trade-off of more ecologically specific impact measures (Woods & Verones, 2019) against product coverage (STECF, 2023) (Table 3).

Table 3. Summary of characteristics of methods originating from LCA and fisheries science regarding the impact of fishing on the seabed.

Originating from LCA	Originating from Fisheries Science
<p>Woods and Verones (2019)</p> <ul style="list-style-type: none"> • Designed to be globally applicable • Accounts for single and repeated impact • CFs consider anthropogenic disturbance, benthic response, hydrodynamic energy and recovery period • Peer-reviewed publication • EF compatible • GLAM recommended unit (PDF.yr) • Model limited to Europe benthic habitats • CF are not fully accessible • Not fully applicable (lack of data and GIS software) • Normalisation factors are not mentioned • Several input parameters are needed to develop new CFs • GIS software is requisite for new CF development 	<p>Normalised STECR (2023)</p> <ul style="list-style-type: none"> • Applicable to any seafood product adhering to EU's CMO regulation (EU reg. 1379/2013) • Generalised based on both gear type interactions with seabed and species associations with benthic habitats • Well documented, transparent and accessible logic for stakeholders' acceptance • Two tier reporting standards encourage more detailed reporting for more accurate scoring • Logic and underlying knowledge assembled and reviewed by STECF • Not inherently quantitative • CF calculations are based on a priori assumptions of risk acceptance applied to categorical scores • Does not account for repeated disturbance • Does not account for community specific sensitivities/vulnerabilities • Operational as it is, but derived CF would need fine tuning and peers' validation

Source: Authors' own elaboration.

In addition to this primary trade-off, there are other strengths and weaknesses of each method, which are detailed in Annex 1 and in the tables above, while Annex 3 include the summary of this work.

1.3 Aims of this report

This report focuses on the implementation of shortlisted methods, shown in Table 2 and Table 3. Using a series of three diverse case studies, the current report illustrates how each method can and cannot directly address different fishery products and provides a comparison of how these methods could contribute to new EF impact categories, or alternately, how they might provide additional information complementary to EF impact categories.

2 General approaches and shared case studies employed across methods

In order to evaluate the applicability of the selected methods (*c.f.* the scientific robustness), this report applies each method in three different case studies (sections 3 to 6). The intention with this case-based approach is to document the process of working through each method to highlight key limitations, assumptions, data requirements and outputs of each method. The cases all consider the production of raw, landed weight fishery products. The questions of combining these suggested impact categories with the other 16 EF impact categories, as well as how to propagate the information in complex supply chains (e.g., mixed seafood products processed with other ingredients) are beyond what could be achieved in the time dedicated for this project, as it would require deeper investigations and discussions. However, some initial elements are presented along those lines in section 8.

Discussion on whether to combine the two fishing impact categories in focus in this report (biotic resource depletion and seabed disturbance) into one score for the purpose of integrating with other impact pathways or categories was discussed by the authors. The outcome of this discussion was the recommendation to keep these two impact categories independent, as they represent very different interactions of fisheries with the environment.

The three cases selected include two different fisheries exploiting one data-rich European stock, and one example of an imported (external to the EU) product with more limited data availability (Figure 1).

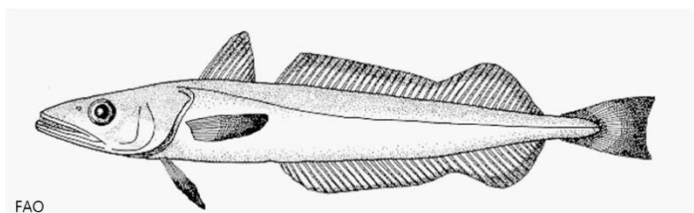
The first case is European hake (*Merluccius merluccius*) from the Bay of Biscay, fished by the Spanish demersal trawl fleet. This case exemplifies a typical data-rich ground-fish fishery where the fishing gear interacts with the seabed.

The second case is based on the same species and area, except the products are harvested by the Spanish mid-water trawl fishery, whose fishing gears are much less likely to interact with the seabed.

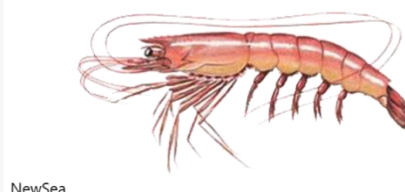
The third and final case is the Argentine red shrimp (*Pleoticus muelleri*), fished by an inshore beam trawl fishery in Argentina. This case provides an example of a less well documented fishery, outside of ICES advice and outside of European habitat databases.

Figure 1. Illustration of the three case studies.

European Hake in the Bay of Biscay



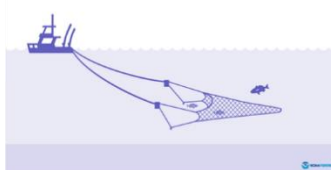
Argentine Red Shrimp



Spanish Demersal Trawl



Spanish Mid-water Trawl



Argentinian In-shore Beam Trawl



Source: Authors' adaptation from FAO and NewSea.

2.1 Target output

Regarding the potential inclusion of methods shortlisted in the life cycle impact assessment (LCIA) phase of EF methods, the experts strived to provide a normalised LCIA indicator for the respective impact category (fishery resource depletion, or seabed disturbance from fishing). They did not discuss the weighting procedure, and thus the report does not aim to provide any examples of single scores. While the normalisation procedure uses data to make individual product impact measures relative to the total global impact for the same impact category, the weighting procedure involves making value decisions on how different impact pathways/categories relate to one another. This value-based approach is beyond the capability of the small expert group and requires a political process.

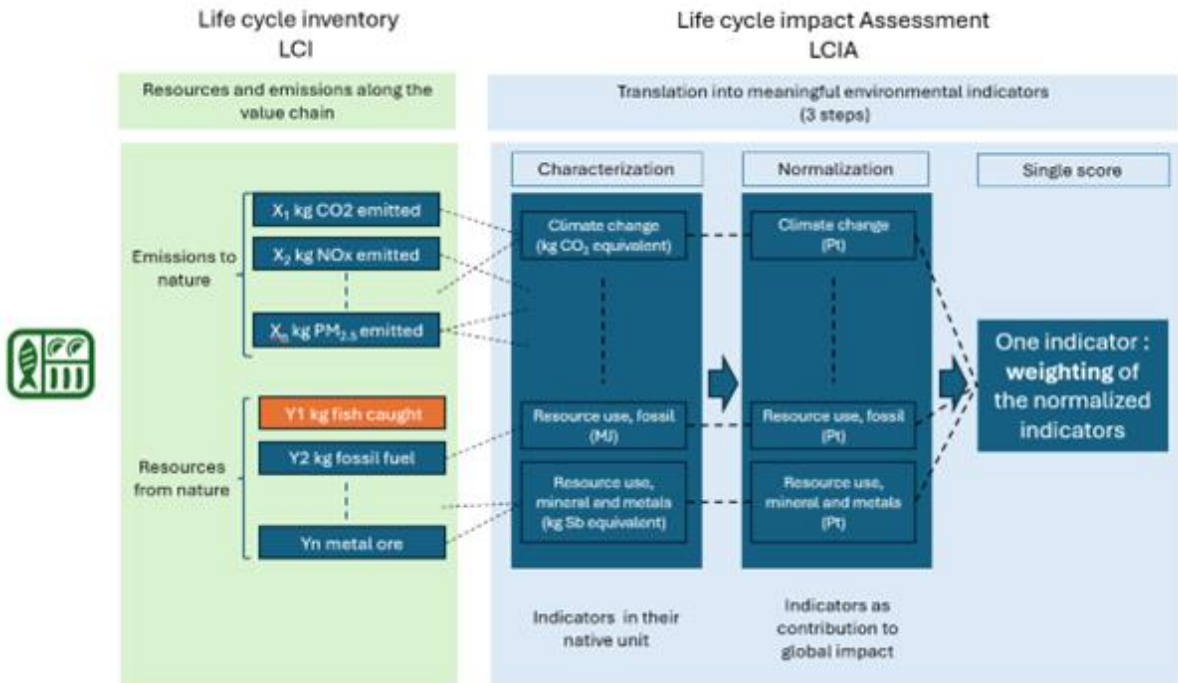
Additionally, this report aims to document the life-cycle inventory requirements, and the detailed method used to characterise the respective indicators (Figure 2). In doing so, this report highlights data needs and gaps, as well as methodological assumptions and limitations to operationalisation.

It is interesting to highlight that at the start of the project, the dichotomy between the simplified wordings LCA vs. non-LCA methods was meaningful and simple to refer to, with LCA methods implying that they are specifically designed to be LCA-compatible in the EF framework, and non-LCA encompassing any method coming from other fields of science, including fisheries and marine ecology science. However, a key challenge discussed in the first report was indeed whether and how non-LCA methods could possibly become LCA-compatible and better aligned with the EF framework. This second report has progressed significantly on that particular issue, proposing paths towards converting and/or integrating the STECF indicators into LCA-compatible values. As such, the convenient wording dichotomy between LCA and non-LCA methods becomes progressively blurred and less discriminating, and the choice of wording might need to be reconsidered in the future.

Finally, the non-LCA expert group reflected upon the fact that more time might be required before decisions are made to include two new indicators in the EF framework, or that none of the proposed impact pathways/categories may be deemed as broadly acceptable by diverse stakeholders. Thus, they discussed alternative ways to integrate and convey the non-LCA information onto fishery products – e.g., via a semi-quantitative pathway based on the combined STECF grades. The outcome

of this method is a single letter grade combining evaluations of stock exploitation and seabed impact, with guidelines for how such a grade should propagate through value chains to consumers. This is presented in section 7.

Figure 2. Overall scheme of the assessment of a food product within the framework of PEF.



It starts with the life cycle inventory that assesses all emissions to nature and depletion of natural resources to produce the product over its whole value chain. The life cycle impact assessment step translates those thousands of inputs into meaningful environmental indicators (characterisation), through the aggregation of substances (i.e., emissions) or raw materials (i.e., resources) that contribute to the same environmental dimension, represented through impact categories. Each indicator is then translated as a relative contribution to the global impact (normalisation), expressed in “points” (Pt)¹. Indicators are then weighted to display a single score.

Source: Authors’ own elaboration.

2.2 Update frequency

The dynamic nature of wild fisheries resource status and fishing activities means that fisheries indicators change over time; and in particular regarding stock status (depletion of biotic resources) which is evaluated every year by stock assessment scientific organisations. In parallel, the data underpinning the calculation of normalisation factors are also of a dynamic nature. This time-variant property of these impact pathways is a major paradigm shift for LCA practitioners, although among the existing 16 EF indicators, a few are already time-varying to some extent (e.g. water scarcity). There are common procedures from the fisheries management field that can be adopted to ensure that these impact pathways remain relevant and accurate over time. typically, the process of monitoring stock status combines regular (annual, biennial) assessments of fisheries resources, applying an agreed methodology to the most recent available data, together with periodic evaluations and updates of the assessment procedure itself (termed a “benchmark” in ICES parlance) every approximately five years, or as deemed necessary by experts. This approach allows regular updates to the perception of stock status with minimal review and auditing, while allowing

¹ A point is equal to the division i) of the total impact over one year due to all emissions and resources from all sources worldwide, ii) by the global population for that same year. It therefore represents the “global average impact per individual over one year”. For EF, computations are made with global 2010 data (for emissions, resources, and population).

for adaptation and improvement to changes in data availability, methods and stock dynamics, under more rigorous review, but less frequently.

As such, the issue of update frequency is central in this question, and many options can be considered. As an example, the MSFD (Marine Strategy Framework Directive) proceeds with an update of the environmental status of EU waters every 6 years. On this basis, the latest implementation of the stock status indicator by STECF Expert Working Group EWG 23-18 (STECF, 2024) suggested that while the database shall and can be updated every year with new stock assessment information, the score itself could be revised at slower frequency. The EWG proposed a six-year average for the grade (average 2018-2023); but faster update of this system could be needed if the stock status fluctuates more, and could also be considered at reduced additional costs, since a large part of the system already includes automatised updates procedures. The update frequency decision should also consider other food labelling processes. In the LCA realm, (Stanford-Clark, Loiseau et al. 2024) recommend an update of the impact factors every five years.

3 On the LCA approach to quantifying the impact of depletion of targeted biotic resources (stock exploitation) – based on Hélias et al. (2023)

3.1 Description of method

This report is primarily interested in the normalized impact results.

3.1.1 Characterization method

After the evaluation of some methods described in Annex 1, the method described by Hélias et al. (2023) was selected to be further analysed. It measures the impact on species, an “ecosystem quality”-like approach. This method is not a measure of overfishing, as they consider an impact from “underfished” species as well. It is rather a measure of potential impact on ecosystem quality (Hélias et al., 2023). It presents an intrinsic approach to stocks: any fish removal from nature has an impact. It is compatible with LCA and EF. However, a normalization has not been performed, but a discussion with authors suggest that it is easily feasible based on FAO catch data. Hélias et al. (2023) compute characterization factors at two levels:

CF reg, at level of (eco)regions, where CFs are expressed either in species*year (number of species potentially disappeared over a period of one year), or in PDFreg*year (Potential Disappeared Fraction of species at regional level over a period of one year).

CFglo, at global level where CFs are expressed in PDFglo*year (Potential Disappeared Fraction of species at global level over a period of one year).

CFs are provided for target fish species. The relation between CFreg and CFglo is computed using Verones et al. (2017).

Numeric values for CFs are publicly available in the Supplementary Information, based on input data that is a combination of species or species group (depending on data availability) and on FAO major area. There are also default values for very broad categories (e.g., “marine species, nei”). The method displays a high number of characterization factors, based on information of species or species group on the one hand (n~1800), and FAO marine area (n=18) on the other (i.e., ~5000 CFs). CFs are applied on a mass of a given species fished in a given FAO area and are computed for year 2018, based on 3-year time series (2017-2019).

The databases from which characterization factors are constructed include well-recognized international institutions (i.e., FAO and FishBase). However, the CMSY algorithm used to compute missing data are controversial within fishing management scientists (ADEME, 2023).

There is no stock assessment per se, but rather an assessment of a given species status in an FAO area. Computing CFs on specific stock data is however feasible and has been done in Hélias et al. (2023).

Uncertainty is reported quantitatively for each characterization factor for confidence interval bounds, based on biomass uncertainty.

3.1.2 Recent methods update

In April 2024, an update on the factors, with a similar method to (Hélias, Stanford-Clark et al. 2023) (2015 fish data), was published by (Stanford-Clark, Loiseau et al. 2024) (2018 fish data).

They propose three ways of converting CFs from regional to global scale. When discussing the method in this report, in this report, we keep using the original way (called Relative Endemicity, see section 3.1.1) as recommended by the authors.

They propose new CFs accounting for discards. The rationale behind discards' contribution to the impact is that, while discards contribute to catches, since they are not landed, there is no direct estimation of their impact. Thus, discards in the different FAO Major Fishing Areas are allocated to landings in order to be accounted for in the total assessment of impact of Biotic Resource Depletion.

Contrary to the former publication, supplementary information provides underlying data contributing to CF computation: Catches (C), Biomass (B), intrinsic growth rate (r), and carrying capacity (K). This enables us to understand why results are obtained whereas without this breakdown of each value, we cannot infer what explains higher or lower impacts. Moreover, provision of biomass estimation for all stocks of all species enables computation of stock-specific global CFs which are calculated by multiplying regional CFs with the ratio of stock to total species biomass.

The authors recommend computation of stock-specific CFs based on C, B, r and K estimates from assessment bodies (Regional Fisheries Management Organizations or state bodies). It has been computed for several stocks of Northeast Atlantic, managed by ICES, in (Wermeille, Gaillet, et al., 2024). Such computation is straightforward for regional CFs thanks to method transparency, but global CFs require the additional data of total biomass of the species which is not available to our knowledge except in CMSY computation by (Stanford-Clark et al., 2024).

Note that these updated factors were not accounted for in the following results (section 3.3) but methodological updates and precisions are discussed in section 3.4.

3.2 Application of method - Normalisation

Normalization method was discussed with Arnaud Hélias (Arnaud Helias personal communication 2024) and established as consistent with EF 3.1 method for normalization (Sala 2017; Andreasi Bassi, Biganzoli et al. 2023). This method was applied to Hélias CFs (Hélias, Stanford-Clark et al. 2023). Between the 4 possible CFs, we chose CF expressed in PDF.year (Potentially Disappeared Fraction of species during a year) at global level.

In Hélias et al., 2023, all characterization factors are provided, along with total catches for metastocks (species in FAO area). Some CFs and catches are provided for dates prior to 2015, referring to superseded stocks (they were either not fished anymore later or their definition had changed). For normalization, only metastocks with data in 2015 were selected. Multiplying individual CFs by catches enables calculation of total impact for each metastock, the sum of which gives the total impact of fisheries worldwide. To obtain the impact per capita, it is divided by 2011 world population (6,895,889,018 habitants), following EF 3.1 normalisation procedure (Sala et al. 2017). This is indicated in equation 1.

Equation 1. Total impact per capita

$$Total\ impact\ per\ capita = \frac{\sum_{stocks} (Catches_{stock} \times CF_{stock})}{World\ population}$$

The resulting normalization factor is $1 / (2.31E-10) = 4.33E+09$ (PDF_{glo.}.year/(capita.year))⁻¹.

3.3 Experience and performance with case studies

Case studies have been developed for products live weight at landing. In line with what is expected to be achieved in the inclusion of this indicator into EF, for these case studies, standard published CFs were used. They apply to “metastocks”, that are aggregation of stocks of a given species in its FAO area. Computation of stock-specific CFs, while possible when data is available, is time-demanding. The exercise has been performed for hake stocks (see section 3.4.) but is not expected to be feasible at the scale of European consumption. For example, no data exist for Argentine red shrimp stocks.

3.3.1 Hake from the Bay of Biscay caught by Spanish fleets, fished with demersal trawl

3.3.1.1 Life cycle inventories

For biotic resource depletion of the European hake, *Merluccius merluccius*, from Atlantic, Northeast, the corresponding characterisation factor from (Hélias, Stanford-Clark et al. 2023) was utilised before and after normalisation computations as described above.

3.3.1.2 Results and discussion

Results for 1 kg live weight hake, at landing, are $1.74E-14$ PDF_{glo.}year, corresponding to $7.54E-5$ Pt after normalization (Figure 3).

3.3.2 Hake from Bay of Biscay caught by Spanish fleets, fished with paired mid-water trawl

Same as above, as fishing 1kg of hake has the same impact on biotic resource depletion, regardless of fishing gear / technique.

3.3.3 Argentine red shrimp from the inshore Argentine fishery

3.3.3.1 Life cycle inventory

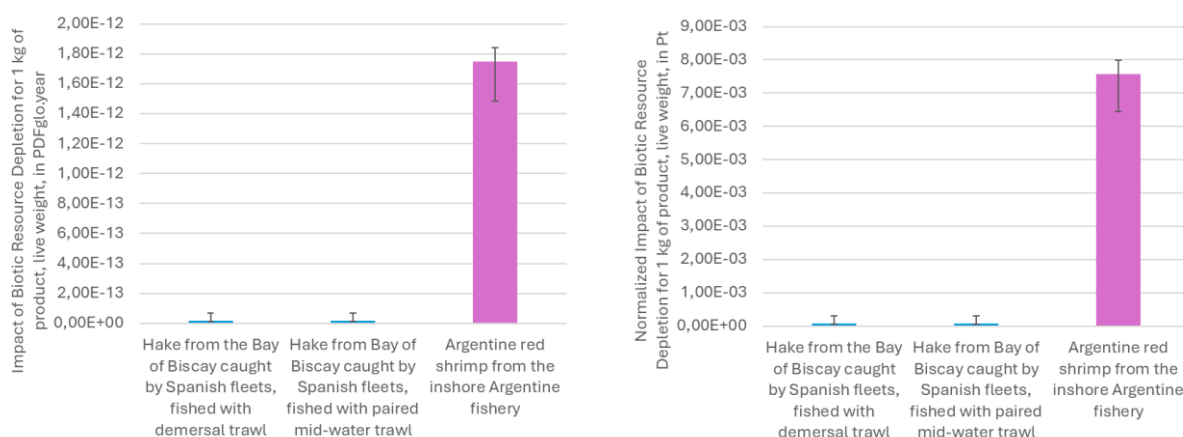
For biotic resource depletion of the Argentine red shrimp, *Pleoticus muelleri*, from the Atlantic, Southwest, the corresponding characterisation factor from (Hélias, Stanford-Clark et al. 2023) was utilised before and after normalisation computations as described above.

3.3.3.2 Results and discussion

Results for 1 kg liveweight shrimps are $1.75E-12$ PDF_{glo.}year, corresponding to $7.57E-3$ Pt after normalization (Figure 3).

For shrimp, the impact of biotic resource depletion is higher than hake by two orders magnitude. This difference is linked to the combined values of input parameters, which unfortunately, are not available in (Hélias, Stanford-Clark et al. 2023).

Figure 3. Impact of Biotic Resource Depletion related to production of 1 kg of each case study.



On the right, characterized impacts in PDF_{glo}.year, and on the left, normalized impacts in Pt. Error bars represent upper and lower limits of 95% confidence intervals, they result from CF computations for which confidence interval was provided by Hélias et al., 2023.

Source: Authors' own elaboration.

3.3.4 Overall performance summary

3.3.4.1 Characterization of fishing stocks

The method of Helias et al 2023 is recent, and has been published in a peer review publication, and characterisation factors are available for a total of 5070 pairs (species x fishing areas). It also displays default values when a specific area is not available and or when species is unknown. The publication proposes to prioritize completeness over precision (i.e. specific identification of stocks with available RFMO data) and hence provides a thorough list of CFs for 2065 species for 18 FAO major fishing areas based on the CMSY algorithm.

Helias also displays uncertainty on the factors, which can also be useful when comparing products and define if one is significantly better than the other. Assessment values and quality of that value are assessed independently in that method.

Results are very contrasted between case studies, for the two metastocks assessed (hake from Bay of Biscay vs Shrimp from Argentina); they display two orders of magnitude difference.

It is worth noting that, for stocks which have precise C and B data from RFMO, Hélias's formula is straightforward and can applied and compared to CMSY results. This application is displayed for hake stocks in Figure 4 and discussed in section 3.4; uncertainty regarding the assessment with precise C and B data is substantially reduced.

3.3.4.2 Normalisation

Normalisation is feasible based on catch data, and has been implemented for the case studies, in full compliance with other EF 3.1 impact categories.

Normalised results display contrasted results in biotic resource depletion impact pathway between the two examples with a difference by two orders of magnitude, unchanged from characterisation results.

It is probable that adding this impact category will significantly change the picture for the most impactful fished products. Indeed, most global impacts of fishing are related to a single economic sector, (marine) food, be it i) directly or ii) indirectly as feed for aquaculture; other economic sectors

use very low fishing resources. As the impacts between products vary by several order of magnitudes, weighting, that influence the contribution of a single impact category in the single score by one order of magnitude (10% weighting is reducing impact contribution by one order of magnitude), will not dwarf the most impactful products.

3.3.4.3 Implementation in PEF computations in LCA software

The examples above have demonstrated the extreme simplicity in implementing the computations in EF. This has also been implemented in our LCA software (SIMAPRO) in the following manner:

- Create elementary flows for these “resources from nature” in the software, for each (species; area) pair characterised in Helias 2023. The elementary flow nomenclature has to be agreed upon but would necessarily contain information on the species name (latin name) and the area. In terms of species naming, and to avoid ambiguity, we highly recommend using latin names (as opposed to English ones).
- Create an LCIA method based on a copy of EF3.1, edited by adding an impact category called “Biotic resource depletion”. The CF values published by Helias 2023 were connected with the elementary flows created above and included for the characterisation model; normalised values computed in line with section 3.2 were added for the normalisation model. The updated method has 17 indicators, instead of 16 for the native EF. It is straightforward to include an updated weighting of those 17 categories, however, it has not been performed within this contract. This has to be done eventually but is beyond the scope of this report.

For the inventory datasets related to fishing activities, an elementary flow has to be added in each of them to trace the removal of the fish from the sea.

Once the updates have been done, computations can be automatically run in the software including the new added impact category, and software will automatically display results at characterisation, normalisation, and (eventually) single score levels. This can be done for fish at landing, but also integrated in downstream value chains, accounting for downstream processes such as processing (energy, water, live to net weight ratio), packaging, distribution, and consumer preparation.

Future method updates on “biotic resource depletion” would be addressed by updating the method, in an integrated fashion together with (potentially) other impact categories of the EF method.

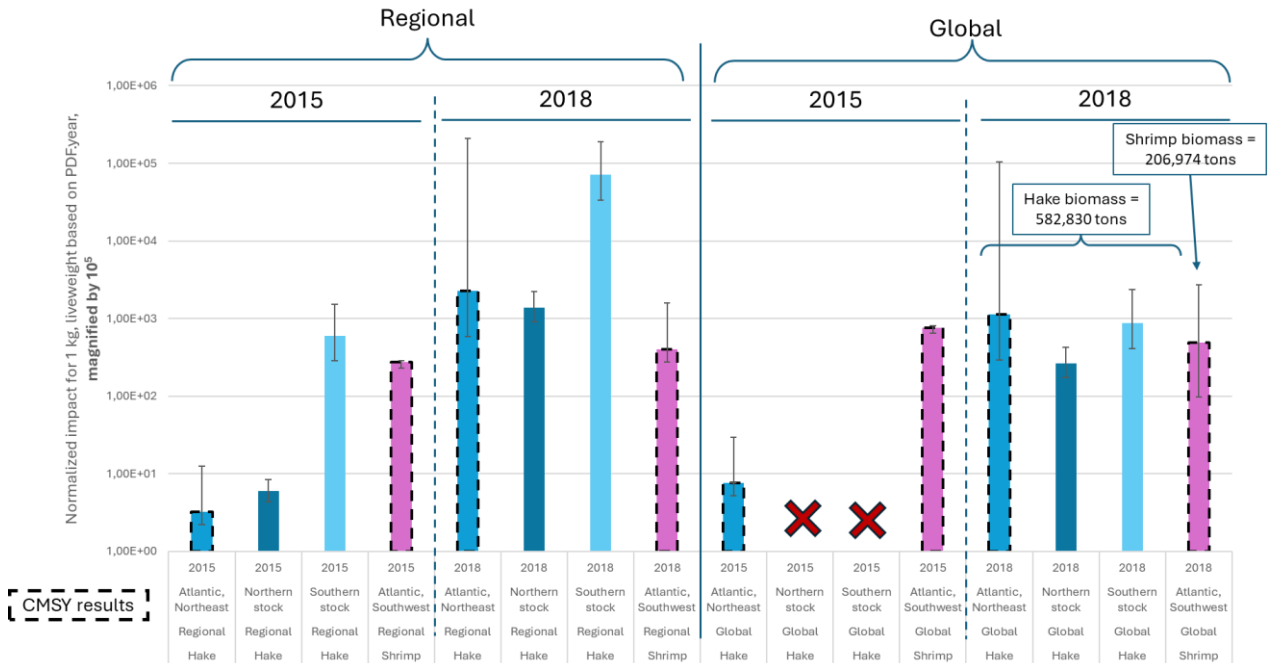
3.4 Fit for purpose?

Different characterization factors can be obtained based on unit (species.year or PDF.year at regional or global level); geographical scale (Metastock: a species in an FAO Major Fishing Area; stock specific); and years of computation (2015; 2018). Figure 4 displays the different normalized CFs that can be obtained in PDF.year (regional or global) for European hake, *Merluccius merluccius*, fished in Bay of Biscay and Argentine red shrimp, *Pleoticus muelleri*, fished in Argentine waters.

For stock-specific CFs, data from the International Council for the Exploration of the Sea was retrieved in order to apply the method as presented in (Hélias, Stanford-Clark et al. 2023). Data correspond to the latest advice on both hake stocks that are studied in Northeast Atlantic, and which overlap in the Bay of Biscay, the Northern hake stock (hke.27.3a46-8abd) and the Southern hake stock (hke.27.8c.9a). Following (Hélias, Stanford-Clark et al. 2023), CFs were calculated based on biological reference points (B_{MSY} and F_{MSY} respectively converted to carrying capacity $K=B_{MSY} * \exp(1)$ and intrinsic growth rate $r = 2 * F_{MSY}$), and time series of biomass and catches, averaged over 3 years: 2014-2016 and 2017-2019. Normalization is calculated based on total impact of global

fisheries for each time period which resulting in a factor of $4.33 \text{ E9 Pt/PDF}_{\text{glo.}}\text{year}$ in 2018 and $1.32 \text{ E11 Pt/PDF}_{\text{glo.}}\text{year}$ in 2015. Error bars result from i) for stock-specific CFs, from 95% confidence interval on biomass estimate by ICES; ii) for standard CFs, from 95% confidence interval on CF calculation by CMSY algorithm.

Figure 4. Comparison of normalised impacts of 1kg liveweight fish.



Modelling based on (Hélias, Stanford-Clark et al. 2023) for bars labelled as 2015 data and Stanford-Clark, Loiseau et al. 2024) for bars labelled as 2018 data for case study s' hake (blue bars) and red-shrimps (pink bars). Bars with dotted lines refer to computations with CMSY algorithm for metastocks within the FAO area based on data from Helias et al. 2023 for 2015 data and Stanford-Clark et al. 2024 for 2018 data. Dark blue bar (■) is for computations done with RFMO data for Northern hake stock (hke.27.3a46-8abd) and light blue bar (■) is for computations done with RFMO data for southern hake stock (hke.27.8c9a). Bars on the left half (labelled "Regional") refer to normalisation based on data in PDFreg.yr. Bars on the right half (labelled "Global") to normalisation based on data in PDFglo.yr. 95% confidence interval is displayed with vertical lines on each bar. Results are presented on a log scale and magnified by a factor 10⁵. Cross (X) indicates values that could not be computed ('total species biomass' values are not available in publication)

Source: Authors' own elaboration.

3.4.1 A compelling need for periodic update of data inputs

Authors of the latest publication (Stanford-Clark, Loiseau et al., 2024) highlight the need for periodical update of the factors, "particularly relevant for wild, biotic resources with reactive system characteristics, such as fisheries, as the balance of internal non-linear responses to external pressures, including exploitation, can cause rapid changes in the degree of impact, through extreme depletion or (positive or negative) changes in stock management". They suggest a five-year period.

For now, only assessments for 2015 and 2018 are available, and have been used. In Figure 4 we compare results computed with (Hélias, Stanford-Clark et al. 2023) based on 2015 data, and (Stanford-Clark, Loiseau et al. 2024) based on 2018 data. We take the numerical values available in the publication, which are based on CMSY computations. Looking into the results, one can see that for hake, characterization factor increases by two to three orders of magnitude between 2015 and 2018, highlighting the need for CF updates in time, to account for the evolution of the living resource status.

3.4.2 Uncertainty – stock assessment

Figure 4 also displays computation specific to the two hake stocks (North and South) present in the Bay of Biscay, with data retrieved from ICES database (Murawski, et al. 2024). The Northern stock is approximately ten times more populous than the Southern one. Hence, the impact of the former is generally closer to the impact computed for the metastock. Uncertainty is substantially reduced for computations with stock specific data, that are in general within range of uncertainty provided for metastock. Therefore, as discussed in (Wermeille, Gaillet et al. 2024) and (Stanford-Clark, Loiseau et al. 2024), it is recommended, when known, to favor computations for specific stocks rather than metastocks (computed CMSY data). However, the specific stock related to a given food product is not always known, and data may not exist as for the Argentine red shrimp which is not assessed by assessment bodies, and metastock data provide a fall-back option.

3.4.3 Regional or global metric?

In Figure 4, one can also compare results between regional (unit PDFreg.yr) and global (unit PDFglo.yr) metrics. Global metrics enable us to account for relative endemicity of species in the metastock. Red Shrimps are only present in the 41 FAO area. Therefore, their “endemicity” is full, and results do not show any difference between regional and global impacts. Conversely, hake is not endemic to the 27 FAO area but is also present in two others (FAO 37 and FAO 34); therefore, we see lower impacts at global than regional level. Computation from regional to global requires the global biomass of species (so called “global species biomass”). It is not available for 2015 data as disclosed in Hélias publication; therefore, for specific stock data, we were not able to compute for the global metrics (see red crosses in Figure 4)². For 2018 data, the “global species biomass” was available in the publication, and accounting for global species biomass was therefore possible. For 2018 data, Figure 4 displays more homogeneous results for stock-related impacts: northern and southern stocks differences in impact are substantially reduced, and of a similar order of magnitude to the metastock impact.

Arnaud Hélias recommends (pers. com³) to use the global metric (rather than regional) as they account for endemicity and therefore are homogeneous to a risk of the species disappearing globally. However, the assumption of global metric appears counterintuitive from a fisheries science point of view, as if a population is depleted in one area, it will not be repopulated again by fish from different populations from the same species. But as discussed above, it is also possible to account for regional metric.

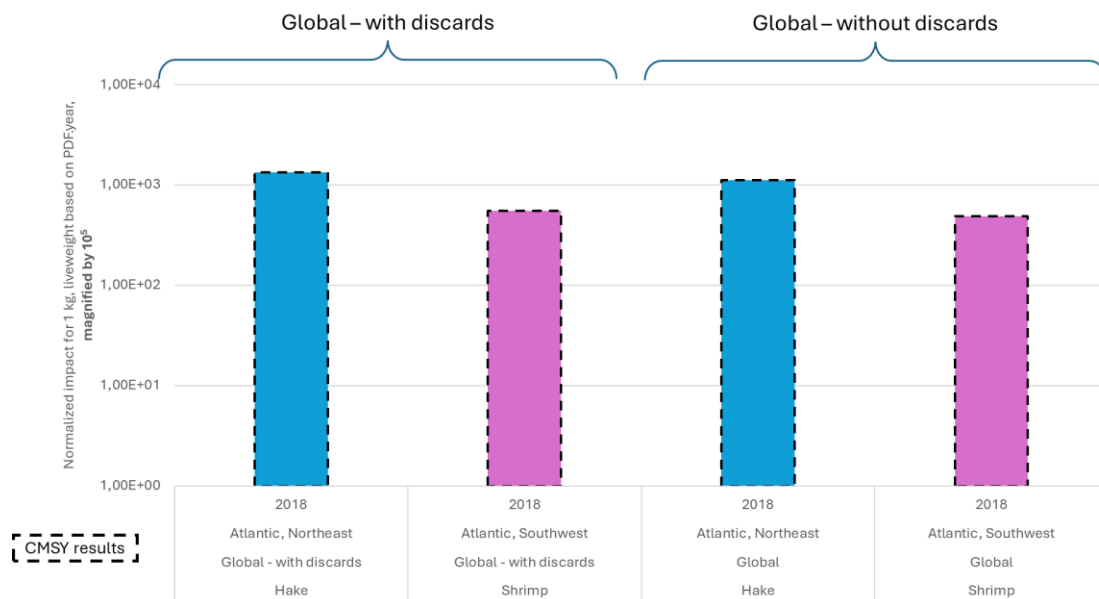
3.4.4 Accounting for discards

Stanford-Clark, Loiseau et al. 2024 recommend favouring CFs with discards rather than without, as they account for damage done to species that are not targeted nor economically valued but however collateral damage of fishery activities. Impacts with and without discards are supplied for metastocks in Stanford-Clark, Loiseau et al. 2024 ‘s supplementary information. Figure 5 displays the differences for the two case studies. As expected, impacts with discards are higher than without, but they stay within the same order of magnitude.

² We would suppose, however, that the authors of publication have them at hand; the computation with there help should be feasible

³ Email and meeting - 25/05/2024

Figure 5. Comparison of normalised impacts of 1kg liveweight fish.



Modelling based on Stanford-Clark, Loiseau et al. 2024 (CFs computed with 2018 fish data) for case study s' hake (blue bars) and red-shrimps (pink bars). All bars are displayed for computations with CMSY algorithm for metastocks within the FAO area in PDFglo.yr. The two left bars display results accounting for discards, and the two right bars without discards. Results are presented on a log scale and magnified by a factor 10^5 .

Source: Authors' own elaboration.

3.5 Development suggestions

In all cases (metastocks or stocks), uncertainty around biomass estimates is substantial (around one order of magnitude for stocks and two for metastocks) but this should not be a reason for disregarding this impact pathway. Choosing this latter option would indeed lead to considering resource depletion to be zero, which is certainly wrong.

Recommendation is to be as specific as possible on stocks, to reduce uncertainty, rather than using metastock data. However, depending on availability of both (final food product) information and RFMO data, it is not always possible, and metastock data is always a fallback option. We recommend however to conduct more case studies to compare outputs and check consistency of results.

As stated before, global CFs are preferable for comparing products fished in different locations, and these would ideally be stock-specific instead of proxies at the FAO Major Fishing Area scale. However, obtaining global CFs requires the estimation of the species' total biomass, which may not be available for many species. Indeed, while for many stocks, biomass is available, it is often not the case for all stocks of a species, thus preventing the precise calculation of global CFs. In their standard CFs, authors tackled this issue by using the CMSY algorithm for all estimations.

Characterization factors have been recently updated and published (Stanford-Clark, Loiseau et al. 2024), these new CFs being based on data from 2018 (2017-2019) instead of 2015 (2014-2016) that have been used in the present application.

There is a compelling need for periodical and regular updates of CFs. Despite being outside of the scientific scope of the study, the authors recommend a governance body to oversee periodical updates and account for possible methodological evolutions.

4 On the non-LCA approach to quantifying the impact of depletion of targeted biotic resource (stock exploitation), based on STECF (2023, 2024)

4.1 Description of method

The letter-based grading system from STECF is not directly convertible into quantitative characterised (and possibly subsequently normalized) indicator scores (“LCA-like”). Indeed, on the one hand, such quantitative scoring was not part of the initial request of the EU Commission to the STECF, as only a grading process was thought of. On the other hand, time constraints did not allow the French *GT Mer* (ADEME, 2023) to explore further scoring options. The authors are not aware of other published initiatives that would have taken that endeavour.

Preliminary and unpublished results ideas developed in late 2023, over the end of the French *GT Mer* (ADEME 2023), by Pr. Didier Gascuel, Chair of the first STECF EWG 20-05, member of the second EWG 22-12, and member of the *GT Mer*, to pursue integration of the STECF grading system, which represents state-of-the art sustainability assessment, into a LCA-like framework. These ideas were informally presented to some of the *GT Mer* stakeholders. The objectives are to:

- seek characterisation to translate letter-grading into a quantitative score for each type of impact; and
- propose a normalisation method to link that CF to impact per kilo seafood.

In the LCA spirit of measuring impact of individual choices, the key conceptual idea proposed by Gascuel is to standardise the impact of extracting/harvesting one kilo of a specific stock relative to the total overfishing impact of global extractions/harvest, per capita in a simple manner (at whole-weight stage, before gutting, filletting and processing). The proposed steps were described in the first report as follows:

- For Step 1: Following the range of F/F_{msy} and/or advice ratios between 0 and >2 of System 2 above, back transform STECF final grade A-E into an impact value between 0 and 2 using the same boundaries – e.g. if B would be the value around $F/F_{msy}=1$, then one could score $A=0.5$ or 0.8 depending on underfishing level, $B=1$, $C=1.2$, $D=1.5$, $E=2$ (see below);
- For Step 2: Gascuel proposed to calculate an individual normalisation Factor based on the individual volume of worldwide overfishing (for example $FAO \text{ total catches} * FAO \text{ share of overexploitation} * \text{average } F/F_{msy} \text{ of overexploitation} / \text{worldwide population}$). Then $\text{Impact score} = \text{Impact value} / SF$. We have slightly appended that initial suggestion to also include the sustainable share of the catch, in order to account for the total quantification of the pressure.

These two steps have thus been applied and further analysed in this second report, as described below.

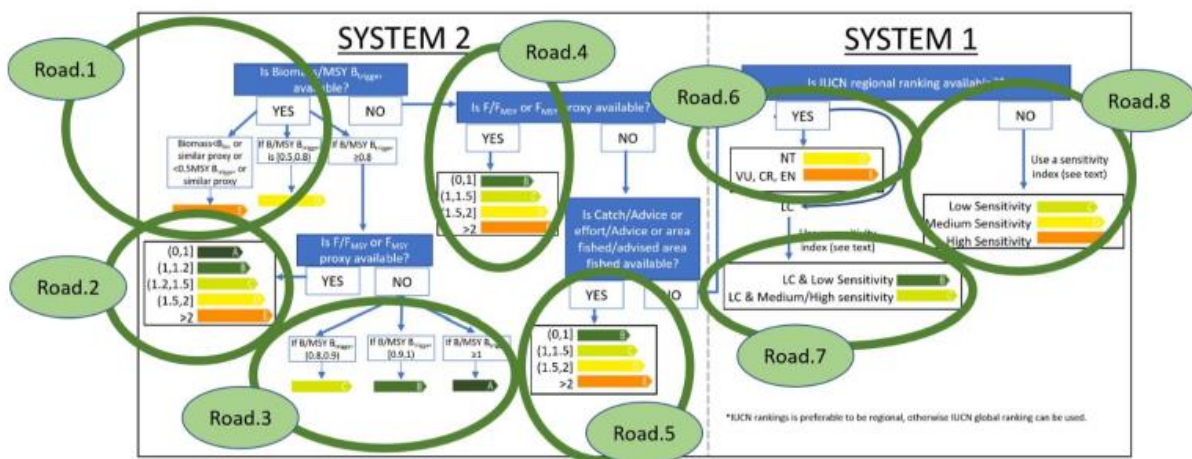
For processed and blended seafood products, the impact can be calculated according to the percentage of seafood for the total weight, alongside other similar calculations for the other ingredients; as such, the impact remains proportional to the amount of seafood in the package.

4.2 Application of method

4.2.1 Step 1: Applying the STECF grading approach.

The STECF roadmap for grading fish products is reproduced here, extracted from EWG 22-12 report (STECF 2023). The main point is that several “roads” of computation have been proposed, depending on the availability and precision of stock status information (Figure 6 and Table 4). System 2 cover cases with some stock assessment data available, ranging from data-rich with available F/Fmsy and B/Bmsy or B/MSYBtrigger (for ICES stocks) estimates, to data limited with more partial information. System 1 covers stocks for which no stock assessment information is available, and stock status proxies are based on IUCN status (roads 6 or 7) or, ultimately, Fishbase sensitivity data (road 8).

Figure 6. Decision tree from EWG 22-12 for the indicator on the status of the stock



Indicator used by the IT tool to evaluate sustainability levels according to stock status (IUCN assessments: LC: Least Concerned; NT: Near Threatened; VU: Vulnerable; EN: Endangered; CR: Critically Endangered).

Source: STECF 22-12(STECF, 2023).

Table 4. Criteria developed for the decision tree to assess the sustainability levels of fisheries products according to the stock status.

Road	System	Criteria	Thresholds	Score
1	2 (Advanced)	B/MSYBtrigger (or other B ref point)	<0.5 or B<B _{lim}	E
			>0.5 and <0.8	D
<i>if <math>\geq 0.8</math> and F/F_{msy} available - road 2</i>				
2	2	F/F _{msy}	<1.0	A
			≥1.0 and <1.2	B
			≥2.0 and <1.5	C
			≥1.5 and ≤2.0	D
			>2.0	E
<i>if F/F_{msy} not available - road 3</i>				
3	2	B/MSYBtrigger (or other B ref point)	≥0.8 and <0.9	C
			≥0.9 and <1.0	B
			≥1.0	A

Road	System	Criteria	Thresholds	Score
		<i>if B/MSYBtrigger not available - road 4</i>		
4	2	F/Fmsy	<1.0	B
			≥1.0 and <1.5	C
			≥1.5 and <2.0	D
			>2.0	E
		<i>if F/Fmsy not available - road 5</i>		
5	2	Catch/Advice	<1.0	B
			≥1.0 and <1.5	C
			≥1.5 and <2.0	D
			>2.0	E
		<i>if Catch/Advice not available - road 6</i>		
6	1 (Basic)	IUCN status	NT	D
			VU, CR, EN	E
		<i>if LC - road 7</i>		
7	1	Sensitivity	Low	B
			Medium and High	C
		<i>if IUCN status not available - road 8</i>		
8	1	Sensitivity	Low	C
			Medium	D
			High	E

NT = near threatened; VU = vulnerable; CR = critically endangered; EN = endangered. The table here is described with MSYBtrigger as the basis reference point for biomass, following ICES approach, but other ref points for biomass exist.

Source: STECF (2023), Grati et al.(2025)

The STECF EWG 23-18 (STECF 2024), whose report was made publicly available in early April 2024, progressed further on the applicability of the stock status grading to a great number of combinations of species*area. And this was again progressed later on by the submission of a scientific article by Grati et al. (2025). Particular effort was dedicated to reducing the number of combinations grading NA in the first place, to improve coverage and operationalisation of the method. These most recent improvements are briefly described here.

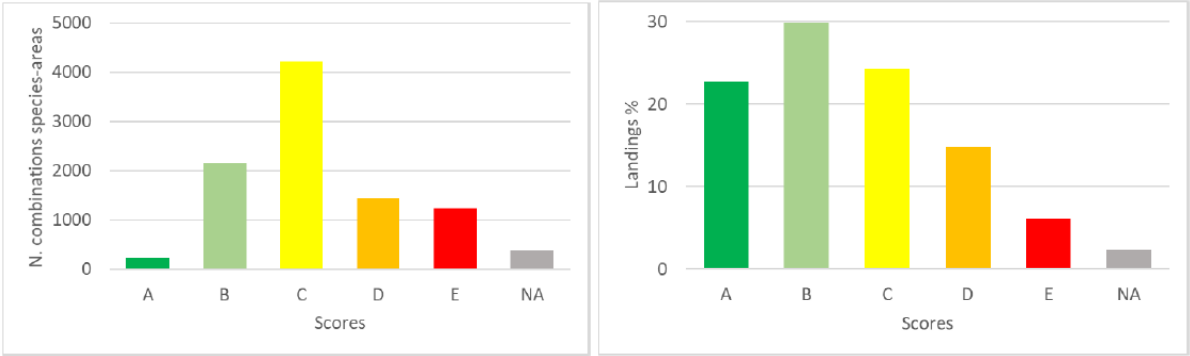
For testing, the EWG 23-18 first compiled a list of taxa and area combinations derived from the information provided by each EU Member State in the 2021 AER, therefore, in this case here only focusing on landings made by EU fleets. A total of 11,727 combinations of taxa-areas, corresponding to 2,267 taxa and 161 areas/subareas (3 563 885t landed), were tested using the IT tool

https://halieut.agrocampus-ouest.fr/discardless_app/fishing_pressure2023/. For information, all data used for grading are available as .csv files, accessible from the “download data” menu.

As a first step, the EWG focused on the combinations that resulted in NA in the IT tool (6,413 in total, i.e. 55% of the occurrences). Extensive cleaning and improvements were made, mainly focused on the rare and sensitive species landed. Ultimately, the number of unique combinations

was decreased to 9,652, and among those the NA grade was reduced to as little as 3.9% of the combinations (and 233 species in total), corresponding to 2.3% of the landings volume (Figure 7).

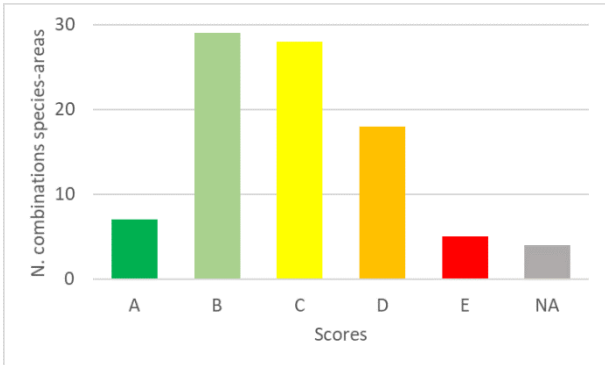
Figure 7. Final distribution of grades for the indicator on stock status calculated for the 9,652 species-area combinations gathered from the AER 2021 data. Left: in number of combinations; right: weighted by landings.



Source: STECF, 2024

A similar exercise was conducted by EWG 23-18 for grading the 20 most important imported species (based on EUMOFA trade data), leading to 91 species-area combinations, which returned a very similar distribution of grades (Figure 8). Source STECF(2024)

Figure 8. Final distribution of grades for the indicator on stock status calculated for the 91 species-area.



Combinations gathered from the EUMOFA database for the 20 most important imported species.

Source: STECF, 2024

This exercise can be considered as a successful proofing of the letter-grading approach, allowing almost complete coverage of any catch, and able to discriminate across the full range of grades. This supports the confidence that the approach is operational for grading, is coherent with worldwide sustainability assessment, and can thus be pursued for further exploration of its integration into the PEF, as presented below.

Regarding discards, since the method uses fishing mortality (F) and biomass data from official stock assessments, when discards data are available, they are included in the total catches that enter stock assessment and contribute to the total F estimate.

4.2.2 Step 2: Characterisation

As described above, it was proposed to scale the letter grade into a numerical value ranging from 0.5 to 2. This is a somewhat subjective proposal, but which is meaningful considering that a large share of worldwide F/F_{msy} estimates for exploited stocks would distribute within that [0.5-2] range (see also the numerical investigations below). One thus assumes that for those stocks for which

that information is missing, system 1 roads provide proxies for F/Fmsy values within the range [1-2] (no grade A can be given in System 1). The empirical distribution of grades shown above tends to support that proposal for transforming grades A-E, as we roughly observe a normalised quantitative scale centred on B (B=1) with a multiplicative lognormally distributed representation of risk: If B is the reference level, then least impactful products would have half the impact than that of the reference (0.5) while the worst (E) would have twice more (2). Inside the A grade, the value is proposed equal to 0.5 if F/Fmsy<0.8, and equal to 0.8 if 0.8 <F/Fmsy < 1, expressed in “unit of overexploitation/kg” (Uex/kg).

4.2.3 Step 3: Normalisation

The initial proposal by Gascuel was investigated further for quantification. Normalization makes a characterisation factor for any given product/activity, correspond to the quantification of impact per inhabitant from a pressure. It is based on the total quantification of the pressure, divided by the world population. The normalisation factor NF, is proposed to be calculated as indicated in equation 2:

Equation 2. Normalisation factor

$$NF = \frac{\text{Sustainable } C \cdot \left[\frac{F}{F_{MSY}} \right]_{\text{sustainable}} + \text{Non sustainable } C \cdot \left[\frac{F}{F_{MSY}} \right]_{\text{non sus.}}}{\text{World population}}$$

With C = worldwide catch and [F/Fmsy] the weighted fishing mortality for sustainable and non-sustainable catch fractions respectively. Here below follows a first trial to quantify this NF based on publicly available information; however, acknowledging the heterogeneity and inconsistencies in the global datasets used for that exercise in the limited timeframe available for this in the ad-hoc contract, it is advised that this NF estimate and computation proposal would need to be validated by a larger group of fisheries scientists before full endorsement of the parameter. This will not change the relative scale of impact between different seafood products, as that is based on the STECF grade itself, but it is highly important to validate the NF value before any further decision can be made on a potential weighting of this indicator in the EF framework. A scientific publication on the scoring method is already on the way (Grati et al., 2025), which should ideally be supplemented by the additional steps proposed here.

4.2.3.1 Total, sustainable and unsustainable catches

The biennial FAO state of world fisheries “SOFIA” report is the key reference for monitoring fish stock status worldwide. In its latest assessment (FAO, 2024), the FAO assesses that in 2022, wild captures fisheries summed up to 91 million tonnes (live weight equivalent) of aquatic animals, including 11.3 million tonnes freshwater fisheries⁴. Meanwhile, The FAO assesses that the fraction of fishery stocks within biologically sustainable levels decreased to 62.3 percent in 2021. This calculation treats all fishery stocks equally, regardless of abundance and catch. When weighted by their production levels, biologically sustainable stocks account for 76.9 percent of the 2021 landings of assessed stocks monitored by FAO. The FAO does not clearly specify whether these data production estimates correspond to catches or to landings only, but we understand it is landings. Worldwide discards values were estimated around 10 million tonnes (Zeller et al., 2018; Perez Roda et al., 2019). In addition, the FAO does not indicate either the percentage of total landings this fraction of assessed stocks corresponds to, which are difficult to assess but can be substantial as discussed in the first report (over 50% of total landings according to Minderoo Foundation (2021).

⁴ <https://openknowledge.fao.org/server/api/core/bitstreams/00f1704f-b092-492c-84da-63c2a88923e2/content/sofia/2022/capture-fisheries-production.html>

On this basis we propose that Total catch = 101 million tonnes, and sustainable Catches = 77 million tonnes, but this latter is most certainly an overestimate. Unsustainable catches are set at 34 million tonnes.

4.2.3.2 Sustainable and non-sustainable F/Fmsy

Gascuel (pers. com) proposed to weight this by the severity of the overfishing occurring worldwide, as the impact is of course worse with F/Fmsy values way above 1 than close to 1, and suggested an average value of $F/F_{msy}=1.5$ for those stocks that are overexploited based on Costello et al. (2016). Unfortunately, that value is not easy to update and proof-check, as we could not find tables and data in the FAO report supporting the above-mentioned estimates of sustainable fishing. Furthermore, standard synthesis reporting on worldwide overfishing (not only FAO, but also SDG 14.4.1, EU MFSD or EU CFP monitoring) are also usually presented in a categorical manner (number of stocks exploited sustainably), without much indication of the actual level of overfishing left.

One exploration was made by extracting the stock assessment data set provided in OurWorldInData website⁵ which shall reflect the worldwide stock assessment data from RAM Legacy Database (<https://ramlegacy.org>). That dataset proved however to be highly incomplete and not updated, with global figures only partly coherent with the FAO figures. As an example, total landings given from the 1329 stock references recorded in the full data set only sum up to a share of the total landings (20 million tonnes in 2015, even less afterwards, far from the 80 million marine fisheries assessed by FAO). Of these 1329 stocks, 396 have landings data for 2015, and only 118 of these do report a F/Fmsy value for that year, representing 54% of the landings reported for that year in the entire database. Selecting only those values where $F/F_{msy}>1$, (52 stocks, 20% of landings for these 118 stocks), the average F/Fmsy value weighted by landings returned indeed a value of 1.56.

However, quoting from OurWorldInData website, “robust stock assessments data are largely missing across Asia, Africa (except South Africa), and much of Latin America. While not much can be said about the status of these stocks, the very fact that there aren’t any good data is itself a bad sign. Managing the sustainability of fisheries is a delicate process – to do it well requires continual monitoring of the fish populations, their reproduction rate, and how much fish are caught. This balance can change from season to season or even month to month as environmental factors change and affect the dynamics of the ecosystem. Regions that manage fisheries well are constantly monitoring and changing catch limits when necessary. Fisheries that are not carrying out these assessments will struggle to maintain this balance. Based on the fishing pressure assumed intense in many places⁶ (measured among other from estimates of trawling intensity), the consensus is that many of these fish stocks are not in a healthy place”. Indeed, Costello et al. (2012) examined what could be concluded for fisheries lacking formal assessment, which they estimated comprised >80% of global catch in the late 2000s. They found that small unassessed fisheries were in substantially worse condition than assessed fisheries, but large unassessed fisheries may be performing nearly as well as their assessed counterparts. Building further on this, Costello et al. (2016) previously estimated the mean global value of F/Fmsy to 1.5, combining RAM database with CMSY estimates for non-assessed stocks, which is consistent with the average value estimated above. These data could be compared with the more recent estimates performed by Helias (2023), while keeping in mind all the caveats.

⁵ <https://ourworldindata.org/explorers/fish-stocks>, accessed 07/05/2025

⁶ data from Global Fishing Watch can be used for such global estimates, <https://globalfishingwatch.org/>

Here, in the absence of firmer or more updated results, we have arbitrarily chosen to buffer the F/Fmsy average slightly above the value obtained for assessed stocks, with penalty factor set up at 1.7.

On the other hand, for the same calculation of the fish stocks exploited sustainably from OurWorldInData in 2015, selecting only those values where F/Fmsy<=1, (66 stocks, 80% of total landings of the 118 assessed stocks), the average F/Fmsy value weighted by landings returned a value of 0.60. Hilborn et al. (2020) had analysed that dataset and had indeed concluded that effective fisheries management is instrumental in improving fish stock status, pointing to a favourable status for many managed stocks.

However, considering that this information in the database covers only 25% of the total marine landings assessed by FAO, and considering that these 2015 estimates are anyway outdated, at this stage it remains difficult to come with a firm estimate for the NF calculation. The FAO estimates that only 11.8 percent of the total number of assessed stocks in 2021 are classified as underfished (biomass above 120 percent of the target level, B/BMSY > 1.2). Therefore it appears overoptimistic to use that value of 0.60 for the calculation of NF. Thus, we do not provide here an alternative estimate for F/Fmsy for sustainable catches, and will keep a value of 1.

Reported to the world population reference value of 6 895 889 018 persons in 2010 would lead to a numeric application of:

Equation 3. Calculated Normalisation Factor

$$NF = \frac{(7.7 \cdot 10^7 \cdot 1) + (3.4 \cdot 10^7 \cdot 1.7)}{6,895,889,018} = 1.95 \cdot 10^{-2} \text{ tons.hab}^{-1}$$

$$NF = 19.548 \text{ kg.hab}^{-1}$$

Utilising this normalisation factor - as indicated in equation 3 - , users can calculate all possible normalised CF values that would be attributed to whole-fish products (Table 5).

Table 5. Table of relationship between STECF grades compared to the quantified values.

STECF Grade	STECF Qualitative Description (see figure)	Proposed Score Quantification	Normalised CF (=score / 1.95E+01)
A	F/Fmsy<0.8	5.00E-01	2.56E-02
	0.8 <F/Fmsy < 1	8.00E-01	4.09E-02
B		1.00E+00	5.12E-02
C		1.20E+00	6.14E-02
D		1.50E+00	7.67E-02
E		2.00E+00	1.02E-01

For description, refer to Figure 6 above. These proposed values are based on a multiplicative log-normal representation of risk across the STECF scores.

Source: Author's own elaboration.

4.2.4 Step 4: Integration with other impact categories

The step above derives a normalised characterisation factor which can be described as a proportion contribution to overfishing per kg product live weight and could thus in theory be fit for integration into EF as an additional indicator. For processed products, this could be proportionally calculated

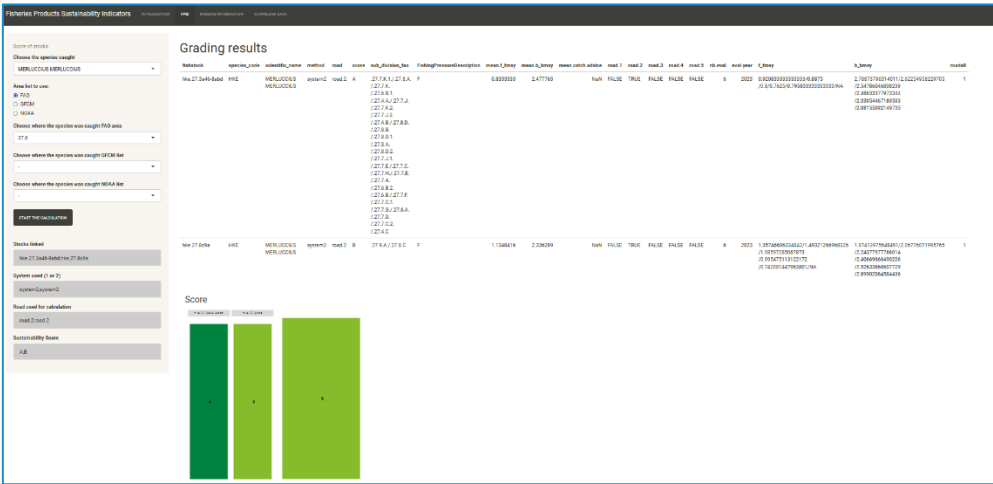
according to the recipe: as such a seafood product with 10% seafood would be calculated as 10% of the impact per kg product.

4.3 Experience and performance with case studies

4.3.1 Hake from the Bay of Biscay caught by Spanish fleets, fished with demersal trawl

Hake (*Merluccius merluccius*) in FAO area 27.8 is available in the pilot tool for two stocks: hke.27.3a46-8abd (Northern hake) and hke.27.8c9a (Southern hake). Spanish fleets would catch both stocks in the Bay of Biscay, and the CMO data would not necessarily allow distinguishing from which stock the catches come. Accordingly, unspecified hake would take the worst score of the two, as agreed by STECF (2023) with DG Mare (Figure 9).

Figure 9. Screenshot of the scoring pilot tool for hake in Bay of Biscay.



Source: https://halieut.agrocampus-ouest.fr/discardless_app/fishing_pressure2023/

With 2023 stock assessment data available from ICES, a 5-year average (2018-2022) of F/Fmsy and B/Bmsy (proxy) can be calculated, leading to following System 2 / road 2 for both stocks.

hke.27.3a46-8abd has average F/Fmsy < 1 (=0.83) and B/Bmsy > 1 (=2.48), scoring A.

hke.27.8c9a has average F/Fmsy > 1 (=1.13) and B/Bmsy > 1 (=2.34), scoring B.

As such, if subarea would not be available and only specified at the level of Area 27.8, the scoring for the product would be B.

Converted into standardised numeric value,

1 kilo of hke.27.3a46-8abd would be scored 0.8 (=score A and 0.8 < F/Fmsy < 1), which then when normalised would score 0.8 / 19.548 = **0.0409**

1 kilo of hke.27.8c9a or of hake unspecified Bay of Biscay would be scored 1 (=score B), which then when normalised would score 1 / 19.548 = **0.0512**

4.3.2 Hake from Bay of Biscay caught by Spanish fleets, fished with paired mid-water trawl

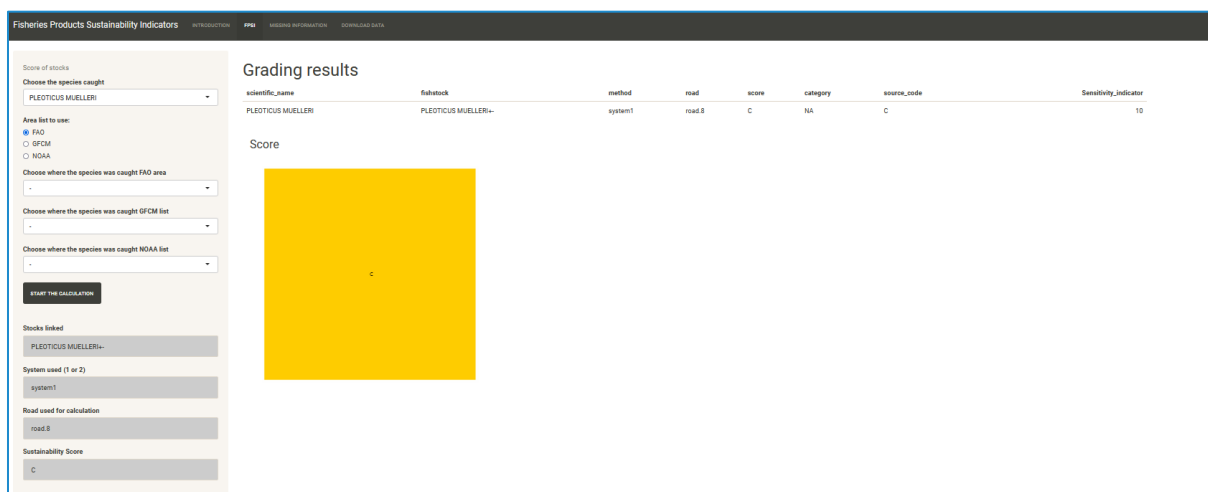
Same as above, as fishing 1kg of hake has the same impact on biotic resource depletion, regardless of fishing gear / technique.

4.3.3 Argentine red shrimp from the inshore Argentine fishery, fished with paired beam trawl

This is a very good example of how far one can go with a fully imported species without much biological data available.

Argentine red shrimp (*Pleoticus muelleri*) is available in the pilot tool, without specification of the area (Figure 10).

Figure 10. Screenshot of the scoring pilot tool for Argentine red shrimp from inshore Argentine waters.



Source: https://halieut.agrocampus-ouest.fr/discardless_app/fishing_pressure2023/

There is no stock assessment data recorded in the tool; the species is assessed as System 1. However, this species is neither assessed by IUCN. Then the calculation road followed by the tool is as described in EWG 23-18 report section 2.1.2: the EWG incorporated in the tool the sensitivity values directly extracted from fishbase.org (for fish species) and sealifebase.org (for other species than fish) and estimated using the method by Cheung et al. (2005). For example, for *Pleoticus muelleri*, <https://www.sealifebase.se/summary/Pleoticus-muelleri.html> informs that vulnerability is low (score 10 of 100). This value leads to a score C in road 8, that is what we can be seen in **Figure 10**Figure 10.

Converting this grade into a standardised numeric value:

1 kilo of the species PLEOTICUS MUELLERI+- would be scored 1.2 (=score C), which then when normalised would score $1.2 / 19.548 = \mathbf{0.0614}$

4.3.4 Overall performance summary

4.3.4.1 Performance of the STECF letter-grading

The examples above have demonstrated the extreme simplicity and facility of use with which a coherent quantitative factor could be proposed for stock status indicator, on widely diverging case studies.

It is noteworthy to underline that the STECF has now completed its 4-years endeavour to (i) develop and operationalise the concepts, (ii) frame them around the diversity in the availability of existing data, and (iii) design, test and validate the pilot tools to ease data access and updating; making this utmost timely today for environmental labelling implementation in real conditions.

It is also noteworthy to underline that, beyond the mere ease of automated data update thanks to the increasing online availability of stock assessment results, the data sets used in the pilot tool are largely building on data collected for other policy purposes, which secures a regular scientific effort dedicated to inspecting and validating the data collected automatically. Indeed, stocks assessment data are annually collected by the STECF in the frame of the CFP “balance-capacity” requirement (STECF, 2023b); as such, it makes it a natural synergy, at little additional marginal cost, to also update the corresponding database for environmental labelling. This is a strong asset for European fish stocks.

Third, it has also been noted above that the STECF grading approach now allows scoring for basically any product, both domestic and imported (provided that the last few remaining NA percent are inspected further and graded – see discussion below); and allows also discriminating across products along the full range of grades from A to E, based both on the status of the stock or species itself (abundance, vulnerability) and on the quality of the available data. The system bears fairness and virtuous incitation to scientific monitoring, in that a product without any assessment cannot score better than C, and a product assessed in System 1 (IUCN data) cannot score better than B; only products with a true assessment and demonstrated good status can score A.

4.3.4.2 Performance of the normalised quantitative impact

We have demonstrated here a simple though meaningful way to convert that qualitative grading into a quantitative impact factor, reported to the kilo thanks to a normalisation factor based on worldwide overfishing impact. This normalisation factor would also need to be updated at regular intervals, knowing that overfishing levels evolve with time and need to be monitored.

However, it is only over the very end of the contract work that the proposed scores were brought together and compared with the 16 LCA indicators for the case studies (section 8), which revealed large differences in scales and orders of magnitude across impact types. Part of this will of course be addressed in the weighting procedure, but refinements in the proposed NF factor might also be re-discussed within a wider group of experts, without changing the core thinking behind it.

4.4 Fit for purpose?

This trial work appears quite fit for the purpose sought and suggests us that it is possible to integrate the worldwide reference approach on sustainable fishing, based on MSY, within the EF framework in a weight-based approach. This is a very interesting development, as it seems to overcome some of the conceptual – even philosophical – challenges raised in the first report.

In addition, the STECF (2023) has itself undergone important reflexions on robustness and operationalisation, which contribute to making this approach pragmatic, transparent, simple to convey, and robust.

Throughout its trials, the STECF has been confronted to situations where several grades could be derived for the same product, if there are several sub-stocks or several habitats (for seabed impact) (cf STECF 22-12 report). Here the decision made, in agreement with DG Mare, is to systematically pick up the worst of all choices as the default value, without discussion about weighting or priority. For example, that would apply here to products based on “hake in the Bay of Biscay”, i.e. referring to subdivision 27.8. Without additional information, the grade would be B, i.e. the worse among hke.27.3a46-8abd (graded A) and hke.27.8c9a (graded B), even though catches of hke.27.3a46-8abd are much higher in 27.8 than those of hke.27.8c9a. This could of course give occasion for dispute, as this could be perceived as unfair labelling of a stock fished sustainably. Nevertheless, choosing the opposite would be deemed equally unfair, but also misleading for environmental

purposes. We recognised that there is no simple best solution for this, and that the approach chosen by STECF is coherent, transparent and simple.

The STECF has reflected on appropriate time periods for averaging and updating the grades, knowing that several stock assessments are not performed every year, but only from time to time (for example, some tuna RFMOs operate with a rotating system where each stock will be assessed every three or five years).

For averaging, STECF (2023) suggested to use a period of 6 years to estimate average fishing pressure, catch and stock status relative to the appropriate reference levels; that will ensure for most stocks a minimum of two assessments and ensure consistency with the most recent MSFD guidelines (European Commission, 2022), which perform update status every 6 years.

For updating, STECF (2023) underlined that stock status may change from year to year for the stocks assessed every year (most ICES stocks). In addition, scientific assessments and reference points such as BMSY or Blim, may also change from time to time, following science improvements and benchmarking process. Therefore, the corresponding assessments also need to be updated on an ongoing basis. As such the database should be nevertheless updated every year with the availability of new stock assessment data, so that grading always builds on the most recent information; meanwhile keeping in mind that the long 6-years average recommended above allows for a stable smoothing, avoiding that grades would fluctuate too often.

4.5 Development suggestions

The STECF EWG 23-18 (STECF, 2024) proposes several suggestions for further improving the grading tool and the data collection (section 2.4 of the EWG report, STECF 2024). These suggestions are directly relevant for robust integration in EF. Some of the points particularly described in the report are:

- Paying more attention to the scale and periodicity of IUCN assessments, considering that some global and/or older assessments may not be fully reliable for grading, and that some case-by-case decisions might be necessary to decide between calculation roads 6-7 or 8 in System 1.
- The main challenges encountered by STECF in implementing the IT tool stemmed from the difficulty in combining several highly heterogenous data sources. This was successfully addressed by a dedicated group of scientific experts; but when the tool is to be scaled up for transparent policy use, there are some ways to smooth and streamline the process, including:
 - additional effort made by RFMOs and stock assessment bodies (NOAA, ICES, GFCM) to further improve and standardised their public assessment databases and web services;
 - a process to collect and integrate national assessments for local coastal stocks, not always easily available and trackable at international levels;
 - improvement in stock unit definition for stocks outside European waters, to limit confusion when several biological stocks exist within the same large FAO areas others than FAO 27 (North Sea Atlantic) and FAO 37 (Mediterranean and Black Sea). This could require a discussion on future upgrading of the basic CMO data requirements regarding area definition

5 On the LCA approach to quantifying the impact of fishing on the seabed based on Woods and Verones (2019)

5.1 Description of method

Access to improving technology in the past two centuries has implied that human societies have shifted from a limited use of ocean resources, which has mainly focused on the use of biotic resources in coastal areas, as well as in pelagic ecosystems throughout the world's oceans. However, with the arrival of fishing vessels propelled with fossil fuels new fishing techniques emerged that allowed targeting fishing stocks that were located at deeper depths in the high seas, in many cases affecting benthic ecosystems on the seabed (Woods et al., 2016). Moreover, oil and gas production, oil spills in the open sea or along coastal ecosystems, coastal infrastructure (e.g., piers, ports), or seabed mining are some additional pressures that affect seabed systems (Woods et al., 2016). Of all these activities that affect the seabed, fishing activities are considered to be the main global contributor to marine seabed habitat destruction (Airoldi & Beck, 2007, Brown et al., 2018), especially considering the vast amounts of area covered as compared to other common activities. However, this does not necessarily mean that the severity of the seabed disruption per area affected is homogeneous among activities, or even between fishing gear when focusing on fishing gear exclusively. In this context, bottom trawling is one of the most disruptive fishing gear, generating habitat modification, as well as affecting the diversity of ecosystems, as well as species richness and abundance in different ecological niches across the global seabed (Thrush & Dayton, 2002).

The inclusion of seabed impacts in environmental management methodologies, such as LCA, has been discussed for the past two decades, with an array of different efforts aiming to integrate these impacts within LCA metrics. While initial efforts by Nilsson and Ziegler (2007) focused on reporting area of seafloor swept by fishing gear per hour of effort, impact categories soon developed into more complex computations to reflect the actual impact pathway exerted on seabed habitats. In this sense, different efforts have developed impact pathways that can be categorized based on their assessment in affected area assessment (Abdou et al., 2017; Woods and Verones, 2019; Pr at et al., 2021), distance to nature (Farmery et al., 2017), or productivity assessment (Langlois et al., 2015).

In the preliminary study described in Annex 1, two of these methods were identified as those with a high level of coherence with Life Cycle Impact Assessment quality and robustness criteria and, therefore, appropriate candidates for their inclusion in the EF method: Langlois et al. (2015) and Woods & Verones (2019). In a second iteration, however, Woods and Verones (2019) was selected as the most reasonable impact category to select based on two elements: i) a more quantitative accountability of uncertainty, and ii) a methodological framework that is more in line with current developments in other impact categories, such as those developed in the GLAM framework. More specifically, Woods and Verones (2019) presented a series of CFs to assess seabed damage from single-impact and repeated-impact perspectives. In the single-impact perspective, the occupation and transformation impact suggested by Mil  i Canals et al. (2007) was considered. The occupation impact CF included the extension of damage and initial benthic response. The transformation impact considered in addition to the described parameters the recovery time of a certain ecosystem, which was based on substrate-specific recovery time, hydrodynamic energy and the stock of potential recolonizers. In the repeated-impact perspective, the main processes are not occupation and transformation, but recoverable and non-recoverable transformation impacts due to potential incomplete ecological recovery in the industry-specific time lapse between different

disturbance episodes. A total of 17 marine ecoregions in Europe were considered and the resulting CFs were presented in $\text{PDF} \cdot \text{year} \cdot \text{m}^{-2}$.

Despite the appropriateness of this method, certain shortcomings were identified when analysing the theoretical principles of the Woods and Verones (2019) method. In this sense, four main limitations were reported. In the first place, a single impact pathway that computes habitat loss/creation is used in the modelling framework of the cause-effect chain that leads to the final disappeared fractions of species (PDFs). This implies that the method does not account for other environmental aspects related to seabed disruption. A second issue is the fact that the method focuses on the impact on the benthic area exclusively, whereas the interrelation with other areas of the water column, namely the different depths of the pelagic zone, but especially the benthopelagic zone, remains out of the methodological scope. A third point is that when assessing the seabed disturbance generated by fishing, it must be considered that other anthropogenic actions, such as marine infrastructure, or seabed mining, should also be eventually included within a seabed impact category, as their actions are contributing to the same disturbance and generating similar if not identical impact pathways as the one's described above. Hence, an LCA method (indicator) to account for the impacts on the seabed, unlike depletion of the targeted resource, which is ultimately a fishing-exclusive action, should also leave the door open for methodological improvements that allow the inclusion of characterization factors related to non-fishing actions on the seabed. A final point is the fact that the method presented by Woods and Verones (2019) only provided a background database to calculate the CFs for European waters. While replicability is possible and the steps to apply the method in other marine areas of the world are described in the publication, doubts remain to whether data quality and availability allow for the calculation of CFs throughout the globe, especially in international waters where governance and access to updated and thorough marine-related data is low.

Based on this brief discussion, the application of the Woods and Verones (2019) method is discussed in Section 5.2 and further elaborated with a case study in Section 5.3. Finally, Section 5.4 provides a short discussion on the appropriateness of including this LCIA method within the current EF 3.1 method.

5.2 Application of method

As mentioned above, the Woods and Verones (2019) developed an area assessment perspective in their modelling in which they assume that every given unit of seabed area contributes equally to the total species richness within each of the 17 marine ecoregions related to European marine waters. In other words, the assumption implies that every square metre of seabed in a smaller ecoregion contributes to a greater proportion of the total species richness, whereas in a larger ecoregion the impact would be lower. Marine ecoregions are obtained from a study by Spalding and colleagues (2018), in which they characterize 232 ecoregions throughout the world. However, a first limitation of this map is that these 232 ecoregions only account for coastal and shelf waters that are shallower than 200 m or within the first 370 km (i.e., 200 nautical miles) from the coast, whichever of the two counts is furthest away from the coast. While this assumption covers most fishing practices in the world in which fishing gear directly disturbs the seabed, there is an increasing trend for trawlers to extend their range to deeper waters in the high seas, especially close to slopes and seamounts, where marine diversity tends to be highest and catches are potentially more succulent and effective than in other areas of the ocean (Althaus et al., 2009), and, in many cases, beyond the coastal areas under national jurisdiction. In this sense, and in terms to guarantee a global applicability of the method, further marine ecoregions beyond the areas recommended by Spalding and colleagues (2018) should be proposed, especially in those areas in

which high fishing effort and ecosystem richness are reported. Having said this, however, the idea of a broader indicator that accounts for other activities beyond fishing (e.g., seabed mining) should be present in method development, for which a higher accountability of seabed areas will eventually be necessary.

The method by Woods and Verones (2019) considers two differing types of impacts on the seabed: single impact and repeated impact. The rationale behind this dichotomous perspective is linked to the anthropogenic disturbance regime, i.e., the frequency with which a specific area of seabed is disrupted, based on the logic that multiple disturbance events in a short period of time may compromise the capacity of a given area of seabed territory to fully recover.

5.2.1 Single impact seabed disturbance modelling

5.2.1.1 Impacts of seabed occupation

When focusing on seabed occupation, the determination of the PDF is relatively straight forward, as only two main datapoints are needed: i) the initial benthic response (P) and, ii) the reciprocal of the area (A). For the former, Woods and Verones use three differentiated values to represent the benthic response: i) high, when there is a total disruption of the seabed, which could be considered for seabed mining or permanent anthropogenic infrastructure, for which a P_{high} value of 1 is fixed; ii) moderate, when the disruption presents a variable level of reversibility without human intervention, for which a P_{moderate} value of 0.5 is assumed; and, iii) null, when no disruption is considered and, therefore, the human intervention is not contributing to seabed occupation (i.e., $P_{\text{null}} = 0$). While this classification is useful and allows disaggregating between the severity of the seabed disruption, it still represents a qualitative scale in which different levels of intervention on the seabed due to, for instance, different types of fishing gear or different seabed types, remain unaccounted for. Hence, through this parameter we are assuming that all bottom fishing gear are contributing to a 50% species loss in the area affected, whereas permanent infrastructure is representing a 100% species loss. For the latter, areas of the different ecoregions in European waters, as well as the remaining ecoregions worldwide, are available at the Digital Observatory for Protected Areas (DOPA) Explorer (European Commission, 2024).

5.2.1.2 Impact of seabed transformation

In terms of seabed transformation, Woods and Verones (2019) assume a linear recuperation of the disrupted seabed. While this is a simplification, it must be noted that it is coherent and homogeneous with equivalent pathways developed for terrestrial land use (Milà i Canals et al., 2007). Moreover, the method considers that the species loss in a given area is geographically confined to the seabed territory under assessment that suffered the damage, and once the disruption starts to recover until a new natural stability, the displaced species commence to recover the area previously damaged. Hence, the transformation impacts on the seabed are calculated as a quotient of the CF for occupation (see previous subsection), a 0.5 coefficient that represents the linear nature of the species richness recovery over time, and the estimated ecological recovery time (T), as shown in Equation 2 of Woods and Verones (2019). Thereafter integer T is dependent on three additional parameters: i) substrate type (s); ii) hydrodynamic energy at the seabed category (h); and iii) ecoregion.

In the first place, a substrate-specific median recovery time (M_s) is needed to represent the influence of each substrate on the recovery time T . A range of values between 0.5 years and 55 years has been provided for different substrates based on a collection of 27 studies in the scientific literature, of which 26 studies were in European waters and 1 in Maine (US). The median recovery

times are visible in Table 6. It can be observed that disrupted seagrass meadows (55 years) and rock or biogenic reef, i.e., corals and other similar substrates (ca. 19 years) present the highest recovery times, whereas sands (4 years) and muds (0.5 years) show the lowest recovery times. The main limitations of this parameter with current level of marine science and LCIA development are multiple. In the first place, computation of recovery times for each type of substrate is based on a very limited amount of datapoints from the scientific literature. For instance, the value reported for muds is based on one single datapoint. Secondly, there is great geographical bias in the reported literature, as all but one study is reporting conditions of recoverability in European marine areas. A third point is whether the classification of substrates is sufficient, or whether further level of disaggregation should be needed, especially if we consider that an updated version of the impact pathway should extend the territorial scope to the entire ocean surface worldwide. In this sense, seamounts and guyots, which represent areas of high ecosystem diversity and a desired target for many fisheries, not only represent fragile ecosystems (Koslow et al. 2016) but are also usually made from volcanic rock rather than the surrounding sedimentary bed that is predominant in world oceans (Dutkiewicz et al., 2015). A final limitation is the heterogeneity of substrates during fishing effort, which are highly variable within ecoregions.

Table 6. Summary of substrate-specific median recovery times.

Statistic	Rock or biogenic reef	Coarse or mixed sediments	Sands	Muds	Seagrass meadows	'Unknown'
Median	18,5	5,5	4	0,5	55	5,5
Number of study sites	4	12	8	1	2	-

Source: Authors' own elaboration adapted from Woods and Verones (2019).

Hydrodynamic energy at the seabed, which can be defined as the kinetic energy embedded in the motion of marine water, is the second integer that influences ecological recovery time T according to the method developed by Woods and Verones (2019), as it is assumed that higher level of hydrodynamic area fosters food availability and recolonization rates. This integer is a fundamental part of the method proposed, as it assumes that recovery rates of seabed territory will be more likely in areas where there is a certain level of horizontal and/or vertical circulation of ocean water, whereas in areas with low circulation the recovery rates expand. Considering that there is a vertical gradient in terms of water circulation, with deeper areas presenting lower hydrodynamic energy than shallower areas, depth acquires an important role in the determination of the CFs proposed in Woods and Verones (2019).

In general terms, the values of the hydrodynamic energy were simplified for the geographical area under study in Woods and Verones (2019) based on depth: i) moderate to high energy in depths shallower than 72 m; ii) low to moderate energy in depths between 303 m and 72 m; and iii) low energy for all depths deeper than 303 m. The reasons for this simplification, among others, were linked to the fact that only 67% of the marine area studied by Woods and Verones (2019) reported hydrodynamic energy. We then assume that these values will be tougher to aggregate in global databases for ecoregions in the Global South and in international waters, where the amount of available data tends to be scarcer. Nevertheless, the use of bathymetry data, which is highly available for most corners of the world's oceans appears as an appropriate proxy to determine different levels of hydrodynamic energy based on depth. However, a discussion could be open to see whether a greater level of disaggregation than that shown in Table 7 would be plausible to provide increased granularity for this parameter.

Table 7. Depth ranges for estimating hydrodynamic energy at the seabed class and modelling parameters for different ranges.

Hydrodynamic energy at the seabed class	Depth (d; m)	Hydrodynamic energy parameter value
Moderate-high	$0 < d \leq -72$	0.75
Low-moderate	$-72 < d \leq -303$	1.25
Low	$-303 < d$	1.50

It should be noted that they also report a parameter value for high hydrodynamic energy (0.50) and for moderate hydrodynamic energy (1.00), but these values are not assigned to a particular seabed class in the model.

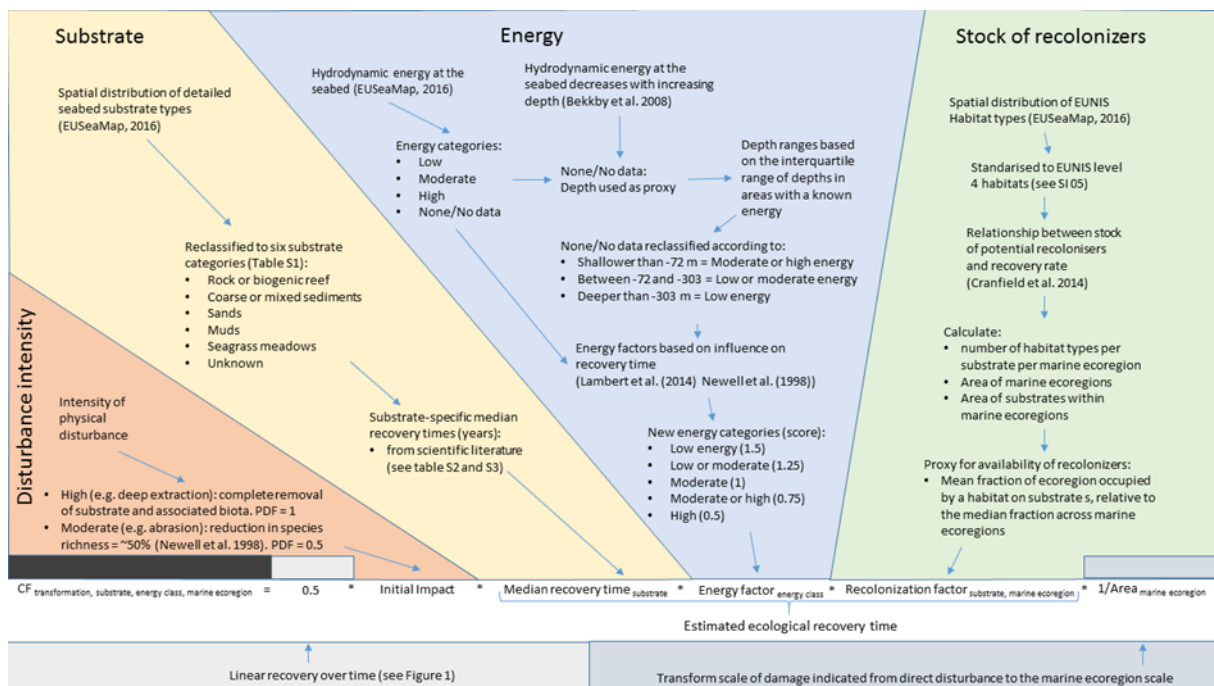
Source: Authors' own elaboration adapted from Woods and Verones (2019).

The third and final integer to calculate the ecological recovery time is the biotic stock of potential recolonisers on a given substrate type in a certain ecoregion. For the calculation of this integer, the EUNIS habitat classification, as described by Moss (2008) and available in the EUSeaMap (Populus et al., 2017), is used. EUNIS is a comprehensive system that is freely available at <http://eunis.eea.europa.eu/habitats.jsp>; including not only habitat types in marine environments, but also terrestrial environments. However, its main limitation is the fact that data availability is geographically confined to the European Union and its maritime territory, whereas no data is available for the rest of the world. This implies an important setback in the effort to extend the validity of the method to a global geographical context. Moreover, although the method computed the number of unique habitats at level 4 categories per substrate type using EUNIS and ArcGIS, the exact computational pathway used by the authors is not available in the published scientific paper or its supplementary material, limiting the possibility to reproduce the computed CFs that were published in the paper. The main reason for this is the number of unique habitats and their scatteredness in the database, which implies that a manual replicability of the exact pathway is unfeasible.

The number of unique habitats is then used to calculate the stock of potential recolonizers. This parameter is calculated based on the average stock of available recolonizers across the 27 ecoregions. In this sense, if in a specific area there is a lower than median stock available of recolonizing individuals, the recovery rate of the areas affected by the disturbance would be slower, resulting in a longer ecological recovery time T . Once again, this implies that at higher depths, where species abundance and density is less densely distributed, the recovery times would tend to be longer than in shallow areas where primary productivity and the photic zone contribute to higher species abundance and richness.

A full overview of the assumptions considered when building the Woods and Verones (2019) seabed impact method can be observed in Figure 11.

Figure 11. Graphical representation of the data sources and the way the Woods and Verones (2019) method was constructed to calculate characterisation factors (CFs) for transformation impacts within the full potential impact perspective.



Source: Woods and Verones (2019).

5.2.2 Repeated impact seabed disturbance modelling

The model developed by Woods and Verones (2019) assumes that full recovery of a given seabed area that has been disrupted by a certain human-induced event will only be possible if no further disruptive events occur prior to the estimated ecological recovery time. If this does not occur, and what is seen instead is recurrent events of disturbance prior to natural recovery times, then a set of recuperation and non-recuperation situations commence to iterate with no actual arrival to prior natural (pristine) states. This can particularly be the case in heavily exploited bottom trawling fisheries, where the harvesting of recurrent seabed areas is common, especially if individual vessels do not have a log of where other vessels have been harvesting, which, on the other hand, is the most probable situation.

In this context of repeated disturbances, two parameters become important. On the one hand, the interval between disturbance events (D_m), which is based on the reciprocal of trawling intensity, as well as on two depth zones (below or above 200 m). For the former, the trawling intensity regimes were extracted from a study by Eigaard and colleagues (2017) which provides data from most ICES fishing zones within European marine waters. However, it remains unclear if this level of granularity will be available in other marine ecoregions in the rest of the world. On the other hand, the non-recovered ecosystem quality is another parameter of importance. This parameter is calculated as the subtraction between the achieved ecological recovery, which is the product of the time available for ecological recovery (D_m), and the rate of ecological recovery.

A final limitation of the method presented by Woods and Verones (2019) is the fact that the final computed CFs in European waters is not available other than visually and a few site-specific values that can be extracted from the text of the publication. Therefore, we currently lack a freely available repository in which these CFs can be downloaded and used. However, for the sake of this report, the

authors of the study were contacted, and the site-specific CFs for the 17 marine ecoregions were made available through shapefiles (see Section 5.4).

5.2.3 Coupling with life cycle inventory interventions

Assuming that the current CFs that have been computed by Woods and Verones (2019) are available, it should be noted that current seafood LCA studies available in the scientific literature tend to lack the level of spatial and temporal granularity of fishing effort (Ruiz-Salmón et al., 2021) that is required to apply the CFs proposed by Woods and Verones (2019). However, this limitation must be interpreted with care. On the one hand, current data quality and availability constraints of the life cycle inventories of seafood products are expected to reduce through time in upcoming years due to improved available datasets in commercial (e.g. ecoinvent, Sphera) and national (e.g. Peru LCA, IBICT) databases. On the other hand, improved computational power should allow obtaining more precise data from fishing vessels through satellite-sourced information (e.g. AIS or VMS). Despite this promising outlook, accessibility to improved data to build the CFs at a global level is necessary. In fact, it is evident that the data used to power the Woods and Verones (2019) method is not always easy to collect from databases, namely when the data necessary refers to the high seas or EEZ's in the Global South. In addition, elementary flows linked to marine processes will need to be upgraded to record this additional granularity in the datasets. In this sense, elementary flows linked to fishing effort represented in nautical miles of propulsion, specific ecoregions of activity, and more precise data on the actual surface area of gears in contact with the seabed will be necessary.

5.2.4 Normalisation

Normalization factors used in LCA are usually based on aggregating inventory emissions or impacts at a regional or global level, as described by Crenna et al. (2019). Usually, global normalization factors are preferred over other perspectives (e.g., regional), as the global ones are considered more appropriate for decision-making (Pizzol et al., 2017). In the case of seabed impacts, as discussed in sections 5.1 and 5.2, however, there are numerous difficulties of applying this perspective, as current methodologies (Woods and Verones) do not have a full quantification of the inventories across the world. In fact, this also has a direct consequence on the computation of the environmental impacts or damages at a planetary level.

Consequently, for the sake of including some sort of normalization for the current environmental indicator, data from Amoroso et al. (2018) were extracted to represent the estimated amount of seabed surface area that was trawled in the period 2010-2012. In this sense, Amoroso and colleagues considered the 7.8 million km² of seabed surface at depths lower than 1000 meters, estimating that 14% of this territory (i.e., 1,092,000 km²) were trawled in the period abovementioned. While this inventory of trawled areas can serve the purpose of being an estimated area of affectation, it has numerous limitations, including the fact that it only accounts for a partial, although probably overwhelming, total seabed area affected by fishing-related activities and does not consider the level of damage exerted based on different parameters, such as substrate, hydrodynamic energy, etc. The human population considered was that of 2010 (i.e. 6,895,889,018), based on Sala et al. (2017). Despite this attempt to include normalization for this impact category, it must be noted that the perspective proposed does not comply with the EC rules for computing normalization in EF.

5.3 Experience and performance with case studies

5.3.1 Hake from the Bay of Biscay, fished by Spanish fleets with demersal trawl

Full CFs from the study by Woods and Verones (2019) were obtained through direct contact with the authors of the study. The Google Earth shapefiles were made available and uploaded to Google Earth Pro. Thereafter, the CFs for the South European Atlantic Shelf, the marine ecoregion in which the Spanish hake fleets operates, were extracted, as shown in Table 8 for single impact modelling.

Table 8. Single impact characterisation factors (CFs) for the South European Atlantic Shelf marine ecoregion.

Description of CFs	Unit	Value
Occupation – moderate intensity	PDF*m ²	7.51E-13
Occupation – high intensity	PDF*m ²	1.50E-12
Transformation – moderate intensity	PDF*yr*m ²	3.05E-12
Transformation – high intensity	PDF*yr*m ²	6.09E-12

Source: Authors' adaptation from Woods and Verones (2019).

Considering that no primary data were available considering the area trawled by this fleet, approximate data for West Iberia in Amoroso et al. (2018) were assumed, which present a factor of 0.54 metric tons of live weight fish per square kilometre and year based on data in the period 2010-2012. This was then converted to a factor of 5.40E-4 kg of hake per square metre of affected area. In other words, it was assumed that 1852 m² of seabed were trawled to fish 1 kilogram of European hake. The results in PDF are reported in Table 9, considering moderate intensity disturbance on the seabed.

Table 9. Final endpoint environmental impact results for the extraction of 1 kilogram of European hake (*Merluccius merluccius*) in the South European Atlantic Shelf marine ecoregion assuming single impact modelling with the Woods and Verones (2019) model.

Description of CFs	Unit	Value
Occupation – moderate intensity	PDF	1.39E-09
Transformation – moderate intensity	PDF*yr	5.64E-09

Source: Authors' own elaboration.

Obtaining the final environmental results for the single impact perspective is relatively straightforward and computable. However, it must be noted that it is assumed that this type of impact is industry-generic and particularly suited for activities that are either new or only occur once. In this sense, although this could be the case in some of the areas that are trawled, the South European Atlantic Shelf is a highly industrialized maritime area in which it is plausible to assume that a wider area is constantly swept by trawlers for fishing activities, whereas a minimal area corresponds to new areas that have never been trawled previously. In contrast, the repeated impact perspective is intended to account for the impacts that occur in areas with multiple interventions from the same activity (usually trawling), in which the fishing intensity and, therefore, the area that is trawled, as well as the ecological recovery time become two important parameters to monitor.

In this sense, it must be assumed that in most highly trawled areas, multiple disturbance events are likely to occur, in some cases with the ecological recovery time being longer than the time between two different interventions, meaning that only incomplete recovery will be attained. Therefore, in most trawling fisheries a sequence of events that are intercalated between recovery times that in

some cases will be longer, and in other will be shorter than the ecological recovery time, will tend to occur.

Hence, in a second step the repeated impact CFs were also extracted from Google Earth Pro (see Table 10). However, it should be noted that in this case the CFs are not common to the specific ecoregion, but vary throughout the ecoregion based on trawling intensity, depth zone, and other parameters which make the computation of these impacts highly site-specific. Moreover, for repeated impact a difference must be made between recoverable and non-recoverable transformation impacts,

In this sense, for the current study the three most common CF values based on total territory within the South European Atlantic Shelf marine ecoregion were used, as shown in Table 10. It must be noted that in two of these, the recoverable CF was equal to the single-impact CF for transformation, whereas the non-recoverable CFs were fixed at a value of zero. The reason behind the latter value is linked to the fact that this area shows a trawling intensity that is lower than that of the ecological recovery time.

Table 10. Repeated impact characterisation factors (CFs) from Woods and Verones (2019) for the South European Atlantic Shelf marine ecoregion.

Description of CFs	Unit	Value
Transformation – recoverable (A)	PDF*yr*m ⁻²	3.05E-12
Transformation – non-recoverable (A)	PDF*yr*m ⁻²	0
Transformation – recoverable (B)	PDF*yr*m ⁻²	3.05E-12
Transformation – non-recoverable (B)	PDF*yr*m ⁻²	0
Transformation – recoverable (C)	PDF*yr*m ⁻²	2.03E-12
Transformation – non-recoverable (C)	PDF*yr*m ⁻²	0

Source: Authors' own elaboration.

The results in PDF are reported in Table 11, considering moderate intensity disturbance on the seabed and the same life cycle inventory data as for the single impact perspective. Considering that no data were available regarding specific geolocations, a weighted mean of different areas was not conducted, but rather it was assumed that each FU (i.e.1 kg of European hake) occurred in specific subareas of the marine ecoregion.

Table 11. Final endpoint environmental impact results for the extraction of 1 kilogram of European hake (*Merluccius merluccius*) in the South European Atlantic Shelf marine ecoregion assuming repeated impact modelling with the Woods and Verones (2019) model.

Description of CFs	Unit	Value
Transformation – recoverable (A)	PDF*yr	5.64E-09
Transformation – non-recoverable (A)	PDF*yr	0
Transformation – recoverable (B)	PDF*yr	5.64E-09
Transformation – non-recoverable (B)	PDF*yr	0
Transformation – recoverable (C)	PDF*yr	3.76E-09
Transformation – non-recoverable (C)	PDF*yr	0

The modelling assumes different scenarios in which the fishing effort occurs in different subareas of the South European Atlantic Shelf.

Source: Authors' own elaboration.

In terms of normalization, a normalization factor of 158.36 m²/per capita was estimated, considering 1.092E12 m² of affected area (Amoroso et al., 2018) and a world population in 2010 of 6,895,889,018 people (Sala et al., 2017). Table 12 and Table 13 detail the normalization results under different modelling scenarios for single (see Table 12) and repeated impacts (Table 13) on the seabed.

Table 12. Final normalized results for the extraction of 1 kilogram of European hake (*Merluccius merluccius*) in the South European Atlantic Shelf marine ecoregion assuming single impact modelling with the Woods and Verones (2019) model.

Description of CFs	Unit	Value
Occupation – moderate intensity	Pt	2.20E-07
Transformation – moderate intensity	Pt	8.93E-07

Source: Authors' own elaboration.

Table 13. Final normalized results for the extraction of 1 kilogram of European hake (*Merluccius merluccius*) in the South European Atlantic Shelf marine ecoregion assuming repeated impact modelling with the Woods and Verones (2019) model.

Description of CFs	Unit	Value
Transformation – recoverable (A)	Pt	8.93E-07
Transformation – non-recoverable (A)	Pt	0
Transformation – recoverable (B)	Pt	8.93E-07
Transformation – non-recoverable (B)	Pt	0
Transformation – recoverable (C)	Pt	5.94E-07
Transformation – non-recoverable (C)	Pt	0

The model assumes different scenarios in which the fishing effort occurs in different subareas of the South European Atlantic Shelf.

Source: Authors' own elaboration.

5.3.2 Hake from the Bay of Biscay, fished by Spanish fleets with paired mid-water trawl

Mid-water trawls, unlike bottom trawling, does not have contact with the seabed. While this statement may be subject to numerous exceptions, we should assume that in general terms, there is no direct physical contact between these trawls and the seabed. However, it is also true that impacts on the seabed are not always limited to direct physical damage, but to other impact pathways, such as accumulation of detritus or shading (Langlois et al., 2015). In any case, for the effects of the methodology that is being applied in this particular case, the impacts on seabed impact would be zero.

5.3.3 Argentine red shrimp from the inshore Argentine fishery, fished with paired beam trawl

The Woods and Verones (2019) method is not applicable for this fishery with the current data quality and availability constraints described in Section 5.2. Table 14 shows a list of the different parameters needed to compute results for a location beyond European waters, with a description of the availability of the data.

Table 14. Table of data/parameters required to calculate the seabed impact characterisation factors for various regions.

Parameter	Europe	Rest of the World (RoW)	Argentina
Marine ecoregion	Marine ecoregions obtained from Spalding et al. (2007).	Marine ecoregions available from Spalding et al. (2007).	Marine ecoregions available from Spalding et al. (2007)
Substrate type	Obtained from Populus et al. (2017) for European waters.	Not available in Populus et al. (2007). Data available in the scientific literature highly scattered for RoW.	Some scientific articles available on specific areas of the Argentinian EEZ (Sánchez-Carnero and Rodríguez-Pérez, 2021).
Ecological recovery times	Obtained from a pool of studies (Woods and Verones, 2019).	Very limited bibliography worldwide, mainly in the United States (Neckles et al., 2005).	As for as we were able to ascertain, there was no evidence in the literature for recovery time of seabed substrates.
Hydrodynamic energy classes	Data obtained from EMODnet (2017)	Bathymetry data only available in EMODnet for certain regions of the world, mainly in Europe. GEBCO (gebco.net/data_and_products/gridded_bathymetry_data/gebco_2021), Seabed 2030 (seabed2030.org) are some initiatives that are aiming to complete a full ocean floor map by 2030.	No data found in the scientific literature.
Habitat richness	European Nature Information System (EUNIS) habitat classification system used (Moss, 2008)	Misiuk and Brown (2024) recently published a review of over 1300 studies on the benthos in the past 30 years, identifying the type of data presented. Could be a good framework to identify studies or databases to use as a benchmark for RoW.	
Fishing intensity data	Data obtained from Eigaard et al. (2017).	Hiddink et al. (2016), Amoroso et al. (2018) and Mazor et al. (2021) provide interesting global data on trawling intensity that could be used.	Alemany et al. (2016) and other prior studies analyse the intensity of trawling in the Patagonian shelf of Argentina.

Source: Authors' own elaboration adapted from Woods and Verones (2019).

5.3.4 Overall performance summary

The main current limitation of the Woods and Verones (2019) model is the fact that it does not provide a full accountability of the seabed throughout the world, which is an important limitation considering that an important fraction of consumed fish in Europe is imported from other regions throughout the world. Anyhow, the single impact model is feasible to be used for the European context, provided that information on swept seabed area is available or can be at least estimated. In contrast, the operability of the repeated impact perspective is more difficult due to the fact that geotracking of vessels is not something that is currently coupled to life cycle inventory data. In fact, the modelling of either approach in commercial LCA software appears to be challenging due to the need to create numerous elementary flows that are highly geographic-specific to areas that are not commonly used in other impact categories and are particular only to seabed impact. Moreover, the addition of the geotracking information of vessels to commercial LCA software seems something still rather remote and implies an important change in the way that commercial software is configured.

5.4 Fit for purpose?

We consider that the use of the Woods and Verones (2019) method is not currently fit for purpose due to limitations in data quality and availability, as well as a lack of elementary flows to support the merge with life cycle inventories in commercial software. While this is the current state of discussion, we do consider that efforts could be channelled to improve the geographical coverage of the model that ultimately could lead to its use in LCIA modelling (this is on-going work in at least one Horizon Europe Project).

In the first place, a thorough work must be conducted to identify data sources that can be used to cover the different parameters that are included in the computational modelling of the method (e.g., hydrodynamic energy or substrate type). In the case that these are not available for wide ecoregions of global oceans, alternative methodological pathways would be necessary to see whether a mathematically viable and scientifically robust alternative is feasible.

Secondly, we hypothesize that a full accountability of marine regions is necessary to account for all fishing grounds on Earth, as well as other human activities on the seabed (e.g., seabed mining). Otherwise, an increasing number of activities would remain unaccountable despite the technological leaps that have allowed humanity reaching.

A third point is transparency and reproducibility. For the former, currently there is no open repository to download the CFs available in European water. For the latter, certain parameters used in the modelling, namely those related to recolonization using the EUNIS database, are complex to recover and reproduce.

Finally, in terms of linking seabed impacts with life cycle inventories, discussion is still to be made in order to add the adequate elementary flows to datasets. In this sense, as abovementioned, elementary flows that grasp the distance of vessels when trawling, the specific ecoregion or ecoregions of fishing events, among other relevant flows. In addition, a revision of the relevant metadata of datasets (i.e., unit processes) should be performed to decide on what additional data should be used to define the main characteristics of datasets when a component of human-induced seabed impact occurs.

5.5 Development suggestions

The current main bottleneck in the Woods and Verones (2019) method is the lack of comprehensiveness from a geographical perspective, with current CFs limited to European waters. It must be noted that the LCA research team at NTNU are currently updating the method in the frame of an active European Horizon Project and aim to extend the geographical scope of the study, to consider, at least, all continental shelf areas across the world (Francesca Verones, NTNU, personal communication, June 2024). However, it remains unclear whether CFs will be made available for seabed areas beyond the continental shelf and if other human activities (e.g., seabed mining or coastal infrastructure) will be modelled in the method.

6 On the non-LCA approach to quantifying the impact of fishing on the seabed based on STECF (2023)

6.1 Description of method

Based on the preliminary evaluation of LCA and non-LCA method (Annex 1) the best non-LCA method available for assessing the impact of fishing for certain seafood species on the seabed is the STECF approach (STECF, 2023). Similar to the approach described in section 4, the STECF method for seabed impacts comprises two alternative “systems” depending on the data that is reported for a given seafood product. Using the reported gear type used in harvesting, combined with knowledge of the habit of the harvested species and its interactions with various seafloor habitats, a qualitative letter score (A -> E), based on a scoring value between 1 and 6, is attributed to the seafood product to reflect the expected impact of that particular fishery (Table 15).

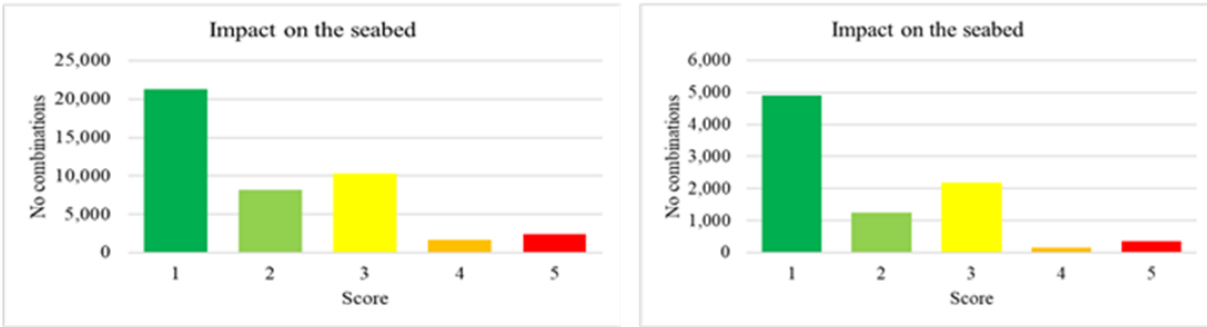
Table 15. Calculation of the final score (sum of fishing gears impact and habitat sensitivity) for the impact on the seabed indicator.

Fishing gear impact - Habitat sensitivity	0	1	2	3
1	1: Very low	2: Very low	3: Low	4: Medium
2	2: Very low	3: Low	4: Medium	5: High
3	3: Low	4: Medium	5: High	6: Very high

Source: Grati et al. (2025).

Since the STECF (2023) method, Grati et al. (2025) have tested the method on a great number of combinations of species, areas and gears landed by EU fisheries from the FAO areas 27 (Northeast Atlantic Ocean) and 37 (Mediterranean and Black Sea (43,810 (FAO 27) and 8,862 (FAO 37) combinations respectively (Figure 12).

Figure 12. Frequency distributions of the fisheries sustainability indicator scores for gear-species-subarea combinations in FAO 27 (Northeast Atlantic Ocean), left and FAO 37 (Mediterranean and Black Sea), right.



Data source EU Fisheries Dependent Information database 2022 (https://stecf.ec.europa.eu/data-dissemination/fdi_en).

Source: Grati et al.(2025)

Because the output of this method of assessment is a qualitative letter score, it is not directly translatable to an LCIA score. However, France’s working group *GT Mer*, have suggested methods to make the STECF qualitative scores into LCA compatible quantitative scores (ADEME, 2023). There were two aims for this endeavour:

- to translate letter-grading into a quantitative score for each component of the seabed impact pathway; and
- to propose a method to normalize this newly quantified score, so that it represents the relative seabed impact of a fishery product compared to the total seabed impact of wild-caught seafood produced, per person, per year, globally.

The key driver of this work, Didier Gascuel, was consulted for the purpose of this report, and provided background information on the *GT Mer* approach, a spreadsheet of methods used and an introduction to these methods.

6.2 Application of method

6.2.1 Step 1: Applying the STECF scoring approach

The first step in the application of this method was to utilise the STECF approach to obtain “internal scores” for the seabed impact, namely a score for the gear used and a score for the habitat linked to the species. In the STECF workflow, the sum of this score determines the qualitative letter score. However, in this application of the method, one must utilise the sum of the scores independently to account for the awarding of the very lowest impact score, which is a subdivision of the STECF “A” grade. The process to get to this stage is described in detail in STECF (2023).

6.2.2 Step 2: Quantification

During the development of the quantification methods for both fishing pressure and seabed impacts, the quantification of STECF scores was considered in the context of risk. As was shown in section 4, applying a multiplicative log-normal representation of risk is inherently understandable. This mean that by making the second impact category (B) a reference category with a score equal to one, those products that have lower impact get a halving of the score to 0.5 and those products that have the worst impact, score double (two). Furthermore, to represent those cases where gear interactions with the seafloor are unlikely, a “very low” impact score is awarded (0.1) (Table 16). As these scores have no inherent units, we apply a nominal unit of “Units of seabed impact” (Usi) which we apply per kilogram of live-caught fish (Usi.kg⁻¹) for the purpose of tracking the score through normalisation etc.

Table 16. Table of relationship between STECF scores, grades and descriptions compared to the quantified values.

Sum of STECF scores for gear and habitat	STECF Grade	STECF Qualitative Description	Proposed Score Quantification (Usi.kg ⁻¹)
<=1	A	Very low	0.1
2			0.5
3	B	Low	1
4	C	Medium	1.2
5	D	High	1.5
6	E	Very high	2

These proposed values were presented to GT Mer and are based on a multiplicative log-normal representation of risk across the STECF scores.

Source: Authors’ own elaboration.

6.2.3 Step 3: Normalisation

The normalisation step was undertaken to make the impact from a given product relative to the total impact of wild-caught seafood production. Furthermore, to make it relevant in the context of individual consumer choice, this impact is scaled to a per-capita level of impact.

As seabed impact from fishing is primarily brought about by the contact of gears physically disturbing the seabed, fishing effort data would be a sensible way to quantify total global impact. However, global data on fisheries are difficult to obtain, and this is especially true for fishing effort, let alone for fishing effort by gear type and/or species-specific catches. Furthermore, this normalisation method necessitates a score that represents impact per unit mass of consumption, and linking the amount of effort per gear to the production of one kilogram of a specific species is not feasible. Therefore, the decision was made to link the total production of seafood products by gear type, rather than the effort expended by gear type, in the calculation of total impact.

Modelled/reconstructed catches by gear type were downloaded from *The Sea Around Us* project (<https://www.seaaroundus.org/data/>) and gear types were mapped to the CMO regulation (EU reg. 1379/2013) Annex III gear categories, and their respective STECF gear scores. The latest global dataset available was for the year 2019, however, following the procedure laid out in Sala *et al.* (2017), the year 2010 was ultimately used as the reference year for calculating normalisation factors.

Total human population to match the latest global fisheries data set (2019) was found to be 7,713,468,000; based on United Nations Department of Economic and Social Affairs documents (<http://www.unpopulation.org/>). The global human population for the reference year of 2010 was reported as 6,895,889,018 in Sala *et al.* (2017).

Catches were aggregated to total annual catches per CMO gear group, before being transformed to kg of catch per person, for each gear group. Catch per person, per gear group was then multiplied by the gear's STECF gear score to get the total global impact per gear.

The sum of all gears' total global impact per year was used as the normalisation factor for Seabed Impact, where the per-product quantified score was divided by the normalisation factor. The R-script used to generate this normalisation factor is provided as an annex and below is an equation 4 representing the overall process:

Equation 4. Normalisation Factor STECF approach

$$NormalisationFactor = \sum_{n_{Gear\ Groups}} \left(Impact\ Score_{Gear\ Group\ n} \times \left(\frac{Annual\ Global\ Catch_{kg.Gear^{-1}}}{Human\ Population} \right) \right)$$

The method described above is a first attempt to quantify a normalisation factor based on publicly available information; however, acknowledging the heterogeneity and inconsistencies in the global datasets used for that exercise in the limited timeframe available for this in the ad-hoc contract, it is advised that this approach and the subsequent estimates should be validated by a larger group of fisheries/benthic ecologists before full endorsement of the parameter. This will not change the relative scale of impact between different seafood products, as that is based on the STECF grade itself, but it is highly important to validate this approach before any further decision can be made on a potential weighting of this indicator in the EF framework. A scientific publication would be the most appropriate process to follow; alternatively, a request to the STECF Plenary could be a possible option for such a validation process.

6.2.4 Step 4: Integration with other impact categories

The final normalised score represents the proportion of impact from one kilogram of the seafood product being assessed (live weight), thus the integration of this value into the EF process only works for landed product. Post-processing to things such as fileted fish or peeled shrimp, requires knowledge of the proportion of the original animal is retained in the product. Furthermore, for products with mixed ingredients, the final proportion of the product that contains the seafood product(s) is also necessary to consider. Examples of this further integration are given in Section 8.

Additionally, the integration of new impact categories, often involves weighting, the process of down- or up-scaling the relative contribution of one impact category compared to others. This process requires value judgements based on societal priorities and desires. This is beyond the scope of this report.

6.3 Experience and performance with case studies

The normalisation factor for all case studies in the non-LCA seabed impacts section was calculated as described in section 0 for both the most recently available global data year (2019; Table 17) and for the EF reference year (2010; Table 18). The overall value was calculated as 33.0811 kg.person⁻¹.year⁻¹ in 2019 (Table 17), and 33.47387 kg.person⁻¹.year⁻¹ in 2010 (Table 18). Both values are comprised of the sum of gear group specific global impacts.

Table 17. Figures used in calculating the normalisation factor for this non-LCA approach to incorporating seabed impact using the most recent global data available (2019).

CMO Regulation Annex III Gear Groups	Tonnes of global catch 2019 ¹	Proportion of global catch by gear	kg of catch per person ²	STECF gear score ³	Relative impact by gear type, per person per year
Dredges	3.88E+05	3.64E-03	5.03E-02	3.00E+00	1.60E-01
Gears unknown or not specified	2.68E+07	2.51E-01	3.47E+00	3.00E+00	1.04E+01
Gillnets and similar nets	8.23E+06	7.71E-02	1.07E+00	1.00E+00	1.07E+00
Hooks and lines	8.06E+06	7.56E-02	1.05E+00	1.00E+00	1.05E+00
Pots and traps	4.78E+06	4.48E-02	6.19E-01	1.00E+00	6.20E-01
Seines	1.93E+07	1.81E-01	2.50E+00	2.00E+00	5.00E+00
Surrounding nets and lift nets	1.71E+06	1.60E-02	2.21E-01	1.00E+00	2.20E-01
Trawls	3.74E+07	3.51E-01	4.85E+00	3.00E+00	1.46E+01
Sum	1.07E+08	1.00E+00	1.38E+01	-	3.31E+01

¹ Source: Pauly D., Zeller D., Palomares M.L.D. (Editors), 2020. Sea Around Us Concepts, Design and Data (seararoundus.org)

² Source: United Nations, Department of Economic and Social Affairs, Population Division (2019). World Population 2019: Wall Chart (ST/ESA/SER.A/434).

³ Source: STECF 2023. Scientific, Technical and Economic Committee for Fisheries (STECF) – Validation of selected sustainability indicators and underlying methodologies for the revision of the EU marketing standards for fisheries products (STECF-22-12), Grati, F. and Druon, J. editor(s), EUR 28359 EN, Publications Office of the European Union, Luxembourg, 2023, ISBN 978-92-76-60480-8, doi:10.2760/214080, JRC132121.

Table 18. Figures used in calculating the normalisation factor for this non-LCA approach to incorporating seabed impact using the JRC’s EF reference year (2010).

CMO Regulation Annex III Gear Groups	Tonnes of global catch 2010 ¹	Proportion of global catch by gear	kg of catch per person ²	STECF gear score ³	Relative impact by gear type, per person per year
Dredges	3.88E+05	3.64E-03	5.03E-02	3.00E+00	1.60E-01
Gears unknown or not specified	2.68E+07	2.51E-01	3.47E+00	3.00E+00	1.04E+01
Gillnets and similar nets	8.23E+06	7.71E-02	1.07E+00	1.00E+00	1.07E+00
Hooks and lines	8.06E+06	7.56E-02	1.05E+00	1.00E+00	1.05E+00
Pots and traps	4.78E+06	4.48E-02	6.19E-01	1.00E+00	6.20E-01
Seines	1.93E+07	1.81E-01	2.50E+00	2.00E+00	5.00E+00
Surrounding nets and lift nets	1.71E+06	1.60E-02	2.21E-01	1.00E+00	2.20E-01
Trawls	3.74E+07	3.51E-01	4.85E+00	3.00E+00	1.46E+01
Sum	1.07E+08	1.00E+00	1.38E+01	-	3.31E+01

¹ Source: Pauly D., Zeller D., Palomares M.L.D. (Editors), 2020. Sea Around Us Concepts, Design and Data (seararoundus.org)

² Source: UNDESA, United Nations, Department of Economic and Social Affairs, Population Division. World Population Prospects: The 2010 Revision. Extended Dataset. 2011. United Nations publication, Sales No. E.11.XIII.7. Cited in Sala S., Crenna E., Secchi M., Pant, R., Global normalisation factors for the Environmental Footprint and Life Cycle Assessment, EUR (28984), Publications Office of the European Union, Luxembourg, 2017, ISBN 978-92-79-77213-9, doi:10.2760/88930, JRC109878

³ Source: STECF 2023. Scientific, Technical and Economic Committee for Fisheries (STECF) – Validation of selected sustainability indicators and underlying methodologies for the revision of the EU marketing standards for fisheries products (STECF-22-12), Grati, F. and Druon, J. editor(s), EUR 28359 EN, Publications Office of the European Union, Luxembourg, 2023, ISBN 978-92-76-60480-8, doi:10.2760/214080, JRC132121.

6.3.1 Hake from the Bay of Biscay, fished by Spanish fleets with demersal trawl

Hake fished from the Bay of Biscay by the Spanish fleet can be caught using a range of gears, including long-lines, gill nets and trawls. In the STECF approach, the gear used to harvest the seafood product is one of the two indicators of seabed impact. The gear used for harvesting is a mandatory reporting component of the CMO regulation, which specifies a mandatory two-letter code or a voluntary three-letter code for more accurate differentiation between gears. Therefore, for demersal trawled hake, the producers may report either the two-letter code “TX”, representing all trawl gears, or “OTB” representing demersal otter trawls. In both of these cases, the score for the gear is three, as the more specific three letter code gear is considered one of the more impacting of the “TX” gear group.

According to FishBase and the ICES stock annex for hake in the Bay of Biscay, Hake utilises a range of substrate types (rock, sand and mud) across different depths (50m – 1000m) at different stages in their life-history and seasonally. Based on the work done by STECF (STECF 2023), the main habitat for Hake was considered to be “offshore circalittoral mixed sediments”. This habitat association would score hake coming from the Spanish fleet in the Bay of Biscay as one.

The combined STECF method score for hake from this area, caught with demersal trawl, would therefore be four (three plus one), corresponding to a STECF grade of “C” and converted to a proposed “quantified score” of 1.2 (see Table 16).

Dividing this score by the reference year (2010) normalisation factor, gives a final impact score for the seabed impact category of 0.03585, before any weightings might be applied.

6.3.2 Hake from the Bay of Biscay, fished by Spanish fleets with paired mid-water trawl

Given the same context as for the hake case study above, hake from a mid-water trawl fishery of the Spanish fleet operating in the Bay of Biscay could submit two possible gear codes for the same product. The first would be “TX”, as the mandatory reporting, which corresponds to all trawls. The second, if the fishers provided the more detailed gear code that is voluntary, would be “OTM”, representing the midwater otter trawls. Unlike the above context, in this case the choice of submitting only mandatory reporting information would result in a higher impact score, than if providing more information, because the “TX” code would receive the most severe score of three, while the “OTM” code would receive a medium impact score of two. As we’ve already run through the example for “TX” and this stock/fishery in the previous section, here we will continue only with the scenario where the producers submit the three-letter “OTM”, code.

Furthermore, using on the predefined STECF habitat associations, the “offshore circalittoral mixed sediments” that hake are attributed to, would receive a score of one.

The combined STECF method score for hake from this area, caught with paired mid-water trawl, would therefore be three (two plus one), corresponding to a STECF grade of “B” and converted to a proposed “quantified score” of 1 (see Table 16).

Dividing this score by the normalisation factor, gives a final impact score for the seabed impact category of 0.029874048, before any weightings might be applied.

6.3.3 Argentine red shrimp from the inshore Argentine fishery, fished with paired beam trawl

Argentine red shrimp caught in the inshore fishery of some Argentine states, have a Marine Stewardship Council assessment, which provides richer information than for stocks from other areas of the Argentine red shrimp fishery. In this fishery, the harvesting method is primarily paired beam trawls, which either operate on circalittoral mud (Landa et al., 2024), or offshore circalittoral sand (STECF 2023).

Independent of whether the producers report a two or three-letter gear code (“TX” for all trawls, or “PTB” for paired beam trawls), products from this fishery would receive a STECF gear score of three.

Based on the STECF defined species main habitats, the offshore, circalittoral sand category would earn this fishery a one for the STECF habitat score.

The combined STECF method score for Argentine red shrimp from this area, caught with paired beam trawl, would therefore be four (three plus one), corresponding to a STECF grade of “C” and converted to a proposed “quantified score” of 1.2 (see Table 16).

Dividing this score by the reference year (2010) normalisation factor, gives a final impact score for the seabed impact category of 0.035849, before any weightings might be applied.

6.3.4 Overall performance summary

Based on the principle of attributing the proportion of impact to provide relative comparisons for consumer choice, this report builds on the work done by Gascuel, and the French working group *GT Mer* (ADEME, 2023), to derive a quantified score based on risk, and to calculate a relevant normalisation factor for this potential impact category.

For all three case studies, the STECF approach for assessing and grading seabed impact according to gear category and species, which are the minimal reporting requirements for seafood products in

the EU market, was possible. In cases where gear types could be reported at the mandatory, two-letter code, or as a voluntary three letter code, the STECF system was able to provide scores for both eventualities. There is an incentive for producers to report the extra, voluntary information for fisheries where the gear type is known to be of lesser impact to the seafloor than its more aggregated category (e.g. mid-water trawling vs demersal trawling for hake).

6.4 Fit for purpose?

The versatility of the STECF method means that it can be applied to all seafood products entering the EU market, according to the EU's own regulations on reporting (CMO regulation: EU reg. 1379/2013), and where an expert panel has had the opportunity to assign a primary habitat to the species. The latter part has been initiated and quite comprehensively undertaken by the STECF (STECF 2023: Annex 1).

The conversion from categorical grades to a numeric score does lead to the situation where impact scores will be clustered around certain fixed values, relative to the changes in category and the proportion of the landed organism that is being sold or the proportion of the final product that consists of the seafood product in question.

The main source of uncertainty for the approach outlined in this section is the normalization factor. The use of total global catch by gear type as a proxy for the global seabed impact by gear type is a significant assumption. However, given effort data by gear type is not feasible to attain, and linking species catch rates to effort on different gears is even more untenable, this seems the only suitable proxy for attaining global, or even regional, levels of impact for normalisation calculations.

6.5 Development suggestions

The STECF report outlining the approach applied here, also critically evaluates its own methods, providing suggestions for development. Of these suggestions, the key developments would be:

1. Making the three-letter gear code reporting mandatory via amendments to the CMO regulation: It would increase the differentiation between different fishing activities and provide more information for the EF and ultimately support consumers' ability to discern between alternative products.
2. Implementing secondary, and potentially even tertiary, habitat associations for species: It could drastically change the perceived impact of a product's harvesting on the seabed. In the current scenario (as applied in the case studies above), the STECF experts have attributed a singular habitat to each species, based on databases of species traits, ecology and life-history. However, even the sources used to make these attributions often list more than one habitat as important for a given species. Furthermore, in the examples used in this report (hake in the Bay of Biscay and Argentine red shrimp from a coastal fishery) there were alternate habitats described as being actively fished by various gears in stock specific assessment reports (ICES, 2023; Landa et al., 2024). When considering the inclusion of these extra potential habitats, the STECF propose retaining the most sensitive habitat (i.e. the habitat that attributes the highest impact score). In our case studies this would change the outcome for all.
3. Validation of the method for calculating the normalisation factor: It would ensure that the proposed method and the resultant normalisation factors stand up to peer review and achieve general acceptance. This could be achieved in two different ways scientific publication, or review of this report in plenary of STECF.

7 On combining the non-LCA approaches to qualitatively convey the impact of fishing on both stocks and seabed

7.1 Background for reflection on suggesting an alternative integration of non-LCA methods

In the two non-LCA approaches described in previous sections, one for each of the impact indicators addressed in this report, a very similar method was used for quantifying the STECF qualitative categories. Furthermore, the same principles were the foundation for generating the normalisation factors for each of these STECF derived impact indicators. Because the non-LCA approaches calculate characterisation factors from categorical, semi-quantitative data, the expert group reflected upon the likelihood that these proposals would not be directly adopted by those responsible for delivering the EF. As a contingency, we provide here a description of an approach in which the standard STECF qualitative grades might be provided to the consumer via semi-quantitative “meta-data”, in the EF framework. This method simply describes guidelines for the inclusion of STECF grades on products and is presented in a situation where this grade is dependent on both indicators, but could easily be adopted for a single indicator, by ignoring the contribution of one or other of the indicators at the first step.

7.2 Application of method

Based on the outcome of the two STECF grading systems described in sections 4 and 6, one overall grade can be applied to the final product to represent the degree of environmental impact derived from the particular fishery.

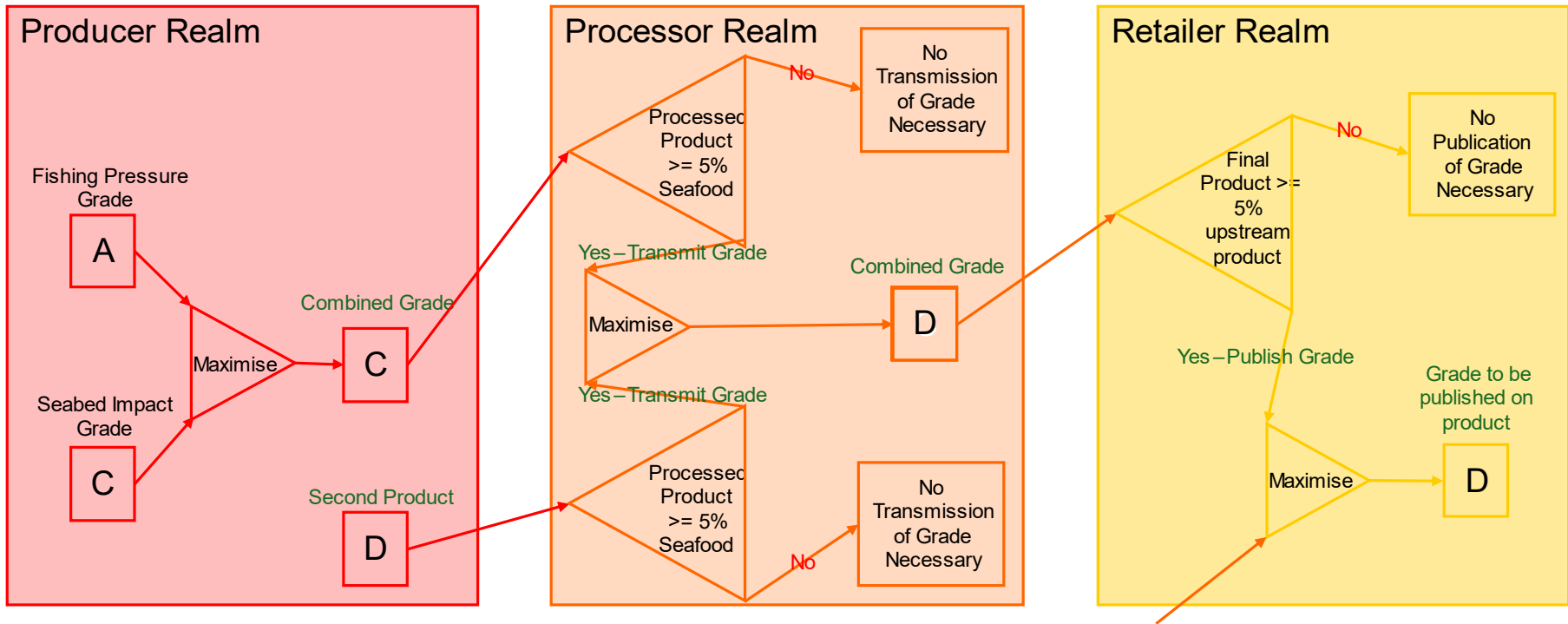
The basis of combining grades is to retain the grade indicating the highest impact (Figure 13). As such, for a raw seafood product with two different grades for biotic resource depletion and seabed disturbance, the worse of the two grades is applied to the raw product (see the “producer realm” box of Figure 13).

If during processing the raw product is mixed with other products, then one applies a threshold (5%), such that if the processed product contains a sum of 5% or more from all seafood derived products, then a seafood impact grade must be applied to the processed product (see the top or bottom half of the “processor realm” box in Figure 13). If during processing, more than one raw seafood product is mixed/combined, then the highest impact grade of all the seafood components are applied to the processed product (see the whole of the “processor realm” box from Figure 13). Such scenarios could include cases where raw product of the same species but from different fisheries (stocks or gears) are mixed/combined, or cases where different species are mixed/combined (e.g. surimi as imitation crab meat).

The “processor realm” box, from Figure 13, may apply to multiple steps in the value chain. In each step, the 5% threshold should apply to the immediately preceding product, raw or processed.

Finally, at the level of the retailer, if the final product contains 5% or more of the immediately preceding product, be that raw or processed, then the highest impact grade must be displayed on the product in some pre-determined graphical representation that supplements the EF score (see the “retailer realm” box from Figure 13).

Figure 13. Suggested decision tree for the inclusion of the STECF method seafood impact grading on consumer products.



Source: STECF (2024).

7.3 Experience and performance with case studies

In the below sections on case studies, the STECF Grades are taken from the same case studies in the earlier non-LCA report sections. This section takes these independent grades and combines them. Furthermore, this section also goes further to four theoretical consumer products, to demonstrate how the grading system would propagate through the value chain.

7.3.1 Hake from the Bay of Biscay, fished by Spanish fleets with demersal trawl

7.3.1.1 Raw product

Demersally-trawled hake from the Spanish fleet operating in the Bay of Biscay received a grade of B for the sustainability of the resource extraction and a grade of C for the impact on the seabed. Therefore, the live-weight product at the point of sale from the producer would be scored with the grade indicating the highest impact i.e. “C”.

7.3.1.2 Processed product to consumer product

For this hypothetical product, we selected frozen, boneless, skinless stakes. The production of this product involves filleting, skinning, boning, freezing and packaging. This reduces the mass of the raw product that is available to the end consumer (discarding of skin and skeletal remains).

The original raw product carried a grade of C. This product is not mixed and can be traced back to the fishery, and hence, the gear that it was fished with; therefore, no secondary product grade needs to be considered. The processing of this product does reduce the mass available to consumers, but as the grade is not dependent on the per unit mass of the raw product, this has no bearing on the grade as it moves through the processing chain. The final product contains only derivatives of the raw product. Hence, it exceeds the 5% threshold and must display the grade that is passed on with the processed product. In this case the final consumer product must display the grade C for seafood sustainability along with the PEF score.

7.3.2 Hake from the Bay of Biscay, fished by Spanish fleets with paired mid-water trawl

7.3.2.1 Raw product

Midwater or pelagically trawled hake from the Spanish fleet operating in the Bay of Biscay received a grade of B for the sustainability of the resource extraction and a grade of B for the impact on the seabed. Therefore, the live-weight product at the point of sale from the producer would be scored with the grades being in agreement, i.e. “B”.

7.3.2.2 Processed product to consumer product

For this hypothetical product, we selected fresh fish cakes. The production of this product involves fileting, skinning, and boning of the hake. Furthermore, it involves the combination of other species that are at the processor’s disposal, together with eggs and breadcrumbs as binding agents. For the sake of argument, the worst scored seafood product in such a fish cake might be the Argentine red shrimp, imported from the Argentine inshore fishery which makes up less than 5% of the final product.

The original raw hake product had a score of B. The final product contains more than 5% of seafood derived products. Irrespective of whether the combination of hake with other seafood products occurs at an off-site processor, or directly at the retail premises, because the product is over the 5% seafood threshold, it must carry the seafood sustainability letter grade, if it is subject to display the

EF score. Regardless of what other species/stocks/fishing methods are used to produce the final product and in what proportions, it must take on the grade of the upstream product that indicates the highest level of impact. In this theoretical example, it is the Argentine red shrimp. Thus, the final consumer product must display the grade C, even if most of the product consists of the mid-water trawled hake that attained a score of B.

7.3.3 Argentine red shrimp from the inshore Argentine fishery, fished with paired beam trawl

7.3.3.1 Raw product

Beam trawled Argentine red shrimp from the inshore Argentine fleet received a grade of C for the sustainability of the resource extraction and a grade of C for the impact on the seabed. Therefore, the live-weight product at the point of sale from the producer would be scored with the grades being in agreement, i.e. "C".

7.3.3.2 Processed product to consumer product

For the third hypothetical product, we selected shrimp flavoured instant noodles. In this product the bulk of the mass is the noodle component with less than 5% of the mass coming from the flavouring powder mixture that contains 50% dried shrimp powder, with this shrimp component coming from Argentine red shrimp from the inshore fishery of Argentina.

The original raw shrimp product had a score of C. During processing, the flavour powder mixture is made by an intermediary. The product they produce contains 50% of the raw shrimp product and no other seafood products. Therefore, this intermediary product, if imported independently to the EU, must contain in the PEF meta-data, a seafood sustainability grade of C. In the packaging and production of the final noodle product, within the EU, the resultant consumer product contains less than 5% of seafood products in total. Therefore, it is not necessary to display the seafood sustainability grade alongside the PEF score, on the consumer product. Thus, the final consumer product need not display the grade C but may choose to do so.

7.3.4 Overall performance summary

As this approach is closer to the original proposals made by STECF, in that it does not try to quantify impact rather indicate relative impacts, there is less structural uncertainty introduced via the selection of different sources used in the normalisation procedures of the other approaches. These grades were possible to derive for all six cases across the three raw seafood products that were selected as case studies (two grades per raw seafood product). The three different theoretical consumer products derived from these raw products exemplify how these grades might be propagated along value chains and the types of rules that may apply to their combination, retention and eventual publishing on consumer products.

The grades do not reflect any mass-specific or volumetric impact. Therefore, the combination of products leads to the retention of the grade indicating the highest impact. While this may over-penalise the use of some products, it may also discourage the abuse of threshold values to hide higher-impacting products among lower-impact products.

7.4 Fit for purpose?

The utility of this approach depends on two main factors: 1) the ability of the EF system to record and propagate meta-data (such as the letter grade); and, 2) the acceptance of the EF labelling

design by the wider LCA community, stakeholders and, ultimately, the companies implementing it, of the extra grade being displayed in conjunction with the overall PEF score.

Furthermore, it is beyond the scope of this report to consider how the processing and combination of products might be monitored and audited. The practicalities and feasibility of this need to be considered, as this is an additional workflow to that of the core PEF score.

7.5 Development suggestions

The threshold values are primarily placeholders to illustrate the decision flow for the retention and combination of letter grades. The 5% value should be considered in the context of other EU regulations on food labelling and aligned to common practice for similar contexts.

8 Comparisons between proposed CFs and across existing CFs for all four methods

Ultimately, most methods were able to produce indices (e.g., CFs) for all case studies that were selected *a priori* (Table 19 and Figure 14). However, for some geographical locations, namely seabed impact using LCA indicators, we were unable to produce indices for some of the case studies that were proposed. As discussed in Section 5, Argentine red shrimp was not computed due to the lack of sufficient data to calculate the necessary CFs for this particular fishery. Moreover, for the other two fisheries evaluated, only the non-normalised CF is shown, as the normalization proposed in Section 5 does not comply with the EC rules for computing normalization in EF. Details on non-normalised impacts for 16 EF indicators and proposed new fisheries indicators are available in Annex 2.

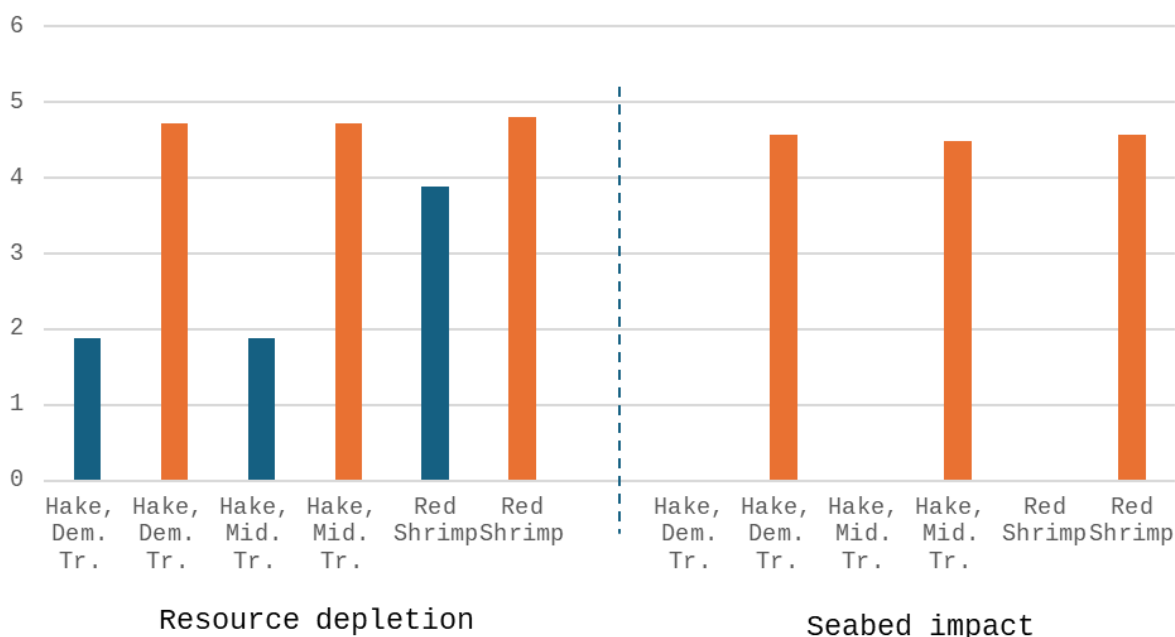
Table 19. Aggregated results of the various methods for calculating characterisation factors (CFs) for fish stock exploitation and seabed impact for use in the European EF framework.

Impact category	Case Study	Method Source	Non-normalised CF	Normalisation Factor computed	Normalised CF
Fisheries Exploitation	Bay of Biscay Hake, Demersal Trawl	Helias et al. (2023)	1.74E-14 PDF _{glo} .year/kg	4.33 E ⁹ Pt/(.PDF _{glo} .year)	7.54E-5 Pt/kg
	Bay of Biscay Hake, Midwater Trawl		1.74E-14 PDF _{glo} .year/kg		7.54E-5 Pt/kg
	Argentine Red Shrimp, Paired Beam Trawl		1.75E-12 PDF _{glo} .year/kg		7.57E-3 Pt/kg
	Bay of Biscay Hake, Demersal Trawl	"normalised" STECF (2024)	1 Uex /kg (Uex="unit of overexploitation")	1/1.95E+01= 5.12E-02 Pt/Uex	5.12E-02 Pt/kg
	Bay of Biscay Hake, Midwater Trawl		1 Uex /kg		5.12E-02 Pt/kg
	Argentine Red Shrimp, Paired Beam Trawl		1.2 Uex /kg		6.14E-02 Pt/kg
Seabed Impact	Bay of Biscay Hake, Demersal Trawl	Woods & Verones, 2019	5.64E-9 PDF*yr/kg	N/Ap	N/Ap
	Bay of Biscay Hake, Midwater Trawl		0	N/Ap	N/Ap
	Argentine Red Shrimp, Paired Beam Trawl		NA	N/Ap	N/Ap
	Bay of Biscay Hake, Demersal Trawl	"normalised" STECF (2023)	1.2 Usi/kg (Usi = "Unit of seabed impact")	3.35E+01 Pt/Usi	3.58E-02 Pt/kg
	Bay of Biscay Hake, Midwater Trawl		1 Usi/kg		2.99E-02 Pt/kg
	Argentine Red Shrimp, Paired Beam Trawl		1.2 /kg		3.58E-02

NA= not available; N/Ap= not applicable.

Source: Authors' own elaboration.

Figure 14. Log of normalized impact in μPt of 1 kg, live weight at landing, for the 3 case studies and the two impact types (left : resource depletion; right : seabed impact), and the two approaches .



Blue: LCA approach, orange : non-LCA approach. It should be noted that for seabed disturbance all the case studies do not have a normalization value. However, it is important to remark that in the case of hake, the absence of a normalization value is due to the lack of a standardized pathway to convert the environmental damage into a normalized value, whereas in the case of red shrimp the lack of a normalization obeys also to the lack of data to compute the damage.

Source: Authors' own elaboration.

8.1 Comparing Helias (2023) and normalised STECF (2024) approaches to fish stock exploitation

8.1.1 Absolute differences within cases, between methods

The two methods have been successfully applied on the contrasting case studies, demonstrating an appropriate technological readiness level, with the methods being operational. The resultant indicators derived from these two methods are both on vastly different scales from other impact categories and each other but also differ in their relative scoring of the different cases explored.

8.1.2 Relative differences within methods, between cases

The STECF normalised scores are narrowly distributed, with scales varying by a factor 4 between the lowest and highest scores for any product (from 0.10, grade E, to 0.025, grade A) On the other hand, the CF from Helias et al. (2023) can have some orders of magnitude difference from one stock to another; the two examples here returning a factor 100 between hake and Argentine red shrimp, which is however less than the examples shown in ADEME (2023), computed with the regional approach (differences in scales up to a factor 10^6).

Summarising from the discussions described in more details in the first report and in ADEME (2023), the scale differences arise from the relative pre-eminence of overfishing status vs stock size, and the importance of accounting for additivity in LCA. The Helias et al. (2023) is fully additive but the outcomes imply that (i) for two similarly depleted stocks, it is considered better to consume

1 kg fish of a larger (than a smaller) stock and (ii) depending on stock and status, it can be better to eat 1 kg of a big stock in a 'depleted' (overfishing) status than 1 kg of a smaller stock with a 'less depleted' (less overfishing) status. Conversely, the STECF approach will always favor the "no overfishing status", regardless of stock size. As such, MSY-based scoring is not strictly additive. However, additivity is implicitly embedded in the annual dynamic and self-regulation of the fisheries management system, where the amount of fish available to consumer is not freely driven by demand but is limited by the regulations such as TACs (Total Allowable Catches) and fishing quotas, and where too much consumption one year will be penalised by lower quotas the following year.

Although weighting is not addressed in this report, the authors discussed whether deciding on an appropriate weighting for the stock status indicator in relation with the 16 other EF indicators would be challenged or not by such differences in scales across products. It is noted that there are other impact categories in EF for which CF vary by several orders of magnitude (e.g. water scarcity, toxicity). Differences in scales will mean that some stocks will hardly have any impact for the category, whereas others would have a major one.

8.2 Comparing Woods & Verones (2019) and normalised STECF (2023) approaches to seabed impacts

The differences between the LCA (Woods & Verones, 2019) and the non-LCA (STECF 2023) methods are derived from their starting out point. The LCA method starts by considering the mechanisms of impact, the frequency of impact, the resilience of different organisms from various habitats, and the recovery times of these species. All these parameters require data from representative, real world observations. This data requirement is a heavy burden for a system such as the EF framework that is trying to be universally applicable, in this case, to all seafood products, and potentially, in the future, to other activities that can potentially generate seabed disturbance.

Conversely, the non-LCA method starts from the outlook that having such detailed data available ubiquitously is not achievable and instead works from the data that is currently available, or is road-mapped to become available, and builds indices from these data. With the lower data requirements, comes much lower resolution and qualitative indicator grades.

8.3 Integration with and comparison to other impact pathways

While comparison with other impact categories / pathways is outside the scope of this project, a single example was taken through to this stage for the purpose of providing context. Hake from the Bay of Biscay, analysed with the LCA based Helias (2023) approach, was selected as this individual example, to illustrate how the additional impact indicator can be taken into account, after normalisation, as a 17th indicator. The full process is described in the below subsections.

8.3.1 Life cycle inventories

In order to capture the impacts from the current EF method, we use datasets describing fishing activities in the ecoinvent 3.9 (Wernet, Bauer et al. 2016) and Agribalyse 3.1 databases. Those datasets describe, per kg fished, the quantity of energy used, and waste generated; they enable to inventory all resources to- and emissions from nature generated from fishing activity, apart from the fishing resources from nature.

8.3.1.1 Hake from the Bay of Biscay, fished by Spanish fleets with demersal trawl

The three datasets used as proxy for hake fishing in the Bay of Biscay are listed in Table 20.

Table 20. List of life cycle inventories used to assess impacts of fishing activity at landing.

Name	Database	Country	Type of fishing	Comment
Landed hake, fresh {ES} hake, capture by trawler and landing whole, fresh Cut-off, U	Ecoinvent 3.9	Spain	coastal	Mix of bottom and mid-water trawling according to documentation
Demersal fish, fresh {ES} hake, capture by trawler and landing whole, fresh Cut-off, U	Ecoinvent 3.9	Spain	Off-shore	Mix of bottom and mid-water trawling according to documentation
Saithe, NorthSea, Bottom Trawl, average, fresh, at landing/EU U	Agribalyse 3.1	France	unknown	Bottom Trawling

Source: Authors' own elaboration.

8.3.1.2 Hake from the Bay of Biscay, fished by Spanish fleets with paired mid-water trawl

There is no life cycle inventory dataset in the databases that cover this case study.

8.3.1.3 Argentine red shrimp from the inshore Argentine fishery, fished with paired beam trawl

Between all two most-used European database: ecoinvent 3.9, and Agribalyse 3.1., the only process for shrimp comes from aquaculture in China (*Fresh shrimps, China production {FR} U*). Aquaculture process was considered to poorly reflect impact of fishing, so it was disregarded. Instead, another inventory was selected to reflect resources and emissions similar to what could be expected for the Argentine beam trawl fishing process: *Great Scallop, BSBrieuc, Dredge, average, at landing {FR} U* from Agribalyse 3.1.

8.3.2 Method

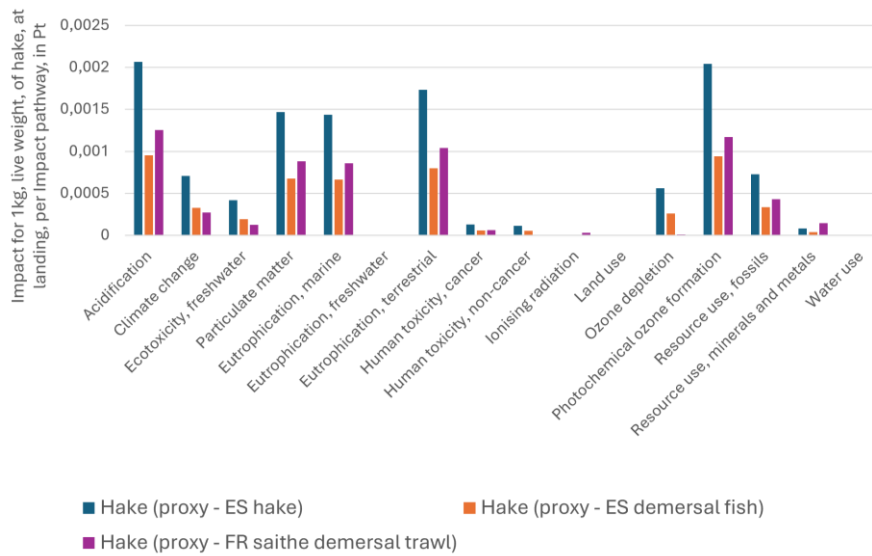
Normalized impact values from the 16 EFs categories are assessed with EF 3.1 using SIMAPRO 9.5.5. Impact values for the indicators studied in the present report were taken from Table 19. However, it should be noted that the Woods and Verones (2019) method was not accounted for as the normalisation factors for this method were not computed, as we do not have a full quantification of the inventories across the world.

8.3.3 Results

8.3.3.1 Hake from the Bay of Biscay, fished by Spanish fleets with demersal trawl

Normalized results for 1kg live weight hake, at landing, are displayed in Figure 15. Among the 16 EF indicators, impact is dominated by energy, mostly from diesel combustion, major contributor of: *Acidification; Particulate matter; Eutrophication, marine; Eutrophication, terrestrial; and Photochemical ozone formation*. Note that the three datasets are providing contrasting results, due to the quantity of energy consumed per kg hake.

Figure 15. Normalized impact of 1 kg, live weight, of hake, at landing, using EF 3.1 method.

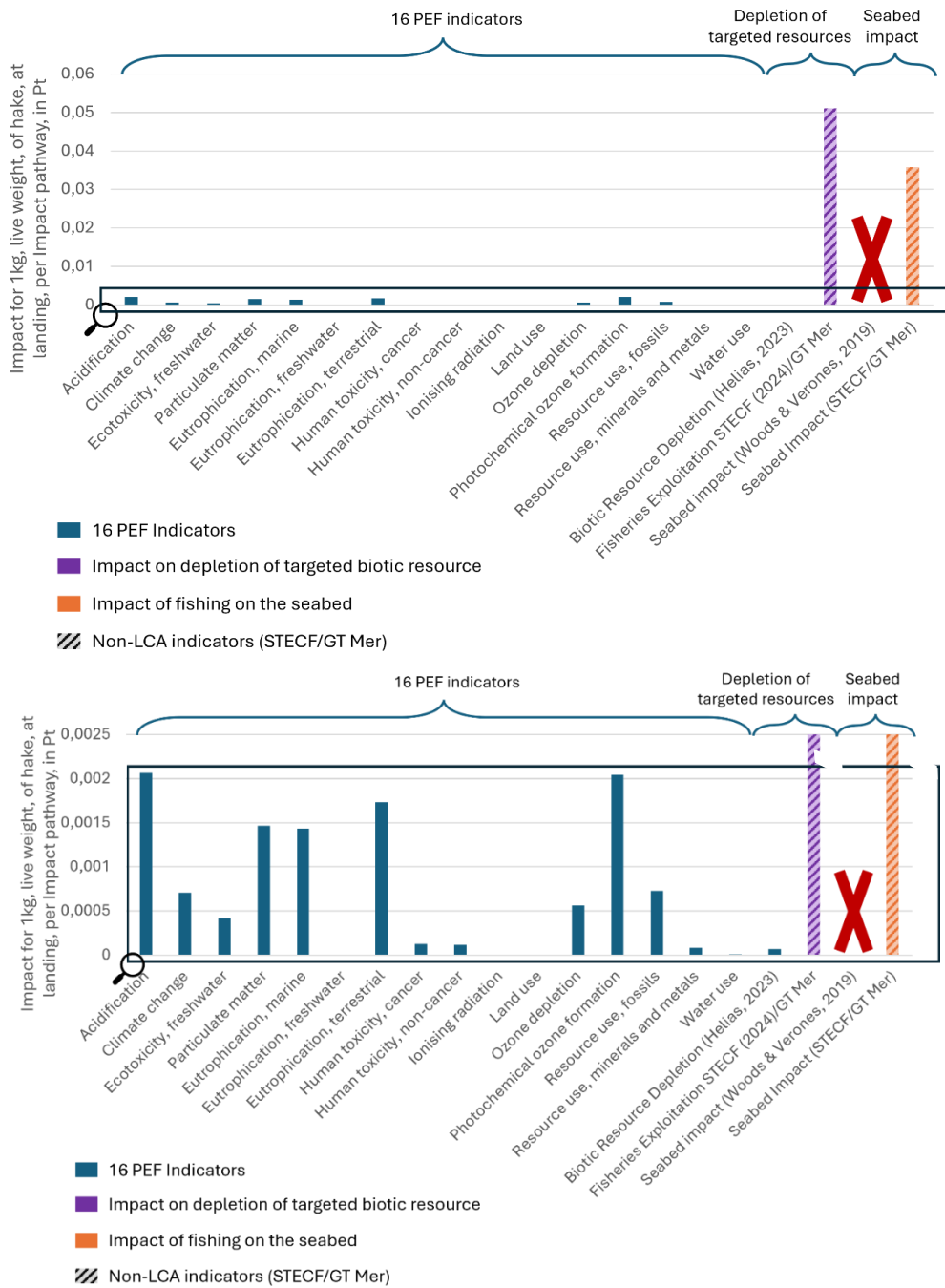


Colours correspond to the 3 proxy datasets used: blue - Landed hake, fresh {ES} hake, capture by trawler and landing whole, fresh | Cut-off, U (proxy – ES hake); orange - Demersal fish, fresh {ES} hake, capture by trawler and landing whole, fresh | Cut-off, U (proxy – ES demersal fish); purple - Saithe, North Sea, Bottom Trawl, average, fresh, at landing/EU U (FR saithe demersal trawl).

Source: Authors' own elaboration.

Normalized results for 1 kg live weight hake, at landing, are displayed in Figure 16, with the 16 EF indicators and the indicators studied in the present report. Results show that non-LCA indicators both give a very high impact compared to EF impact pathways, whereas LCA indicators both give a low impact compared to EF impact pathways. Biotic resource depletion (Helias et al., 2023) is at the same order of magnitude as EF indicators: Resource use, minerals and metals; Human toxicity, cancer; and Human toxicity, non-cancer.

Figure 16. Normalized impact of 1 kg of hake, at landing, for the 16 EF indicators, and for the 4 indicators studied in the present report.



Only one proxy dataset was used to calculate EF indicators: Landed hake, fresh {ES} hake, captured by (bottom) trawler and landing whole, fresh | Cut-off, U. For both seabed impact indicators (Woods & Verones (2019) and STECF/GT Mer); the case study “Hake from the Bay of Biscay, fished by Spanish fleets with demersal trawl” was selected. The red cross symbolizes that no result was obtained for indicator Seabed impact (Woods & Verones, 2019), due to absence of normalisation values. Bottom plot is a zoom of top plot, with indicators Fisheries Exploitation STECF (2024)/GT Mer and Seabed Impact (STECF/GT Mer) exceeding the upper limit of 0.0025 Pt for 1 kg of hake.

Source: Authors' own elaboration.

8.3.3.2 Hake from the Bay of Biscay, fished by Spanish fleets with paired mid-water trawl

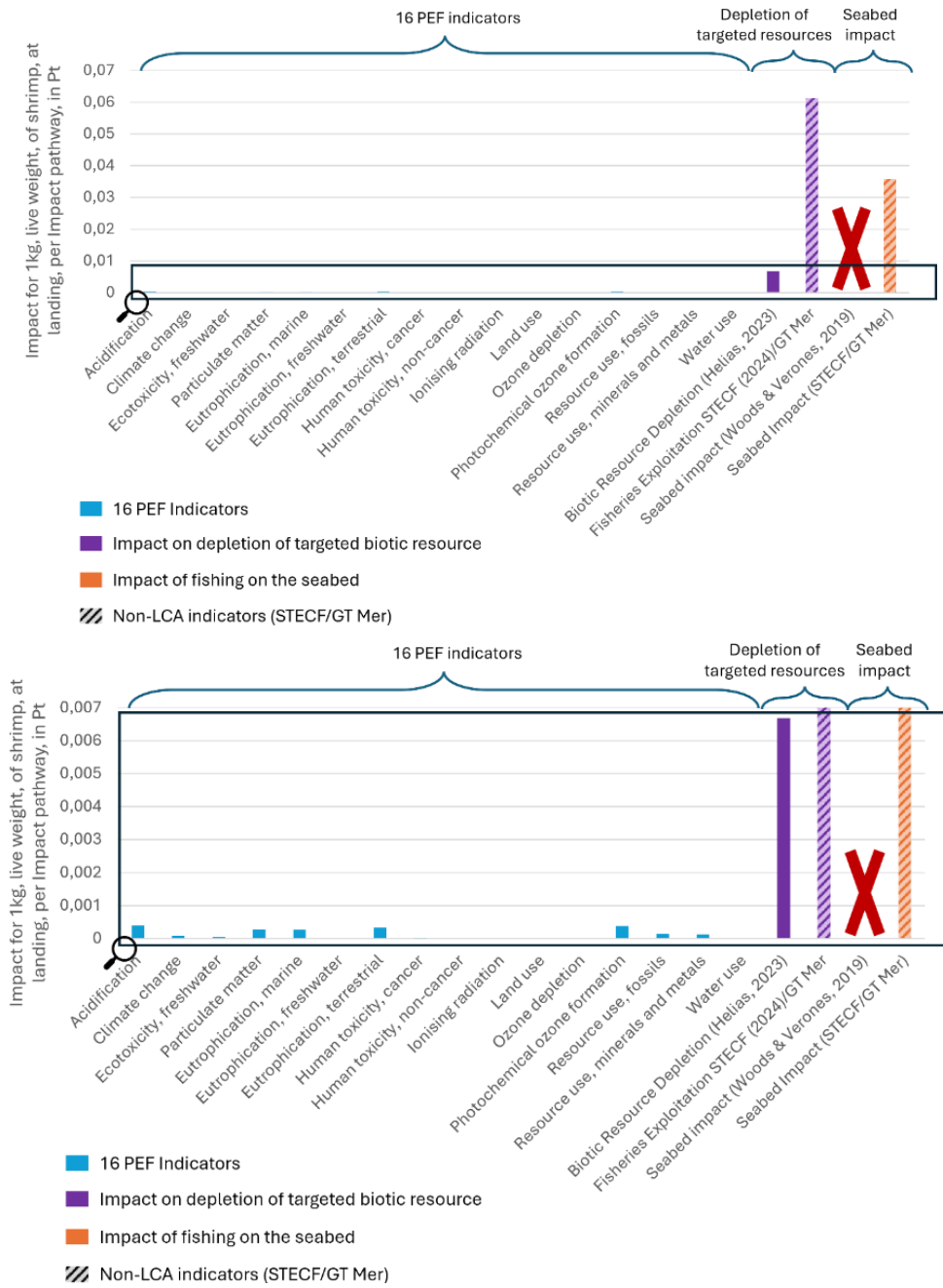
For the second hake study with a midwater gear, we could not find any dataset related to that fishing technique. However, mid-water trawls are known to consume less fuel than bottom trawls

(Parker and Tyedmers 2015), we would therefore expect that this fishing technique would yield to lower impacts on the 16 EF indicators compared to the one displayed in Figure 16.

8.3.3.3 Argentine red shrimp from the inshore Argentine fishery, fished with paired beam trawl

Normalized results for 1 kg, live weight, of shrimp, at landing, are displayed in Table 17 with the 16 EF indicators and the 4 indicators studied in the present report. Results show that non-LCA indicators both give a very high impact compared to EF impact pathways. Biotic resource depletion (Helias et al., 2023) also gives a very high impact. Seabed impact (Woods and Verones, 2019) is not yet computable for non-European fisheries.

Figure 17. Normalized impact for 1 kg of shrimp, at landing, for the 16 PEF indicators, and for the 4 indicators studied in the present report.



The red cross symbolizes that no result was obtained for indicator *Seabed impact (Woods & Verones, 2019)*. Bottom plot is a zoom of top plot, with indicators *Fisheries Exploitation STECF (2024)/GT Mer* and *Seabed Impact (STECF/GT Mer)* exceeding the upper limit of 0.01 Pt for 1 kg of shrimp.

Source: Authors' own elaboration.

9 Overall Recommendations

9.1 Fish stock exploitation – recommendation and justification

Fundamentally, the two methods explored to quantify the impact of stock exploitation on the stock itself are based on very different definitions of “impact”. This difference between definitions is an expression of two different philosophies; thus, a recommendation on which is better, could not only be grounded in scientific rigour or validity as it also depends on the values of those seeking to use the output of the methods.

More specifically, the LCA-rooted Helias et al. (2023) method, considers that any removals or exploitation of a fish stock constitute an impact. The Helias et al. (2023) approach was designed to be in line with other LCA indicators linked to the “ecosystem quality” area of protection; this method has been demonstrated in section 3, including a critique of limitations.

Conversely, the STECF (2023, 2024) based approach, derived from the field of fisheries science, considers that as long as the stock is not overfished, fishing removals are aligned with biomass renewal. A stock that is not overexploited implies that a sizeable number of adult fish are left in the sea to secure the basic biological processes of biomass growth and generation renewal (“recruitment”). As such, non-overfished stocks are more productive than overfished stocks, also ensuring more productive fisheries and higher yields for human consumption. This concept has been a foundation of fisheries management for decades. No overfishing (sensu MSY) remains the primary objective of European Common Fishery Policy and is the only explicit indicator in the Sustainable Development Goal for fisheries (SDG 14.4). The application of this STECF approach to inform the EF is documented in section 4, together with a discussion of its limitations.

In conclusion, because both of these approaches are evaluated to be sound (applied reasonably to our case studies and with available data to support operationalisation), the expert group did not conclude on a recommendation, but has strived to present foundations, outcomes and limitations in the most transparent way. In summary, the key messages regarding the resource depletion indicator are given in Box 1.

Box 1. Key messages regarding the resource depletion indicator.

Major challenges

- Coherence of various public policies (EF and Common Fisheries Policy)
- Ecological relevance and accounting for specificities of fishing (living resources, renewable, wild, shared)
- Additivity (direct vs. “intrinsic”)

Other challenges

- Stock status change over time (annual assessments), and there is a need to discuss periodic update of the scores
- Completeness vs. precision
- Dealing with uncertainty

Need for future work

- Expand case studies application to more diverse cases and further explore the methods’ operational capabilities.
- Continue scientific exchanges across communities, fine tune and peer-validate the normalised STECF proposal
- Since the two methods differ widely and assess two different aspects of fishing (biodiversity on the one hand, resources for human consumption on the other), it would be useful to consult for acceptance and meaningfulness of the targeted method.

Thus, suggestions are to pursue computation of various case studies over the whole value chain to put some concrete elements at hand and objectivize decisions, and that stakeholders (namely policy makers and fishery managers, fisheries and value chain representatives, environmental NGOs, and consumer organisations) are consulted as to which approach they deem most relevant/appropriate for use in the EF and ultimately consumer labelling.

9.2 Seabed impact – recommendation and justification

Despite comprehensive consideration of all available methods, the only suitable LCA based method for calculating CFs of seabed impacts remains currently underdeveloped and is thus not yet suitable for operationalisation in the EF at a worldwide scale. Therefore, our recommendation is to adopt the STECF approach for seabed impacts. While this approach is a quantification of a previously categorical indicator, its data reliance is limited to data that is either already collected and reported, or data held in extant, quality-controlled databases. It was easily applied to all three case studies selected and is broadly applicable to all EU derived and imported products.

This clear recommendation is relevant for the current situation. However, to future proof the EF framework and to prepare for improvements, EF developers and practitioners should pay particular attention to the comprehensive descriptions of data requirements (inventory requirements) described for the LCA approach, provided in section 5. Moreover, the STECF indicators for seabed impact only apply for fishing, and yet other economic activities impact the seabed, such as mining or energy production (e.g. offshore windpower). The Woods and Verones (2019) method is applicable to all types of seabed disturbance (e.g., seabed mining, coastal and open sea infrastructure...), not just fishing, although case studies for other activities are yet to be developed in the scientific literature. The method is based on complex relationships between sensitivity, disturbance magnitude and frequency, as well as recruitment/re-colonisation opportunities. These characteristics make it a comprehensive indicator of seabed impact, which can place fisheries in the same context as other human activities. Once this approach matures to a point where it is based predominantly on empirical data, available for most of the oceans, and returns results coherent with standard area-based approaches such as those used in the Marine Strategy Framework Directive (cf review in Deliverable 1), then it will be a potentially far superior indicator than STECF approach recommended here, allowing for finer description of impact than the broad brush approach associating a targeted species with an average potential impact regardless of the actual footprint of the fishery. However, the timeframe for that becoming a reality is unknown.

9.3 Qualitative scores

The use of qualitative scores (combined from various letter grades from the STECF (2023, 2024), as described in section 7, is provided as an alternative to the incorporation of quantitative scores into the EF.

For stock exploitation, there are two suitable methods that provide quantitative or semi-quantitative indicators for incorporation into the EF, and thus, we do not recommend a qualitative grading be used to reflect stock exploitation, independent of the other CFs. However, should there be issues with weighting whichever of these methods are selected via a political process, then the qualitative pathway described here remains a fall-back.

Conversely, for seabed impacts, there is only one feasible method (i.e., the STECF method), which converts qualitative grades to quantitative scores and “normalises” them using only a proxy for true global impact. Therefore, while we recommend the use of this semi-quantitative CF, it can be envisaged that this approach may be challenged or questioned by either LCA developers or

practitioners. Should this be the case, we propose the use of the STECF letter grading as complementary data with the propagation rules outlined in section 7, as the fallback method. However, we reiterate our primary recommendation is the semi-quantitative STECF approach detailed in section 6.

10 Conclusions

Following the comprehensive review of potential methodologies for incorporating fish stock exploitation and seabed impacts of fishing into the EF framework undertaken in as preliminary study and detailed in Annex 1, this report applies the most appropriate methods selected from the fields of LCA and from fisheries science to the impacts of fishing on exploited stocks and the seabed. Based on the experience of applying these methods to various case studies, this report investigates their operational feasibility and provides recommendations for which method(s) should be carried forward in the development of the EF (Table 21).

For fish stock exploitation, the methods from both the LCA and fisheries science fields are fit for purpose and operationally mature. Both methods were successfully applied across a range of predefined case studies. However, the main differentiating point between them is the fundamental difference in philosophy regarding what an impact consists of. In the LCA method, any extractions are an impact, considering as a reference state the ecosystem without any human intervention, while from the fisheries science field, a stable exploitation implies a stable impact, and the stock can in principle indefinitely sustain itself; overfishing is described at the point beyond which too much fishing implies too low levels of adult biomass left in the sea, and thus risks of impaired growth, reproduction (thus, renewal of generations, so called recruitment), and consequently lower long-term yields for human consumption. This fundamental difference limits direct objective comparisons, and, therefore, this report recommends that relevant stakeholders are introduced to these two fundamental differences of approach and asked to select which fits best with the intended application (i.e., the EF framework).

For seabed impacts, the method from the LCA field is not mature enough and not supported by globally available data to be used operationally in the EF. This is reflected in the inability to calculate CF values for some of the selected case studies (essentially any case study beyond European waters), although in the future it is expected for this method to be usable also to assess the seabed impact of other activities beyond fishing effort. Therefore, the modified, semi-quantitative STECF approach is recommended as an interim indicator in the EF. This recommended method, which utilises already reported, recorded or collated data, was successfully applied to the three case studies selected, and is applicable to all seafood products produced in or imported into the EU. Caution must be kept however, in the future, to harmonize the seabed impacts disturbances due to fishing with other types of seabed disturbances such as granulate extractions, seabed mining, and sea energy installations (e.g. windmills), that cannot be assessed with the STECF metric.

Table 21. Summary conclusions of the analysis.

	Originating from LCA	Originating from Fisheries Science
Impact of Depletion of the biotic resource (stock exploitation)	Directly applicable in EF (including normalisation) Not aligned with fishery policy and science	Possibly applicable in EF (peer validation / publication) Aligned with fishery policy and science
Impact of Fishing on the Seabed	Not applicable in EF	Possibly applicable in EF (peer validation / publication) Aligned with fishery policy and science

Source: Authors' own elaboration.

Finally, an approach to utilise only qualitative letter grades based on the STECF methods for both stock exploitation and seabed impact is described together with rules for combining and propagating these scores through value chains. This qualitative approach should only be considered as a fall-back option and is only the tertiary recommendation for stock exploitation and the secondary recommendation for seabed impact.

As a concluding word, the experts wish to highlight that this project represented, to their best knowledge, the first in-depth encounter between European LCA science and fisheries science communities, triggered by the EF requirements. It is obvious that norms, methods and concepts differ between these communities, and it is difficult and time-consuming to apprehend and get acquainted with one another technicalities and specificities. This report expressed the major progresses achieved but more collective thinking still necessary to achieve fully mature, robust and coherent methods for the inclusion of fisheries impacts in EF.

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List of abbreviations and definitions

AIS	Automatic Identification System
BMSY	Biomass at Maximum Sustainable Yield
CF	Characterisation Factor
CFP	Common Fisheries Policy
CMSY	Catch Maximum Sustainable Yield
CMO	Common Market Organisation
CTU	Comparative Toxic Unit
DG	Directorate-General
DOPA	Digital Observatory for Protected Areas
EEA	European Environment Agency
EF	Environmental Footprint
EMODnet	European Marine Observation and Data Network
EUNIS	European Nature Information System
EWG	Expert Working Group
FAO	Food and Agriculture Organization
FMSY	Fishing Mortality at Maximum Sustainable Yield
GEBCO	General Bathymetric Chart of the Oceans
ICES	International Council for the Exploration of the Sea
JRC	Joint Research Centre
kBq	Kilobecquerel
kg	Kilogram
LCA	Life Cycle Assessment
LCIA	Life Cycle Impact Assessment
m ³	Cubic meter

MJ	Megajoule
MSFD	Marine Strategy Framework Directive
MSY	Maximum Sustainable Yield
OEF	Organisation Environmental Footprint
PEF	Product Environmental Footprint
PDF	Potentially Disappeared Fraction
Pt	Points
RFMO	Regional Fisheries Management Organization
SDG	Sustainable Development Goal
STECF	Scientific, Technical and Economic Committee for Fisheries
UN	United Nations
UNDESA	United Nations Department of Economic and Social Affairs
VMS	Vessel Monitoring System

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Annexes

Annex 1. Baseline analysis

This work was led by Ian Vázquez-Rowe and prepared together with the other three experts on the matter– Elliot Brown, Anne-Claire Asselin, and Clara Ulrich.

1. Background of the study

Back in 2011, the Roadmap to a Resource Efficient Europe (EC, 2011) developed a set of actions regarding sustainable consumption and production to establish a common methodological approach to enable Member States and the private sector to assess, display and benchmark the environmental performance of products, services and companies based on a comprehensive assessment of environmental impacts over the life cycle ('environmental footprint'). In addition, this roadmap was focused on ensuring a better understanding of consumer behaviour and provide better information on the environmental footprints of products, including preventing misleading claims and refining eco-labelling schemes.

The Product Environmental Footprint and Organisation Environmental Footprint (also named, Environmental Footprint - EF) methods (EC, 2013⁷; EC, 2021⁸) were developed as the first implementation step. Since 2010 JRC has undertaken to further develop and implement them and draw lessons from pilot and transition phases and case studies, especially as related to the development/review of impact assessment methods, and tackling underlying data needs, to respond to the policy needs indicated on various policies as:

The COMMUNICATION on Building the Single Market for Green Products (COM(2013) 196 final)⁹, The Circular Economy Action Plan¹⁰, The farm to fork strategy¹¹, The sustainable carbon cycles¹² The proposed Ecodesign for Sustainable Products Regulation¹³, The proposal for the Green Claims Directive¹⁴, The Taxonomy Regulation¹⁵,

⁷ C(2013) 179/EU COMMISSION RECOMMENDATION of 9 April 2013 on the use of common methods to measure and communicate the life cycle environmental performance of products and organisations Text with EEA relevance, <https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32013H0179>

⁸ C(2021) 9332 final COMMISSION RECOMMENDATION on the use of the Environmental Footprint methods to measure and communicate the life cycle environmental performance of products and organisations [https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=PI_COM:C\(2021\)9332&qid=1689667489548](https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=PI_COM:C(2021)9332&qid=1689667489548)

⁹ COM(2013) 196 final. COMMUNICATION FROM THE COMMISSION TO THE EUROPEAN PARLIAMENT AND THE COUNCIL Building the Single Market for Green Products Facilitating better information on the environmental performance of products and organisations. <https://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=COM:2013:0196:FIN:EN:PDF>

¹⁰ COM(2020) 98 final COMMUNICATION FROM THE COMMISSION TO THE EUROPEAN PARLIAMENT, THE COUNCIL, THE EUROPEAN ECONOMIC AND SOCIAL COMMITTEE AND THE COMMITTEE OF THE REGIONS A new Circular Economy Action Plan For a cleaner and more competitive Europe

¹¹ COM(2020) 381 final COMMUNICATION FROM THE COMMISSION TO THE EUROPEAN PARLIAMENT, THE COUNCIL, THE EUROPEAN ECONOMIC AND SOCIAL COMMITTEE AND THE COMMITTEE OF THE REGIONS A Farm to Fork Strategy for a fair, healthy and environmentally-friendly food system.

¹² COMMUNICATION FROM THE COMMISSION TO THE EUROPEAN PARLIAMENT AND THE COUNCIL Sustainable Carbon Cycles COM(2021) 800 final; {SWD(2021) 450 final} - {SWD(2021) 451 final}

¹³ Proposal for a REGULATION OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL establishing a framework for setting eco-design requirements for sustainable products and repealing Directive 2009/125/EC <https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:52022PC0142&qid=1689668067347>

¹⁴ COM/2023/166 final Proposal for a DIRECTIVE OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL on substantiation and communication of explicit environmental claims (Green Claims Directive). <https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:52023PC0166>

¹⁵ C(2023) 3851 final. Commission Delegated Regulation EU of 26.6.23. https://finance.ec.europa.eu/system/files/2023-06/taxonomy-regulation-delegated-act-2022-environmental_en_0.pdf

The sustainable Food System legislative framework¹⁶.

Regarding fishing activities, resource sustainability is crucial when assessing their environmental footprint. However, this is currently not reflected in the 16 EF impact categories³. Recital 25 of the proposal for a Directive on substantiation and communication of explicit environmental claims states that an economic actor making an explicit environmental claim about a fishery product should have an obligation of diligence to find evidence substantiating the sustainability of the targeted fish stock⁹. The recital refers to stock assessments by the International Council for the Exploration of the Sea (ICES) and similar stock assessment bodies that can be used for that purpose. Therefore, an assessment of the targeted stock(s) is a mandatory and critical requirement that needs to be reflected in any EF study on marine fish products. To assess the impacts on fish stock, key elements, such as the ones listed below, should be carefully analysed:

- Biomass and fishing mortality assessments across a range of scientific bodies and fisheries management organisations, including the calculation of relevant averages, including aspects of self-regeneration vs. depletion (especially due to catching for human consumption);
- Catch/advice and effort/advice data where quantitative biomass and fishing mortality assessments are not available.
- Consider the regional/global IUCN classification and sensitivity indices of certain species, where none of the above data is available.

To address the identified need of better capturing stock sustainability in fishing activities, and the need to update the current Guide for EF compliant data sets - which should be updated within early 2024 to improve and expand current Environmental Footprint databases¹⁷ - the JRC runs research and coordinating activities covering different tasks to further improve the EF methods. This includes updating some of the recommended Life Cycle Impact Assessment methods, to better capture impacts due to fishing (including impacts due to overfishing, fish stock alteration and disturbance of the seabed). These research topics require specific scientific expertise in the relevant fields, for which JRC requested external expertise assistance. The recommendations provided by the STECF¹⁸ should be considered when drafting the recommendations to evaluate the impacts due to fishing activities. Results from this research will be considered to shape the updates of the Guide for EF compliant data sets.

Objectives of the study presented in the annex

In this context, the JRC has gathered a pool of collaborators to engage in a wide array of activities, including, yet not limited to, literature reviews, research work and contribution to ongoing JRC studies. The overall purpose of this pool is to further improve the EF methods by addressing, from a life cycle assessment perspective, when meaningful and appropriate, the assessment of impacts due to fishing activities. In this context, the LCA methodology has experienced a slow but gradual upgrade in terms of the environmental impact categories that are used linked to marine resources and ocean conservation. Environmental aspects such as marine eutrophication, marine eco-toxicity, or ocean acidification, for instance, have been available in the LCA literature prior to the

¹⁶ Proposal for a legislative framework for sustainable food systems (FSFS) https://food.ec.europa.eu/horizontal-topics/farm-fork-strategy/legislative-framework_en

¹⁷ Simone Fazio, Luca Zampori, An De Schryver, Oliver Kusche, Lionel Thellier, Edward Diaconu. Guide for EF compliant data sets: Version 2.0, EUR 30175 EN, Publications Office of the European Union, Luxembourg, 2020, ISBN 978-92-76-17951-1, doi:10.2760/537292, JRC120340.

¹⁸ STECF - *Scientific, Technical and Economic Committee for Fisheries*

development of the EF method, although currently the EF method only considers marine eutrophication within its framework. In contrast, up to very recently, a set of marine-related environmental aspects, such as targeted resource use, anthropogenic impact on the seafloor, among others, have not been included in the major multi-category LCIA methods that are available in most computational software, such as ReCiPe (Huijbregts et al., 2017), CML (Leiden University, 2024) or the International Reference Life Cycle Data System – ILCD (ILCD, 2011). Efforts to develop and improve indicators to measure the latter impacts in an LCIA framework have multiplied in recent years, and some of these have become more widely used in the LCA community.

Notably, impacts on the targeted biotic resource or seabed result from one single activity in the product life cycle and, therefore, an end-to-end analysis is not as essential as for other impacts (e.g., climate change). Consequently, improvement of the EF methods could be achieved either by updating the recommended Life Cycle Impact Assessment (LCIA) methods or including other additional non-LCA-based information or indexes that are appropriate to capture impacts on targeted stocks and impacts of fishing activities on the seabed.

To reach the purpose, the objective of this pool is to: a) identify and assess Life Cycle Impact Assessment (LCIA) methods and other non-LCA based scientific methods to reflect the depletion of targeted biotic resources and the alteration/disturbance of the seabed due to fishing activities. This research, thereafter, links the LCIA assessment to the analysis of the Life Cycle Inventory (LCI) data and information necessary to assess these impacts and methods; and proposes a preliminary approach for the integration of these methods and indicators within the product life cycle. ⁽¹⁹⁾.

The pool of experts is composed of 4 experts with different backgrounds and institutions, and their tasks have included:

1. To identify LCA-based and non LCA-based methods that can be relevant to quantify the impacts of fisheries activities on the targeted biotic resource and the seabed, as derived from the technical and scientific literature.
2. To develop an assessment framework for the evaluation of the mapped methods for their purpose of capturing the above impacts.
3. To evaluate the mapped methods by using the proposed criteria and provide recommendations on the most appropriate method(s).

This report represents Deliverable D1, a report containing recommendations on the most scientifically robust LCA and non-LCA method(s)/indicators to assess the impacts of fishing activities, including methodological challenges and opportunities.

A comment on the type of fishing impacts investigated here

The present study was commissioned to work on two key sources of environmental impact of fishing activities, i.e. the impact of fishing on the targeted biotic resource, and the impact on the seabed. However, it is important to keep in mind that there are several other types of impact, which are not dealt with here in the present study.

First, the standard EF method currently uses LCA to analyse 16 indicators of environmental impact regarding resource use, GHG, pollution etc. The LCA community has identified a major gap in assessing impacts from (marine) macro- and microplastic pollution, including from fishing gears disposed of to the sea (Woods et al., 2021).

Second, in fisheries science, describing the environmental impact of fishing is more focused on the diversity of types of impacts on biodiversity, and there are many other important aspects beyond

¹⁹ This may include the need to capture, for example, the impacts on fish stock depletion or biodiversity due to the production of feed used for other animal farming.

the two addressed here. For example, STECF EWG 20-05 and Danto et al. (2022) have defined other types of impact than those two investigated here:

- Impact on Endangered, Threatened or Protected (ETP) or otherwise sensitive species.
- Unwanted landings and discards.
- Impact on marine food webs and growth
- Carbon footprint.
- Waste and pollution.

Of these types, carbon footprinting is indeed already included in most LCA studies, including the EF method, and efforts to include the effects of marine plastic on ecosystem quality and human health are already being developed within the frame of the MarILCA project (Boulay et al., 2021). However, the other aspects, and especially the concern on bycatch of protected and sensitive species, should not be omitted and must be investigated in future iterations of the environmental scoring of fish products.

Furthermore, the authors underline that the present study is limited to marine fisheries, and do not cover neither freshwater fisheries nor aquaculture activities. Nevertheless, it should be noted that impacts on biodiversity also exist related to terrestrial ecosystems, for which environmental impact categories are also being currently developed. In fact, it is plausible to assume that in the future, cross-compartment biodiversity indicators will be developed in which biotic or abiotic environmental damages will be measured jointly without the need of discriminating between marine, aquatic or terrestrial compartments.

2. Methodology

Sources of information

Sections 2.1.1 and 2.1.2, below, describe the sources of information for LCA and non-LCA methods respectively. For studies comparing both approaches, the authors report on a relevant investigation conducted by the French Agency for Ecological Transition (ADEME) over 2022-2023, hereby referred to as “GT Mer” (Gaillet et al., 2023). This study compares the results obtained by LCA and non-LCA methods for measuring depletion of targeted resources.

For LCA methods

LCA methods refer to impact categories that have been developed following a life cycle impact assessment (LCIA) logic, in which a cause-effect chain is identified from the anthropogenic interventions in nature to the potential damage that can be created by the former. In the frame of this report, however, LCA methods are linked to the marine-related indicators, namely depletion of the targeted biotic resources and seabed disruption, that consider an LCIA logic as described above.

The main sources of information are studies available in the international scientific literature linked to the development of these marine-related impact categories. However, it should be noted that two different periods of time have been identified in this literature. On the one hand, studies published prior to 2015 in which the cause-effect chains proposed were not mature and, therefore, these indicators were not aggregated to the mainstream LCIA multi-indicator methods. On the other hand, studies that were developed in 2015 and after, mainly built as improvements of existing indicators, in which the level of sophistication, robustness and scientific rigour becomes closer to that of other environmental impact pathways. In this sense, it should be noted that it is an objective of the Global Life Cycle Assessment community to include some of these impact categories within its scope through the United-Nations-hosted, named GLAM initiative (LCI, 2023).

For non LCA methods

The wording “non-LCA” methods refer to the whole body of literature and work coming from “traditional” fisheries and marine ecology science. Analysing and quantifying the impact of fishing has been the core objective of fisheries science for decades, and that body of literature is huge. Considerations on seabed impacts are though more recent than considerations on the depletion of targeted resources, so a systematic review of primary literature on indices for the seabed impacts of fishing was undertaken. The review utilised a PRISMA-like approach and the search terms, exclusion criteria and data extraction schema are provided in Section 4.

Given the importance of these topics for fisheries management, international organisations play a forefront role in piloting and organising the development of operational methods, testing, and implementing them at large scale, to ensure a full and general applicability across all wild capture fishery products. As such, the review of non-LCA methods mainly builds on published reports from

ICES – International Council for the Exploration of the Sea

STECF – Scientific, Technical and Economic Committee for Fisheries from the European Union

FAO – Food and Agricultural Organisation

MSC – Marine Stewardship Council

Regarding non-LCA methods, it is highly important to clarify that a key aspect of the review was to consider the ability to measure impact as intended in the EF thinking, i.e., at the scale of a food product (i.e. a product with wild seafood sold to consumers on the EU market, either from domestic

fleets or imported), and not at the scale of a fish population, a fishing fleet and/or an area, which are the most common scales of analysis for methods coming from fisheries science and marine ecology. This specific angle of view proved thus challenging for non-LCA methods and was preeminent in their selection. Indeed, we draw attention to other approaches currently gaining attention, in particular in France, about global sustainability assessments at the scale of the production operator, i.e., the vessel/the fishing fleet, and not at the scale of the catch and consumers' products. These new approaches aim to take into account various dimensions of ecological and economic performance of fisheries into a limited set of synthetic indicators (e.g. Dewals and Gascuel, 2020, Danto et al., 2022, Ye and Link, 2023, Quemper et al., 2024). Such approaches have not been included in the present review, but they represent interesting developments within non-LCA methods for proposing alternative holistic and operational indicators for impact assessment.

Criteria analysis

Considering the overall purpose of assessing the environmental performance of seafood products available on the European Market, the methods are described in chapters 3 (targeted resources depletion) and 4 (seabed impact), respectively, highlighting the following operational information:

- Generality of the method, such that it can apply to all products, both domestic and imported.
- Ability to cover data-poor situations and missing stock assessments.
- Type, availability, and quality of input data.
- Type and format of outcome (scoring, value, etc.).
- In Section 5, a more detailed evaluation of the most pertinent methods is proposed.

Following the European Commission (2011) recommendations, a three-stage procedure to assess the life cycle impact assessment indicators has been proposed to systematically determine the most scientifically solid models quantifying the impacts of resource depletion and seabed impact on the marine environment. The first stage, criteria definition, required the identification of crucial standards the impact assessment would need to have to better reflect the marine environmental burden. In the second stage, analysis of indicators, the models were subjected to evaluation in each criterion according to levels of compliance. Finally, in the recommendation stage, we retain those indicators, LCA and non-LCA, that we consider the most scientifically robust and operationally feasible, pointing out their main benefits and drawbacks. There are several ways of evaluating the suitability of the impact assessment methods. This report is focused on criteria from scientific and stakeholder natures.

The scientific criteria include:

Completeness of scope. This criterion refers to the mode the impact pathway covers an information gap to calculate the full cause-effect pathways of environmental impact from intervention to actual damage (i.e., midpoint or endpoint), the mode the elementary flows are bound to the cause-effect chain, and the ecosystem coverage of the model.

Environmental relevance. This criterion refers to whether the method makes sense from an ecological / environmental point of view, i.e. whether it captures key features of population and ecosystem dynamics, applies to relevant taxonomic groups and whether the models are applied or tested in case studies at an adequate scale.

Scientific robustness and certainty. This criterion represents how the model includes up-to-date and validated knowledge, data with potential to be improved and further developed, verification of indicators using monitoring data, and uncertainty and sensitivity analysis.

Documentation, transparency, and reproducibility. This criterion denotes the accessibility of the model information, the specification of choices made for parameters' value, the availability of input data and characterization factors, and the ease for stakeholders and third parties to understand, reproduce, discuss and eventually contribute to these outputs' values.

Applicability. This criterion shows if the input data for inventory flows can be directly gathered from relevant actors (e.g., producing industry). It also evaluates the implementation of the model in the market relevant LCA software tools and the ease to conform with ILCD nomenclature.

Temporal representativeness, resolution, and time horizon. This criterion assesses the temporal resolution of the input and output factors, the temporal representativeness (e.g., outdated information, information not older than 10 years, 5 years, etc.), and the definition of time horizon in the characterization model.

Spatial resolution. This criterion rates the geographic area considered and transferability, as well as the spatial resolution of input and output factors.

Compatibility with Environmental Footprint (EF) method. This criterion judges the compatibility of the model with the analysis of the product and organisation life cycles, including comparative analysis. This criterion also evaluates the conformity of the model with EF data nomenclature, whether the spatial resolution of the model matches with EF method, normalization processes and factors availability, and ratio of EF flows covered.

The stakeholder criteria include: Stakeholder acceptance and suitability for communication in a business and policy contexts. This criterion evaluates the understanding of indicators and the relying principles, body supporting the model principles and usability of output factors in practitioners' context.

The general criteria, which have been defined above, as well as the subcriteria that can be considered are summarized in Table 1a.

Table 1a. Scientific and stakeholder general criteria and subcriteria used in the current report to evaluate the LCA and non-LCA indicators retained on depletion of targeted biotic resources and seabed impact for detailed evaluation.

Criteria	Subcriteria	Description
Introduction		Timeframe, discounting, etc.
		Marginal (M) or Average (A) defined, if not described (ND)
		Total number of individual substances covered by specific provided characterisation factors
Completeness of scope		Cause-effect chain completeness
		The impact indicator covers the majority of impact mechanisms and relevant elementary flows for the AoP Natural Environment - Ecosystem coverage
		The impact indicator covers the majority of impact mechanisms and relevant elementary flows for the AoP Natural Resources

Criteria	Subcriteria	Description	
		The midpoint indicator is chosen in a way that all LCI are appropriately aggregated as early as possible in the cause effect chain	
		The characterisation model is adaptable to spatial and temporal explicit evaluation	
		Global geographical validity preferable, separate validity for Europe beneficial (geographic coverage of output factors)	
		The method is compatible with, or developed specifically for, the comparative assessment scope of LCA (e.g. factors do not include security factors/precautionary principle)	
		When empirical data is used, double counting is avoided	
Environmental relevance		Unit of measure of input	
		Unit of measure of output	
		Underlying approach (Intrinsic ecosystem functioning/instrumental value for humans as defined in GLAM)	
		All critical parts of the environmental mechanism describing the cause-effect chain are included with acceptable quality given current scientific understanding --> provide a list of specific criteria per impact category	
Scientific robustness & certainty	Scientific robustness	The critical part of the model including the parameters used in the model have been peer reviewed (journal, panel, book, etc.)	
		The model reflects the latest knowledge for the cause-effect chain (the critical links are covered) --> provide a list of specific criteria for each impact category	
		The model including the underlying data have a good potential for being consistently improved and further developed including regarding geographical/emission situation and temporal differentiation	
	Certainty	Indicators can be confirmed and verified against monitoring data, if available	
		Uncertainty estimates of the indicators are provided, justified, and reported in statistical terms	
		Scenario and model uncertainty as well as substance data and parameter uncertainty are taken into account	
		Sensitivity analysis included	
		The category indicator and characterisation models are science based	
	Documentation, transparency & reproducibility		The model documentation is published and accessible (incl. description of the mechanism, the model, temporal and spatial scale, etc.)? This must support the development of new, consistent factors by third parties.
			The set of characterisation factors/models is published and accessible
		The input data are published and accessible	
		The characterisation model is published and accessible	
		Ability for third parties to freely generate additional, consistent factors and to further develop models e.g. incorporating further geographical/emission situation, temporal and speciation differentiation	

Criteria	Subcriteria	Description
		Value choices are explicitly stated
Applicability		Ease to update to conform e.g. with the ILCD nomenclature and units
		The characterisation factors are straightforward to apply for general LCA practitioners and in most market relevant LCA software tools
		Life cycle inventory figures for the distinguished emission compartments or resource types can be directly made available by the relevant actor such as the producing industry
Temporal representativeness, and resolution, time horizon		Temporal resolution of the input
		Frequency of updates from modeling body
		Modeling body in charge of updates
		Temporal representativeness of the underlying biophysical models
		Temporal resolution of the output factors
		Definition and relevance of the output time horizon
Spatial resolution		Spatial representativeness of the underlying biophysical models
		Spatial resolution of the input used in the model
		Spatial resolution of output factors
Stakeholder acceptance criteria		The indicator is easily understood and interpretable
		There is an authoritative body behind the general model principles like the IPCC model (consensus/international endorsement)
		The principles of the model are easily understood by non-LCIA experts
		The covered elementary flows and impact models do not inappropriately favor or disfavor specific industries, processes, or products
		Compatibility with LCA and EF methodology
		Conform with EF nomenclature and units or adaptability
		Spatial resolution of the model in relation to the spatial resolution of EF elementary flows
		Is data for normalization accessible?
		Coverage of elementary flows
Output factors		Usability of factors for practitioners
		Tests of method already conducted

Source: Authors' own elaboration.

3. LCA and non-LCA methods for quantifying the depletion of targeted biotic resources

Introduction: Concepts, methods, and sustainability metrics used in fisheries management

It should be noted that in this report we have used the wording “depletion of targeted biotic resources”, which from this moment onwards we simplify to “depletion of targeted resources”, understanding that it is not the objective of this study to deal with marine abiotic resources and their potential depletion. Having said this, it is important to highlight that other wordings in different scientific communities – such as “biotic resource use”, “depletion of natural resources” or “sustainable fishing”, may exist and can be used interchangeably with the wording used here.

Different approaches to “sustainable fishing”

The LCA and non-LCA approaches presented here represent two contrasted conceptual approaches for considering ecosystem values, which imply major differences in how they assess impact on living resources (UNEP SETAC GLAM 1, 2017):

The “instrumental approach” (as labelled in GLAM 1) is based on the concept of ecosystem services describing how living resources are useful for humans. From this standpoint, the ocean and seas are provisioning a resource (fish) to humans and the aim of reducing environmental impacts is to guarantee a Maximum Sustainable Yield (MSY). It is used in fisheries management worldwide and through the concepts of F_{MSY} , B_{msy} and Kobe diagrams (see explanations below). This approach considers that if the MSY criteria are satisfied, the stock is in good environmental status and can sustain the impact of fishing without deterioration²⁰

The “intrinsic approach” (as labelled in GLAM 1) is that resources and ecosystems have a value for themselves, and that any disturbance from the “human-free” state of nature has an impact. It has historically been used in LCIA biodiversity assessment through the Area of Protection “ecosystem quality” with a unit expressed in “PDF” (Potentially Disappeared Fraction of species). Any removal of fish by humans from the ocean is considered impactful, even within sustainable fisheries that comply with MSY criteria.

Between those two approaches, there is no “good” or “bad” one, as they are assessing different aspects with different foundational concepts: in the first case, the continuity of “ecosystem services” from a resource standpoint; and, in the second case, the changes from a “human-free” ecosystem quality. It is however important to clearly distinguish the two when evaluating the methods.

This conceptual difference appears though as a major challenge for finding a coherent method satisfying the two approaches for the EF framework. In fact, it could be the case that a decision must be made and acknowledged when selecting the method for depletion of targeted resources. This decision appears more important for the issue of depletion status of targeted resources than for the issue of seabed impact or for the other types of impacts described in section 1.3. The reason of this is that considering targeted resources from either the point of view of the production or from the point of view of the impact affects the shape of the relationship between the indicator and the level of human activity. The production yield curve displays a typical bell-shaped form with a maximum found at a non-zero level of human activity (red line on Figure 1a), whereas an impact curve is typically uni-directional (more human activity means more impact – blue line on Figure 1a;

²⁰ Although MSY is the worldwide reference level for sustainable fishing, some discussions exist that considering a lower reference exploitation level, for example the one corresponding to maximum economic yield F_{MEY} , would be more appropriate for the good environmental and economic status of fisheries.

which would also be the case for seabed impact). In contrast, when assessing seabed impact or other impacts, the disturbance is inherently based on an “intrinsic approach” in which the impact is measured on a “human free” state of nature.

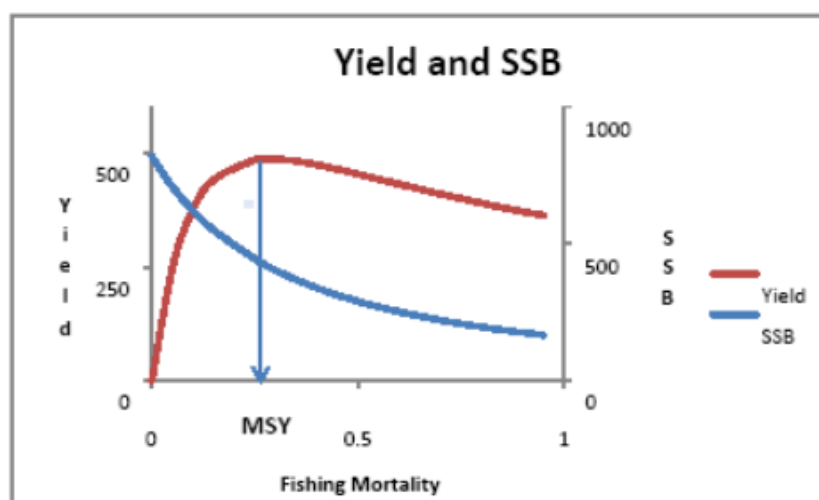
Overfishing vs overfished²¹

The contract refers to various wording such as “overfishing”, “fish stock depletion”, “biomass and fishing mortality assessments”, but these different words refer to different key concepts. For instance, “overfishing” and “overfished” are two different occurrences measured by two different units. In brief, “overfished” refers to the state of the stock, while “overfishing” refers to the pressure exerted on it, in relation to the worldwide reference concept, called Maximum Sustainable Yield (MSY), “the highest theoretical equilibrium yield that can be continuously taken on average from a stock under existing average environmental conditions without significantly affecting the reproduction process” (EU, 2013):

Overfished refers to the biomass (B) of a population, or stock, of fish. This is the number of fish in the water. There is some amount of biomass, B, that will produce MSY—this is B_{MSY} . If the biomass of fish in the water is well below B_{MSY} , the stock is overfished, or depleted. If the number of fish in the water is more than would produce MSY, it is underfished. The ratio of B/B_{MSY} is commonly used; however, the way to calculate it varies depending on the governing body, as discussed below.

Overfishing refers to fishing mortality (F), or the rate of fish killed by catching them (approx. as the proportion of fish caught). There is an ideal proportion of fish to catch that will produce MSY—this is called F_{MSY} . If the proportion of fish caught (F) is greater than F_{MSY} , overfishing is occurring. If F is less than F_{MSY} , underfishing is happening. Fishing mortality is usually given as a ratio of F/F_{MSY} ; a ratio over 1 means there is some degree of overfishing happening.

Figure 1a. Idealised graph of yield and Spawning Stock Biomass (SSB) for a stock against increasing fishing mortality).



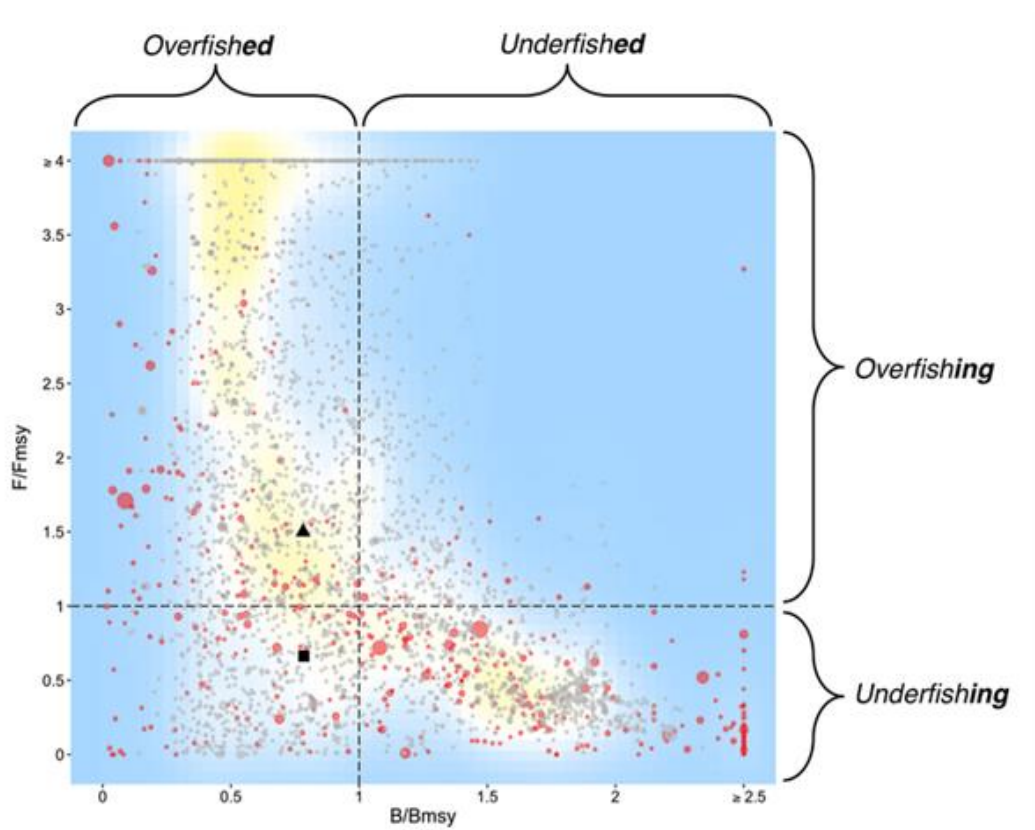
Source: ICES (2010)

Fishing mortality is the only variable that can be directly controlled by fisheries management. Overfishing, if persistent, generally leads to a stock being overfished and thus overexploited, so the

²¹ Part of this section is copied from <https://sustainablefisheries-uw.org/seafood-101/overfished-overfishing-rebuilding-stocks/>

two concepts are of course linked but are not interchangeable. Where available, stock status is measured as a combination of the two concepts, as we describe below. To illustrate the complementarity between the two, (Sustainable Fisheries, 2023) plotted the F/F_{msy} ratio against the B/B_{msy} ratio to get a sense of the sustainability of a fishery. Below (see Figure 2a) is one of these “Kobe plots” of most of the world’s fisheries. Each dot represents a stock. The red dots represent data from the RAM Legacy Stock Assessment Database²², and the gray dots represent estimates for unassessed fisheries (Costello et al., 2016). Dot size scales to fishery MSY. The triangle is the median and the square is the MSY-weighted mean. Figure 2a shows that all situations occur across the world and the size of the MSY (dot scale), linked to the size of the stock, has no direct relationship to the state of the stock.

Figure 2a. Aggregated Kobe plot for worldwide fisheries



Source: <https://sustainablefisheries-uw.org/seafood-101/overfished-overfishing-rebuilding-stocks/>²³

Management objectives

In line with the requirements from the United Nations Convention on the Law of the Sea (UNCLOS) and the United Nations Fish Stocks Agreement (UNFSA), the depletion status of targeted resources in relation to MSY (hereby called MSY approach through this report) is thus the worldwide reference since many states committed to it at the Earth Summit in Johannesburg in 2002. Worldwide, fisheries are called “sustainable” or “unsustainable” according to the United Nations FAO’s conclusions on which fisheries are overexploited and which are not, thus based on the biomass

²² <https://www.ramlegacy.org/>

²³ Right to copy figure agreed

criteria. And the UN SDG 14.4 requires “to restore fish stocks in the shortest time feasible, at least to levels that can produce maximum sustainable yield as determined by their biological characteristics”.

However, for EU fisheries, both the biomass and the mortality criteria are mentioned. One objective of the CFP 1380/2013 is, according to Article 2, “to ensure that the exploitation of living marine biological resources restores and maintains populations of harvested species above levels which can produce the maximum sustainable yield” (EU, 2013). This objective is, thus, defined in terms of population biomass with regards to the Bmsy. However, Article 2 also states that “in order to reach the objective of progressively restoring and maintaining populations of fish stocks above biomass levels capable of producing MSY, the MSY exploitation rate shall be achieved by 2015 where possible and, on a progressive, incremental basis at the latest by 2020 for all stocks.” This article thus explains that while Bmsy is the ultimate goal, Fmsy is the operational objective that the CFP pursues to achieve that goal.

Another key EU policy frame is the EU Marine Strategy Framework Directive (MSFD), intended to “...contribute to coherence between different policies and foster the integration of environmental concerns into other policies, such as the Common Fisheries Policy (CFP)”. MSFD Descriptor D3 of Commission Decision (EU) 2017/848 of 17 May 2017 aims that populations of all commercially exploited fish and shellfish are within safe biological limits, exhibiting a population age and size distribution that is indicative of a healthy stock. Three criteria have been originally identified, but only the two first have an agreement on valid indicators, so the third (“The age and size distribution of individuals in the populations of commercially exploited species is indicative of a healthy population”) is not assessed and described further.

D3C1 – The Fishing mortality rate of populations of commercially-exploited species is at or below levels which can produce the maximum sustainable yield (MSY).

D3C2 – The Spawning Stock Biomass of populations of commercially-exploited species are above biomass levels capable of producing maximum sustainable yield.

Stock assessment methods and processes

Measuring the metrics F/F_{msy} and B/B_{msy} is a statistical process called “stock assessment”, which requires two major steps:

Being able to quantify (proxies of) fishing mortality and/or biomass, and their trends over time.

Being able to define reference points in order to assess over/underfishing and/or over/underfished status.

The two steps require different type of information, and achieving these depend on the quantity and quality of the fishery data available on the fish population of interest, as well as on the length of the time series, including data on

Catch (landings, discards) and fishing effort of the fisheries, usually coming from compulsory reporting like logbooks.

Biological information on the fish taken from scientific sampling (age, length, weight, maturity...).

Fishery-independent information such as biomass or abundance indices coming from routine scientific surveys or from reference “tuning” commercial fleets.

The availability of these data is highly variable across exploited fish populations (stocks), ranging from data-rich to data-moderate and data-poor situations, and performing accurate stock assessment requires thus finding the most appropriate statistical method for each data situation. This is a complex process, since there are many methods available, and many pitfalls. To summarize this complexity, we quote here part of the introduction by Punt (2023), one of the worldwide leader scientists in fisheries stock assessment, in his summary article: “*The ‘ideal’ stock*

assessment is based on a model of the population dynamics that adequately matches the reality of the system being modelled, capturing just the correct number of processes to enable accurate and precise outcomes to be created. The ideal stock assessment would be able to estimate all of the key parameters related to natural mortality, growth, recruitment, selectivity and movement, within a framework that assigns appropriate weight to the data, fits the data adequately and captures all sources of uncertainty related to estimation, including model uncertainty, process uncertainty, and observation uncertainty. Reality naturally seldom matches this ideal (...), given the limitations of system understanding, data quality and quantity, and computational limitations.

The earliest quantitative model-based stock assessments were based on simple production models and virtual population analysis. These methods have a limited number of options (...). For example, for production models the decisions to make relate to the form of the production function (e.g., Schaefer, Fox or Pella-Tomlinson), whether the errors are assumed to be observation or process, how to quantify uncertainty, and how to weight alternative indices of abundance.

Contemporary methods of stock assessment are based on the ‘integrated analysis’ paradigm, in which the model of the population dynamics is developed separately from that of the model that relates the data to the population dynamics model (the observation model). Stock assessments based on this paradigm have increased in sophistication (...) and several major stock assessment software packages now exist that implement various forms of integrated analysis assessment (...). These packages contain myriads of options for the model of the population dynamics and that of the observation process, meaning that given a single data set, the final assessment may depend on the philosophy of the analyst. Hence, the technical and statistical sophistication of the methods used to construct and fit integrated stock assessments means that given a data set, different analysts may construct quite different assessments. (...).” (Punt, 2023)

This introduction is here to recall that estimating a “stock status overfishing indicator” or a “fish stock depletion indicator” is not, and shall probably never be, an automatized “press button” one-size-fits-all statistical process. This is work tailored to each fish population, where best estimates are provided after careful selection of (i) model structure (population model, spatial structure, and process error), (ii) modelling of key biological processes of a population (growth, selectivity, natural mortality, and recruitment) and (iii) analysis of diagnostics and data weighting.

Worldwide, such work takes place within dedicated collective scientific structures (“working groups”) usually operated by States or RFMOs (Regional Fisheries Management Organisations), with established processes for ensuring that stock status estimates are robust through best practices, transparency, and peer review. This is a process mobilizing major scientific, financial, and human resources. Considering EU fisheries in Northeast Atlantic waters as an example, the status of individual stocks is assessed by ICES, with more than 80 experts working groups affiliated to the “fishery resource steering group”²⁴. Strict rules have been defined to cover all possible situations of data availability and reliability (ICES, 2023)²⁵, so that the choice of methods to be used to define the reference points and to assess status with regards to them is fully transparent, generic, and robust. In 2023, ICES produced status information for 276 stocks²⁶, distributed into 6 methodological categories according to data availability.

The major issue of fisheries products lacking stock assessment data

A major challenge for the purpose of identifying the environmental impact of wild caught products with regards to stock depletion is the absence of official stock assessment for a great number of these products worldwide. This is true even for many species sold and consumed on the EU market,

²⁴ <https://www.ices.dk/community/groups/Pages/FRSG.aspx>

²⁵ <https://doi.org/10.17895/ices.advice.22240624>

²⁶ <https://sid.ices.dk/products/reports.aspx>

and both as domestic production and imported products. For instance, in France it was estimated that 19% of domestic landings in 2022 came from non-assessed fish populations, and 5 % came from populations assessed but without proper reference points allowing to assess their sustainability (“non classified”) status (Biseau, 2023)²⁷. For the North-East Atlantic areas, ICES provides annual reporting of the knowledge status on stocks, displayed and described for each ecoregion in the so-called “fisheries overviews”²⁸. Taking the Greater North Sea region as an example, which is considered as one of the most data-rich areas in the world, ICES (2022)²⁹ described that stock status in 2022 relative to the EU Marine Strategy Framework Directive (MSFD) good environmental status (GES) is missing for 73 out of 118 stocks (62%) regarding assessment criteria of fishing pressure (D3C1) and for 66 stocks out of 118 (52%) regarding assessment criteria for stock reproductive capacity (D3C2).

In Europe, though, these many stocks with missing data represent only a small part of the total catches compared to assessed stocks (9% and 5% respectively for the two criteria), but that proportion may vary substantially across regions, both within EU fishing areas and outside. In fact, worldwide, the Global Fishing Index of the Minderoo Foundation (2021) estimates that 52% of the global catch is from stocks that lack sufficient data to determine if they are sustainably exploited, These are found in many parts of the world, especially, but not limited to, the high seas beyond the Exclusive Economic Zones (EEZs) of nations (Jarvis and Young, 2023). To fill these assessment gaps, models are being developed to estimate stock status where there are no fishery-independent (survey) abundance indices to inform on stock density (due to various reasons may them be bio-ecological or socio-economical). One of the models receiving the most attention is CMSY++ (Froese et al. 2023), developed and improved since its first version (Catch-MSY: Martell and Froese (2013)) by an international team of fisheries scientists. This “catch-only” model is based on the Schaefer (1954) model. Although such model cannot replace stock-specific field-based data collection, it can be used as a fallback option to infer required data for data-poor stocks and is already widely used in low-income countries and in the Southern Hemisphere as well as for other stocks without stock assessments.

We report though that such catch-only methods (i.e. lacking independent time series of population abundance and density from surveys) remain a source of controversy in fisheries science, because doubts may remain on the robustness and non-bias of the results obtained. Explained plainly, if F (fishing mortality) reflects a (kind of)³⁰ ratio between catches and abundance (B), then conversely catches reflect a (kind of) ratio between F and B , and thus a given amount of fish caught by a fishery could in theory correspond to an infinity of F and B combinations, e.g. either from a healthy stock with sustainable exploitation level (High B , low F), or from an overexploited depleted stock (low B , high F). The model thus needs additional information to fit among that infinity of combinations, and the parameterization assumptions made for the additional parameters fitting the models is crucial (Froese et al., 2023). Without any fisheries-independent information on population abundance and density (i.e., catch-per-unit-of-effort or scientific survey), a risk remains that stock status results are not fully robust (Ovando et al., 2021), and the previous version (CMSY) has been reported to be at risk of being over optimistic (Bouch et al., 2020).

²⁷ <https://www.ifremer.fr/fr/actualites/bilan-2022-en-france-hexagonale-la-moitie-des-volumes-de-poissons-peches-provient-de>

²⁸ <https://www.ices.dk/advice/Fisheries-overviews/Pages/fisheries-overviews.aspx>

²⁹ https://ices-library.figshare.com/articles/report/Greater_North_Sea_ecoregion_fisheries_overview/21641360?file=38379173

³⁰ “kind of”, as the mathematical formulation is not quite as simple as a simple ratio..

In summary, this introduction chapter highlighted some of the key conceptual differences underpinning the LCA and non-LCA methods regarding the measurement of the depletion state of targeted resources. These differences are discussed again in further details in section 5.3.

Review of LCA methods reflecting the depletion of fish stocks

Current LCA indicators, as shown in **Table 22a**, quantify overfishing mainly from three different approaches (i.e., stock, distance-to-target and productivity assessments).

Table 22a. Overfishing indicators that have been proposed in Life Cycle Impact Assessment.

Assessment	Approach	Unit of metric	Meaning of characterization factor	Reference
Stock assessment	LPY: midpoint	$\text{kg}_{\text{lost yield}} \cdot \text{kg}^{-1}_{\text{current yield}}$	Kilograms of lost yield per kilogram of current yield	Emanuelsson et al. (2014)
	OF: midpoint	$\text{kg}_{\text{fished in excess}} \cdot \text{kg}^{-1}_{\text{sustainable catch}}$	Kilograms fished in excess to what should be fished following MSY per kilogram landed.	
	OB: midpoint	$\text{kg}_{\text{biomass in excess}} \cdot \text{kg}^{-1}_{\text{remaining biomass}}$	Kilograms of biomass extracted in excess per kilogram of biomass remaining.	
Stock assessment	Midpoint	tonnes ⁻¹	Depleted stock fraction (an analogue of PDF and PAF in ecosystem quality AoP) per tonne removed	Hélias et al. (2018)
Stock assessment	Endpoint	Species \cdot year \cdot $\text{kg}^{-1}_{\text{catch}}$	Number of species loss during one year per kilogram of catch	Hélias et al. (2023)
Stock assessment	Endpoint	$\text{PDF}_{\text{reg}} \cdot \text{year} \cdot \text{kg}^{-1}_{\text{catch}}$	Potentially disappeared fraction of species lost at (eco)regional level during one year per kilogram of catch	
Stock assessment	Endpoint	$\text{PDF}_{\text{glo}} \cdot \text{year} \cdot \text{kg}^{-1}_{\text{catch}}$	Potentially disappeared fraction of species lost at global level during one year per kilogram of catch	
Distance-to-target (DtT) assessment	Midpoint	Eco-points \cdot $\text{kg}^{-1}_{\text{catch}}$	Environmental damage in eco-points due to biotic resource depletion per kilogram of catches	Frischknecht et al. (2021)
Distance-to-target (DtT) assessment	Stock in the sea (SIS): midpoint	Eco-points \cdot $\text{kg}^{-1}_{\text{catch}}$	Environmental damage in eco-points that reflects available biomass	Bach et al. (2022)
	Fish manager (FM): midpoint	Eco-points \cdot $\text{kg}^{-1}_{\text{catch}}$	Environmental damage in eco-points that reflects fishing effort	
	Target pressure (TP): midpoint	Eco-points \cdot $\text{kg}^{-1}_{\text{catch}}$	Environmental damage in eco-points that reflects economic exploitation perspective	

Assessment	Approach	Unit of metric	Meaning of characterization factor	Reference
	Production based regional midpoint approach (RMA): midpoint	Eco-points · kg ⁻¹ _{catch}	Environmental damage in eco-points due to regional production	
	Consumption based regional midpoint approach (RMA): midpoint	Eco-points · kg ⁻¹ _{catch}	Environmental damage in eco-points due to regional consumption	
Distance-to-target (DtT) assessment at species level	Midpoint	year · t ⁻¹ _{wet weight}	Maximum potential regeneration time (years) per kilogram of catch	Langlois et al. (2014)
Productivity assessment at ecosystem level	Midpoint	year · t ⁻¹ _{wet weight}	Time (years) required to regenerate the biomass extracted per kilogram of wet weight	
Productivity assessment	Midpoint	kg _{C-eq} · kg ⁻¹ _{catch}	Kilograms of carbon equivalents per kilogram of catch	Langlois et al. (2015)
Distance to nature assessment	Midpoint	m ² · kg ⁻¹ _{catch}	Squared meters affected by catches and influenced by natural degradation potential	Farmery et al. (2017)

LPY= loss potential yield; OF= overfishing through fishing mortality; OB= overfishedness of biomass.

Source: Authors' own elaboration

Emanuelsson et al. (2014) presented the first LCA method linked to assessing the depletion of targeted resources, through the development of the lost potential yield (LPY), overfishing through fishing mortality (OF) and overfishedness of biomass (OB), as three complementary midpoint indicators to assess overfishing. The LPY indicator, previously named wasted potential yield in Emanuelsson et al. (2012), accounts for the difference in the average annual yield between an optimal scenario (Y_{opt}) and a scenario based on the current fishing pressure (Y). The current yield is calculated from fishing mortality and biomass values documented by the International Council for the Exploration of the Sea (ICES). Although lost yields (i.e., overfishing) are addressed based on the maximum sustainable yield (MSY) indicator, the LPY can identify collapsed stocks at heavily overfished conditions, as well as underfished stocks that are a loss of opportunity for biomass provision. The OF and OB indicators are suggested as a complement to LPY (Emanuelsson et al., 2014). On the one hand, the OF indicator expresses the relationship between fishing mortality at the current pressure (F) and fishing mortality at the maximum sustainable yield (F_{MSY}), corresponding to the F/F_{msy} indicator described above. It represents the proportion of kilograms fished in excess to what should be fished under sustainable conditions. On the other hand, the OB indicator expresses the relationship between biomass at a sustainable state (B_{MSY}) and the biomass at the current pressure (B), corresponding to the B/B_{msy} indicator described above. It represents the kilograms of biomass missing per kilogram of biomass remaining. The OB parameter is important when it comes to evaluating the recovery of the biomass of a stock even if it is sustainably exploited. Although the authors recommend the joint use of the three indicators (i.e., LPY, OF and

OB), they acknowledge that data for LPY is more difficult to obtain and recommend using OF and OB alone in the worst-case scenario of data quality and availability.

In accordance with European Union recommendations (JRC-EIS, 2011), Hélias et al. (2018) proposed the depleted stock fraction as a midpoint indicator in the natural resource area of protection pathway. This metric is considered an analogue to potentially disappeared fraction of species (PDF) or potential affected fraction (PAF). The characterization factors generated can be interpreted as the years a stock needs to recover from biomass withdrawal per unit of resource extracted. Characterization factors were computed for > 4000 FAO stocks in 26 FAO major fishing areas using the CMSY method, with input data such as catches, intrinsic rate of population growth (from Fishbase) and current biomass of a given stock. By assuming similar fishing mortality, the authors calculated characterization factors when data are available at multi-stock level. Hélias et al. (2023) refined the previous method to marginal and average approaches within the ecosystem quality Area of Protection. Now, 5000 populations from 25 FAO major fishing areas are covered. A regional endpoint in $\text{PDF} \cdot \text{year}$ units is achieved through the depleted stock fraction from Hélias et al. (2018) as an effect factor, and the inverse of the growth rate constant multiplied by the inverse of relative biomass as a fate factor, following a similar logic in USEtox[®] framework (Fantke et al., 2018). Global characterization factors were also addressed by considering the proportion of the ecoregion to global biomass, complying with LC-Initiative guidelines (Verones et al., 2017). This means that global characterization factors are lower than regional characterization factors based on the endemism of the assessed species. In general terms, the median values for regional and global characterization factors are $9.4 \cdot 10^{-10} \text{PDF}_{\text{reg}} \cdot \text{year} \cdot \text{t}^{-1}$ and $1.9 \cdot 10^{-10} \text{PDF}_{\text{glo}} \cdot \text{year} \cdot \text{t}^{-1}$, respectively (Hélias et al., 2023). The environmental impacts assessed by this method are carried out at population-level based on one single elementary flow of fish mass caught. If various populations are caught, then each of them is separately assessed and the related impact is aggregated.

The LCA distance-to-target (DtT) approach assesses the proximity of the current situation of the environment to a science-based target or political target (Muh et al., 2023). Frischknecht et al. (2021) quantified multiple human activities pressures in their Ecological Scarcity Method (ESM). The method includes emissions to air, emissions to surface waters, emissions to groundwater, emissions to soil, resources, wastes, non-substance emissions (i.e., noise). Regarding marine fish resources use, the characterization factors were derived from Hélias et al. (2018) work and some adjustments were made by Itten & Stucki (2022). Peruvian anchovy (*Engraulis ringens*) was set to be the reference unit to indicate relative intensity of the fishing intervention. Normalization and weighting factors were used to achieve an impact unit of ecopoints. The ESM indicator applies only to the Switzerland region.

Bach et al. (2022) developed stock in the sea (SIS), target pressure (TPA), fish manager (FM) and two regional midpoint approach (RMA) indicators to assess the impact of biotic resource extraction by fisheries. Environmental damage is presented in ecopoints units. Therefore, comparison with other impact categories could be carried out. The SIS indicator is related to biodiversity conservation because biomass is included in the computation. The TPA indicator integrates the ratio of the current fishing mortality (F) to the fish stock maximal sustainable pressure (F_{MSY}). The FM indicator consists of the ratio of the catch to the maximum sustainable yield (MSY) for a given stock. All of them are global indicators and their normalization factor is the inverse of the catch. The RMA indicators were proposed to assess the production and consumption of a stock in a region. Herring stock in the Northeast Atlantic is used in the characterization factor as a reference unit. Normalization factor requires the total catch in the area. Weighting factor includes the proportion of the catch to the overall fully fished stock. The case study in Bach et al. (2022) addressed only the

SIS, TPA, FM and RMA based on consumption. Given the difference between the global and regional approaches, it is impossible to compare them directly even though they have the same unit (i.e., ecopoints).

The Langlois et al. (2014) study present biotic natural resource depletion due to fishing at species level as a DtT approach. If a stock is underexploited, moderately exploited, or fully exploited (i.e., catch below the MSY), then the impact is calculated by the mass uptake of a given species tempered by its maximum sustainable yield (MSY). Otherwise, an additional factor (i.e., MSY to catch ratio) is added to the equation if a stock is overexploited (i.e., catch above the MSY or fishing effort above the fishing effort at MSY). MSY is a frequently and easily used reference point for fisheries management agencies (Ricard et al., 2012). Characterization factors were calculated for 58 depleted stocks and 69 sustainably fished stocks for the species in the RAM Legacy Database.

Only two studies applied a productivity assessment approach. Langlois et al. (2014) evaluated the biotic resource extraction impact now at an ecosystem level. The impact is computed by the proportion of NPP of a fishery and the NPP of an ecosystem, which is addressed with its area and the average productivity in mass of organic carbon per square meter. The classification of marine ecoregions from Spalding et al. (2007) allowed the computation of characterization factors at a province level.

The second method was reported by Langlois et al. (2015), in which the biotic natural resource depletion at ecosystem level was determined by the net primary production (NPP) as the amount of carbon equivalents required to sustain the production of fishing activities. The NPP indicator is also referred to as net primary production used (NPP_{used}), primary production required (PPR) or biotic resource use (BRU) in the literature (Avadí & Fréon, 2013; Cashion et al., 2016). Given the carbon mass units in NPP, comparison of biotic resource use on different food systems is now possible. The model requires trophic level and trophic efficiency, which have been reported in detail by Pauly & Christensen (1995), and a factor representing a 1:9 ratio for carbon mass to wet mass proportion. The NPP indicator has been used in several seafood LCA studies (Aubin et al., 2006; Papatryphon et al., 2004; Parker & Tyedmers, 2012; Pelletier et al., 2009).

In the study from Farmery et al. (2017) the Naturalness Degradation Index (NDI) is presented as a part of a distance to nature assessment (i.e., naturalness). The NDI is explained as the square meters affected by catch and influenced by the hemeroby classification (i.e., the distance of the current environment to its potential natural state induced by human activity). A multidisciplinary expert group judgment with expertise in ecology, biology, population modeling, marine habitats, biodiversity, and ecosystem functioning was required to describe the hemeroby classes. This kind of qualitative assessment is suitable for circumstances where data is limited. Nevertheless, in the opinion of Farmery et al. (2017), this method requires further research.

Review of non-LCA methods reflecting the depletion of targeted resources (“MSY methods”)

As described in the introduction (section 3.1), MSY approaches are the worldwide reference standard for sustainable fisheries management (e.g. FAO, RFMOs, UN SDG 14).

Based on the availability of individual stock status data issued from the stock assessment process described above, several methods used by various international organisations are described here that provide global assessment on the depletion of fish stocks through various approaches of categorisation and scoring. For each of these we briefly describe the underlying requirements according to the criteria described in Section 2.2.

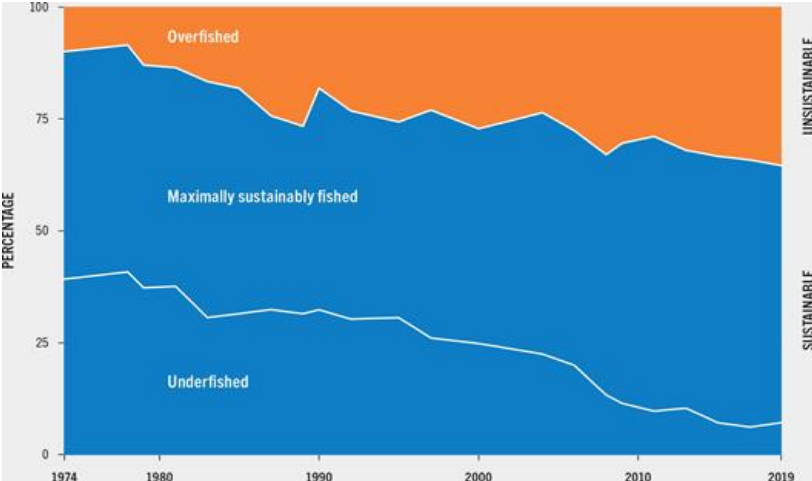
Additionally, as was disclaimed in section 2.1.2, we underline that the non-LCA methods presented here are restricted to those meant to be applicable at the scale of a food product (i.e. any product involving any wild seafood species sold to consumers on the EU market, either from domestic fleets or imported), and as such the review does not cover all approaches, including novel approaches designed for assessing global sustainability at the scale of the production operator, i.e., the vessel/the fishing fleet (e.g. Dewals and Gascuel, 2020, Danto et al., 2022, Ye and Link, 2023, Quemper et al., 2024).

FAO worldwide assessment

The UN body FAO oversees producing a biennial worldwide status of the world fisheries (FAO, 2022). In 2019, 64.6% of marine fishery stocks were considered within biologically sustainable levels (“maximally sustainably fished” + “underfished”). This percentage treats all stocks equally regardless of their abundance and the volume of their catch. When weighted, FAO estimates that 82.5% of the 2019 fishery landings were from biologically sustainable stocks. See figure 3a.

Notably, since the three categories below sum up to 100%, this implies that only stocks for which a stock status could be defined are included in this report. As explained above, this means thus that a great number of fishery products cannot be classified in this way.

Figure 3a. FAO global trends in the state of the world’s marine fishery stocks, 1974–2019.



Source: FAO (2022)³¹

The methodology to allocate a given stock status to a given category is described in the Appendix of FAO (2011). The classification uses four major indicators, whose use varies according to their availability. These are: stock abundance (the primary attribute), spawning potential, catch, and size/age composition. The FAO acknowledges that there is no clear consensus or precise estimates of suitable thresholds for defining status of stocks and has chosen to center status on stock biomass (abundance) and loosely based on the standard Schaefer (1954) model.

As a result, “fully exploited” is defined as a biomass within a band of 40–60% of the virgin stock, considering various uncertainties. Stocks above 60% of the unfished biomass are classified as non-fully exploited, and those under 40% as overexploited (Table 3a).

³¹ Copyright agreement - <https://www.fao.org/contact-us/terms/en/>

Table 3a. FAO criteria for the classification of fish stock status

Category	Characteristics
Overexploited	<ol style="list-style-type: none"> 1. Stock abundance <ul style="list-style-type: none"> • Estimates of current stock biomass are < 40 percent of the estimated unfished stock size. • Catch rates (CPUE) are < 40 percent of the initial levels. • Survey abundance indices are < 40 percent of the initial values. 2. Spawning potential <ul style="list-style-type: none"> • Spawning stock biomass is < 20 percent of the unfished biomass. 3. Catch trend <ul style="list-style-type: none"> • Catches have dropped significantly from a peak without a clear cut in fishing effort. • Current catch is < 50 percent of the maximum after a 5-year smoothing. 4. Size/age composition <ul style="list-style-type: none"> • Size/age composition unstable (excessively affected by recruitment, too few size classes in the exploited population given a species' life history). • Trends in size/age compositions are evident that indicate increasing (and/or excessive) fishing mortality.
Fully exploited	<ol style="list-style-type: none"> 1. Stock abundance <ul style="list-style-type: none"> • Estimates of current stock biomass are between 40 and 60 percent of the estimated unfished stock size. • Catch rates (CPUE) are between 40 and 60 percent of the catch rates of the initial fishery stage. • Survey abundance indices are between 40 and 60 percent of the initial values. 2. Spawning potential <ul style="list-style-type: none"> • Spawning stock biomass is between 20 and 40 percent of the unfished biomass. 3. Catch trend <ul style="list-style-type: none"> • Catches have stabilized at or close to the peak values in the last 5–10 years although there may be interannual fluctuations. 4. Size/age composition <ul style="list-style-type: none"> • Size/age composition is stable (not excessively affected by recruitment, enough age or size classes in the exploited population given a species' life history).
Non-fully exploited	<ol style="list-style-type: none"> 1. Stock abundance <ul style="list-style-type: none"> • Estimates of current stock biomass are > 60 percent of the estimated unfished stock size. • Catch rates (CPUE) are > 60 percent of the initial catch rates. • Survey abundance indices are > 60 percent of the initial values. 2. Spawning potential <ul style="list-style-type: none"> • Spawning stock biomass is > 40 percent of the unfished biomass. 3. Catch trend <ul style="list-style-type: none"> • Catches increased over time when fishing effort has increased. 4. Size/age composition <ul style="list-style-type: none"> • Size/age composition of the catch has been stable and has not shown large changes in comparison with that of the initial stage of the fishery.

Source: FAO (2011).

EU CFP and MSFD

A few initiatives provide regular updates of the status of EU stocks and have thus developed operational translations of the EU CFP and MSFD regulations into decision rules for allocating stocks in different categories.

For example, the STECF provides an annual “CFP monitoring report”, estimating the number of stocks overfished ($F > F_{MSY}$), or not overfished ($F \leq F_{MSY}$), but also inside ($F \leq F_{pa}$ and $B \geq B_{pa}$) and outside ($F > F_{pa}$ or $B < B_{pa}$) safe biological limits (SBL) in the NE Atlantic (both EU and non-EU waters). Unknown SBL refers to stocks whose status regarding SBL could not be assessed. (STECF 2023b).

For the MSFD, ICES classify the stocks according to whether they are fished below or above F_{MSY} for criteria D3C1, and whether stock size is greater or lower than $MSY B_{trigger}$ for D3C2 respectively.

None of these two reports provides options for alternative status statements for non-assessed and non-classified stocks. As such, these categorisations cannot be deemed operational for generic environmental assessment of EU market, especially for imported products.

MSC Scoring system

The Marine Stewardship Council is the most widespread fisheries label worldwide, with 500 certified fisheries, 19% of marine wild catch currently engaged with the MSC program (either as certified, in assessment, suspended and in-transition), and over 20,000 MSC labelled products on sale in over 60 countries. MSC has developed generic scoring standards to classify stocks; the latest version (MSC Fisheries Standard v3.0) has been revised in October 2022 (MSC 2022b).

The depletion indicator requested in the present contract would mainly correspond to the MSC Principle 1 Sustainable target fish stocks: *“A fishery must be conducted in a manner that does not lead to over-fishing or depletion of the exploited populations and, for those populations that are depleted, the fishery must be conducted in a manner that demonstrably leads to their recovery”*,

6 performance indicators (PI) are used to score a stock under that principle:

— Outcome:

- PI 1.1.1: Stock Status.
- PI 1.1.2: Stock rebuilding.

— Harvest strategy (management)

- PI 1.2.1: Harvest Strategy.
- PI 1.2.2: Harvest control rules and tools.
- PI 1.2.3: Information/monitoring.
- PI 1.2.4: Assessment of stock status.

Each of these 6 indicators shall be scored on a scale 0-100, leading to an aggregated score for the Principle 1. MSC provides extensive guidelines for what corresponds to the thresholds scoring 60, 80 and 100³², where it sets that 60 is the minimum acceptable performance, 80 is global best practice and 100 is state of the art performance. MSC weights some PIs more than others in scoring (with default weighting values provided by the MSC), so a fishery can receive an overall “fail” even if it has only one individual PI failing score. In order to be as generic as possible and cover various levels of stock information available, the MSC approach is largely based on a risk framework. The scores reflect the likelihood that the objectives set in the various PIs are complied with, with methods to measure this described in the technical guidelines.

For the present review, one can assess that the MSC scoring approach is:

Genericity of the method to apply to all products, both domestic and imported: In theory the scoring method would be applicable to any product if data are available. The method includes provision for data-poor situation in a risk-based framework. However, the overall process for scoring the PIs involves some complex likelihood assessment of the objectives’ achievement level, following a detailed methodology and depending on the quality/quantity of the data available. This is a

³² https://www.msc.org/docs/default-source/default-document-library/for-business/program-documents/fisheries-program-documents/msc-fisheries-standard-v3-0.pdf?sfvrsn=53623a3_31

comprehensive work requiring extensive human resources; and it cannot be used in an automatized way.

Type and availability of input data: In practice, scoring data will logically only be available for the fisheries and products that have engaged with the MSC. For example, Melnychuk et al. (2022) have analysed the status of 82 MSC-certified stocks. Input data used in the scoring assessment are described in the individual reports³³ but not directly and easily available in public database.

Type and format of outcome (scoring, value etc) For the fisheries and products engaged with the MSC, scoring outcomes for the different PIs and Principles, in the form of a value between 0 and 100 are available in individual assessment reports, but not directly and easily available in public database.

STECF Sustainability Indicators for Fisheries Products

1.1.1.1 Principles: System 1 and System 2

The topic of sustainability indicators has been subject to a suite of expert working groups (EWG)³⁴ commissioned by the European Commission to the STECF³⁵. The objective is to develop an operational framework to assign a grade or rating to any fisheries product based on its sustainability performance. The grades mentioned (e.g., A to E) indicate different levels of fisheries sustainability, with A being the most sustainable and E being the least. In May 2021, the STECF released a report (STECF 20-05) proposing methods of measuring and communicating along the supply chain some sustainability aspects of fisheries and aquaculture products (FAPs), based on scientifically sound, simple, and verifiable criteria and indicators. The STECF suggested developing a rating system step by step (see Table 4a), starting with a System 1 based on simple indicators for data-limited fishery and aquaculture products, while products benefiting from additional data would allow for a more reliable assessment of sustainability criteria under a System 2.

Table 4a. Definitions of systems 1 and 2 to determine sustainability aspects of fisheries and aquaculture products along seafood supply chains.

System 1	Based only on data which are mandatory (art. 35 CMO Regulation) under the consumer information provision of the CMO Regulation (i.e., species, capture area, gear category).
System 2	Based on additional voluntary information (art. 30 of the CMO Regulation) provided by producers (e.g., more detailed information on the type of fishing gear, that caught those products, etc.).

Source: STECF (2020).

STECF 20-05 proposes 8 indicators of environmental sustainability (see Section 1.3). After this first proposal, the Commission selected 3 of these as priority indicators: (1) the state of the targeted stock/species, (2) the impact of the fishing activity on the seabed and (3) the impact on sensitive species. Follow-up works were supported work through ad-hoc contracts in 2022, and a EWG was convened again in September 2022 to review the findings of the ad hoc contracts, to finalize grading methods for the first two indicators, and to initiate a preliminary process that could progress on the third indicator (STECF-22-12)

³³ <https://fisheries.msc.org/en/fisheries/>
³⁴ Two published by the time of the writing of this study, a third being published in April 2024, t
³⁵ <https://stecf.jrc.ec.europa.eu/reports/strategic-issues>

1.1.1.2 Decision tree

Regarding indicator (1) on stock status, the EWG 22-12 aimed to i) develop an indicator for fishing pressure applicable to stocks with quantitative assessments by ICES, STECF and RFMOs based on the proposal in the STECF report under system 2; ii) develop an indicator for fishing pressure for stocks not covered above based on the proposal in the STECF report under System 1.

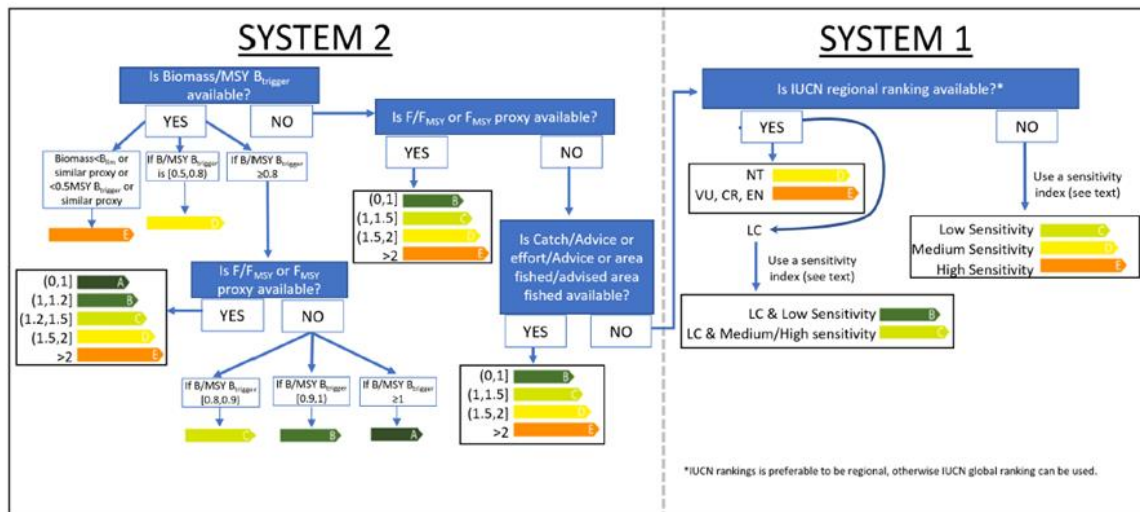
The EWG worked towards maximizing utilization of all available knowledge from scientific committees of advisory bodies and stock assessments. Concerning data availability and sustainability assessment, the key methodological choice made by STECF is that when no official stock assessment exists and System 2 is not possible, other sources of public information must be used, i.e. primarily IUCN and vulnerability assessments. As such this System 1 approach for the grading system is not based on fishing pressure or stock data but on global biological status on species, which widely increases coverage.

In summary, the main results of the two-system approach are:

- System 1 offers a simple and large coverage but an imprecise vision of the status of the stocks. This approach is however relatively easy to implement and can be considered as a low-resolution grading for a seafood product. It can also create an incentive for producers and retailers to provide the relevant information allowing the grading of their product with System 2 to a potentially higher grade.
- System 2 presents a more accurate method for a limited number of stocks for which full information exists. System 2 requires preparing and maintaining a database on a regular basis with updated stock assessment data.

The large coverage of System 1 allows grading a high proportion of existing seafood products including imports. System 2 presents yet a relatively good coverage in terms of volume for the main commercial species consumed in the EU, with more than 80% of the imports being graded with that more detailed system for cod, skipjack tuna, haddock, herring. Another key imported species covered under System 2 include Saithe (90% of the products in volume coming from the 5 main EU providers are graded under system 2), Blue whiting (100%), Mackerel (100%), Sardine (97%), Yellowfin tuna (100%), Bigeye Tuna (100%), Octopus (67%). Due to the prevalence of imports in the seafood consumption (see EWG 22-12 Annex 2 for description), further analysis could however be performed to improve the System 2 coverage as it varies among the imported species. See Figure 4a with the decision tree.

Figure 4a. Decision tree to evaluate sustainability level according to fishing pressure.



IUCN classification: LC = least concerned; NT Near Threatened; VU Vulnerable; EN: Endangered; CR: Critically Endangered.

Source: STECF 22-12³⁶

1.1.1.3 Operationalization: data needs, database structure and data collection

It must be noted that the work described in this section is posterior to the comparison of LCA/non LCA methods on overfishing assessment performed by France in its GT Mer (Gaillet et al., 2023). In this French report, the comparison was performed on the preliminary grading and data collected in 2020 (EWG 20-05). The work below represents thus one step further, not yet published at the time of writing this report, but that shall be published after the STECF Plenary meeting in March 2024.

In 2023, the EU Commission tasked again an ad-hoc subgroup to pursue the operationalization and the implementation of this grading process, with the aim to

- Identify the relevant publicly accessible data sources that should be used to set up the database. These include stock assessment data (biomass and fishing mortality), catch/effort advice, IUCN classifications and sensitivity indices.
- From the various data sources, extract the data that should populate the sustainability indicator database. If possible, implement a first version of the decision tree.

Document the above findings for the contractor setting up the database, incl. instructions for calculating averages of assessment data as required in the approach described in the EWG report.

The ad-hoc group completed its report in October 2023 (Guitton et al. 2023), and its work has been discussed and validated by another dedicated STECF EWG, STECF 23-18 taking place in 11-15 December 2023 (report available from April 2024). The ad-hoc group has developed pathways to implement the decision tree and has also gathered a pilot comprehensive database and RShiny interface for grading³⁷

³⁶ Reuse of this document is authorised under the Creative Commons Attribution 4.0 International (CC BY 4.0) licence (<https://creativecommons.org/licenses/by/4.0/>)

³⁷ Currently available at https://halieut.agrocampusouest.fr/discardless_app/fishing_pressure2023/

System 2: stocks with stock assessment data

For System 2, data are provided by the advisory bodies in the various RFMOs. The necessary information to feed the System 2 grading comes from stock assessment models where biological reference points are defined and different values are compiled or from stock advised catches or efforts compared to realized catch or effort. This information is not available for all stocks for two main reasons, lack of data or issues in fitting stock assessment models to data.

The necessary information is:

- F: fishing mortality yearly estimated by the models.
- FMSY: fishing mortality consistent with achieving Maximum Sustainable Yield (MSY).
- SSB: spawning stock biomass (total weight of all sexually mature fish in the stock) yearly estimated by the models.
- Bpa: precautionary reference point for spawning stock biomass (SSB).
- MSY Btrigger: a biomass reference point that serves as an action threshold. When the spawning stock biomass (SSB) falls below this level, fishery managers are expected to implement measures to reduce fishing mortality to ensure the stock can rebuild to sustainable levels, consistent with achieving Maximum Sustainable Yield (MSY).

Alternatively, time series of advice and realized catch or effort can be used.

In the prototype database, these data are stored in specific tables called Fishdata (Time series of different data (F, SSB, landings, discards) by each stock, year and working group) and Limits (Reference points or limits (FMSY, MSY Btrigger, Blim, Bpa) defined for each stock, year and working group).

System 1: Other stocks

For System 1, two additional sources of data are needed:

- IUCN data for species status for large areas.
- Sensitivity data, reflecting individual species/stocks sensitivity to fishing mortality, as collected by the EWG from the literature (Cheung et al., 2007; Osio et al., 2015; Rindorf et al., 2020).

The ad-hoc group (Guitton et al., 2023) aimed also to collect data and grade a great number of stocks, with a specific focus on the most important commercial species on the EU market, both domestic and imported, based on data from EUMOPA³⁸.

Data for species caught in EU waters were expected to be available through ICES, GFCM, STECF and ICCAT assessments. For species caught outside EU waters, other data sources were investigated. To identify these, the species quantities imported from non-EU waters by country of origin were examined by the ad-hoc group. Key species coming from Norway, Iceland, UK, Greenland and Faroe Islands are assessed by ICES. This means that most catches of the various stocks of e.g. Atlantic cod, Atlantic herring, Atlantic mackerel and saithe are expected to be covered by data obtained from ICES. As skipjack tuna and yellowfin tuna are assessed by tuna RFMOs (ICCAT in Atlantic Ocean, IOTC in Indian Ocean and WCPFC/IATTC in Pacific Ocean), common sardine (Morocco) by CECAF and are exploited also by EU fleets, data are available from STECF balance-capacity database. Catches

³⁸ <https://eumofa.eu/>

and stocks of Alaska pollock in US waters are assessed by NOAA and available from the NOAA database. The ad-hoc group recommended that species for which area information is insufficient to identify stocks (for example for Cape hake), the lowest grade is used.

To feed the database tables, information was thus collected from three different sources (ICES, NOAA and STECF balance capacity working group data) and using different means regarding data and service availability. ICES and NOAA are useful data sources, as they provide web services allowing for automated and standardized data extraction.

As described in Guitton et al. (2023), as of December 2023, the pilot tool developed by the ad-hoc group already contains:

- .991 stock entries with various time series available in the Fishdata file.
- .728 stock entries in the limits file.
- .792 stock entries in the stockdef file.
- .1050 stock entries in the sensitive species file

Taking the next step: a proposal to move from environmental grading to “LCA-like” quantitative scoring

None of the two main sources of information on this grading system, STECF EWG 22-12 and French GT Mer (Gaillet et al. 2023) proposed methods to convert STECF letter-based grading into normalized quantitative score (“LCA-like”). Indeed, on the one hand, such quantitative scoring was not part of the initial request of the EU Commission to the STECF, as only a grading process was thought of. On the other hand, time constraints did not allow the GT Mer to explore further scoring options. The present authors are not aware of other published initiatives that would have taken that endeavour.

What follows below are, thus, preliminary – and thus unpublished - ideas developed in late 2023, after the French GT Mer, by Pr. Didier Gascuel, Chair of the first STECF EWG 20-05, member of the second EWG 22-12, and member of the GT Mer, to pursue integration of the STECF grading system, which represents state-of-the art sustainability assessment, into a LCA-like framework. These ideas were informally presented to GT Mer. The objectives are to:

seek normalization to translate letter-grading into a quantitative score for each type of impact; and propose a method to link that normalized factor to impact per kilo seafood.

In the LCA spirit of measuring impact of individual choices, the key conceptual idea proposed by Gascuel is to normalize “1 kg of fish eaten by an individual person” with regards to “worldwide average individual impact on overfishing” in a simple manner. The proposed steps are:

-For Step 1: Following the range of F/F_{msy} and/or advice ratios between 0 and >2 of System 2 above, back transform STECF final grade A-E into an impact value between 0 and 2 using the same boundaries – e.g. if B would be the value around $F/F_{msy}=1$, then one could score A=0.5 or 0.8, C=1.2, D=1.5, E=2;

-For Step 2: Gascuel proposes to calculate an individual Standardization Factor based on the individual volume of worldwide overfishing (for example $FAO \text{ total catches} * FAO \text{ share of overexploitation} * \text{average } F/F_{msy} \text{ of overexploitation} / \text{worldwide population}$). Then Impact score = Impact value / SF

For processed and blended seafood products, that impact can be calculated according to the percentage of seafood for the total weight, alongside other similar calculations for the other ingredients; as such, the impact remains proportional to the amount of seafood in the package. Examples of this are available in an exploratory spreadsheet.

NB This novel proposal is explained here for the depletion indicator, but it can be equally used for the seabed impact. As of today, it is clear that this approach is not mature enough to be considered operational. However, it appears as a simple and promising proposal that could function as an “LCA-like” indicator, that deserves further investigation.

Summary

The main outcome of this Section 3 is that regarding the assessment of the depletion status of the targeted resources, several methods have been described both in the LCA and non-LCA sections, but only few comply with the requirement of applicability to any type of catch, both domestic and imported, with or without stock assessment data. As such, in Section 5, the only methods retained for evaluation are those from Hélias (2018, 2023) for LCA, and from STECF (2023) for non-LCA – the same methods as those compared in Gaillet et al. (2023), except that updated and more comprehensive data are now available from STECF, whereas Gaillet et al. (2023) worked on the earlier dataset from (STECF, 2020).

4. LCA and non-LCA methods for quantifying seabed impact

Introduction

Seafloor or seabed are terms used to describe the bottom of the sea or ocean whose characteristics are determined by volcanic and tectonic processes (Conrad, 2015). The importance of such an environment relies on the diverse organisms it can sustain the sink effect it exerts on carbon dioxide due to microscopic marine algae and the source of mineral resources it contains. In recent decades, seafloor damage events have been noticed more frequently (e.g., destructive fishing, deployment of anthropogenic physical structures and non-biotic resource extraction activities). Fishing activities impact upon the marine environment via multiple mechanisms (Beukhof et al, 2022). One of the best studied mechanisms, aside from the direct extraction of species, is the disturbance to the seabed and seabed habitats (Scibberas et al, 2017). This disturbance comes from any and all interactions of fishing gears with the seabed, and while demersal trawls and dredging are both intuitively obvious contributing practices and the most studied in terms of quantifying real-world impact (Fernandez-Arcaya et al., 2024), all fishing practices may ultimately impose a pressure on the seabed and its habitats.

Other impacts can occur when construction of structures is perpetuated for long periods. Inger et al. (2009) found that pipeline installation can alter organism habitats and aquatic vegetation. Nonetheless, not every construction produces negative impacts. Artificial reefs exert positive effects on the environment since they provide habitat for the organisms, yet its efficiency is not comparable to natural reefs (Fujii et al., 2023). Within non-biotic resource extraction pressures, we found dredging and smothering from oil and gas production, deep-sea mining (Woods et al., 2016) and use of explosives (Langlois et al., 2014b). Although these impacts are not linked to fishing and, therefore, are not subject to evaluation in the current report, it should be noted that broader indicators for seabed impact that include these sources of damage may arise in the near future.

Mobile, bottom contacting gears, such as demersal trawls (e.g. otter and beam), dredges and anchored seines disturb the seabed via abrasion from various moving components of the gear. While the magnitude of these pressures varies by gear and sub-gear components, the pressures themselves are the same: sediment resuspension, crushing biogenic structures, removing macro-algae and vascular plants, and otherwise reducing structural complexity (Hiddink et al, 2017; Bradshaw et al., 2022). While less studied, passive gears with bottom contacting components impose similar pressures, albeit with lower severity (McLavery et al, 2023). For example, set gill nets over reef structures, entangle and dislodge macroalgae, while heavy pots and anchors crush biogenic structures and reduce sediment rugosity while being set and retrieved. Even pelagic based fishing practices, such as pelagic trawls and floating gill nets or longlines can impose pressure on the seafloor when gears are lost and sink.

The impact of pressures from different fishing practices, is mediated by the sensitivity of the habitat (and its associated community) to the form and magnitude of the disturbance, the frequency of the disturbance and the longevity/regeneration time of the habitat and species within it. Thus, any evaluation of the impact of fishing on the seabed must include estimations of the pressure exerted and the sensitivity of the habitats in which it is exerted.

To make accurate estimations of fisheries' impacts on the seabed requires reliable spatial information on the activity of vessels, the gears they deploy and the underlying habitats in which they are operating. Furthermore, the sensitivity of these habitats to the various gears needs to be known and decisions need to be made about the time scale over which fishing activities are to be integrated. Alternatively, more generic, but less accurate, approaches assume some generic

relationships between habitat sensitivity to different gears and the habitat associations of different fishing practices or the species being targeted.

In this section we review the existing available metrics/indicators/methods for quantifying or otherwise evaluating the impact of fisheries on the seabed. We do this for two categories of approaches, those methods derived to inform LCA exercise, and those methods derived from fisheries ecology more generally.

Review of LCA methods on seabed impact

There have been different efforts in the past 15 years to account for these impacts within LCA metrics. Current LCA indicators, as shown in Table 5a, can be mainly categorized according to their assessment in affected area assessment and productivity assessment.

Table 523a. Seafloor disturbance indicators that have been proposed in Life Cycle Impact Assessment.

Method	Metric	Approach	Unit of metric	Meaning of characterization factor	Reference
Area assessment	Seafloor contact index	Midpoint	$\text{km}^2 \cdot \text{h}^{-1}_{\text{effort}}$	Seafloor contact index: seafloor area swept per hour of effort	Nilsson and Ziegler (2007)
Productivity assessment	Construction on seafloor: occupation	Midpoint	$\text{kg C} \cdot \text{m}^{-2}_{\text{affected area}}$	Kilograms of carbon equivalents per square meter affected by construction	Langlois et al. (2015)
Productivity assessment	Construction on seafloor: transformation	Midpoint	$\text{kg C} \cdot \text{m}^{-2}_{\text{affected area}} \cdot \text{yr}^{-1}_{\text{affected}}$	Kilograms of carbon equivalents per square meter affected by construction over a time horizon (20, 100 and 500 years)	Langlois et al. (2015)
Productivity assessment	Seafloor destruction by fishing gear: benthic stock	Midpoint	$\text{kg C} \cdot \text{m}^{-2}_{\text{affected area}}$	Kilograms of carbon equivalents per square meter affected by fishing gears	Langlois et al. (2015)
Productivity assessment	Seafloor destruction by fishing gear: transformation	Midpoint	$\text{kg C} \cdot \text{m}^{-2}_{\text{affected area}} \cdot \text{yr}^{-1}_{\text{affected area}}$	Kilograms of carbon equivalents per square meter affected by fishing gears	Langlois et al. (2015)
Area assessment	Sea use impact category: impacted area	Midpoint	$\text{m}^2_{\text{affected area}} \cdot \text{yr}^{-1}_{\text{affected area}}$	Area required for fish production	Abdou et al. (2017)
Area assessment	Sea use impact category: solid matter deposition	Midpoint	$\text{kg C} \cdot \text{m}^{-2}_{\text{affected area}} \cdot \text{yr}^{-1}_{\text{affected}}$	Kilograms of solid matter from the fish farm on the seabed	Abdou et al. (2017)
Distance to nature assessment	Natural Degradation Indicator (NDI)	Midpoint	$\text{m}^2 \cdot \text{kg}^{-1}_{\text{catch}}$	Square meters affected by catches and influenced by natural degradation potential	Farmery et al. (2017)

Method	Metric	Approach	Unit of metric	Meaning of characterization factor	Reference
Area assessment	Seafloor disturbance: single impact	Endpoint	PDF · m ⁻² affected area	Potentially disappeared fraction of species per square meter	Woods and Verones (2019)
Area assessment	Seafloor disturbance: repeated impact	Endpoint	PDF · m ⁻² affected area	Potentially disappeared fraction of species per square meter	Woods and Verones (2019)
Area assessment	Seabed disturbance: single impact - regional and global	Endpoint	PDF · year	Potentially disappeared fraction of species year aggregated	Préat et al. (2021)

Source: Authors' own elaboration.

Nilsson & Ziegler (2007) calculated an indicator based on the ratio of area swept by trawling systems per hour (i.e., seabed contact index in km² · h⁻¹) for gear type used. In this study, the index was multiplied by the fishing effort and a GIS map was obtained for the Kattegat, a fishing ground shared by Sweden and Denmark. The seafloor contact index was developed using technical information collected from fishermen, such as gear width, size of trawl doors and common towing speed. Some conclusions about habitat conditions were drawn based on this information combined with MarLIN assessment for each habitat in the Kattegat system.

Langlois et al. (2015) has been already cited in the overfishing LCIA method. In the same article, the authors proposed a way to assess impacts of seafloor destruction by two main interventions (i.e., moored construction and fishing gear). In each intervention, the disturbance on the seafloor is assessed in two phases, designated as occupation and transformation. The moored construction intervention includes the destruction of standing biomass in the seafloor structure (SI_{LSF, seafloor}) as the occupation impact and the impact after the construction was carried out as the transformation impact (TI_{LSF, seafloor}). The fishing gear intervention is calculated with the same figure. There is an initial impact over the biomass (SI_{LSF, seafloor_trawl}) and the corresponding transformation impact (TI_{LSF, seafloor_trawl}) after the fishing activities are carried out. Occupation impacts (SI) quantify the removal of the standing biomass in the ecosystem (i.e., kg C · m⁻²), while transformation impacts (TI) are calculated based on the net primary production required, which units are expressed in kg C · m⁻² · year⁻¹. Langlois et al. (2015) computed 26 characterization factors for different ecosystem types including polar, temperate, tropical, rocky, sedimentary and mud habitats.

Abdou et al. (2017) utilized the sea use concept applied in Langlois et al. (2015) to assess environmental impacts in seabream (*Sparus aurata*) and seabass (*Dicentrarchus labrax*) in a Tunisian aquaculture farm. The novel approach proposed by Abdou and colleagues (2017) was the incorporation of the MERAMOD® model (Cromey et al., 2022) to calculate solid matter generated from fish farm to sea use impact category in kg · m⁻² · year⁻¹ unit. They followed a two-step procedure. In the first step a grid was generated using bathymetry and cage layouts. In the second step initial deposition was determined considering wastage rate, faeces production and hydrodynamics. Some of the input data Abdou et al. (2017) required were grid size, grid cell resolution, number of cages, shape of cages, diameter of cages, depth under the cages, current velocity, among others.

Farmery et al. (2017) applied the Natural Degradation Indicator (NDI) for seabed damage. The method relies on the evaluation of the ecosystem using criteria and scores to determine how far

the current state of the ecosystem is from a pristine condition (i.e., naturalness approach). For seafloor metrics, the criteria addressed the level and reversibility of harm to (i) habitat structure and (ii) key elements of ecosystem structure and function. The scores were computed to give an average score that was assigned to a hemeroby classification for seafloor. Then, this classification was related to a naturalness degradation potential that was used in the NDI calculation. The NDI metric was developed specifically for Australian fisheries and the evaluation required scientists with expertise in many fields, including biology, ecology, fisheries, marine habitats, and biodiversity.

Woods & Verones (2019) presented a series of characterization factors to assess seabed damage from single-impact and repeated-impact perspectives. In the single-impact perspective, the occupation and transformation impact by Milà i Canals et al. (2007) suggested were considered. The occupation impact characterization factor included the extension of damage and initial benthic response. The transformation impact considered in addition to the described parameters the recovery time of a certain ecosystem, which was based on substrate-specific recovery time, hydrodynamic energy, and the stock of potential recolonisers. In the repeated-impact perspective, the main processes are not occupation and transformation, but recoverable and non-recoverable transformation impacts due to potential incomplete ecological recovery in the industry-specific time lapse between perturbation. A total of 17 marine ecoregions in Europe were considered and the resulting characterization factors were presented in $\text{PDF} \cdot \text{year} \cdot \text{m}^2$. It should be noted that although centred on fishing impacts to seabed impact, the methodology leaves the door open to further expansion to account for impacted derived from other activities (e.g., seabed mining). Finally, an interesting methodological outcome of this study is the fact that the method attributes higher environmental impact when seabed disturbance is occurring at greater depths, based on the concept of hydrodynamic energy, through which wave exposure tends to decrease at higher depths (Bekby et al., 2018). This, consequently, leads to lower energy environments at deeper depths and, hence, the seabed requires longer times to recover its natural state. In fact, this outcome opposes the methodology proposed by Langlois and colleagues (2015), in which they assume that below a depth of 60 meters photosynthetic production is close to zero, given that the productivity at this level of the water column depends exclusively on the arrival of suspended particulate matter from more superficial zones of the water column. To a certain extent, this apparent discrepancy is due to the fact that the two methodologies base their impact pathways of differing approaches: area assessment in the case of Woods & Verones (2019); and productivity assessment in the case of Langlois et al. (2015).

Finally, Pr at et al. (2021) developed a framework to quantify biodiversity loss in deep-sea environments. The work had basis on the cause-effect chain from Langlois et al. (2014b), the classification of impacts due to different sea-floor interventions (i.e. occupation, transformation, and permanent impacts) from Mil  i Canals et al. (2007), the regional biodiversity loss computation from Woods & Verones (2019) and suggestions from Koellner et al. (2013) when considering permanent impacts. Basically, the regional biodiversity impacts are calculated regionally and then are converted into global values which can be compared amongst other systems. The final unit obtained with this method is $\text{PDF} \cdot \text{year}$. As well as the previous method, 17 marine ecoregions were assessed. Unlike Woods & Verones (2019), however, this method recommends assessing non-recoverable damages to pristine areas. Since the deep-sea system is mainly unexplored and the number of discovered species continues to grow, this method does not consider species endemism and species threat level for evaluation of global scale impacts (Chaudhary et al., 2015), but it includes ecosystem vulnerability and scarcity.

Review of non-LCA methods on seabed impact

The impact of fishing activities on the seabed has received a lot of attention in both the fields of applied fisheries ecology and the more fundamental marine ecology over the last approximately eight years. This has both driven, and been driven by, increased interest in accounting for these effects in the management of human activities in marine space. These impacts can be considered from two different angles: First, the degree of impact a given geographic area or habitat area has sustained or is experiencing from all forms of fishing activities (area-based). Alternately, one can consider the degree of impact that a given fishery or fishing activity imposes upon the seabed (fishery-based). Our interest is in documenting the environmental impact on the seabed that is associated with the harvest of a given seafood product. Therefore, for the purpose of this report, it is the fishery-based approach that is appropriate for quantifying fishing impacts on the seabed.

Below we identify and describe the indicators and systems of indicators that have been identified in the recent primary literature or are being operationalised by various institutions. First, we describe a systematic review of primary literature, followed by those systems being operationalised, that are known to the authors.

Review of recent primary literature on seabed impacts of fishing

A systematic review of literature was undertaken with the aim to identify, select and extract information from all papers that describe the impact of various fishing practices on the seafloor, from the perspective of a per-fishery / fishing activity impact.

This review was framed in a PRISMA Eco-Evo like approach (O'Dea et al., 2021) to ensure that objective decisions about search criteria (Table 6a), exclusion criteria (Each column represents a subject, where all members of the column are joined together with "OR" statements as synonyms. The columns/subjects are then joined together with "AND" statements as required information. Together this combination of OR and AND level means that all papers containing at least one search term from each column will be returned in search results.

The database selected for the search was SCOPUS, and the search was undertaken on January 3rd, 2024. Within SCOPUS advanced search the field code "TITLE-ABS-KEY" was utilised, which looks for the specified search terms within article titles, abstracts and key words.

Table 6a. Search terms utilised in the primary literature search.

Activity	Context	Impact	Desired Information
fishing	sea*floor	disturbance	index
fishery	sea*bottom	scour*	indices
fisheries	reef	break	
	benth*	damage	
	kelp	extract*	
	macro*alga*	resuspen*	
	coral	dislodg*	
	sediment	remov*	

Each column represents a subject, where all members of the column are joined together with "OR" statements as synonyms. The columns/subjects are then joined together with "AND" statements as required information. Together this combination of OR and AND level means that all papers containing at least one search term from each column will be returned in search results.

Source: Authors' own elaboration.

Table 7a. Exclusion criteria applied to the search results returned from the SCOPUS Database search. First applied to article titles and abstracts, then subsequently applied to the full text of those articles that were retained through the initial screening.

Exclusion Code	Exclusion Criteria	Exclusion Justification
	Retain	Meets no other exclusion criteria.
A	Not Marine	Only interested in marine fisheries.
B	Not about the interaction of bottom contacting fishing gears and the seafloor	Only interested in the impacts of fishing gear with the seafloor.
C	No general index or measure proposed	Only concerned with effect within study, no generalisable index.
D	Index is location based, not fishery based	Only interested in the impact of a fishery on the seafloor, not the cumulative impacts of fisheries more broadly, on a particular area of seafloor.
E	Index or measure is not specific to gear-seafloor interaction	Only interested in the impact of a fishery on the seafloor, not the cumulative impacts of fisheries more broadly, via other mechanisms.
F	Not enough detail	Article does not contain enough information to repeat the exercise of producing the index/indices.
G	Published prior to 2019	Limited to the most recent years, to focus on the state of the art.

Source: Authors' own elaboration.

Table 8a. Data fields to be extracted from articles retained after screening.

Retain	Context	Data Requirements
Exclusion_FullText	Country or Jurisdiction	Data Required 1
	Gear or species specific	Data Required 2
	Gear groups	Data Required 3
	Species Groups	Data Required 4
	Description of Index Methodology	Data Required
	Comments	Data Nice to Have

Columns are groups of similar data, while each row of each column represents a column in the database to be filled.

Source: Authors' own elaboration.

The selected search terms returned 251 records. Of these, 152 were excluded due to being published before 2019 (criterion G). The remaining records were screened via their titles and abstracts. Of these, 71 were excluded due to being off-topic, i.e. not about the impact of fishing gear interacting with the seabed (criterion B), and a further 10 were excluded as the indices or measures they proposed/utilized were based on impact at a location ("area-based", criterion D). Five records were excluded for not being from a marine context (criterion A), three because they dealt with indirect impacts of fishing, such as trophic cascades from over-fishing (criterion E), and two were excluded as they did not propose or utilize a generalized index or metric and were concerned with their specific context (criterion C).

After title and abstract screening only eight records were retained for full text screening and data extraction. At the full text screening stage, six records were excluded as their indicators were "area-based" (criterion D), and the final two records were excluded because they were off-topic and not concerned with the interaction of fishing with the seabed (criterion B).

In summary, it is a striking outcome of this literature search that no records were retained through to data extraction as there were no recent papers addressing indices or metrics of gear/fishery specific impacts on the seabed, which were not concerned only with the per unit area impact. All published articles deal with "MSFD style" monitoring, assessing and reporting on environment status indicators and seldom on the intensity of the pressure. Those articles that do so estimate the intensity metric, only use the swept area ratio of all bottom contacting mobile gears together, not differentiating between many different fishing practices.

According to the selected criteria for assessing potential methods, none of the methods in this review were thus suitable. Specifically, these methods all failed to appropriately address:

- Completeness of scope
- Applicability

Following this result, we rely on the methods being operationalized by various international institutions, as outlined below.

Marine Strategy Framework Directive Common Implementation Strategy

A technical group was assembled under the Common Implementation Strategy for the EU's Marine Strategy Framework Directive, with the aim to establish threshold values and limits for assessing seabed status under the MSFD (TG SEABED, 2023). This technical group reports on thresholds for the MSFD's Descriptor six, specifically criteria four and five (Table 9a). Indeed, all criteria under the MSFD descriptor six refer to the cumulative impacts on seabed habitats. This is akin to the "area-

based”, approach described above. The closest criterion to a “fishery-based” approach within descriptor six is criterion three: “Adverse effects of physical disturbance on benthic habitats”. Within this criterion, fishing may be one pressure that needs to be quantified, however, the methods for establishing thresholds for this criterion are not addressed by this technical group and are instead left to regional cooperation according to regional sea conventions.

Table 9a. Relevant Marine Strategy Framework Directive Criteria addressed in the MSFD's Common Implementation Strategy technical group on seabed integrity.

MSFD Criterion	Description of the criterion
D6C4	<p>The extent of loss of the habitat type, resulting from anthropogenic pressures, does not exceed a specified proportion of the natural extent of the habitat type in the assessment area.</p> <p>Member States shall establish the maximum allowable extent of habitat loss as a proportion of the total natural extent of the habitat type, through cooperation at Union level, considering regional or subregional specificities.</p>
D6C5	<p>The extent of adverse effects from anthropogenic pressures on the condition of the habitat type, including alteration to its biotic and abiotic structure and its functions (e.g. its typical species composition and their relative abundance, absence of particularly sensitive or fragile species or species providing a key function, size structure of species), does not exceed a specified proportion of the natural extent of the habitat type in the assessment area.</p> <p>Member States shall establish threshold values for adverse effects on the condition of each habitat type, ensuring compatibility with related values set under Descriptors 2, 5, 6, 7 and 8, through cooperation at Union level, taking into account regional or subregional specificities.</p> <p>Member States shall establish the maximum allowable extent of those adverse effects as a proportion of the total natural extent of the habitat type, through cooperation at Union level, taking into account regional or subregional specificities.</p>

Source: Authors' own elaboration.

Based on the assessment criteria agreed for the methods being considered in this, the indices described in this report do not satisfy:

- Generality of the method.
- Ability to cover data poor situations.
- Type and format of outcome.
- Completeness of scope.
- Applicability.
- Compatibility with the Environmental Footprint method.

While none of the metrics proposed are relevant in the context of this report, the examples of what metrics are to be used when assessing the state of seabed biology/ecology are. These include typical species composition, relative abundance and size structure, sensitive species or species providing key functions, recoverability and functioning of habitats and ecosystem processes. It is

metrics and indices based on these types of measures that will promote synergies across the EF and the MSFD indicators, reporting and monitoring work.

ICES Evaluation of Methods and Thresholds for Assessing Seabed Habitats

The ICES “Workshop to Evaluate Proposed Assessment Methods And How To Set Thresholds For Assessing Adverse Effects On Seabed Habitats” (WKBENTH3), evaluated a suite of empirical and model-based indicators, for their suitability in assessing the MSFD descriptor six, which is described above (ICES, 2022). This workshop was specifically addressing indicators for criteria three and five.

As the MSFD calls for evaluating the state of the seabed, all the indicators evaluated in this workshop were “area-based”. Despite this, we investigated the detail of the various indicators to identify any “fishery-based” indicators that were used to calculate these area-based indices, with specific reference to the indicators for criterion three.

Table 10a. Relevant Marine Strategy Framework Directive Criteria.

MSFD Criterion	Description of the criterion
D6C3	<p>Spatial extent of each habitat type which is adversely affected, through change in its biotic and abiotic structure and its functions (e.g. through changes in species composition and their relative abundance, absence of particularly sensitive or fragile species or species providing a key function, size structure of species), by physical disturbance.</p> <p>Member States shall establish threshold values for the adverse effects of physical disturbance through regional or subregional cooperation.</p>

Criteria addressed in the ICES “Workshop To Evaluate Proposed Assessment Methods And How To Set Thresholds For Assessing Adverse Effects On Seabed Habitats” (WKBENTH3), not included in Table a.

Source: Authors’ own elaboration.

None of the area-based indices were deemed directly relevant for the specific purpose of this report. Furthermore, the WKBENTH3 report considers only these indices’ responses to a gradient of combined trawling effort, thus there is no differentiation between gear types or fishing activities that any of the indices employ. Thus, none of the cited indices were investigated any further.

Specific to the evaluation criteria set forth for methods in this report, the methods presented in this workshop report do not satisfy:

- Generality of the method.
- Type and format of outcome.
- Completeness of scope.
- Applicability.
- Compatibility with the Environmental Footprint method.

One step up from this evaluation of methods, a different ICES working group considers the operational assessment of benthic impact from fishing, WGFBIT.

ICES Fisheries Benthic Impact and Trade-offs Tools

The ICES “Working Group On Fisheries Benthic Impact And Trade-Offs” (WGFBIT), provides a tool for assessing benthic impact that is harmonized across all EU regional seas. In their report of their 2022 meeting (ICES, 2023) updates to the methodology behind the tool they produce are defined. Here it can be seen that the relationship of benthic impact to fishing pressure is made via either a simple “time since last disturbance”, or an aggregate of all bottom trawl fishing effort under a swept area ratio. Therefore, as with the previous two cases, the methodology employed by this working group is not relevant for the current report’s aim of finding indices in the “fisher-based” approach.

However, the WGFBIT report extends upon the methods described in the previous two sections, by including the types of benthos, and their biological traits in their index of impact. This accounts for the varying responses of different organisms and communities to physical disturbance from fishing gears. It is the combined characteristics of the fishing activity and the environment (physical or biological) in which they are operating that determines the overall impact. In the WGFBIT report, the variability in the biological environment is taken into account, however the variability in the fishery is limited to the relative effort of all bottom contacting trawls.

Thus, while the methods utilized in WGFBIT are more complete than the previous sections, they still fail to address the key driver of the impact of various fishing activities, the unequal impact of various gears and practices.

STECF Sustainability Indicators for Fisheries Products

Unlike the benthic impact indicators designed for, or described in conjunction with, assessing the state of the environment (i.e. for the purposes of MSFD descriptor six), the work on benthic impacts of fisheries from the STECF Expert Working Group meeting (STECF EWG-22-12) that was reviewed and reported as an STECF report (STECF, 2023) is “fishery-based” (corresponding to the “productivity assessment” mentioned for the LCA methods in section 4.2), and not “area-based”. The methodology consists of two “systems”, similar to what was described above for the indices of fishing pressure from the same source. The first system involves more basic data reporting and is thus more inclusive of fishery products from data poor contexts. The second system utilizes a higher resolution of data reporting to achieve more accurate estimations of impact of fishing on the seabed, and rewards this with access to better scores overall. The STECF review also proposes some extensions to these systems (namely system 2) that would improve accuracy even further, but which requires more work before they can be implemented.

Both systems and the proposed extensions are based on the interaction of different categories of gear type with different seabed habitat types. That is the severity of the pressure from the activity of any given gear combined with the sensitivity of the habitat in which the gear was operating. The differences between the two systems and the extensions are not procedural, but instead are the resolution with which the gear types are reported and the use of more than one potential impacted habitat to select the final score.

System 1 – Seabed Impacts

In both systems, the first consideration is the degree of interaction/impact of the gear type with/on the seabed. System 1 relies on the two-character gear category that is presently reported as mandatory under the EU’s CMO regulation (EU reg. 1379/2013). These gear categories have been assigned impact scores according to their level of interaction with the seabed (Table a). The range of these scores under system 1 is from one to three, where 1 has the lowest interaction with the

seafloor (e.g. pots and traps), while three has the highest (e.g. dredges). Under this system and the current implementation of the CMO regulation, the low resolution of the gear categories means that some gears with dissimilar interactions with the seafloor are grouped, and hence scored, together. For example, the category “Trawls”, includes both “demersal trawls” with bottom contacting gears, and “pelagic trawls” that have no interaction with the seabed. In order to maintain a precautionary approach, where such aggregation introduces ambiguity, the most severe score of the category is applied (i.e. all trawls are scored 3, even if they may be pelagic trawls that do not impact the seabed).

Table 11a. Proposed impact scores by the EWG per mandatory gear category defined in the CMO Regulation (Annex III column 1) together with the respective FAO category code. Taken from STECF, 2023.

Gear category	Gear score	FAO Category
Surrounding nets and lift nets	1	PX
Gillnets and similar nets	1	GX
Hooks and lines	1	HX
Pots and traps	1	FP
Seines	2	SX
Dredges	3	DX
Trawls	3	TX

Source: Authors’ own elaboration.

The second consideration focuses on the habitats that are being impacted. Both systems utilize known relationships between the species that are harvested and their habitat associations. Under system 1, the coarsest resolution, the single most associated habitat is used as the habitat that is being impacted. The identification of species or species groups from the variety of common names has been undertaken for all commonly marketed fish in the EU, and the association of these species to habitat type was made by expert evaluation, with reference to primary literature and FishBase. The habitat categories that are assigned are based on the European Nature Information System (EUNIS) (<https://emodnet.ec.europa.eu/en/seabed-habitats>) marine habitat categories (namely level 2 of the hierarchy). The scores themselves range from zero to three; zero indicating no sensitivity to bottom disturbance (i.e. pelagic habitats), while three indicates a high sensitivity (e.g. biogenic habitats). These scores were attributed by expert evaluation with reference to primary literature and publicly available aggregating databases.

Both the severity of the gear and the sensitivity of the habitats have been scored according to physical, abrasive disturbance, and not to other mechanisms of disturbance (e.g. biogeochemical perturbations or trophic cascades).

The two scores, derived from the severity of the gear and the sensitivity of the habitat are added together to derive the final impact score for a given seafood product and these scores are assigned a letter score across a range of five levels:

— A score – total ≤ 2 : very low impact

- B score – total of 3: low impact
- C score – total of 4: medium impact
- D score – total of 5: high impact
- E score – total of 6: very high impact

System 2 – Seabed Impacts

As described above the system 2 approach for seabed impacts utilizes the same structure as system 1, in that two scores are derived for a given seafood product, which are then combined to deduce a final overall impact score. The first of the two scores reflects the severity of the interaction of the gear with the seabed and the second reflects the sensitivity of the most likely fished habitat to physical disturbance from bottom contacting gears.

In system 2, the gear categories used to derive the first score are more detailed. The initial proposal is to use the three letter codes that are reported voluntarily under the CMO regulation. This provides 28 different categories and differentiates between things such as bottom contacting demersal trawls and non-bottom-contacting pelagic trawls. With this resolution, an extra score value is added, namely zero, such that the range of gear severity scores in system 2 goes from zero to three, instead of one to three. The score of zero reflects those gear categories that do not interact with the seabed. A proposed extension to this system 2, is to utilize the FAO's gear definitions that include a list of 88 categories. This level of differentiation would allow the scoring of intermediate gears (e.g. semi-pelagic trawls); however, it is questionable whether this level of detail would be traceable from capture. This trade-off can be considered in future iterations but is relevant for designing future-proof data flows and storage.

Habitat sensitivity under system 1 utilizes only the most commonly associated habitat of the species in question. This may be misleading in cases where the fishery targets habitats utilized for specific life-history stages (e.g. spawning aggregations or migratory corridors). Therefore, under system two a secondary habitat should be associated with each species and the most sensitive of the two habitats utilized in the calculation of the final benthic impact score. The association of species to secondary habitats has not yet been made, but a specific expert contract has been proposed to populate such a field.

Under system 2, the final numeric score is derived from a simple addition of the gear severity and the habitat sensitivity scores, and allocated to a letter score, in the same manner as for system 1.

Marine Stewardship Council Benthic Impacts Tool

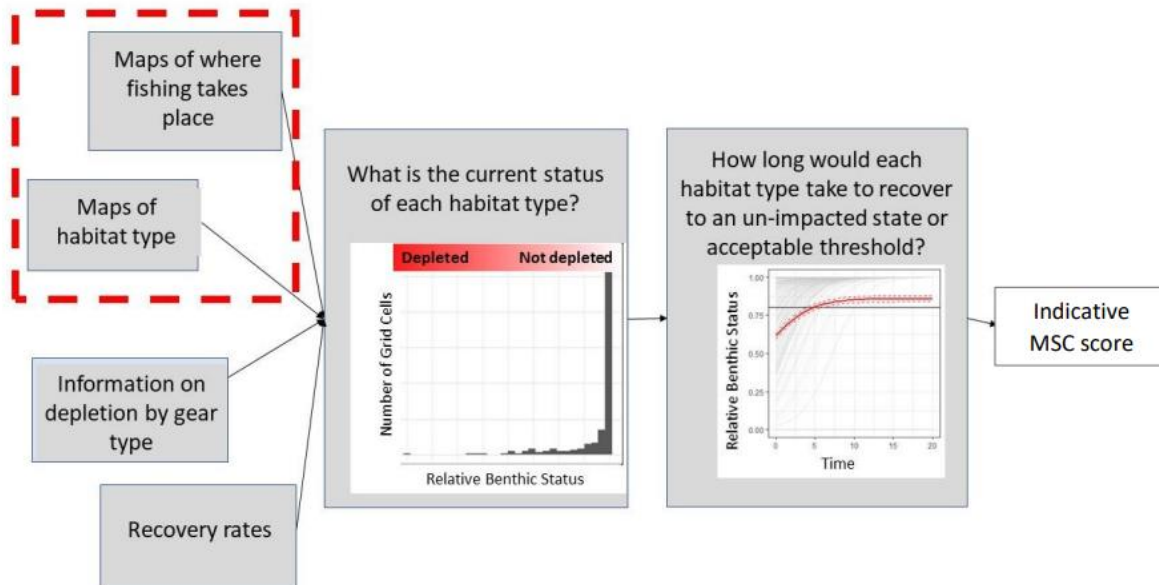
Description of the tool

The Marine Stewardship Council's Benthic Impacts Tool is again a "fishery-based" approach, like the STECF method and different to all other non-LCA methods described so far. A major difference between the MSC Benthic Impacts Tool and the STECF method, is that the former incorporates some quantitative data, as opposed to solely qualitative assessments made by expert panels in the STECF approach.

The MSC tool also combines information on gear severity with information on habitat sensitivity, but this is done in two different ways (Figure 5a). Based on reviews and meta-analyses of primary literature, the depletion rates of various key benthic organisms by different gear types is pre-determined across various habitat types. The recovery rates of these key organisms from within the

various habitats is also determined from primary literature and are pre-determined in the tool. Users of the tool provide the more quantitative data by submitting spatial data layers specific to their fishery.

Figure 5a. Overview schematic of the inputs and outputs of the Marine Stewardship Council's Benthic Impact Tool.



The red dashed box indicates input data required per fishery.

Source: Figure is taken directly from MSC (2022) with explicit permission from the copyright holder.

The per-fishery data required are maps covering the full spatial extent of the fishery, and complimentary maps of the habitat types across this extent. Furthermore, to ensure appropriate estimates of recovery times, context specific benthic community biological information can be provided to override the generic “precautionary” values that are provided as default within the tool.

With these input data the MSC tool estimates community biomass recovery times utilizing a simple population model that incorporates species’ sensitivity to specific gears, and life-history traits such as longevity. These are assessed across different habitats and across space within the domain of the user provided maps, and the ratio of impact and recovery are used to report a metric named Relative Benthic Status (RBS) that can be compared internally across the study domain for individual gears. Finally, the recovery times per habitat are calculated and the probability distributions of these are utilized to determine the MSC scores per habitat. There are four levels of scores, according to the probability that each habitat in the assessment area will not recover to 80% of the estimated unfished value (Table 12a).

Table 12a. MSC Benthic Indicator Tool final

Performance indicator(s)	SG60	SG80	SG100
2.3.1 (FS v3.0) & 2.4.1 (FS v2.01)	“Unlikely” = < 40th percentile	“Highly unlikely” = < 30th percentile	Evidence of “highly unlikely” = < 20th percentile

Scores (column headers) and the corresponding labels and probability classifications used in the different versions of the tool.

Source: MSC (2022).

Limitations to the method

The MSC Benthic Indicator Tool begins to quantify the impact of individual fisheries on the seabed, which is a step forward from the other methods reviewed here. However, there are some important limitations to the tool. Namely, the tool is built on a limited set of gears, the pre-parameterization of the population growth models is primarily derived from temperate, sedimentary habitats, the simple population growth models do not account for migration or dispersal, nor do they account for habitat suitability or trophic interactions, and the model assumes that the fishing footprint provided is stable over time, not accounting for any temporal variation.

The limited set of gears for which impacts have been evaluated (namely: hydraulic dredges, towed dredges, beam trawls or otter trawls) precludes this method from providing scores for many different fishing activities that are sources of seafood for the European market, at least until comprehensive studies on their benthos depletion rates are available.

The reliance on the default biological data has two drawbacks, first these data are only representative of sedimentary habitats and most likely best suited to temperate clines. This means that this tool is not appropriate for biogenic reefs, or rocky habitats. Second, even in temperate, sedimentary substrate contexts, the default parameters are conservative in nature and, thus, may not reflect the true community response in each context. To overcome this limitation users of the tool would have to have access to substantial amounts of benthic sampling data from the context of their fishery, and species longevity defined for the local benthic organisms. This is unlikely to be available in many locations.

The main indicator of overall impact derived from the tool is based on probability of habitat recovery over time; however, the model excludes any effect of migration / reproduction across grid cells or from outside the assessment area, thus overestimating recovery time, which will be aided by dispersal of various life-history stages. Furthermore, the suitability of habitat across all grid cells for the given communities are assumed equal, which may not reflect the natural variation in recovery rates.

Finally, the assessment does not consider changes in fishing patterns over time, thus the assessment is sensitive to the amount of data that is aggregated into the effort map which may mask local temporal peaks in effort or over represent the spatial extent of the fishery, depending on how long of a timeseries is used.

In addition to these inherent limitations to the modelling framework and assumptions, the data requirements may be too onerous for some fisheries, as the tool requires benthic habitat maps, as well as detailed, per-gear category, fishing effort data.

Summary

The key message of this Section 4 is that (and similarly to findings for the depletion status dealt in Section 3), regarding the assessment of the seabed impact, several methods have been described both in the LCA and non-LCA sections, but only few comply with the requirement of applicability to any type of catch, both domestic and imported, based on CMO data. As such, in Section 5, the only methods retained for evaluation are Langlois et al. (2015), Woods & Verones (2019) and Pr eat et al. (2021) for the LCA, and the STECF (2023) method and the MSC benthic impact tool (MSC, 2022), for the non-LCA.

5. Analysis of the methods with regards to evaluation criteria

This section presents a detailed evaluation of some of the methods described in Sections 3 and 4, based on the evaluation criteria described in Section 2.

Evaluation

LCA methods – depletion of targeted resources

We presented and assessed 2 publications: Hélias et al. (2018) measures the impact on natural resources; Hélias et al. (2023) version presents the measure of impact on species, an “ecosystem quality”-like approach. Both methods are not a measure of overfishing, as they consider an impact from “underfished” species as well. It is rather a measure of natural resource depletion (Hélias et al., 2018) or potential impact on ecosystem quality (Hélias et al., 2023). Both methods present an intrinsic approach to stocks: any fish removal from nature has an impact. Both versions are compatible with LCA and EF. However, a normalization has not been performed, but a discussion with authors suggest that it is easily feasible based on FAO catch data. Hélias et al. (2023) compute characterization factors at two levels: the level of (eco)regions, where CFs are expressed either in species*year or in PDFreg*year; and the global level where CFs are expressed in PDFglo*year. CFs are provided for target fish species. The relation between CFreg and CFglo is computed using Verones et al. (2017). An upcoming publication suggests providing default CFs (at regional and global level) that include discards (Stanford-Clark et al., submitted). This publication is also discussing further alignment with GLAM using the global extinction probability (GEP) (Verones et al., 2022).

Numeric values for CFs are publicly available in the Supplementary Information of Hélias et al. (2023), based on input data that is a combination of species or species group (depending on data availability) and on FAO major area. There are also default values for very broad categories (e.g., “marine species, nei”). The method displays a high number of characterization factors, based on information of species or species group on the one hand (n~1800), and FAO marine area (n=18) on the other (i.e., ~5000 CFs). CFs are applied on a mass of a given species fished in a given FAO area, information that has to be collected in LCI from the relevant actors (i.e., fishing company, fisheries, or background LCI databases).

The databases from which characterization factors are constructed include well-recognized international institutions (i.e., FAO and FishBase). However, the CMSY algorithm used to compute missing data are controversial within fishing management scientists (Gaillet et al., 2023). Stanford-Clark et al. (submitted) recomputes CFs for a total of 5575 FAO stocks and 1866 non-spatialised species using the CMSY+ algorithm (Froese et al., 2017, 2019).

There is no stock assessment per se, but rather an assessment of a given species status in an FAO area. Computing CFs on specific stock data is however feasible and has been done in Hélias et al. (2023), as shown in Figure 6a, displaying, in a few examples, consistency of results and in Wermeille et al. (2024). Stanford Clark et al (submitted) explicitly states that the method can be applied with (RFMO) stock data.

Uncertainty is reported in two ways:

Quantitatively for each characterization factor: uncertainty on each characterization factor is reported for confidence interval bounds, based on biomass uncertainty.

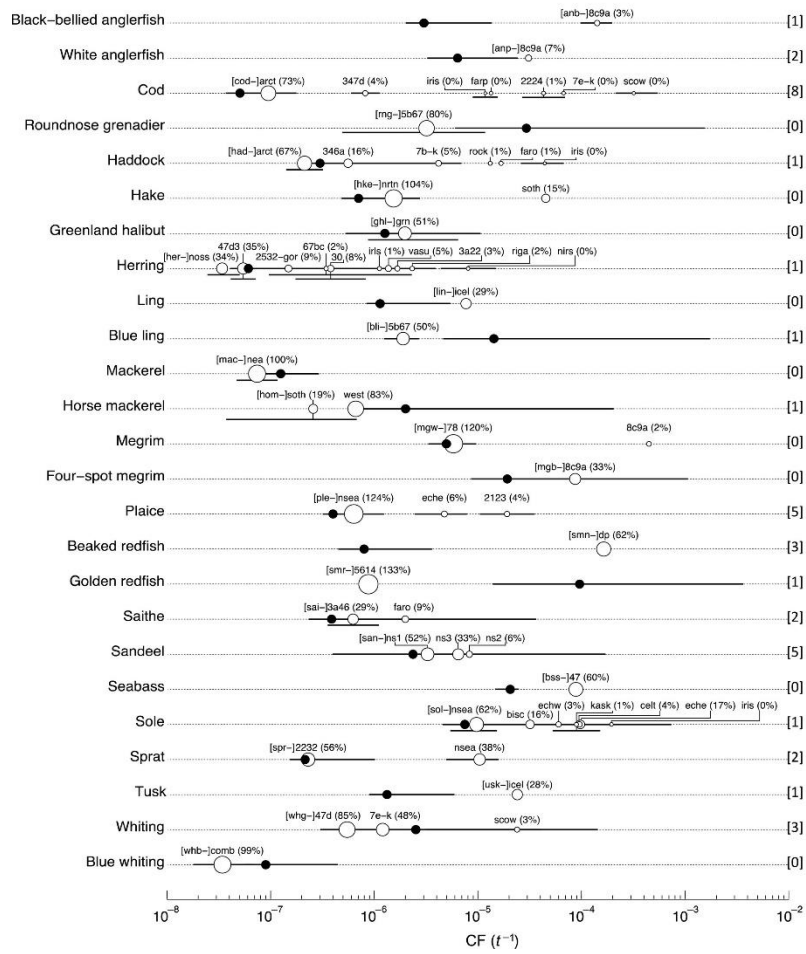
Qualitatively for each pair (species /species group; FAO area): they are classified into 4 classes (I, II, III, IV) depending on the robustness of underlying data for computing CMSY calculation, class I being the least uncertain one. 75% of fished species/species groups are in Class I, II and III.

The consistency of time series is ensured as follows: Hélias et al. (2018) works with 2015 (2014-2016) average data, whereas Hélias et al. 2023 works with 2018 (2017-2019) average data. In an upcoming publication, the authors recommend an update of characterization factors every 5 years to ensure a balance between representativity and applicability (i.e., delivery to LCA software tools).

Temporal and spatial resolution are of vital importance in LCIA. Temporal resolution is anticipated to be every 5 years, for an acceptable trade-off between changes in a “living resources” and relative stability of characterization factors for practitioners (Stanford-Clark et al., submitted). Spatial resolution is based on the areas defined by FAO (i.e., n=18). The method is applicable to stocks when fishery data is available (C and B). This is an open possibility in the method, applied in Hélias et al. (2018) for 25 species. Application to stock specific data is further suggested in an upcoming publication (Stanford-Clark et al. submitted) for more precise computation. This suggestion has been applied e.g. as well as in Wermeille et al. (2024).

It should be noted that the Hélias et al. (2023) model for natural biotic depletion was not included in GLAM 3 due to a series of methodological discrepancies with other environmental impact categories included in the GLAM method. More specifically, it was the belief of team of experts in GLAM that the computational framework of the Hélias et al. (2023) indicator did not comply with other categories being developed in which species richness was depicted rather than species abundance. In this sense, GLAM experts decided in September 2023 to leave the indicator on hold in terms of its inclusion in GLAM awaiting further methodological advances.

Figure 6a. Comparison of CFs computed from FAO 27 area data (black circles) and from ICES data (white circles) for 25 species.



Source: Figure extracted from Hélias et al. 2018. ³⁹

³⁹ Right to copy figure agreed by Arnaud Hélias on 18 January 2023

Table 13a. Evaluation of Hélias et al 2018 and Hélias et al. 2023 according to criteria defined in section 2.2.

	Hélias et al. 2018 "Biotic Depletion Potential"	Hélias et al. 2023 CFs regional impacted species at regional level	Hélias et al. 2023 CFs global Impacted species at global level
Generality of the method, such that it can apply to all products, both domestic and imported	2089 unique names (species/species groups) are covered across 18 FAO marine areas yielding 5070 CFs.	1804 unique names (species/species groups) are covered across 18 FAO marine areas yielding 4498 CFs.	1804 unique names (species/species groups) are covered across 18 FAO marine areas yielding 4498 CFs.
Ability to cover data-poor situations and missing stock assessments	<p>The method refers to the main stock parameters of B biomass and C (catches) and Hélias's formula for CFs applies to stocks.</p> <p>In the operationalization of his method, the publication proposes to prioritize completeness over precision (specific identification of stocks with available RFMO data) and hence provides a thorough list of CFs for x species and 18 FAO major fishing areas based on the CMSY algorithm. For stocks which have precise C and B data from RFMO, Hélias's formula is straightforward and has been applied and compared to CMSY.</p>	<p>The method refers refer to the main stock parameters of B biomass and C (catches) and Hélias's formula for CFs applies to stocks.</p> <p>In the operationalization of his method, the publication proposes to prioritize completeness over precision (specific identification of stocks with available RFMO data) and hence provides a thorough list of CFs for x species and 18 FAO major fishing areas based on the CMSY algorithm. For stocks which have precise C and B data from RFMO, Hélias's formula is straightforward.</p>	<p>The method refers refer to the main stock parameters of B biomass and C (catches) and Hélias's formula for CFs applies to stocks.</p> <p>In the operationalization of his method, the publication proposes to prioritize completeness over precision (specific identification of stocks with available RFMO data) and hence provides a thorough list of CFs for x species and 18 FAO major fishing areas based on the CMSY algorithm. For stocks which have precise C and B data from RFMO, Hélias's formula is straightforward.</p>
Type and availability of input data	<p>The relevance of the assessment is determined qualitatively following the approach of Hélias et al. (2018) and Hélias (2019). Results are briefly presented here, ranging from the most reliable to the least trusted:</p> <ul style="list-style-type: none"> • Class I corresponds to marine fish stocks with only one species, which have been fully assessed with the CMSY algorithm (47% of global catches according to Hélias et al 2018) • Class II brings several groups together, also assessed with CMSY (15% of global catches according to Hélias et al 2018). Class II.a is composed of multispecies marine fish stocks with not more than five species. Class II.b lists non-fish 	<p>The relevance of the assessment is determined qualitatively following the approach of Hélias et al. (2018) and Hélias (2019). Results are briefly presented here, ranging from the most reliable to the least trusted:</p> <ul style="list-style-type: none"> • Class I corresponds to marine fish stocks with only one species, which have been fully assessed with the CMSY algorithm (47% of global catches according to Hélias et al. 2018) • Class II brings several groups together, also assessed with CMSY (15% of global catches according to Hélias et al. 2018). Class II.a is composed of multispecies marine fish stocks with not more than five species. Class II.b lists non-fish 	<p>The relevance of the assessment is determined qualitatively following the approach of Hélias et al. (2018) and Hélias (2019). Results are briefly presented here, ranging from the most reliable to the least trusted:</p> <ul style="list-style-type: none"> • Class I corresponds to marine fish stocks with only one species, which have been fully assessed with the CMSY algorithm (47% of global catches according to Hélias et al 2018) • Class II brings several groups together, also assessed with CMSY (15% of global catches according to Hélias et al 2018). Class II.a is composed of multispecies marine fish stocks with not more than five species. Class II.b lists non-fish mono-species stocks (crustacean, mollusc...). Class II.c encompasses mono-species inland stocks.

	Hélias et al. 2018 "Biotic Depletion Potential"	Hélias et al. 2023 CFs regional impacted species at regional level	Hélias et al. 2023 CFs global Impacted species at global level
	<p>mono-species stocks (crustacean, mollusc...). Class II.c encompasses mono-species inland stocks.</p> <ul style="list-style-type: none"> • Class III (13% of global catches according to Hélias et al 2018) is similar to class II but with multispecies marine fish stocks with more than five species (III.a), non-fish multispecies stocks (III.b) and multispecies inland stocks (III.c). • Class IV stocks (25% of global catches according to Hélias et al 2018) are not directly assessed due to poor data quality. Global aggregated values are used, at species level (IV.a) or group level when values for species are not available (IV.b)," 	<p>mono-species stocks (crustacean, mollusc...). Class II.c encompasses mono-species inland stocks.</p> <ul style="list-style-type: none"> • Class III (13% of global catches according to Hélias et al 2018) is similar to class II but with multispecies marine fish stocks with more than five species (III.a), non-fish multispecies stocks (III.b) and multispecies inland stocks (III.c). • Class IV stocks (25% of global catches according to Hélias et al 2018) are not directly assessed due to poor data quality. Global aggregated values are used, at species level (IV.a) or group level when values for species are not available (IV.b)," 	<ul style="list-style-type: none"> • Class III (13% of global catches according to Hélias et al 2018) is similar to class II but with multispecies marine fish stocks with more than five species (III.a), non-fish multispecies stocks (III.b) and multispecies inland stocks (III.c). • Class IV stocks (25% of global catches according to Hélias et al 2018) are not directly assessed due to poor data quality. Global aggregated values are used, at species level (IV.a) or group level when values for species are not available (IV.b),"
Type and format of outcome (scoring, values etc)	The CFs span over nine orders of magnitude, but the interquartile range is two orders of magnitude.	The CFs span over ten orders of magnitude, but the interquartile range is less than two orders of magnitude.	The CFs span over 13 orders of magnitude, but the interquartile range is more restrained, also covering two orders of magnitude.
Completeness of scope	<p>AoP: natural resources. Covering Marine (<i>and Freshwater- not studies here</i>) fisheries.</p> <p>2089 unique names (species/species groups) are covered across 18 FAO marine areas yielding 5070 CFs. Default values are provided for Marine fishes nei, Marine crustaceans nei, Marine molluscs nei, enabling total coverage (albeit not with same precision) of all reported FAO stocks.</p> <p>Impact is modelled at stock level (FAO areas assumed to be "stock" level for CFs computed with CMSY)</p> <p>CF is equal to percentage of Depleted Stock Fraction (DSF)</p>	<p>AoP: ecosystem quality. Covering Marine (<i>and Freshwater- not studies here</i>) fisheries. Method has been developed in the context of GLAM 3, in accordance with GLAM guidelines.</p> <p>1804 unique names (species/species groups) are covered across 18 FAO marine areas yielding 4498 CFs. Default values are provided for Marine fishes nei, Marine crustaceans nei, Marine molluscs nei, enabling total coverage (albeit not with same precision) of all reported FAO stocks.</p> <p>Impact is modelled at stock level (FAO areas assumed to be "stock" level for CFs computed with CMSY based on an analogy with "Usetox" model, using fate and effect factor.</p> <p>- Environmental mechanism for fate factor (FF): model is linearized at steady state; for each stock,</p>	<p>AoP: ecosystem quality. Covering Marine (<i>and Freshwater- not studies here</i>) fisheries. Method has been developed in the context of GLAM 3, in accordance with GLAM guidelines.</p> <p>1804 unique names (species/species groups) are covered across 18 FAO marine areas yielding 4498 CFs. Default values are provided for Marine fishes nei, Marine crustaceans nei, Marine molluscs nei, enabling total coverage (albeit not with same precision) of all reported FAO stocks.</p> <p>Impact is modelled at stock level (FAO areas assumed to be "stock" level for CFs computed with CMSY) based on an analogy with "Usetox" model, using fate and effect factor.</p> <p>- Environmental mechanism for fate factor (FF): model is linearized at steady state; for each stock, FF is the inverse of the growth rate constant tempered by the inverse of relative biomass.</p>

	Hélias et al. 2018 "Biotic Depletion Potential"	Hélias et al. 2023 CFs regional impacted species at regional level	Hélias et al. 2023 CFs global Impacted species at global level
		<p>FF is the inverse of the growth rate constant tempered by the inverse of relative biomass.</p> <ul style="list-style-type: none"> - Environmental mechanism for Depleted Stock Fraction (effect factor - EF) is looking into two models: average and marginal ones: - Impact of removing biomass is the biomass derivative of Schaefer's equation (marginal approach). - The average approach is based on variation of stock for the time constant τ of the dynamical model (species dependent) and yields similar CFs to marginal. 	<ul style="list-style-type: none"> - Environmental mechanism for Depleted Stock Fraction (effect factor - EF) is looking into two models: average and marginal ones: - Impact of removing biomass is the biomass derivative of Schaefer's equation (marginal approach). - The average approach is based on variation of stock for the time constant τ of the dynamical model (species dependent) and yields similar CFs to marginal.
Environmental relevance	<p>input in kg biomass removed from the sea of given stock. Stock is defined as a species in a given FAO area (18 areas).</p> <p>Underlying approach is "instrumental" ecosystem functioning, based on "use-to-stock/use-to-availability" approach (Sonderreger et al. 2017)</p> <p>CFs based on average 2013-2015 data are provided in Supplementary material</p>	<p>input in kg biomass removed from the sea of given stock. Stock is defined as a species in a given FAO area (18 areas).</p> <p>The underlying approach is intrinsic biodiversity meaning: "any biomass removed from the sea by humans is changing the ecosystem from a 'natural (no human intervention)' state". Cause-effect relationship between biomass removal of a single fished species and biodiversity impact in PDF is not straightforward, as only effect on single fished species is accounted for. On-going research is making the link of one species removal with the entire ecosystem disturbance through species interactions (predation, competition...) (Stanford-Clark et al., 2023)</p> <p>Regional CFs are expressed either in</p> <ul style="list-style-type: none"> - species .year /kg fished biomass (or number of individuals depending on FAO reporting nomenclature). - CFs do not account for discards when fishing. - or in PDF reg.yr/kg fished biomass (or number of 	<p>input in kg biomass removed from the sea of given stock. Stock is defined as a species in a given FAO area (18 areas).</p> <p>The underlying approach is intrinsic biodiversity meaning: "any biomass removed from the sea by humans is changing the ecosystem from a "non-human-intervention" state.</p> <p>Global CFs are expressed in in PDF glo.yr/kg fished biomass. Global PDF is equal to regional PDF times the share of total species biomass present in the FAO area.</p> <p>CFs based on 2015 data are provided in Supplementary Material</p> <p>CFs do not account for discards when fishing.</p>

	Hélias et al. 2018 "Biotic Depletion Potential"	Hélias et al. 2023 CFs regional impacted species at regional level	Hélias et al. 2023 CFs global Impacted species at global level
		individuals depending on FAO reporting nomenclature) (total marine species ~233 000). CFs based on 2015 data are provided in Supplementary material	
Scientific robustness	Peer-reviewed paper. Underlying input data is well documented. For the sake of completeness, model input parameters are based on CMSY algorithm. Thorough computation and reporting of uncertainties.	Peer-reviewed paper. Underlying input data is well documented. For the sake of completeness, model input parameters are based on CMSY algorithm. Missing link is species interaction (upcoming). Computation and reporting of uncertainties.	Peer-reviewed paper. Underlying input data is well documented. For the sake of completeness, model input parameters are based on CMSY algorithm. Missing link is species interaction (upcoming). Computation and reporting of uncertainties.
Documentation, transparency, and reproducibility	Model well documented, accessible, and reproducible. New CFs can be computed with the same model, which is important based on the need for timely update of CFs based on the evolution of stock status and biomass. Model based on average 2013-2015 FAO Catch (C) and Biomass (B) data, and FishBase maximum intrinsic rate of population increase (r)	Model well documented, accessible, and reproducible. New CFs can be computed with the same model, which is important based on the need for timely update of CFs based on the evolution of stock status and biomass. CFs values are accessible in supplementary information, based on FAO 2015 Catch (C) and Biomass (B) data, and FishBase maximum intrinsic rate of population increase (r), extrapolated with CMSY algorithm.	Model well documented, accessible, and reproducible. New CFs can be computed with the same model, which is important based on the need for timely update of CFs based on the evolution of stock status and biomass. CFs values are accessible in supplementary information, based on FAO 2015 Catch (C) and Biomass (B) data, and FishBase maximum intrinsic rate of population increase (r), extrapolated with CMSY algorithm.
Applicability	Input data in kg of catches (of a given species within an FAO area) has to be included in inventories, either from primary data or from secondary databases. In European markets, this information is mandatory for fishing vessels. Information (species, FAO area) must be provided to the end consumer for raw	Input data in kg of catches (of a given species within an FAO area) has to be included in inventories, either from primary data or from secondary databases. In European markets, this information is mandatory for fishing vessels. Information (species,	Input data in kg of catches (of a given species within an FAO area) has to be included in inventories, either from primary data or from secondary databases. In European markets, this information is mandatory for fishing vessels. Information (species, FAO area) must be provided to the end consumer for raw products (excluding

	Hélias et al. 2018 "Biotic Depletion Potential"	Hélias et al. 2023 CFs regional impacted species at regional level	Hélias et al. 2023 CFs global Impacted species at global level
	<p>products (excluding recipes) - live weight/net weight conversion enables to retrieve catches.</p> <p>Implementation in market relevant-software tools is feasible (constraint is # of species, but this is indicator dependent).</p>	<p>FAO area) must be provided to the end consumer for raw products (excluding recipes) - live weight/net weight conversion enables to retrieve catches.</p> <p>Implementation in market relevant-software tools is feasible (constraint is # of species, but this is indicator dependent).</p>	<p>recipes) - live weight/net weight conversion enables to retrieve catches.</p> <p>Implementation in market relevant-software tools is feasible (constraint is # of species, but this is indicator dependent).</p>
Temporal representativeness, resolution, and time horizon	<p>Resolution of the input is clearly stated based on average 2013- 2015 stock data (catches and biomass).</p> <p>Modeling body is INRAe unit in Montpellier</p>	<p>Resolution of the input is clearly stated based on 2015 stock data.</p> <p>Modelling body is INRAe unit in Montpellier</p>	<p>Resolution of the input is clearly stated based on 2015 stock data.</p> <p>Modelling body is INRAe unit in Montpellier</p>
Spatial resolution	<p>Strong assumption in operationalisation of the characterization factors provided is that FAO area corresponds to stocks. Multiple stocks of same species in each FAO area have the same CFs. Spatial resolution of inputs and outputs factors is FAO areas.</p> <p>It is possible to compute more precise CFs for stocks based on RFMO input data for C and B and r values (Fishbase). Computations are straightforward, provided data series are available. examples for ICES stocks are illustrated in the publication.</p>	<p>Strong assumption in operationalisation of the characterization factors provided is that FAO area is an ecoregion (for PDF_{reg} computational purposes) and that stocks are defined by the couple {species-FAO Major area}.</p> <p>Spatial resolution of inputs and outputs factors is FAO areas.</p> <p>It is possible to compute more precise CFs for stocks based on RFMO input data.</p>	<p>Strong assumption in operationalisation of the characterization factors provided is that FAO area is an ecoregion (for PDF_{reg} computational purposes) and that stocks are defined by the couple {species-FAO Major area}.</p> <p>Spatial resolution of inputs and outputs factors is FAO areas.</p> <p>It is possible to compute more precise CFs for stocks based on RFMO input data.</p>
Compatibility with Environmental Footprint (EF) method	<p>LCIA method, fully compatible with EF.</p> <p>Normalisation factors are missing. However, based on oral exchanges with authors, it is easily feasible.</p> <p>Elementary flows are not included yet in LCA inventories, but the CFs cover all type of (species x FAO area) flows, as it also provides default values in case no specific CF exists for species /species group.</p>	<p>LCIA method, fully compatible with EF.</p> <p>Normalisation factors are missing. However, based on oral exchanges with authors, it is easily feasible.</p> <p>Elementary flows are not included yet in LCA inventories, but the CFs cover all type of (species x FAO area) flows, as it also provides default values</p>	<p>LCIA method, fully compatible with EF.</p> <p>Normalisation factors are missing. However, based on oral exchanges with authors, it is easily feasible.</p> <p>Elementary flows are not included yet in LCA inventories, but the CFs cover all type of (species x FAO area) flows, as it also provides default values in case no specific CF exists for species /species group.</p>

	Hélias et al. 2018 "Biotic Depletion Potential"	Hélias et al. 2023 CFs regional impacted species at regional level	Hélias et al. 2023 CFs global Impacted species at global level
		in case no specific CF exists for species /species group.	
Stakeholder acceptance and suitability for communication in a business and policy contexts.	It is not easy for fishery scientists or professionals to get a concrete meaning for CFs, as they are used to stock-based approaches which do not take into account the size of the stocks (biomass volume) in the calculation. They are used to an instrumental approach (MSY, Kobe). CMSY algorithm is subject to controversies in fisheries science.	It is not easy for fishery scientists or professionals to get a concrete meaning for CFs, as they are used to stock-based approaches which do not take into account the size of the stocks (biomass volume) in the calculation. They are used to an instrumental approach (MSY, Kobe). CMSY algorithm is subject to controversies in fisheries science.	It is not easy for fishery scientists or professionals to get a concrete meaning for CFs, as they are used to stock-based approaches which do not take into account the size of the stocks (biomass volume) in the calculation. They are used to an instrumental approach (MSY, Kobe). CMSY algorithm is subject to controversies in fisheries science.
Others	Marginal approach	Marginal and average CFs are provided (same values) 1800 marine species and species groups, 18 FAO areas covered	Marginal and average CFs are provided (same values) 1800 species and species groups, 18 FAO areas covered
	CFs computed with CMSY are provided in SI Global coverage and straightforward usability	Test of method conducted within the French GT Mer (Gaillet et al. 2023) and in Wermeille et al. 2024. Global coverage and straightforward usability	Tested internally. Global coverage and straightforward usability

Hélias et al. 2023 is split into 2 columns depending on the assessment level of method: (fraction of) species impacted are considered at regional level (CFs regional) and at at global level (CF global).

Source: Authors' own elaboration.

Non-LCA methods – depletion of targeted resources

As was described in Chapter 3, all methods classifying fish stocks with regards to overfishing and stock depletion status fail on the major criteria of completeness of scope, in the sense that they can only apply to fish products for which an official stock assessment status is available from a dedicated RFMO. This precludes providing environmental assessment for a potentially great number of products entering EU market, both imported and domestic.

As such, we acknowledge that the proposal made by STECF (EWG 20-05, 22-12 and 23-18) is the widest ranging and most generic method – which makes also intuitively sense since this approach was exactly developed for the purpose of environmental grading of seafood products entering the EU market, on request from the EU Commission. Table 14a summarizes the evaluation criteria.

Table 14a. Evaluation of STECF (2023) method according to criteria defined in Section 2.2.

Criteria	STECF (2020, 2022, 2023) Stock status indicator
Type and availability of input data	<p>All data come from public information.</p> <ul style="list-style-type: none"> - Extraction of data from stock assessments: Where possible (ICES, NOAA), data extraction is automatized using webservices. For other assessments, data collection is mutualised with other routine STECF data work (e.g. (Balance-capacity EWG database). - Data sources for existing catch or effort advice and extraction of data: Heterogeneous across RFMOs. Data exists in public assessment reports, but their automatic extraction and update is still challenging - Data sources for regional/global IUCN classifications, species sensitivity and extraction of data (System 1): IUCN data is available through a webservice easily usable through a dedicated R package (rredlist -https://github.com/ropensci/rredlist) and can also be downloaded from https://www.iucnredlist.org/search. Where the assessment was either not available or the assessment was Data Deficient (DD), the sensitivity of the species was assessed using assessments based on life history traits. The list of species sensitivity available for download at https://doi.org/10.11583/DTU.21063193. <p>As of December 2023, data are already available for several hundreds of stocks and more than 1000 sensitive species in the prototype database.</p>
Type and format of outcome (scoring, values etc)	<p>The STECF aims a providing a Letter-categorical scoring (A to E), regardless of product weight. Exploratory work is underway for translating this into a quantitative “LCA-like” indicator accounting for weight (Gascuel, pers. com)</p>
Completeness of scope	<p>Impact is modelled at stock level (FAO areas assumed to be “stock” level). The method is designed to provide a grading letter for any type of fisheries product, both domestic and imported, and with a without official stock assessment, and based on public and standardised information from international databases. As of December 2023, data are already available for several hundreds of stocks and more than 1000 sensitive species in the prototype database</p>
Environmental relevance	<p>The MSY approach for describing stock status is the worldwide reference used in all international standards, including FAO, EU CFP and EU MSFD, RFMOs and UN SDG 14.4. IUCN is also a global worldwide reference for describing the environmental conservation status of a species in a given area.</p>
Scientific robustness	<p>The grading method has been recently developed across a suite of 3 Experts Working Groups gathering EU fisheries experts, in collaboration with the EU Commission (DG Mare). Not published in peer-review article yet</p> <p>The method makes only use of official and validated stock assessment data (from RFMOs and IUCN) when available, and of simple life history traits sensitivity assumptions when not.</p> <p>The exploratory “LCA-like” step is still in development and has not been tested yet for robustness.</p>

Criteria	STECF (2020, 2022, 2023) Stock status indicator
Documentation, transparency, and reproducibility	The grading model is well documented in EWG reports, accessible and reproducible. All data sources are from official organisations, are public and, for a great number of them, are easily updatable through automated webservices. A prototype database is already available and public. The exploratory “LCA-like” step is still in development and would require further documentation.
Applicability	A grading letter (A-F) can be provided for any product entering EU market, as the mandatory CMO information allows linking with a stock, or if not, to an area where IUCN information can be retrieved. The exploratory “LCA-like” method linking stock status uses Input data in kg of catches (of a given species within an FAO area), which has to be included in inventories, either from primary data or from secondary databases
Temporal representativeness, resolution, and time horizon	Stock status can be updated every year, thanks to the already existing automated built-in facilities, although some comprehensive work is still needed for manually updating the few data that cannot be automatically retrieved yet. Sensitive data do not require regular update as these are based on life history traits
Spatial resolution	Official stock (species / area) data is used, without assumption for missing values. The system 1 approach allows providing species-based information in the case of missing spatial information
Compatibility with Environmental Footprint (EF) method	The letter-grading is categorical, providing a qualitative information, thus not compatible with EF method. An exploratory “LCA-like” approach in development (Gascuel, pers. com) is proposed for providing a quantitative indicator linking with product weight.
Stakeholder acceptance and suitability for communication in a business and policy contexts	All systems engaged in sustainable fisheries management worldwide (fishers, managers, NGOs, scientists, international organisations) are fully familiar with the stock-based MSY reference approach and with the IUCN. The letter-grading method is simple and transparent, easy to explain and to understand, without gap-filling or assumptions for missing data. All data sources are public and from official database, so it is easy to track and check the computations. Previous iterations of the work (STECF 2020, 2022) have already been presented in a number of stakeholders for a, advisory councils etc.

Source: Authors' own elaboration.

LCA methods – seabed impact

Three of the six LCIA models that were identified above for seabed impact were quantitatively analysed according to the criteria and subcriteria presented in previous sections (see Table 15a). However, two additional models, Abdou et al. (2017) and Farmery et al. (2017) were also retained in this section for the sake of comparability. The main reason behind this choice was the year the model was developed (i.e., from 2015 onwards) and the degree of sophistication the LCIA models are expected to show (i.e., not only the impacted area or trawled area as output factors). As mentioned above, in contrast to overfishing indicators, seabed impact models did not present specific names in most of the cases, so they are described when discussed.

Table 15a. Evaluation of Langlois et al. (2015), Woods & Verones (2019) and Pr at et al. (2021) according to criteria defined in section 2.2.

Criteria	Langlois et al. (2015)	Woods & Verones (2019)	Pr�at et al. (2021)
Completeness of scope	-Midpoint LCIA model -Impact pathway with interventions, direct impacts, midpoint and endpoint level indicators, and areas of protection.	-Endpoint LCIA model -Impact pathway based on Mil� i Canals (2007). -One mechanism of ecosystem quality AoP covered in the model	-Endpoint LCIA model -Impact pathway based on Langlois et al. (2014b) and Mil� i Canals (2007).

Criteria	Langlois et al. (2015)	Woods & Verones (2019)	Préat et al. (2021)
	<p>Based on Langlois et al. (2014b) and Milà i Canals (2007).</p> <ul style="list-style-type: none"> -One mechanism of ecosystem quality AoP covered in the model. -Input data aggregated in LCI since the extraction of biotic resources from an ecosystem area. -Model adaptable to spatial and temporal explicit evaluation. 	<ul style="list-style-type: none"> -Input data aggregated in LCI since the extraction of biotic resources from an ecosystem area. -Model adaptable to spatial explicit evaluation. 	<ul style="list-style-type: none"> -One mechanism of ecosystem quality AoP covered in the model -Input data aggregated in LCI since the extraction of biotic resources from an ecosystem area. -Model adaptable to spatial and temporal explicit evaluation.
Environmental relevance	<ul style="list-style-type: none"> -Intrinsic ecosystem functioning approach -Input data must be in m². -Output factors in primary production units (kg C · year⁻¹) -Biotic primary production mechanism in the impact pathway is included with acceptable quality. 	<ul style="list-style-type: none"> -Intrinsic ecosystem functioning approach -Input data must be in m². -Output factors in potential disappeared fraction of species (PDF) -Regional species diversity mechanism in the impact pathway is included with acceptable quality. 	<ul style="list-style-type: none"> -Intrinsic ecosystem functioning approach -Input data must be in m². -Output factors in potential disappeared fraction of species (PDF · year) -Global species diversity mechanism in the impact pathway is included with acceptable quality.
Scientific robustness & certainty	<ul style="list-style-type: none"> -Peer review science-based model -Uncertainty qualitatively discussed 	<ul style="list-style-type: none"> -Peer review science-based model -Impact pathway reflects the latest knowledge in the field and can be further improved. -Sensitivity analysis carried out 	<ul style="list-style-type: none"> -Peer review science-based model -Impact pathway reflects the latest knowledge in the field and can be further improved. -Characterization factors are not computed and so cannot be verified. -Uncertainty qualitatively discussed
Documentation, transparency & reproducibility	<ul style="list-style-type: none"> -Published and accessible documentation: -Model -Characterization model -Modelling parameters -Set of characterization factors -Additional characterization factors can be generated by updating or integrating optimized primary production required for ecosystem and benthic response or by including FAO major fishing areas instead of typos of ecosystems. 	<ul style="list-style-type: none"> -Published and accessible documentation: -Model -Characterization model -Modelling parameters -Set of characterization factors -Additional characterization factors can be generated by expanding the study area to ecosystems beyond European region and beyond fishing activities. 	<ul style="list-style-type: none"> -Published and accessible documentation: -Model -Characterization model -Multiple characterization factors for diverse systems can be generated with the framework.

Criteria	Langlois et al. (2015)	Woods & Verones (2019)	Pr�at et al. (2021)
Applicability	<ul style="list-style-type: none"> -Characterization factors are ready to be applied in LCA software. -Direct resource extraction actors can produce LCI data. 	<ul style="list-style-type: none"> -Nomenclature must be conformed to ILCD rules (e.g., scientific names in Latin) -Characterization factors are ready to be applied in LCA software, but only for Europe. -Direct resource extraction actors can produce LCI data. 	<ul style="list-style-type: none"> -Characterization must be developed by specialized LCA practitioners. -Direct resource extraction actors can produce LCI data.
Temporal representativeness, and resolution, time horizon	<ul style="list-style-type: none"> -Temporal coverage of the characterization model not stated. -Time horizon: 20, 100 and 500 years -Inputs and outputs are based on annual time resolution -Modelling parameters gathered from sources from years 1980 – 2006 	<ul style="list-style-type: none"> -Temporal coverage of the characterization model not stated. -No time horizons considered -Seafloor transformation impact resolved on an annual basis -Modelling parameters gathered from sources from years 1985 – 2014 	<ul style="list-style-type: none"> -Temporal coverage of the characterization model not stated. -Time horizon: 500 years -Impact integrated over time -Modelling parameters not available
Spatial resolution	<ul style="list-style-type: none"> -High level of spatial representativeness -26 ecosystem types covered -Output factors resolved at ecosystem level 	<ul style="list-style-type: none"> -High level of spatial representativeness (Europe only). -17 marine ecoregions covered -Output factors resolved at marine ecoregion level 	<ul style="list-style-type: none"> -Low level of spatial representativeness -Potential to cover multiple ecoregions -Output factors potential resolution at ecosystem level
Stakeholder acceptance criteria	<ul style="list-style-type: none"> -Indicator (kg C) understood by LCA and non-LCA practitioners with little expertise in the field -No international scale bodies supporting the model or in charge of updates -Model requires normalization factors and weighting factors development in order to be compatible with EF method -Elementary flows units and resolution are compatible with EF method 	<ul style="list-style-type: none"> -Indicator (kg C) understood by LCA and non-LCA practitioners with little expertise in the field -No international scale bodies supporting the model or in charge of updates -Model requires normalization factors and weighting factors development in order to be compatible with EF method -Elementary flows units and resolution are compatible with EF method 	<ul style="list-style-type: none"> -Indicator (kg C) understood by LCA and non-LCA practitioners with little expertise in the field -No international scale bodies supporting the model or in charge of updates -Model requires normalization factors and weighting factors development in order to be compatible with EF method -Elementary flows units and resolution are compatible with EF method
Output factors	<ul style="list-style-type: none"> -Output factors are usable in LCA and non-LCA context -No case study conducted 	<ul style="list-style-type: none"> -Output factors are usable in LCA context -Case study: Capture of 1 kg of Norway lobster (<i>Nephrops norvegicus</i>) 	<ul style="list-style-type: none"> -Output factors are usable in LCA context -Case study: Deep-sea mining at Clarion Clipperton Fracture Zone

Source: Authors' own elaboration.

The completeness of scope criterion involves the cause-effect chain and the impact pathway mechanisms. Interventions such as shading, biotic-resource uptake, seabed destruction and artificial reef creation were thoroughly described in Langlois et al. (2015) within the sea use impact category they presented. In this study, a (free) biotic primary production midpoint category was selected to calculate environmental impacts in the ecosystem quality area of protection, based on a more general land use framework proposed by Milà i Canals (2007). The seafloor disturbance model presented by Woods and Verones (2019) is another of the models that has described a detailed impact pathway from disturbance of benthic communities to potentially disappeared fraction of species per area unit. The description of the mechanisms includes different types of intervention (i.e., single and repeated impacts) and variables such as habitat type, potential recolonizing individuals and hydrodynamic energy, which are later integrated in the computation of environmental impacts. In the same line, the global seafloor disturbance model by Pr at et al. (2021) is based on the work by Langlois et al. (2014b), which follows the same guidelines from Mil  i Canals et al. (2007). However, Pr at et al. (2021) also use the method from Woods & Verones (2019) to amplify the model to deep seafloor ecosystems and to assess global biodiversity impacts. These two models are very similar, and their ecosystem quality-oriented impact pathway is considered solid. No other models have presented or discussed cause-effect chains. Something to be considered is that the models presented by Abdou et al. (2017) and Farmery et al. (2017) depict midpoints in contrast to the Woods & Verones (2019) and Pr at et al. (2021) models that reach endpoints. In these two last models, double counting is avoided according to a brief discussion in their studies.

One important element to bear in mind when assessing environmental relevance criteria is the degree of mechanism inclusion in the LCIA model for seabed impact. Within the sea use impact category (Langlois et al., 2014b), the intervention seafloor destruction can be divided into two direct impacts: ecosystem fertility and habitat loss/creation. These impacts lead to biotic primary production, oceanic water purification and local/regional species diversity midpoint level indicators, which are expected to be related to the endpoint level indicators ecosystem services damage potential and biodiversity damage potential. Langlois et al. (2015) focused on ecosystem fertility direct impact, (free) biotic primary production as a midpoint indicator and ecosystem services damage potential as an endpoint indicator (i.e., for the ecosystem quality area of protection).

The models by Woods & Verones (2019) and Pr at et al. (2021) only address the direct impact related to habitat loss/creation. The difference between these models consists in the extension from regional environmental impacts (Woods & Verones, 2019) to global environmental impacts (Pr at et al., 2021) that are included in the computational framework. Concerning the input/output units of measure, it was observed that once characterization factors are determined, input data for every model requires an impacted area in m². In contrast, the study by Farmery et al. (2017) requires the mass of five-year average catches. Although the data are easily accessible in most cases, it must be noted that access to these data may be uneven throughout geographic locations and fisheries, as discussed in Section 3.

Output factors vary from one to another model. Langlois et al. (2015) and Abdou et al. (2017) present impact results in net primary production units kg C*year⁻¹; Farmery et al. (2017), in m²*year⁻¹; Woods & Verones (2019) and Pr at et al. (2021), in PDF and PDF*year. The last unit, PDF or potentially disappeared fraction of species, is the metric most commonly applied in more recent studies, especially those that aim at harmonizing the model constructed with current LCIA practice (e.g., the GLAM method). However, reporting in PDF presents some limitations, such as accounting for certain essential biodiversity variables, such as genetic composition, species populations, species

traits, and ecosystem functioning and structure from essential biodiversity variables (Pereira et al., 2013) are not covered by PDF metric indicators.

Regarding *documentation*, all the LCIA models analysed in this section were cited by other authors in academia. Considering that Langlois et al. (2015), Woods & Verones (2019) and Pr  at et al. (2021) are the only models with a clearly defined impact pathway, it could be stated that those models reflect the latest knowledge in the field. Although every other model is also prone to be improved, in the case of Abdou et al. (2017) and Farmery et al. (2017) this is related to further research in critical parts of the model. For instance, in Abdou et al. (2017), resuspension of solid matter previously deposited and benthic communities' response to that are lacking in the MERAMOD method employed. In Farmery et al. (2017), the scoring of an individual fishery may underestimate the impact in a collective fishery zone, which in theory might have higher impact. In addition, the transformation of hemeroby class to Naturalness Degradation Potential (NDP) is presented as linear but it is expected to find non-linear responses in some marine environments, especially after critical points (Selkoe et al., 2015). Regarding *certainty analysis*, sources of uncertainty are qualitatively discussed in Langlois et al. (2015), Abdou et al. (2017), Farmery et al. (2017) and Pr  at et al. (2021). Woods and Verones (2019), in contrast, presented sensitivity analysis and found correlation between varying parameters and characterization factors in the single impact approach with a strong influence of median recovery time and the recolonisers availability. Time between perturbation episodes firmly influenced repeated impact sensitivity. This suggests that Woods & Verones (2019) is the most appropriate LCIA models in terms of scientific robustness and certainty.

Regarding *input data*, information was found to be published and accessible in 5 LCIA models. Some models are quite limited in their data collection and CFs development since the information they use are gathered from specific case studies (e.g., 2 farms from Tunisia (Abdou et al., 2017) and prawn fisheries from Australia countries (Farmery et al., 2017). Although the models can be extended globally by modifying input data, the absence of CFs prevents non-expert practitioners from using the method. This is partially observed in Woods and Verones (2019) because CFs were created specifically for maritime areas in Europe. In this case, a wide range of modelling parameters are needed in order to extend the method beyond the geographical area of study (i.e., marine ecoregion area, benthic response, recovery times, hydrodynamic energy, stock of potential recolonisers, habitat substrate type and interval between disturbance events). The framework provided by Pr  at et al. (2021) only describes the modelling parameters, in which CFs are used and potential sources of this information can be found. No input data and no CFs are provided with exception to those linked to the Clarion Clipperton Fracture Zone case study. Langlois et al. (2015) stands out in this criterion since they present characterization factors for different ecosystem types. Hence, this last model, together with Woods & Verones (2019), satisfy the required standards.

Concerning *applicability*, the MERAMOD employed in Abdou et al. (2017) is a software that needs to be downloaded. Although it is freeware, it implies that time is required to learn and run experiments. Farmery et al. (2017) model requires stakeholders to have fishing/trawling/dredging and ecosystem background knowledge to evaluate hemeroby classification and, thus, allow CFs to be developed for a specific fishery. Parameters for characterization factor calculation in the model by Woods & Verones (2019) can be retrieved from the Supplementary Information referred to in the article, coming up as a more friendly method. As mentioned above, Pr  at et al. (2021) present a framework, and this means practitioners must quantify CFs and must collect input parameters to compute final environmental impacts. In contrast, our analysis suggests that mainly Langlois et al. (2015), but also Woods & Verones (2019), present a friendly framework for non-experts in LCA and seabed impact methods to apply. Characterization factors are available for use in Langlois et al. (2015) and they are easily downloadable and applicable by practitioners in case studies. Although

characterization factors are not directly presented in Woods & Verones (2019), modelling parameters can be retrieved from the Supplementary Information referred to in the article. In general, applicability criteria involve nomenclature aspects in the models. The ILCD nomenclature rules (EC et al., 2010b) mandates m^2 as reference, which is easily achieved since information from many industries can be retrieved in this unit. If the impacted area is not reported directly from industries, it can also be calculated with information from machinery, trawl nets and fishing effort. The latter two models already comply with these issues, making Langlois et al. (2015) and Woods & Verones (2019) convenient methods when assessing applicability.

Temporal coverage is not stated in Langlois et al. (2015), Woods & Verones (2019) or Pr at et al. (2021). Nevertheless, modelling parameters are gathered from multiple sources and from diverse years. Abdou et al. (2017) indicate the impact assessment considers 1 year of production and data was collected through field trips in 2015. Farmery et al. (2017) does not indicate temporal coverage, but their work shows that data were collected from 2010-2015. As can be seen, this latter model reflects better temporal representativeness since a recent period is considered. Woods & Verones (2019) method does not include time or time horizons from input flows. The time property is included in the repeated impact calculation by using mean recovery times to give time integrated PDF units as output factors. Pr at et al. (2021) stated the utilization of a time horizon of 500 years. Langlois et al. (2015) considered 20, 100 and 500 years for time horizons in their characterization factor modelling; hence, it appears as a suitable model in terms of temporal coverage.

Regarding spatial representativeness and resolution, Langlois et al. (2015) and Woods & Verones (2019) are useful when it comes to representing spatial scales of ecosystems. The former includes 26 ecosystem types, whereas the latter includes 17 marine ecoregions in Europe. Pr at et al. (2021) does not give a number, but the system resolution is lower than ecoregions because deep-sea regions are fragmentary and defined for higher levels than ecosystems. Abdou et al. (2017) and Farmery et al. (2017) have a spatial resolution of farms/fisheries. This can be advantageous since results could be expressed at a high level of resolution, but at the same time it could be a disadvantage because the representativeness of the characterization factors is low. With corresponding adjustments, these models can be extended to fisheries in other parts of the world. Expert practitioners are required for this task.

The indicators analysed share a common trait in terms of easiness to understand, as the concepts of biotic primary production, impacted areas and potential disappeared fraction of species are concepts that are reasonable to grasp by practitioners with limited experience in marine resources and ocean conservation, especially when linked to impacts related to fishing. In fact, some of these indicators are used in other contexts as well. For instance, human appropriation of net primary production (in mass of C * year⁻¹) is a provisional indicator used as a proxy to evaluate biosphere functioning in the framework of planetary boundaries (Richardon et al., 2023).

Compatibility with the EF methodology is of utter importance since LCIA models are planned to be incorporated into these recommendations to evaluate product and organisation environmental performance. In this sense, it is mandatory to use per capita normalization factors based on a global value (Sala et al., 2017) and EF weighting factors (Sala et al., 2018), both available online, to comply with EF norms. Regarding EF nomenclature, rules suggest elementary flows shall be aligned with an up-to-date reference package available in the developer's website (DG ENV, 2021). However, these suggestions are still under development since no fishing species were found in the elementary flow list. In light of this, Latin names for species and FAO major fishing areas are highly recommended to unequivocally identify inputs in order to make the models compatible with EF method. In the case of Woods & Verones (2019) it should be noted that additional marine

ecoregions should be added to the flows in anticipation of the expansion of the method to a global perspective, in which maritime areas beyond Europe are included.

Based on the above discussion, two important issues must be considered. Two methods appear to be those with a highest level of appropriateness with respect to their inclusion in the EF method: Langlois et al. (2015) and Woods & Verones (2019). While the two methods are reasonably compliant with the criteria that were set in Section 2, the latter includes a more quantitative accountability of uncertainty, and provides a methodological framework that is aligned with current developments in other impact categories, such as those developed in the GLAM framework, which leads us to recommend Woods & Verones (2019) as the preferred LCA method for seabed impact due to fishing. Having said this, it is important to highlight that Woods & Verones (2019) still presents some shortcomings to be improved, or complemented, with further research. Firstly, one single impact pathway that accounts for habitat loss/creation is considered in the modelling of the cause-effect chain that leads to the PDFs. In this sense, accounting for other environmental aspects related to seabed disruption is a pending assignment to account for in future improvements of this method. Secondly, the method focuses on the impact on the benthic area, whereas the interrelation with other areas of the water column, namely the different depths of the pelagic zone, remains unexplored. A third and final point of consideration is that when assessing the seabed disturbance generated by fishing, it must be considered that other anthropogenic actions, such as marine infrastructure, or seabed mining, should also be eventually included within a seabed impact category, as their actions are contributing to the same disturbance and generating similar if not identical impact pathways as the one's described above. Consequently, in relation to this final point, an LCA method (indicator) to account for the impacts on the seabed, unlike depletion of the targeted resource, which is ultimately a fishing-exclusive action, should also leave the door open for methodological improvements that allow the inclusion of characterization factors related to non-fishing actions on the seabed.

Non-LCA methods – seabed impact

According to the needs of this contract, a strict “fishery-based” approach to seabed impact is required. This automatically discounts many of the methods and approaches that are currently being operationalised to satisfy environmental policy, as these are primarily concerned with the state of the environment (“area-based”) as opposed to differentiating the degree of pressure exerted by different activities. Indeed, our systematic review of recent primary literature returned no relevant results that explicitly investigated the pressure different gears imposed on different seafloor habitat, or if they did, they only considered a few variations on mobile, bottom-contacting gears, not considering passive gears or more varied towed gears.

After applying this restriction, we are left with two methods that are currently being operationalised, the STECF Sustainability Indicators for Fishery Products and the MSC Benthic Impact Tool. From a high-level perspective, the STECF approach is qualitative, while the MSC tool incorporates some quantitative data into a method that relies fundamentally on qualitative relationships. The STECF approach is more comprehensive in its applicability, while the MSC tool currently only works for specific gears and habitat types but is extensible by users with enough data from their own specific context. Below, we qualitatively evaluate the relevance of these two methods according to our predefined evaluation criteria.

Generality of the method, such that it can apply to all products, both domestic and imported

The STECF approach is widely applicable. By utilising only qualitative relationships, the data reporting requirements are minimal, and the two-system approach rewards supply chains that can provide more data without excluding those that are more limited. The use of documented relationships between gears and the seabed, as well as between species and habitats mean that these methods can be applied to many different contexts where seabed maps and fishing effort are limited.

The MSC tool is based on science that has been developed in Europe, primarily for European waters, but does not preclude its application in other contexts. The inbuilt default parameters are Eurocentric and may not be suitable in other contexts. Thus, seafood products coming from other areas require more data on benthic organisms' biological traits and responses to various gear types. Furthermore, the gears that the method are tuned for are limited to some mobile, bottom-contacting gears that are common in Europe, but do not cover all fishing activities in Europe, let alone from around the world.

Ability to cover data-poor situations and missing stock assessments

The state of the stocks is irrelevant for the impact of gears on the seabed. However, the STECF method does not require explicit fishery data, other than the type of gear utilised and the species included in the seafood product. This makes it very applicable to data-poor situations.

The MSC tool requires a-priori established relationships between benthic organisms and the disturbance caused by different gears. These are provided for many European contexts but will need to be provided for each region where products are derived from. Furthermore, for every combination of species assessed and gear used, there are specific data requirements, namely the distribution of effort from that gear category. This spatial data may be difficult to source or verify in some situations, especially for small-scale fisheries. Additionally, there is no defined time period over which this effort data should be integrated, which may lead to gaming the systems by varying the length of the time series that are collapsed to improve the overall score.

Type and availability of input data

As stated above, the data required for the STECF method are already defined and available, with clearly defined roadmaps for extension and improvement. Because these data are of qualitative relationships, they are static (only changing under the new evidence, not naturally varying with time).

The MSC tool requires annual updates to fishing effort data, which may be empirical from some fisheries, or modelled for others. The availability of these data is also variable. For many (non-Mediterranean) European fisheries, effort data will be available in annual data products assembled under the International Council for the Exploration of the Seas, while others will need to be aggregated and assembled on an ad-hoc basis.

Type and format of outcome (scoring, value etc)

The STECF method assigns an individual impact category to different products, which only varies if one of the underlying qualitative relationships is updated according to new scientific evidence. This is currently a letter code across five levels (A-E), which are aggregated from seven numeric scores (0-6) based on the sum of gear pressure score and habitat sensitivity scores.

The MSC tool assigns one of three categories (Table 16a) based on an underlying probability that a habitat's benthic community will recover within 20 years. This score will vary as fishing effort varies over time; thus, the outcome category may vary from year to year (or over whatever period an update is made).

Completeness of scope

The STECF approach provides a method for all marketed products in the EU and considers both the pressure from different gears (comprehensively) and the sensitivity of habitats in which they are likely operating, to assign an impact score.

The MSC tool also considers gear pressure and seabed sensitivity independently, however, the current implementation of the tool does not consider a comprehensive list of gear types.

Environmental relevance

Both methods have large assumptions about the natural occurrence and variability in species-seabed interactions. In the case of the STECF method, the assumption that any given species is caught only in one (or two, in the extension of the method) habitats, limits the reality of the findings. In the MSC tool it is the simple population dynamic model that is employed that ignores many external drivers of recovery such as environmental variability and trophic interactions.

Despite these limitations both methods ultimately reflect the relative severity of different fishing practices' interactions with the seabed.

Scientific robustness and certainty

The quantitative nature of the MSC tool means that its final evaluation categories are defined by probabilities, however, these probabilities are only relevant within the assumptions about the ubiquitous nature of different benthos in different habitats and the types of response curves they would have. The STECF method does not provide certainties or uncertainties, because it is designed with the recognition that the data and knowledge do not exist to support quantitative approaches to this question at such universal scales (all seafood products from anywhere in the world).

Documentation, transparency, and reproducibility

Both methods are well documented, have transparent methodologies and are easily reproducible. The MSC tool, being more complex, will likely only be reproducible by technical experts. The STECF relies on published scientific studies to justify their gear-pressure and species-habitat relationships, so the reproducibility of these studies varies, but theoretically all should be reproducible, and the resultant scores are easily updated according to new findings. The same thing can be said for the MSC tool's data on benthos longevity and gear penetrations.

Applicability

The STECF method is broadly applicable to globally sourced, wild, seafood products. The MSC tool is built to be broadly applicable, however, it was developed in a western European context, the default underlying data and configurations are all from western Europe, and the assumptions in the population models may not fit other contexts.

Temporal representativeness, resolution, and time horizon

The STECF method is not temporally resolved. The MSC tool integrates data over different timescales, which are determined at the data aggregation stage by the user.

Spatial resolution

The STECF Method is not spatially resolved. The MSC tool applies generic population traits to maps of habitat for which the users dictate the spatial resolution based on input layers. The same is true for the aggregation of fisheries pressure data, although the recommended/default settings use a grid of 0.05 degrees of latitude.

Compatibility with Environmental Footprint (EF) method

Uncertain, but the novel exploratory “LCA-like” approach by Gascuel (pers. com) proposing quantitative indicators from the STECF letter grading, described in section 0, is meant to be equally applicable for the seabed impact.

Table 16a. Comparison of non-LCA methods of evaluating seabed impact of various fishery products for two operationalised methods, namely the STECF, 2023 method and the MSC method.

Criteria	STECF (2022) Benthic Impact Scoring	MSC Benthic Impact Tool
Type and availability of input data	All data come from public information. The method technical paper outlines all data sources and how they have been utilised by experts. For example, fish habitat data are taken from FishBase and are matched with EUNIS marine habitat categories. Gear impacts are scored based on expert knowledge and various sources of primary and grey literature. These are available in STECF technical reports.	Data are supplied by those seeking the assessment. The method uses both temporally static data on habitat distributions and benthic species traits, as well as temporally dynamic information on fisheries activities. The information on habitat distribution is available publicly throughout European waters but varies across the rest of the world. The information on species traits, including longevity and response to different gear pressures, is reliant on suitable empirical evidence being available for the context from which the product is derived. Basic default values are available for the European, NE-Atlantic Shelf. Data on fishing effort by gear type needs to be provided by nation states. There exists a mechanism to get this data for European waters from ICES, however, the rest of the world is unknown.
Type and format of outcome (scoring, values etc)	The STECF aims a providing a Letter-categorical scoring (A to E), regardless of product weight. Exploratory work is underway for translating this into a quantitative “LCA-like” indicator accounting for weight (Gascuel, pers. com)	The MSC tool assigns three scores, based on categories of probability that the benthic organisms in each habitat will return to pre-impact state with 20 years of no impact. These probabilities are available as well as the categorical assignment.
Completeness of scope	The STECF approach provides a method for all marketed products in the EU and considers both the pressure from different gears (comprehensively) and the sensitivity of habitats in which they are likely operating, to assign an impact score.	The MSC tool accounts for the spatial heterogeneity of both habitats and fishing practices. This approach also considers gear pressure and seabed sensitivity independently.

Criteria	STECF (2022) Benthic Impact Scoring	MSC Benthic Impact Tool
		However, the current implementation of the tool does not consider a comprehensive list of gear types.
Environmental relevance	The STECF method assumes that any given species utilised in a seafood product has been harvested on a certain habitat type (maybe two, in the extension). This approach also does not account for any form of temporal variability in habitat rejuvenation or frequency of fishing. However, the score still reflects the relative severity of different fishing practices' interactions with the seabed.	The MSC approach assumes uniform and temporally non-varying environmental conditions. The simple population dynamic model that is employed ignores many external drivers of recovery such as environmental variability and trophic interactions. However, the score still reflects the relative severity of different fishing practices' interactions with the seabed.
Scientific robustness	In order to be broadly applicable, the STECF method relies heavily on simple relationships between species, habitat and fishing gears. However, these relationships are based on the best available science and have been established from literature at specific, multinational for a: namely the STECF technical working groups. Because the scores from this approach are based on qualitative evaluations, there is no certainty/uncertainty reported for each score. However, the two-system approach awards access to better scores for those fisheries that can provide the more resolved data on fishing practices. Thus, the uncertainty in the less resolved data is penalised with weaker scores.	The quantitative nature of the MSC tool means that its final evaluation categories are defined by probabilities. However, these probabilities are only relevant within the assumptions about the ubiquitous nature of different benthos in different habitats and the types of response curves they would have. The tool itself is hosted by a private organisation but the mechanics of it are published in peer-reviewed journals.
Documentation, transparency, and reproducibility	The grading model is well documented in EWG reports, accessible and reproducible. All data sources are public and, for a great number of them, are easily updatable through automated webservices. The exploratory "LCA-like" step is still in development and would require further documentation	The MSC tool, being more complex, will likely only be reproducible by technical experts. The tool relies on published scientific studies to justify their gear-pressure and species-habitat relationships, so the reproducibility of these studies varies, but theoretically all should be reproducible, and the resultant scores are easily updated according to new findings.
Applicability	A grading letter (A-F) can be provided for any product entering EU market, as the mandatory CMO information allows linking with a stock, or if not, to an area where IUCN information can be retrieved.	The MSC tool is built to be broadly applicable, however, it was developed in a western European context, the default underlying data and configurations are all from western Europe, and the assumptions in the population models may not fit other contexts.
Temporal representativeness, resolution, and time horizon	Sensitivity data do not require regular updates, only as new knowledge becomes available from primary literature and as requested, as these are based on life history traits which remain relatively static (scale of evolutionary time). Fisheries effort data is not utilised, therefore there is no temporal resolution, nor need for annual updates.	The MSC tool integrates fishing pressure over time. The period over which the pressure is condensed is not specified but guidelines are provided. While the output of the tool is based on recovery time; it does not account for environmental change in that period.
Spatial resolution	This method is not spatially resolved. The link between species and habitat is based on ecological knowledge, not spatial data of habitat and fisheries effort.	The MSC tool applies generic population traits to maps of habitat for which the users dictate the spatial resolution based on input layers. The same is true for the aggregation of fisheries pressure data, although the

Criteria	STECF (2022) Benthic Impact Scoring	MSC Benthic Impact Tool
		recommended/default settings use a grid of 0.05 degrees of latitude.
Compatibility with Environmental Footprint (EF) method	The letter-grading is categorical, providing a qualitative information, thus not compatible with EF method. An exploratory “LCA-like” approach in development (Gascuel, pers. com) is proposed for providing a quantitative indicator linking with product weight.	The letter-grading is categorical, providing a qualitative information, thus not compatible with EF method. An exploratory “LCA-like” approach in development (Gascuel, pers. com) is proposed for providing a quantitative indicator linking with product weight. Furthermore, the underlying probabilities of recovery are available for adaptation into an LCA-like framework. However, the methodology for this does not exist.

Source: Authors’ own elaboration.

Selection of final methods to be brought forward

Based on all the methods presented in Sections 3 and 4, and the critical discussion provided in subsections 5.1 (biotic resources) and 5.2 (seabed impact), a small group of methods were retained considering the criteria than were described in Chapter 2 and the expert judgment of the authors. Table 17a shows the methods that have been selected for further evaluation.

Table 17a. Depletion of targeted resources and seabed impact methods retained for further description and evaluation.

Resources	LCA methods	Non LCA methods
Depletion of targeted resources	Hélias et al. (2018, 2023)	STECF (2023)
Seabed impact	Woods & Verones (2019)	STECF (2023)

Source: Authors’ own elaboration.

Discussion on key conceptual differences between the LCA and non-LCA methods for the indicator on targeted resources depletion

As already pointed out in section 3.1.1, various points of attention must be highlighted regarding key conceptual differences between LCA and non-LCA (MSY) methods for overfishing assessment criteria. We reiterate and discuss them here in more details:

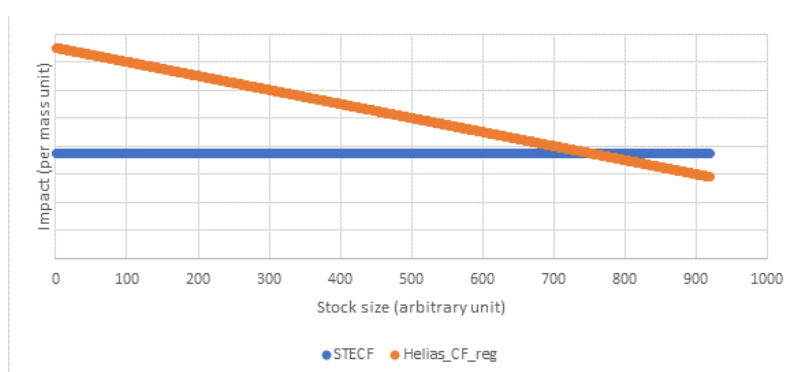
The LCA and non-LCA approaches presented here represent two contrasting approaches to considering ecosystem values, which imply differences in assessing impact on living resources (UNEP SETAC GLAM 1, 2017):

The “instrumental approach” (as labelled in GLAM 1), based on the concept of ecosystem services viewing living resources as being (only) useful for humans, aiming at guaranteeing the Maximum Sustainable Yield (MSY). This is the approach of STECF (non-LCA)

The “intrinsic approach” (as labelled in GLAM 1) based on accounting for disturbance to the “human-free” state of nature. Any removal of fish by humans from the ocean is considered impactful, even within sustainable fisheries. This is the approach of Hélias et al 2023 (LCA)

Another difference is additivity which is, by design, a feature of all LCA methods. This is specifically the case for target resource depletion. Hélias et al. (2023) work by mass (weight of catch within the value chain assessed against stock biomass weight); conversely, the non-LCA MSY approach works by stock unit irrespective of mass, as each fish population is considered on its own, with MSY reference points calculated in a way that reflects the productivity and the ability of the population to sustain fishing pressure. Previous comparison explorations made in the French GT Mer (Gaillet et al., 2023) have shown the major impact of this conceptual difference, with large size stocks more likely to display a lower impact per mass removed than small stocks when using Hélias et al. (2023) (catching 1 kg of a bigger stock has a lower impact than for a smaller stock), while this information is not used in MSY-based sustainable fisheries assessment (see Figure 7a).

Figure 7a. Schematic representation of the relationship between non-LCA STECF method (blue) and Hélias et al. 2023 regional (orange) for stocks at MSY: $B = B_{MSY}$; $F = F_{MSY}$.



STECF impact grading is constant regardless of stock size (A grading - system 2), whereas Hélias 2023 (regional) impact decreases with population size meaning that fishing rare species has more impact than fishing abundant species.

Source: Authors' elaboration from Hélias 2023 and STECF.

Providing a quantitative impact indicator based on product weight which would be compatible with the EF approach is thus not straightforward in the MSY approach, and indeed STECF (2023) was designed to provide a qualitative letter-grading, not a quantitative impact indicator. The GT Mer (Gaillet et al. 2023) performed extensive comparisons of the two methods based on the earlier versions of STECF (2020) and raised many challenging questions on how the STECF method could be adapted to fit the LCA framework (conversion from discrete units to numeric values; duality of the system; additivity etc). The same issue occurs with the non-LCA seabed method. One of the major issues is to deal with the top (A grading) and bottom grading (E grading), as both are defined with only one limit of an interval. For example, in the STECF overfishing approach, “ $F/F_{msy} > 2$ ” yields an E⁴⁰; this describes a large variability of situations as shown in Figure a, where this value can reach a high ratio (≥ 4). This diversity of overfishing values would need to be considered.

Based upon these discussions initiated during the French GT Mer, a novel exploratory “LCA-like” method (Gascuel, pers. com) is briefly proposed (in section 3.3.4) to address that and convert grades

⁴⁰ or “ $B/MSY_{trigger} \geq 1$ yields an A”

to quantitative scores, but this will require further development, testing and documentation before this approach can be considered operational and robust. This subject will be key to discuss in the next deliverable D2, when designing an operational roadmap for EF integration.

Another important difference between the LCA and the STECF MSY approach is how they make use of official stock assessment data coming from RFMOs, and the processes for filling gaps when that information is missing. Both Hélias and STECF methods refer to the main stock parameters of B (biomass) and C (catches) and Hélias's formula for CFs applies to stocks. In the operationalization of his method, Hélias et al. proposes to prioritize completeness over precision (specific identification of stocks with available RFMO data) based on the CMSY algorithm with assumptions on population parameters r and K fitted using values from the scientific literature (FishBase). For stocks which have precise C and B data from RFMO, Hélias's formula is straightforward and has been applied as an example for 25 species in the publication Hélias et al. (2018). Overall, to date, Hélias et al. (2023) CMSY computations cover 5000 pairs {species/species groups; FAO major fishing area}. STECF chooses to put precision as a priority and bases its computations on stock RFMO data when available. For those stocks with existing stock assessment, using these as raw information may not return exactly the same outcomes as fitting CMSY with official stock assessment data, although, as also shown by Hélias (2018), the results are close and consistent. The fallback option for missing data is of IUCN /sensitivity system 1 instead of CMSY. Overall, to date, STECF database covers around 2000 species in both system 2 and 1 (Guitton et al., 2023) and already include many automated updating facilities using webservice provided by key international organizations (ICES, NOAA, IUCN) (Guitton et al., 2023)).

Hélias et al. (2023) LCA method for impact from natural resources presents an approach of "natural resources depletion", with the assumption that a first kg removed from biomass has an impact ("intrinsic" ecosystem-quality approach). The method displays the advantage of aligning with several aspects intrinsic to an LCA framework, as it was designed for that (e.g., completeness, compatibility with LCA, additivity, transparency, reproducibility) but poses the question of stakeholder acceptance. The French GT Mer has evidenced the lack of familiarity of fishery management experts (stakeholders, scientists, managers, NGOs) with the "intrinsic approach", and the difficulty to reconcile the LCA approach with the MSY-based worldwide sustainability standards.

These discussions that occurred in France are likely to prefigure similar methodological and conceptual concerns at a European level, reflecting the fact that EF takes place at the interface of different scientific and stakeholders' communities with different standards. How to move forward will be the purpose of the next deliverable.

Finally, dealing with a renewable living resource, a species, or a stock, requires specific temporal mindset for depletion impact assessment. Humans take their supply of this resource and based on its status, it has variable capacity to replenish itself: methods must account for both catches and renewal capacity. This is dependent on "stocks" (species and location), but vary in time as well, the time component being thus key in the assessment. A large majority of stock assessments are updated each year. Any overfishing indicators are thus only applicable for a short time frame so they will have to be regularly monitored and potentially updated. Updating capability of method developers is important to keep in mind within the assessment criteria.

It is also acknowledged that uncertainty levels are high for LCA and non-LCA approaches, and this is inherent to fisheries (e.g. in the stock assessment or even the absence of stock or other type of information). This explains why the STECF approach is precautionary, including an important part of the uncertainty linked to the lack of information into the grading from A to E.

Operational integration of chosen methods in EF product assessment value chains

When LCIA methods are developed it is important for this to be in accordance with the elementary flows that are available in Life Cycle Inventory (LCI) databases, but also in the LCA software that is therefore adapted to take into consideration these flows. The lack of elementary flows also affects the two environmental aspects that are analysed in the current report, depletion of targeted resource (i.e., fish stock depletion) and seabed impact. Current research however converges to highlight the importance of three defining information items regarding overfishing and seabed impact:

-Species accepted Latin name will unequivocally identify species. Other Latin names (known as synonyms) can be linked to the accepted Latin name thanks to FishBase and SeaLifeBase databases (FishBase, 2023). For some species groups (e.g., tunas) it is sometimes challenging to link the fishery product with the specific Latin name for a fishing activity, and in addition there are thousands of elementary flows in LCI beyond fishing, and many data used for these does not have the degree of granularity that fisheries managers would expect. In this sense, in LCI practice it is sometimes necessary to group, hence in. In this case, a fallback option would be creating “species groups”. A common EC nomenclature defining species groups with the list of all related species Latin names would also be an added value, so that a common name would be used with the same meaning across products. This would be particularly useful for e.g. the “tuna” group.

-Fishing zone is the administrative area where fishing occurs, defined by FAO or FAO-supported Regional Fisheries Management Organizations (RFMOs). Species are often divided in several stocks that are usually geographically determined. Hence, species and fishing zone usually identify the stock which is the relevant unit of management and assessment. We recommend the zone to be defined according to FAO nomenclature (UNSTAT, 2023). FAO nomenclature defines 4 levels: areas, subareas, divisions, and subdivisions. FAO is only responsible for setting the first level, Major Fishing Areas. The levels below are set by different interested organizations (for example, regional fishery bodies) and thus the subsequent levels are heterogeneous. However, definitions are set by the EC or subsequent levels for FAO Major Fishing Areas 27 (North-East Atlantic) and 37 (Mediterranean and Black Sea) (EC, 2023) and we strongly recommend using FAO nomenclature with the highest level of granularity possible.

In some cases, the most granular level of subdivision will not suffice to identify stocks. It can be either due to higher granularity in stock segregation, like the case of Norway lobster (*Nephrops norvegicus*) for which up to 34 different stocks are defined in FAO 27, including 3 cohabitating in subdivision 27.7.j.2 (Dobby et al., 2021). It can also be due to other definitions of stocks, like Atlantic herring (*Clupea harengus*) for which stocks in Icelandic waters (Division 27.5.a), the North Sea (subarea 27.4), and Western Newfoundland (Division 21.4.R) are defined by spawning season: spring or fall (FishSource, 2023). However, even with various or very fine stock definitions, proxies

of stock status either using worst status in the fishing zone or (weighted) average can be used to calculate an overfishing impact.

-Fishing Gear. we recommend using gear type information as defined in Commission Regulation (EC) No 26/2004 (EC, 2004) and Commission Implementing Regulation (EU) No 404/2011, annex XI (EC, 2011). STECF (2023) states that this nomenclature characterizes gears with sufficient granularity regarding their impacts on seabed (and potential additional catches than the target species). However, this information is not used per se in current LCIA methods (e.g. Woods and Verones (2017)). We suggest basing all fishing life cycle inventories (LCIs) on those three elements. We recommend putting the three in the name of the process output flow.

Table 18a summarizes a first proposal of life cycle Inventory, based on i) either Hélias et al (2023) or STECF 2023 for fishing pressure and ii) STECF 2023 for seabed impact. In the case seabed impact STECF, as adapted for EF, is additive (e.g., it is twice more impactful for 2 kg of fish than for 1kg), we propose to include the intensity of seabed impact as an elementary flow. Otherwise, the mention of the intensity has to be discussed with the commission: in our view, the mention in metadata is, to date, not prescriptive enough. Regarding input flows and metadata in fishing LCI, our rationale is to have a short-term proposal enabling operationalization of either Hélias et al 2023 or STECF 2023 for stock depletion indicator and STECF 2023 for seabed impact. However, we also make additional suggestions in flows from technosphere and metadata to facilitate uptake of an LCA-based seabed impact, basing our proposal on parameters that Woods and Verones (2019) highlighted. In terms of flows from the ecosphere, the Hélias et al. (2023) method requires including the quantity of catches for the pair (i.e., Latin name & FAO fishing zone). The STECF 2023 methods requires additional data on stock advice and/or IUCN status, depending on data availability, which do not necessarily fit in the LCI flow (see details in section 3). However, it must be noted that a database structure has been proposed by Guitton et al. (2023) to collect and store data on stocks. In fact, the tool presented Guitton et al. (2023) has already parametrized those stocks already included and, therefore, a score is already available for them.

We propose to include a second elementary flow that would bear the quality rating of STECF 2023 for seabed impact. We suggest including a flow from the technosphere with effective fishing time with given gear. This flow could help verify consistency of energy consumption for fishing, and mention gear type, linked with the value of the elementary flow on seabed impact. Additionally, this flow could be used in a later stage to compute, when combined with “average fishing speed” and “gear width”, the surface of the “trawled area”, one of the inputs in the Woods and Verones (2019) publication.

We listed additional elements in metadata. Those could be mobilized in the near future for an evolution of the inputs in LC Inventories, when methods evolve.

Table 18a. Summarized proposal of the guidance needed for the indicators selected to be operational within an LCA and EF framework, with baseline assumption of using either Hélias et al. (2023) or STECF 2023 for fishing pressure and STECF 2023 for seabed impact.

Where	What	Unit	Recommendation level *	Comments
In process output name	Species - accepted Latin name	N/A	II	If group of species (e.g., tunas), we recommend defining all Latin species names related to this group in an EF annex
	Species - fishing zone	N/A	II	Most precise FAO subdivision # available
	Species - gear	N/A	II	Gear nomenclature according to 28 EC names
In flows from techno sphere	Time fishing with gear X	Time unit	II	Gear nomenclature according to 28 EC names <i>Could help approach a quantitative trawling surface and damage ultimately</i>
In flows from ecosphere	Quantity of a given stock A	kg	I	Target stock Nomenclature of stocks TBD according to method (>5000 in Hélias et al. 2023)
	Quantity of discarded species caught	kg	III	Default value TBD
	Intensity of seabed impact **	discrete values 1 to 6	I	for use of STECF 2023 for seabed impact based on information in process output flow/flow from technosphere (species name and gear)
In metadata	Stock unique identifier	N/A	II	Nomenclature (TBD) e.g., from Fishsource
	Seabed impact surface trawled	m ²	III	In the further perspective of integrating a simplified version of Woods & Verones (2019)
	Discards	kg	II	with indication of type of discards
	Average depth of trawling	m/discrete value	III	Additional information for intensity of seabed impact. Could be a discrete value according to Woods and Verones (2019) specifications

Where	What	Unit	Recommendation level *	Comments
	Gear width	m	III	Additional information for intensity of seabed impact. Could be a discrete value according to Woods and Verones (2019) specifications
	Bycatch	kg	III	Species accepted Latin name and quantity (available e.g., certification assessments for labels)

(*) Recommendation levels are classified from I to III, I being highest recommendation and III being lowest

(**) in case of additivity of seabed impact indicator.

Source: Authors' own elaboration.

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List of abbreviations and definitions

AoP: Area of Protection

B: current biomass of stock (in tons)

Bmsy: Biomass at Maximum Sustainable Yield (in tons)

CFP: Common Fisheries Policy

CMO: Common Market Organisation

EF: Environmental Footprint Method: an LCIA method developed by JRC to assess and compare consumers' products environmental footprints within the EC

F: current fishing pressure

FAO: Food and Agricultural Organisation

Fmsy: Fishing pressure at Maximum Sustainable Yield

GLAM: Global LCA Access to Methods ; A UN consensus-based initiative on LCA methods

GT Mer: Groupe de Travail Mer - Seafood working Group in English. An ad-hoc working group set up by the French Agency for Ecological Transition (ADEME) in 2022-2023

ICES: International Council for the Exploration of the Sea

ILCD: International Reference Life Cycle Data System

LCA: Life Cycle Assessment

LCI: Life Cycle Inventory

LCIA: Life Cycle Impact Assessment

MSC: Marine Stewardship Council

MSFD: Marine Strategy Framework Directive

MSY: Maximum Sustainable Yield (in tons)

MSY Btrigger: a biomass reference point that it is an action points; when SSB is below this level managers are expected to take measures to reduce fishing mortality

NOAA: US National Oceanographic and Atmospheric Administration

PEF: Product Environmental Footprint

RFMO: Regional Fisheries Management Organisation

SDG: Sustainable Development Goal

SSB: Spawning Stock Biomass: total weight of all sexually mature fish in the stock

STECF: Scientific, Technical and Economic Committee for Fisheries of the European Commission

Stock: A marine biological resource that occurs in a given management area

Annex 2. Non-normalised impacts for 16 EF indicators and proposed new fisheries indicators

Table 1b. Non-normalised results for the case studies: hake, shrimp for the 16 EF indicators.

Impact category	Unit	HAKE			SHRIMP
		Demersal fish, fresh (ES) hake, capture by trawler and landing whole, fresh Cut-off, U	Hake - Saithe, NorthSea, Bottom Trawl, average, fresh, at landing {EU} U	Landed hake, fresh (ES) hake, capture by trawler and landing whole, fresh Cut-off, U	Shrimp - Great Scallop, BSBrieuc, Dredge, average, at landing {FR} U
Acidification	mol H+ eq	0,052884908	0,069698658	0,114741709	0,02213274 1
Climate change	kg CO2 eq	2,457108852	2,059865859	5,331064702	0,67362146 7
Ecotoxicity, freshwater	CTUe	10,92094513	7,13089045	23,69462185	2,55280036
Particulate matter	disease inc.	4,02445E-07	5,24549E-07	8,73165E-07	1,66974E- 07
Eutrophication, marine	kg N eq	0,012948289	0,016784517	0,028093247	0,00530753 2
Eutrophication, freshwater	kg P eq	4,53443E-06	8,25049E-06	9,83813E-06	5,06935E- 06
Eutrophication, terrestrial	mol N eq	0,141134791	0,183855323	0,306213012	0,05813692 9
Human toxicity, cancer	CTUh	1,01721E-09	1,06627E-09	2,20699E-09	3,85386E- 10
Human toxicity, non-cancer	CTUh	6,80905E-09	7,0428E-10	1,47733E-08	4,08876E- 10
Ionising radiation	kBq U-235 eq	0,003316965	0,119315282	0,00719665	0,03910100 6
Land use	Pt	1,543288848	3,725825268	3,348395696	1,71650431
Ozone depletion	kg CFC11 eq	1,35475E-05	4,88053E-07	2,93933E-05	1,46452E- 07
Photochemical ozone formation	kg NMVOC eq	0,038446684	0,047900909	0,083415824	0,01526631 5
Resource use, fossils	MJ	21,79214836	27,89729943	47,28132123	9,29283866 3
Resource use, minerals and metals	kg Sb eq	2,36855E-06	9,22759E-06	5,13893E-06	7,64584E- 06
Water use	m ³ depriv.	0,042149331	0,044355595	0,09144927	0,02426908 5

Method EF 3.1 on SimaPro 9.6.0.1.

Source: Authors' own elaboration.

Table 2b. Non-normalized results for the case studies: hake, shrimp for the three methods.

Impact category	Unit	HAKE	HAKE	SHRI MP
		Demersal Trawl	Midwater Trawl	Paired beam trawl
Fisheries exploitation (Helias et al. 2023)	PDF _{glo.} year	1,74149 0E-14	1,74149 0E-14	1,747 300E- 12
Fisheries exploitation (STECF/GT Mer)	Uex /kg (Uex="unit of overexploitation")	1	1	1
Seabed impact (Woods and Verones, 2019)	PDF*yr/kg	5.64E-9	0	NA
Seabed Impact (STECF/GT Mer)	Semi-quantitative indicator based on multiplicative lognormally distributed representation of risk	1	1.2	1.2

Stock exploitation (Helias et al. 2023) are taken from Helias et al., 2023 Supplementary information with Argentine red shrimp, *Plueoticus muelleri*, in FAO 41 and Atlantic hake, *Merluccius merluccius*, in FAO 27. CFs for Stock exploitation (STECF/GT Mer). CFs for Seabed impact (STECF, 2023; ADEME, 2023). CFs for Seabed Impact (Woods and Verones, 2019).

Source: Authors' own elaboration.

Annex 3. Summary of the report and Annex 1

On October 16, 2024, co-authors Clara Ulrich and Anne-Claire Asselin delivered a presentation to the EF Technical Advisory Board titled "Environmental Impacts of Fisheries – Integrating Fisheries Impacts into the European Commission's Environmental Footprint Framework." Slides can be downloaded from the following link:

<https://ec.europa.eu/transparency/expert-groups-register/core/api/front/document/112357/download>

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