



HAL
open science

Assessing overfishing based on the distance-to-target approach

Vanessa Bach, Arnaud Hélias, Marco Muhl, Aurélie Wojciechowski, Henk Bosch, Michael Binder, Matthias Finkbeiner

► **To cite this version:**

Vanessa Bach, Arnaud Hélias, Marco Muhl, Aurélie Wojciechowski, Henk Bosch, et al.. Assessing overfishing based on the distance-to-target approach. *International Journal of Life Cycle Assessment*, 2022, 27 (4), pp.573-586. 10.1007/s11367-022-02042-z . hal-03630698

HAL Id: hal-03630698

<https://hal.inrae.fr/hal-03630698>

Submitted on 5 Apr 2022

HAL is a multi-disciplinary open access archive for the deposit and dissemination of scientific research documents, whether they are published or not. The documents may come from teaching and research institutions in France or abroad, or from public or private research centers.

L'archive ouverte pluridisciplinaire **HAL**, est destinée au dépôt et à la diffusion de documents scientifiques de niveau recherche, publiés ou non, émanant des établissements d'enseignement et de recherche français ou étrangers, des laboratoires publics ou privés.



Distributed under a Creative Commons Attribution 4.0 International License



Assessing overfishing based on the distance-to-target approach

Vanessa Bach¹ · Arnaud Hélias^{2,3} · Marco Muhl¹ · Aurélie Wojciechowski⁴ · Henk Bosch⁵ · Michael Binder⁶ · Matthias Finkbeiner¹

Received: 3 August 2021 / Accepted: 16 March 2022
© The Author(s) 2022

Abstract

Purpose Overfishing has been a global challenge for several decades with severe impacts on biodiversity and ecosystem services. Several approaches for assessing overfishing in life cycle impact assessment exist, but do not consider scarcity in line with current policy and science-based targets. Furthermore, comparisons of results with other impact categories, e.g., climate change, are not possible with existing methods. Therefore, five approaches to assess overfishing based on the distance-to-target approach are introduced.

Method Three global species-specific approaches (stock in the sea, target pressure, and fish manager) and two regional midpoint approaches were developed. For the stock in the sea, the weighting factor was derived as the relation of available biomass of the considered species to biomass at sustainable limits. Within the target pressure, the current pressure on fish stocks is set to the maximal sustainable pressure. For the fish manager, the catch is set in relation to the maximum sustainable yield. The catch is used for normalization in all three approaches. The two regional midpoint approaches consider production and consumption based catch of fish stocks in relation to the fully fished share. The overfishing indicator based on pressure on fish stocks serves as the characterization factor. Normalization occurs with the characterized catch.

Results and discussion To demonstrate the applicability of the approaches, a three-level case study was derived: (i) determining ecofactors for ten specific fish species in specific oceans; (ii) deriving ecopoints for production of fish meal and oil in Europe; (iii) comparison of fish oil with rapeseed oil for the categories overfishing, climate change, land use, and marine eutrophication. The highest ecofactors for the global approaches are characterized by high normalization and weighting factors. For the regional approaches, high overfishing characterization factors determine the result. The species contribution increases with rising amounts. Main challenges are data collection and interpretation which limit the overall applicability. The sensitivity analysis shows that the overall results vary significantly depending on the composition of the fish oil and meal.

Conclusions It was shown that four of the five approaches are able to account for overfishing. However, only the production-based regional midpoint approach allows for comparison with other impact categories and is therefore most suitable for integration into life cycle assessment. The developed approaches can be used for a more comprehensive assessment of environmental impacts of different diets as well as aquaculture feed solutions.

Keywords Overfishing · Distance to target · Ecological scarcity method · LCIA

Communicated by Ian Vázquez-Rowe

✉ Vanessa Bach
vanessa.bach@tu-berlin.de

¹ Chair of Sustainable Engineering, Technische Universität Berlin, Strasse des 17. Juni 135, 10623 Berlin, Germany

² ITAP, University Montpellier, INRAE, Institut Agro, Montpellier, France

³ Elsa, Research Group for Environmental Lifecycle and Sustainability Assessment, Montpellier, France

⁴ Evonik Industries AG, Essen, Germany

⁵ DSM Nutritional Products, Kaiseraugst, Switzerland

⁶ Evonik Operations GmbH, Hanau, Germany

1 Introduction

Impacts of the fishery sector including overfishing have been a global challenge for several decades due to severe consequences such as loss of biodiversity and reduction of ecosystem services. Furthermore, decreased availability of fish leads to social and economic consequences as fisheries play a significant role in the provision of food and nutrients (Rogers et al. 2015; Coll et al. 2016; Teixeira et al. 2016; Pope et al. 2016; Winter et al. 2017; Pacoureaux et al. 2021). Currently, around one-third of the global fish stocks are classified as overfished. Overfishing refers to depletion of stocks of fish by (over)exploitation due to excessive fishing. Thus, more fish is caught than can regenerate naturally, leading to a stock, which is fished above the maximum sustainable yield (MSY) (Food and Agriculture Organization of the United Nations 2020). The MSY concept refers to the largest yield (or catch) that can be taken without depleting the considered species stock.

Overfishing is addressed by the Sustainable Development Goal (SDG) 14 (Life below water: conserve and sustainably use the oceans, seas and marine resources for sustainable development) with the original aim to end overfishing by 2020 (United Nations 2016). This goal unfortunately could not be reached and the issue of global overfishing remains. The SDG goal was implemented into many regional and national strategies, e.g., for Europe, where overfishing is supposed to be phased out by 2030 (European Commission 2020).

Several life cycle assessment (LCA) approaches have been developed in the last years to account for fishing impacts and especially overfishing (e.g., Langlois et al. 2014a; Woods et al. 2016; Cashion et al. 2016; Stucki et al. 2018). Several of these methods are based on the net primary production (NPP), which is an indicator accounting for productivity (overall biomass needed within the system to produce a certain amount of fish) (Schneider et al. 2016). The MSY concept and population dynamic models are used to address overfishing. While Langlois and colleagues (Langlois et al. 2012, 2014b) determine overexploitation by comparing MSY in relation to current yields, Emanuelsson et al. (2014) use model simulation estimating yield losses due to current pressure on fish. Next to NPP and MSY, further methods are used to account for overfishing, including an approach to assess marine fish as a resource for the reference area Switzerland based on the ecological scarcity method (ESM) applying political targets (Frischknecht et al. 2021). Ziegler and colleagues (Ziegler and Valentinsson 2008; Ziegler et al. 2011, 2016) propose to use catch rates, discard rate ratios, and amounts of catch for non-commercial use to account for discard any by-catch.

Even though some of these methods address overfishing and overexploitation, results of existing methods cannot be (easily) compared to results of other LCIA categories in LCA on midpoint level. This is an issue for many LCIA methods, but is especially relevant for overfishing, because marine- and agricultural-based product systems are often compared to identify food products and/or diets with the lowest environmental impacts (e.g., comparison of different diets Tyszler et al. 2016; Ridoutt et al. 2017; Bossek et al. 2021). The above mentioned approach by Frischknecht et al. (2021) uses science-based targets to determine overfishing by applying the ESM and therefore allowing for a comparison with other impact categories, but only for the reference region Switzerland.

As overfishing is a multifaceted challenge going far beyond single issues, the need for further methodological development was highlighted within the LCA community (e.g., Sonderegger et al. 2017; Ghamkhar et al. 2020). Additional to overfishing, fishery also leads to other challenges, e.g., discard and by-catch, influencing biotic resource depletion. Thus, within this paper, a contribution is made by introducing additional assessment approaches based on the distance-to-target (DtT) approach to account for overfishing considering political and science-based targets. Deriving results based on political and science-based targets most likely provides additional knowledge regarding overfishing.

In the DtT approach, the distance of a current environmental state is set in relation to a target, which is derived based on political or science-based regulations and findings. For the science-based targets the envisioned target value is determined based on state-of-art scientific knowledge (Muhl et al. 2021). Several DtT approaches have been developed in recent years based on political targets (e.g., for Russia (Grinberg 2015), for the European Union (Ahbe et al. 2017; Muhl et al. 2019), for Japan (Büsser et al. 2012)) as well as science-based targets (Tuomisto et al. 2012; Sandin et al. 2015). DtT approaches are transparently documented, because publicly available environmental data as well as legitimated policy targets for the considered regions exists (Muhl et al. 2021).

The ESM as one implementation of the DtT approach derives targets from environmental legislation and has been gaining attention in the last years as it allows for direct comparison of different environmental impacts (Ahbe et al. 2017; Lecksiwilai et al. 2017; Muhl et al. 2019, 2020; Frischknecht et al. 2021). It however is also applied using science based targets (e.g., Bjørn and Hauschild (2015), Vargas-Gonzalez et al. (2019)). Thus, the ESM is selected as a basis to address overfishing, because it allows the integration of science based and policy based targets as well as a comparison of different environmental impacts within the same approach.

In the following, the ESM and fishing parameters are explained in more detail as they are the basis of the newly developed approaches.

1.1 Background

The ESM was originally developed by Müller-Wenk (1978) to be applied on Switzerland. The method is continuously updated since then, with its latest update published in 2021 (Frischknecht et al. 2021). As shown in Eq. (1), the ecofactors for a resource/emission i is structured in the following four factors: characterization, normalization, weighting, and a constant. For the characterization, (if applicable) emission or resource specific characterization factors (based on reference substances) are applied as common in LCIA. In the normalization step, the current environmental situation of the considered reference system (e.g., country or region) is taken into account. The weighting factor is determined by setting the actual flow (i.e., current annual quantity of considered emission/resource in considered reference area) into relation to the critical flow. The critical flow is a target defined by binding and non-binding legislation (e.g., reduction targets for Europe regarding greenhouse gas emissions (European Council 2014; European Union 2020)) or other targets provided in recommendations of national and international working groups (e.g., tolerable water stress is defined at 20% of the available resource as a target for water consumption (OECD 2004)). In a last step, an identical constant for all ecofactors is multiplied in order to align the range of the numerical values. The ecopoint is the common unit of each environmental impact and can be aggregated into a single-score result.

$$\text{Ecofactor}_i = \underbrace{CF_i}_{\text{Characterization}} \times \underbrace{\frac{1 EP}{F_n}}_{\text{Normalization}} \times \underbrace{\left(\frac{F}{F_k}\right)^2}_{\text{Weighting}} \times \underbrace{c}_{\text{constant}} \tag{1}$$

where

- CF is the characterization factor of emission or resource
- EP is the ecopoint (unit of environmental impact)
- F_n is the normalization flow: current annual quantity of considered emission/resource of considered reference area
- F is the current flow: current annual quantity of considered emission/resource of considered reference area
- F_k is the critical flow: limit value in the reference area

As MSY refers to the largest catch to be fished without depletion, the stock is considered overfished when the catch is higher than the MSY. If the catch is equivalent to MSY, it is considered fully fished. Next to the MSY, also other parameters based on this concept are used to describe the current situation of fish stocks (see Table 1). These fishing parameters can be extracted, for example, from the Sustainable Fisheries Partnership (2019) database and are usually updated at least once per year, sometimes even once per month.

2 Methods

In the following, the developed approaches for overfishing are described, which are initially based on the idea of Emanuelsson et al. (2014), which sets B in relation to B_{MSY} to determine overfishing; as well as Stucki et al. (2018), which applied the DtT approach by setting F in relation to the F_{MSY} . Overall, five different options are developed and are presented in the following: the global species-specific approaches are introduced in Sect. 2.1; the regional midpoint approaches are introduced in Sect. 2.2.

2.1 Global species-specific approaches

Overall, three global species-specific approaches (stock in the sea, fish manager, and target pressure) are developed

Table 1 Overview of fish stock parameters

Parameter (measured in t)	Abbreviation	Explanation
Catch	C	Catch of considered fish species/stock (in t)
Maximum sustainable yield	MSY	Largest catch to be taken without depletion of stock of considered fish species (in t)
Pressure on fish stocks (also referred to as fishing mortality)	F	Proportion of fish species caught and removed by fishing (in t)
Fishing mortality of the maximum sustainable yield	F_{MSY}	Maximal sustainable rate of fish species caught and removed by fishing without depletion of stocks (in t)
Biomass of fish stock	B	Body-weight of all the fish in the water for a specific fish species stock (in t)
Biomass of the maximum sustainable yield	B_{MSY}	Biomass that enables a fish stock to deliver the maximum sustainable yield for a specific fish species (in t)

based on the fishing parameter introduced in the background. They are called species-specific approaches, because specific ecofactors based on specific weighting and normalization flows are applied. Since there is no specific regional reference, but a variety of regions considered, the approach is to be globally applied. This also means that the determined ecofactor is only valid for this specific fish species in the area (here: oceans) considered.

The ecofactor of a fish stock i within a certain area a is determined by considering species-specific current flows, target flows, and normalization flows. Within Table 2, the three global species-specific approaches are displayed, showing the equations to calculate the ecofactors as well as the applied parameter. No characterization factors are used, because overfishing of the species is reflected in the individual normalization and weighting flows. The target values are science-based targets based on common fish parameter (see Table 1).

The *stock in the sea* approach (SIS) is based on available biomass of the considered species i in the considered area a (see Eq. (2)). The current biomass is calculated as the difference of the maximal potential biomass without human pressure (considered as twice $B_{MSY,i,a}$, see Schaefer (1954)), subtracted by the currently available biomass ($B_{i,a}$). The current biomass is set in relation to the biomass at sustainable limits ($B_{MSY,i,a}$) (target) and normalized with the catch of species i in area a ($C_{i,a}$). The SIS focuses on biomass and can therefore be interpreted as a biodiversity conservation-oriented approach.

In the *target pressure* approach (TPA), the current pressure on fish stocks ($F_{i,a}$) is set in relation to the maximal sustainable pressure on fish stocks in a certain area ($F_{MSY,i,a}$) and normalized with the catch ($C_{i,a}$) (see Eq. (4)). The TPA rather reflects an economic exploitation perspective as it is based on the fishing effort deployed.

In the *fish manager* approach (FMA), the ecofactor is determined by setting the catch ($C_{i,a}$) in relation to $MSY_{i,a}$ (which is the maximum catch to stay within sustainable catch limits) and normalized with the catch ($C_{i,a}$) (see Eq. (3)). In this approach, the catch is also used as the current flow. The FMA can be seen as a compromise between SIS and TPA, and therefore reflects fishing effort deployed. The catch ($C_{i,a}$) was chosen as the normalization flow in all three approaches, because it is the current annual quantity of used resources (here: fish species) of the considered reference area (here: oceans).

The underlying fish parameters are interlinked with each other, but their interpretation differs. For example, a catch can be higher than the MSY (which is reflected in the FMA), or a catch can be above MSY, but determined by a stock that is still in good condition compared to the biomass at sustainable limits (which is reflected in the SIS) or conversely by a high fishing effort (which is measured by the TPA).

Table 2 Overview of species-specific and regional midpoint approaches, including equation for ecofactor, current flow, target flow, normalization flow, and characterization factor

	Global species-specific approaches			Regional midpoint approaches (RMA)	
	Stock in the sea (SIS)	Fish manager (FM)	Target pressure (TP)	Production based	Consumption based
Equation for ecofactors	$= \left(\frac{2 \times B_{MSY,i,a} - B_{i,a}}{B_{MSY,i,a}} \right)^2 \times \frac{1}{C_{i,a}} \times \epsilon$ (2)	$= \left(\frac{C_{i,a}}{MSY_{i,a}} \right)^2 \times \frac{1}{C_{i,a}} \times \epsilon$ (3)	$= \left(\frac{F_{i,a}}{F_{MSY,i,a}} \right)^2 \times \frac{1}{C_{i,a}} \times \epsilon$ (4)	$= CF_{i,a} \times \left(\frac{\text{overall catch}_a}{\text{overall catch} - \text{overfished species}_a} \right)^2 \times \frac{1}{\sum_{i=1}^n (\text{catch}_a \times CF_{i,a})} \times \epsilon$ (5)	
Current flow (in t)	Biomass	Catch	Pressure on fish stocks	Overall catch in the considered area	
Target flow (in t)	B_{MSY}	MSY	F_{MSY}	Fully fished stock (in t) in the considered area, calculated by subtracting the amount of overfished species (in t) from the overall catch (in t)	
Normalization flow (in t ^{*herring-equivalent})	Catch _{i,a} ($C_{i,a}$)			Characterized catch in the considered area	
Characterization factor (in herring equivalents)	N/A				$CF_i = \text{overfishing-indicator}_i = \left(\frac{F_{i,a}}{F_{MSY,i,a}} - 1 \right) / \left(\frac{F_{\text{herring,NEA}}}{F_{MSY,\text{herring,NEA}}} - 1 \right)$ (6)

i specific species, a considered area, C catch, ϵ mathematical constant, NEA North East Atlantic

*Considering all fish species caught for European consumption/production, not just the ones of the case study, based on data for the years 2013–2015 by Hélias (2019)

2.2 Regional midpoint approaches

Overall two regional midpoint approaches (RMA) are developed: one considering regional production and one taking regional consumption into account. The ecofactors include species-specific CFs, but since normalization and weighting factors for a specific area (here: Europe) are applied, they are regional approaches.

The calculation formula for both is presented in Eq. (5). The indice “area” is applicable for area-specific consumption and production, respectively. The ecofactor is determined by setting the area-specific catch (regarding production or consumption) in relation to fully fished stocks, which are determined by subtracting the overfished species from the overall catch in the reference area. Furthermore, for the CF, the overfishing indicator of Ziegler et al. (2011, 2016) (see Table S3) is applied and set in relation to herring equivalents (see Table S4). The indicator is determined by setting the pressure on fish stocks ($F_{i,a}$) in relation to the maximal sustainable pressure ($F_{MSY,i,a}$) on fish stocks (similar to the *pressure on fish stock*) and subtracting one (see Eq. (6)). The normalization occurs by the characterized catch. The target is based on the political agenda by the European Commission (2020) to phase out overfishing by 2030.

3 Methods demonstrated with a case study

In this section, a three-level case study is introduced to show the applicability of the developed approaches and to demonstrate how the results can and should be interpreted. First, ecofactors are determined for ten specific fish species and the parameters of the developed approaches are analyzed (see Sect. 3.1). Second, the ecopoints of fish oil and fish meal are presented (see Sect. 3.2). Last, the consumption-based approach is applied to compare fish oil with rapeseed oil over the categories overfishing, climate change, land use, and marine eutrophication (see Sect. 3.3). The production-based RMA cannot be applied in the case study, because some of the fish species used in fish meal and fish oil are caught outside of European waters/oceans (e.g., Gulf of Mexico). It is still introduced as a methodology in Sect. 2, because it might be applied to future case studies.

3.1 Deriving of ecofactors and analysis of underlying parameter

In the case study, 10 species were considered to be used for fish oil and meal: anchoveta (from South Eastern Pacific (SEP)), blue whiting (from North East Atlantic (NEA)), herring (from NEA), sandeel (from North Sea (NoS)), sprat (from NoS), capelin (from Barents Sea (BaS)), menhaden

(from Gulf of Mexico (GoM)), mackerel (from NoS), boar fish (from NoS), and whitefish (from NoS). These fish species were chosen as they are used within fish oil and fish meal production in Europe (Winther et al. 2009). It should be noted that the determined cofactors are only valid for these specific fish species in the considered area (here oceans), e.g., the ecofactor of herring only applies for herring in the NEA, but not for herring in the Celtic Sea.

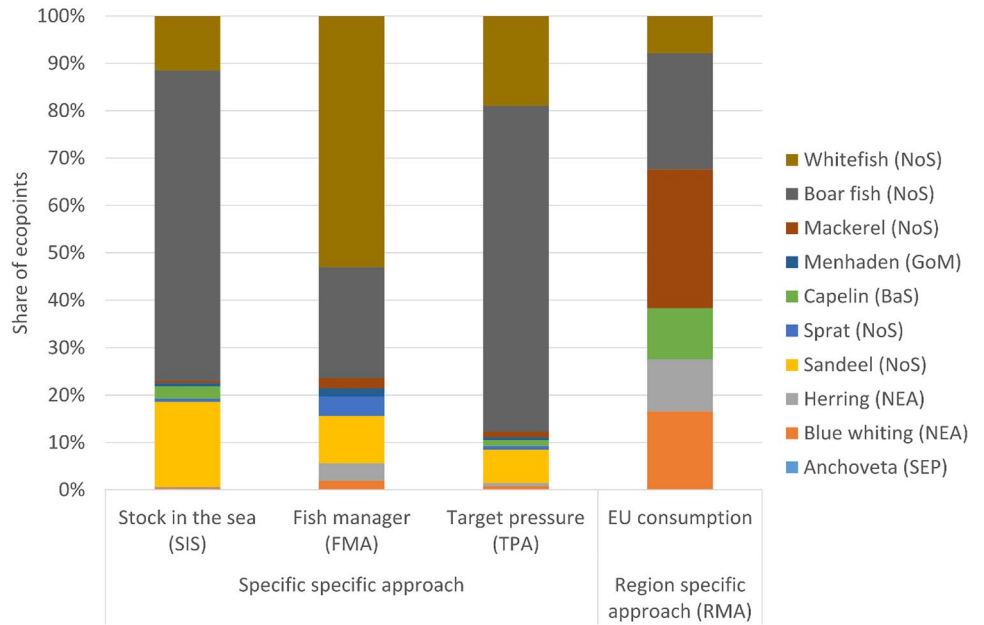
The underlying fish parameters (C , F , MSY , and F_{MSY}) are based on data from Hélias et al. (2018), Sustainable Fisheries Partnership (2019), and Hélias (2019), whereas the parameter B had to be calculated based on data for F and C . All background data of the case study can be found in the supporting information.

The ecofactors of the considered species are shown in Table S8. As the ecofactors itself cannot be compared directly, 1 kg of each fish is assumed as an elementary flow, so that the comparison is based on ecopoints, but the influence of the individual fish species can be analyzed. The shares of different ecopoints are compared on a percentage basis (see Fig. 1).

It can be seen that the hotspot fish species for the global species-specific approach are whitefish (NoS), boar fish (NoS), and sandeel (NoS). To better understand why these fish species are hotspots, the weighting and normalization factors are closely analyzed (see Fig. 2). Whitefish (NoS), boar fish (NoS), and sandeel (NoS) have a high normalization factor. A high normalization factor translates to a low catch, which is seen as a threat to overfishing, as low catch is assumed to correlate with high impacts. This assumption is the same as in normalization in LCA and has been challenged by several authors (more details are presented in Sect. 4). Furthermore, boar fish (NoS) and whitefish (NoS) also show comparatively higher weighting factors. High weighting factors indicate that the set target is exceeded and therefore overfishing takes place: more biomass is extracted, more fish is caught and higher fishing efforts occur than sustainable limits allow.

The SIS has the highest weighting factors for five species (anchoveta (SEP), sandeel (NoS), capelin (BaS), menhaden (GoM), and boar fish (NoS)), compared to the FMA with four species (blue whitening (NEA), herring (NEA), sprat (NoS), and whitefish (NoS)) and the target pressure with only one species (mackerel (NoS)). This means that for the considered species, the biomass exceeds the sustainable limits, whereas sustainable yield and pressure on fish stocks for many species are less overstretched. This difference is more significant for some species (e.g., capelin (BaS), where the weighting factor of the SIS is much higher compared to the other approaches) than for other species (e.g., whitefish (NoS), where the weighting factors are similar for all three approaches).

Fig. 1 Ecopoints of the ten considered fish species for the global species-specific as well as the regional midpoint approaches; relative view as overall results are scaled to 100%



Capelin (BaS) has the highest weighting factor in SIS, meaning that for the fish species capelin in the Baltic sea, the exceedance of the biomass is the highest for the

considered fish species in the introduced case study. Even though the biomass is exceeded, the impacts shown by FMA are close to zero, thus implying that fishing effort is low.

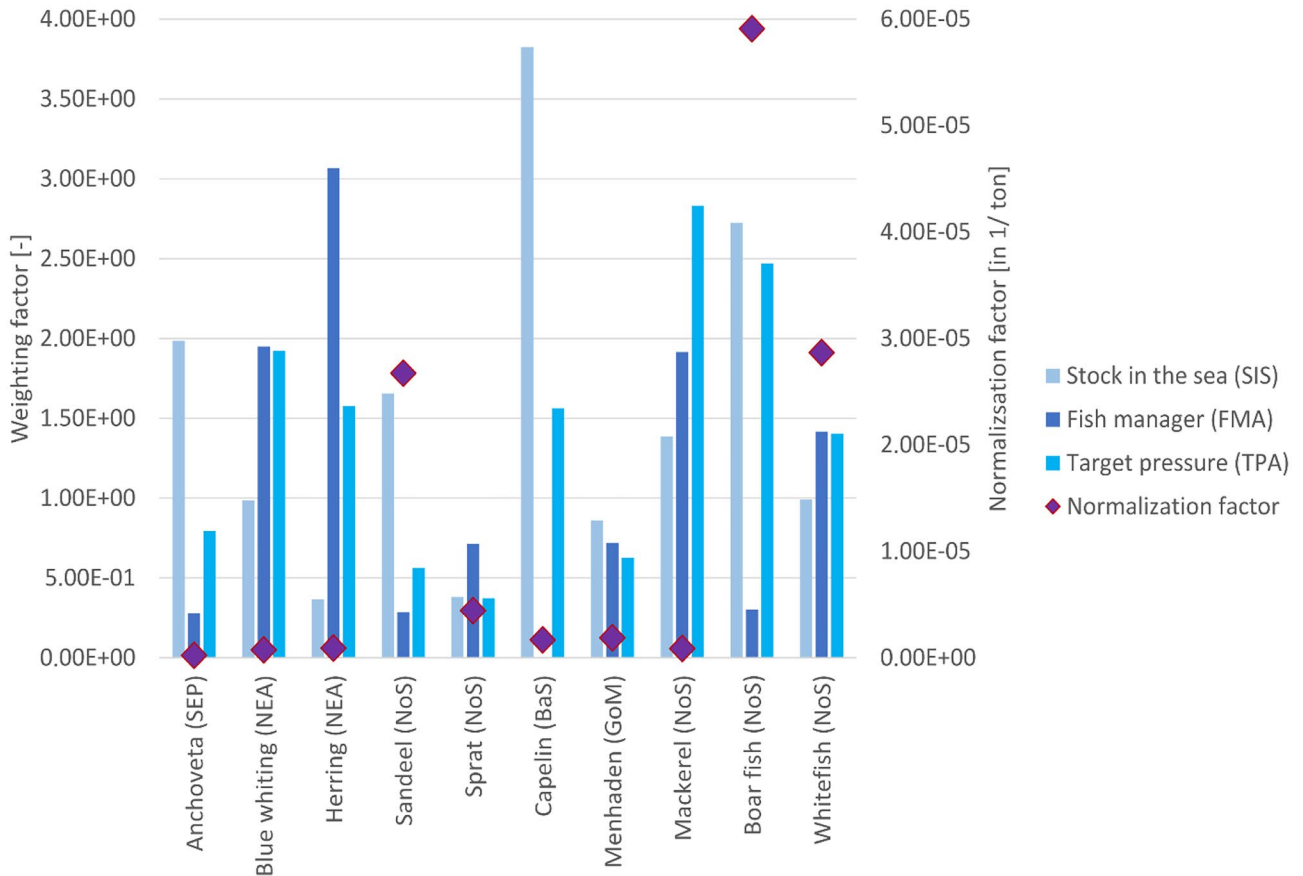


Fig. 2 Weighting factors of the three global species-specific approaches (SIS, FMA, and TPA — blue colored bars) and normalization factors (violet rhomb) for the ten considered species

Considering the results of the three global species-specific approaches, the SIS and TPA are more similar compared to the FMA. This can be explained by the influence of the catch, which is used as the current flow as well as the normalization flow in the FMA. The catch therefore has a higher impact on the results compared to the other two approaches.

For the RMA, the hotspots are mackerel (NoS), blue whiting (NEA), and boar fish (NoS). Due to the calculation approach (see Eq. (6)), the ecofactors of the fish species only differ due to the overfishing indicator. As seen in Table S4, the hotspot fish species show the highest overfishing CFs, meaning that they face the highest risk of overfishing. Weighting and normalization factor are the same for all species in this approach. The calculation of the overfishing indicator shows similarities to the TPA (applying the fishing parameter F and F_{MSY}). However, the hotspots differ significantly compared to the species specific approaches.

3.2 Results for fish oil and fish meal

Next, the determined ecofactors are applied to a specific case study to determine the ecopoints for fish oil and fish meal production in Europe. Thus, the mass of the fish used within fish oil and fish meal is considered. The functional unit of 1-kg fish meal and oil was chosen. Fish meal and oil consist of a variety of different fish species caught in specific regions and is mainly used in aquaculture as fish feed. In 2018, around 22 million tons of fish were used to produce fish meal and oil globally (Food and Agriculture Organization of the United Nations 2016).

The considered fish species and their associated meal and oil yield are based on data of Cashion et al. (2016) and Bosch et al. (2019). Results of fish oil are presented in the paper (see Table 3), whereas the results for fish meal are

shown in the Supporting information (Sect. 7). It should be pointed out that the comparison of the ecopoints for fish oil (and fish meal) only allows to determine the hotspot fish species of each approach. No conclusions can be made which approach is more feasible, as the approaches have different scopes. Whereas the global species-specific approaches have a global scope, the RMA has a European focus. Meaning the overall result of the global species-specific approach and RMA cannot be compared directly. The lower overall result of the RMA therefore does not indicate lower overfishing impacts than shown in the global species-specific approaches.

When the mass is considered as well, the main contributions for the SIS are made by whitefish (NoS), sandeel (NoS), and boar fish (NoS) (Fig. 3). The influence of the boar fish is reduced, because the mass used in fish oil is comparatively low (0.3%). The impact of sprat (NoS), anchoveta (SEP), and menhaden (GoM) increases due to the high mass used in fish oil. These three fish species together with the herring (NEA) sum up to 90% of the overall used amount. Thus, their overfishing impact is enhanced by the high amount used.

A difference can be seen in comparison to the FMA, where the herring (NEA) and whitefish (NoS) have the highest contribution, and sandeel's (NoS), caplein's (BaS), and boar fish's (NoS) impact is smaller, because the used amount of herring (NEA) in fish oil is the second highest. The amount of whitefish is rather low compared to herring and the other species being a hotspot in SIS.

For the TPA, whitefish (NoS), boar fish (NoS), and herring (NEA) show the largest impacts. Whereas for whitefish (NoS) and boar fish (NoS), the overall used amount in fish oil is low, the herring (NEA) is a hotspot because of its high amount used.

Table 3 Results for 1 kg of fish oil (in ecopoints)

Fish species	Mass shares (in %)	Global species-specific approach (ecopoints)			RMA—consumption based (ecopoints)
		Stock in the sea (SIS)	Fish manager (FMA)	Target pressure (TPA)	
Anchoveta (South Eastern Pacific)	40.48	705.66	98.45	282.07	0
Blue whiting (NE Atlantic)	0.37	10.01	19.80	19.52	46.1
Herring (NE Atlantic)	24.19	307.29	2570.36	1321.90	1960
Sandeel (North Sea)	1.45	2423.08	418.25	821.75	0
Sprat (North Sea)	8.66	553.55	1035.39	541.68	0
Capelin (Barents Sea)	2.35	573.53	0.46	234.35	189
Menhaden (Gulf of Mexico)	17.47	1061.00	888.04	772.48	0
Mackerel (North Sea)	2.45	111.33	153.76	227.30	539
Boar fish (North Sea)	0.32	1944.33	215.73	1763.12	58.9
Whitefish (North Sea)	2.28	2445.39	3496.77	3464.81	136
Overall ecopoints		10,135.15	8897.02	9448.99	2903

Mackerel (NoS) shows no hotspot in any of the approaches, because of its small amount used in fish oil, which counteracts the high weighting factors, leading to an overall low result. For the boar fish (NoS), the low amounts do not lead to low ecopoints due to the high weighting for the SIS and TPA and the high normalization factor. The rather low contribution of anchoveta (SEP) despite its high amount used can be explained by the low weighting and normalization factor. Anchoveta has the lowest normalization factor due to its high availability in the South Eastern Pacific despite its high catch. The weighting factors are low as well, because the sustainable limits are less exceeded, especially for pressure on fish stocks and sustainable yield.

When analyzing the absolute results of the global species-specific approach (see Table 3 and Fig. S3), it can be seen that the overall result of the SIS is higher compared to the other two approaches, meaning that the remaining biomass to guarantee repopulation is not within sustainable limits, therefore being the most crucial aspect with regard to overfishing for the case study. The lowest overall result is associated with the FMA, meaning that the MSY is less exceeded compared to the biomass and fish stocks. Overall, it can be stated that similar weighting factors lead to similar hotspots in the results, e.g., whitefish (NoS) has similar weighting factors for all three approaches and similar hotspots in the results. For capelin (BaS) where the weighting factors vary between the approaches, its overall contribution also differs.

When comparing the impacts of the global species-specific approach with the RMA, it can be seen that herring (NEA) has the main hotspot, followed by mackerel (NoS).

3.3 Comparing fish oil with rape seed oil

Last, to shortly demonstrate the possibility to compare ecopoints across different impact categories, fish oil is set in comparison with rape seed oil (EU: 28 Rapeseed oil, raw (economic allocation) dataset from GaBi (Sphera 2021)). Rapeseed oil was chosen as it could roughly be used for similar purposes as fish oil. However, the omega-3 content, on which the functional unit should be based on, is not comparable. One kilogram rape seed oil is therefore not a functional equivalent for 1 kg fish oil. It has to be highlighted that the comparison is mostly done to demonstrate the application of the developed approach for two similar product systems. The result should not be used to make any claims with regard to the comparable environmental performance of these product systems.

The modelled dataset for fish oil is based on an unpublished case study. Thus, detailed inventory data cannot be presented. For the rape seed oil ecopoints of climate change, and land use are determined using the EU ecofactors provided by Muhl et al. (2019). As ecofactors are not available for marine eutrophication, an own approach was derived, using the CFs for eutrophying substances from Frischknecht and Büsser Knöpfel (2013), the distance flow based on Sala et al. (2015), and the target flow proposed by Bjørn and Hauschild (2015) (see Supporting information Sect. 9).

In Fig. 4, the overall results of comparing fish oil and rape seed oil are shown. It can be seen that the impacts of climate change, marine eutrophication, and land use are comparatively low. Land use impacts are higher for the rape seed oil

Fig. 3 Ecopoints of 1 kg of fish oil of the four introduced approaches; scaled to 100%

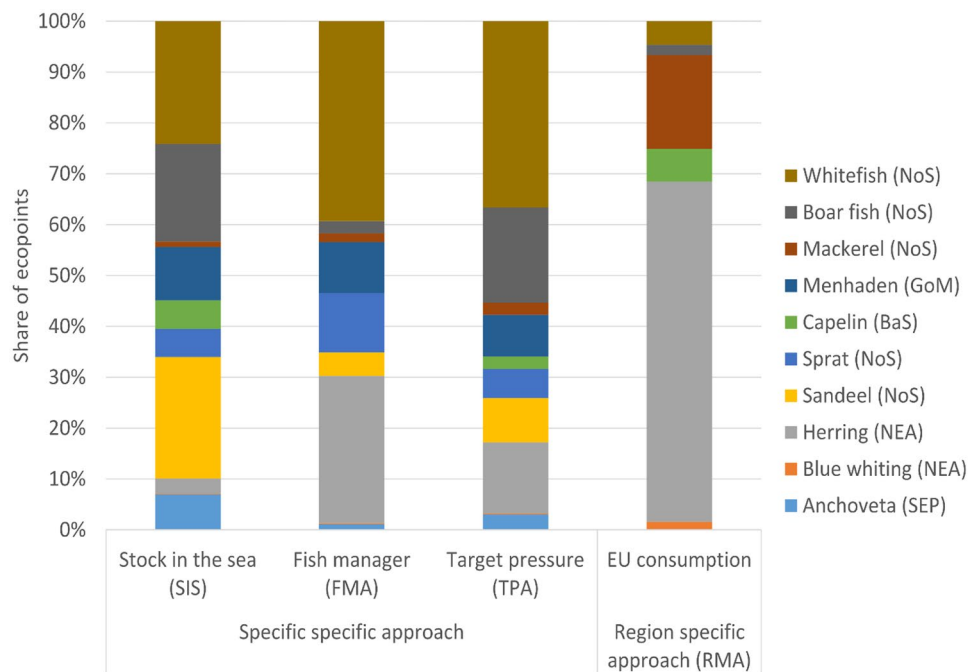
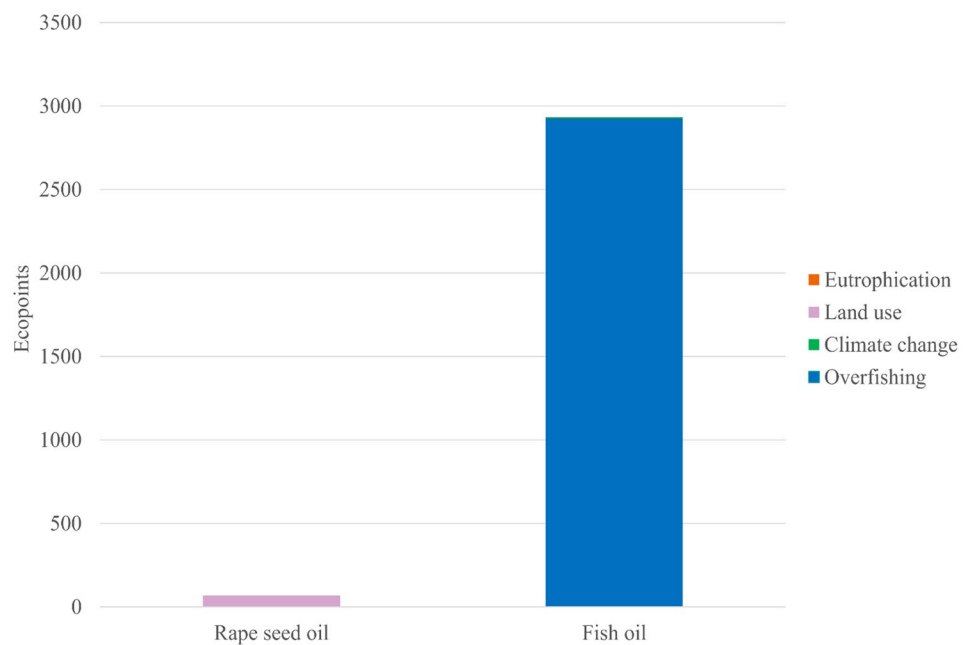


Fig. 4 Comparison of ecopoints of fish oil and rapeseed oil considering the impact categories marine eutrophication, land use, climate change, and overfishing



compared to fish oil, because it is an agricultural product. Impacts of climate change are higher for fish oil due to the fuel needed for the fishing boats as well as energy to produce the fish oil. Lastly, the impact of the category overfishing is the highest for the fish oil, dominating the overall result with 99%. Rapeseed oil does not have any overfishing impacts. Since there is no adequate functional equivalence between fish oil and rape seed oil, the results have to be considered with caution. However, the careful conclusion can be drawn that using fish oil instead of rape seed oil leads to more environmental impacts. To determine how the results would change if the fish oil is composed differently, a sensitivity analysis is carried out and presented in Sect. 4.2.

4 Discussion

The introduced approaches as well as the case study results face several challenges, which are discussed in the following.

4.1 General challenges

Within this paper, ecofactors for only 10 species in specific areas are provided. To apply the approaches also within other case studies with other fish species in additional ocean, ecofactors have to be determined, which can be time consuming.

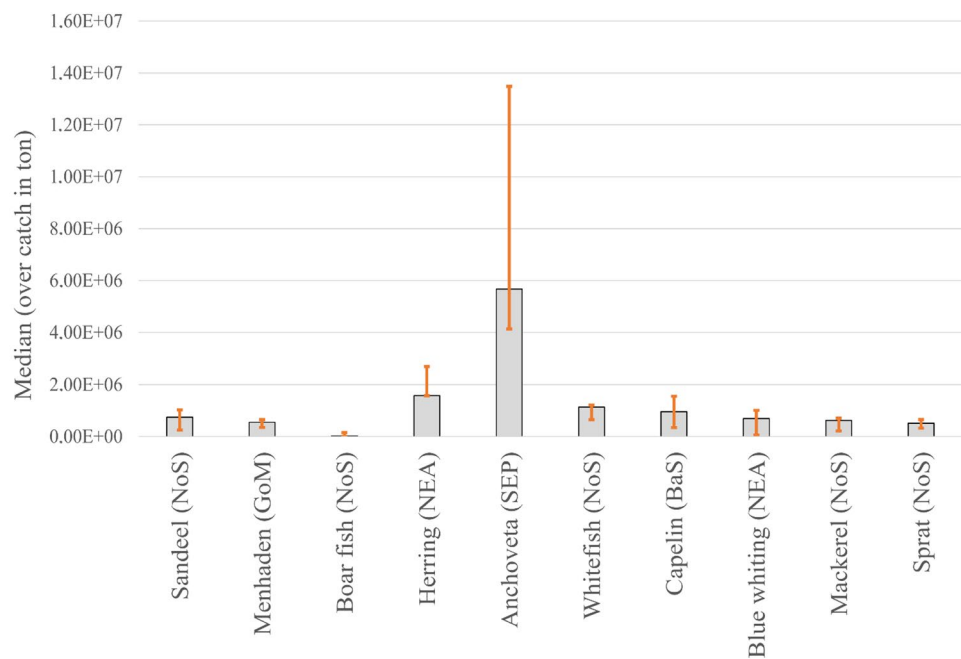
It should also be highlighted that the current approaches relate every species within a specific region, e.g., North East Atlantic, to a specific ecofactor due to the particular characteristics of each species within each region. Another option could be to determine ecofactors for a specific species, e.g.,

herring, within all oceans to derive only one value for every species.

Furthermore, often species- and region-specific data is missing. Thus, for the species considered in the case study, two data sources (Sustainable Fisheries Partnership 2019; Hélias 2019) are used, which already lead to certain inconsistencies in the results. Additionally, the applied data is partly fragmented, because data for many species is not regularly collected. The variability of data is shown in Fig. 5, where catch data over the time frame from 1980 to 2016 is presented. It can be seen that anchoveta (SEP) has the highest median, because it is the fish species mostly fished in the considered oceans. However, it also has the highest variability (presented by orange bar), because the overall amount caught differed significantly over time. This variability over time is smaller for the other fish species, especially for sprat (NoS) and boar fish (NoS). However, data for boar fish has only been collected for the last 10 years, compared to the other fish species where data is available since the 1980s.

The presented variability is not only due to lacking data, but also because the considered system is a living environment, where variability occurs natural. Thus, when interpreting overfishing results using fish parameters like C , B , and F (and associated parameters), one has to keep in mind that the results can vary significantly between years. Therefore, when determining which fish species to use from an overfishing point of view, e.g., in fish oil, several years should be taken into account. Additionally, monitoring of species over the years could be enhanced to better identify changes within the months.

Fig. 5 Variability of global catch data over the years 1980–2016, grey bars showing the median; orange bars showing statistical uncertainty



The applied target values for overfishing are set quite ambitiously as it was defined at phasing out overfishing by 2030 (European Commission 2020). Thus, for many fish species used in fish oil and meal, this target is reached and often exceeded (many fish species are already overfished or at the border of being overfished). By applying such strict target values and comparing overfishing result to other impacts to, e.g., climate change, a dominance of the overfishing category might occur as shown by Bach et al. (2020) as well as in the case study comparing fish oil with rape seed oil.

From a methodological point of view, the biggest challenge is associated with the normalization factor. In the case study, it was demonstrated how significantly it influences the results. High normalization flows refer to high emissions or resource use which leads to a comparably lower ecofactor, whereas lower emissions or resource use result in higher ecofactors. Thus, the results have to be carefully interpreted considering the influence of the region-specific normalization factors. For an enhanced comparability of the different ecofactors on a global scale, the development of ecofactor considering global normalization flows for each species would be desirable.

The introduced and applied approaches can reflect the scarcity of fish species by overfishing. However, several other relevant aspects are not considered, such as damage to the ecosystem as well as to the ocean floor, bycatch, ghost fishing, and discard rate ratios. These shortcomings have been discussed in science throughout the years

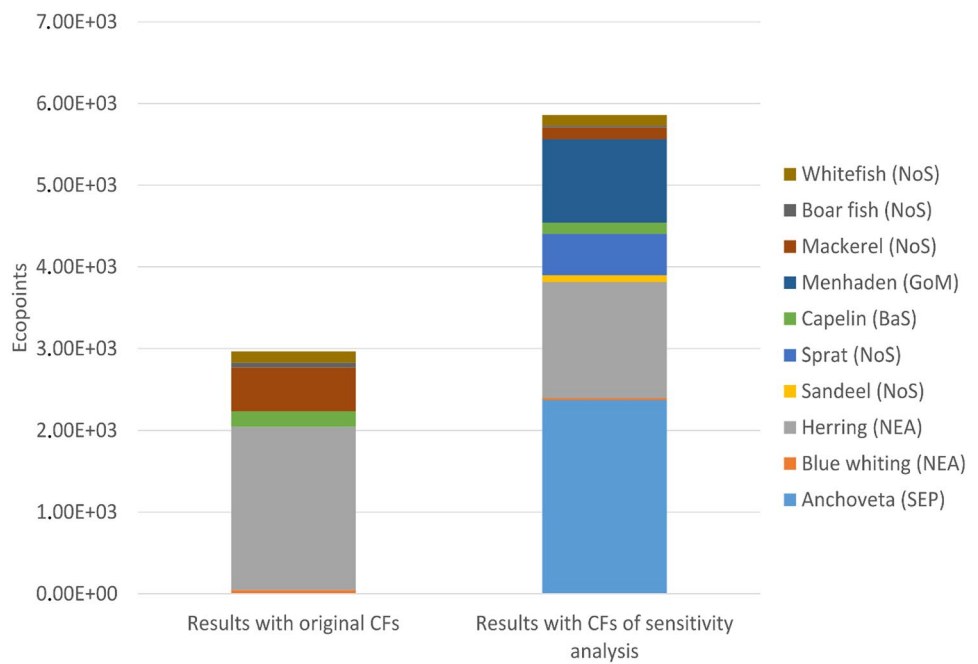
(e.g., Schneider et al. 2016), because interconnections and trophic interactions of species in multispecies systems are often not considered as well as impacts on non-harvested species (e.g., Reynolds 2008; Ghosh and Kar 2014). Finally, the developed approaches for overfishing are new and have only been applied for one case study. To make sure that they provide reliable results and to gain further knowledge on how to interpret the results, they have to be tested on more case studies.

4.2 Sensitivity analyses

For some species, the region-specific ecofactors and therefore ecopoints are zero due to the CF being set to zero due to its negative values. Negative values mean that the pressure of fish stocks is below sustainable limits. Within the RMA, this leads to a zero result; therefore, reflecting that resource depletion does not occur for these species (e.g., anchoveta (SEP)). However, within the global species-specific target pressure approach, the ecopoints are determined without subtracting one and ecofactors being set to zero.

Thus, the applied CFs for the region-specific approach are further analyzed by calculating them without subtracting one (see Eq. (7)). Therefore, the CFs are not negative and no value is set to zero (see Table S5). The sensitivity CFs show higher result for all species, except herring (NEA), because it is the reference species. For the species, which have a zero value in the original CFs, the difference is most noticeable.

Fig. 6 Comparison of results for 1 kg fish oil applying the original CF and CF of the sensitivity analysis for the regional mid-point approach (consumption)



$$CF_{sensitivity_{i,a}} = \frac{\left(\frac{F_{i,a}}{F_{MSY_{i,a}}} \right)}{\left(\frac{F_{herring(NEA)}}{F_{MSY_{herring(NEA)}}} \right)} \left[\frac{1}{\text{herring} - \text{equivalents}} \right] \tag{7}$$

When comparing the results of 1 kg fish oil (see Fig. 6), it can be seen that the results with the CFs of the sensitivity analysis lead to an overall higher result. This can be explained by the influence of the fish species which have been set to zero in the original approach, i.e., anchoveta (SEP), sandeel (NoS), sprat (NoS), and menhaden (NoS). Especially anchoveta (SEP), which also has a high mass share in fish oil, contributes significantly.

By using the CFs according to Eq. (7), all fish species are considered in the analysis. This concept has also been applied in the global species-specific approach, where all species contributions are taken into account. There are arguments made that every potential resource depletion should be presented and therefore setting CFs to zero is not recommended. However, due to their high mass in fish oil, the fish species not facing overfishing influence the results significantly, which leads to an overestimation of the overall result as well as of these species contributions. Furthermore, the overfishing risk of these species is low or not even existing. This is accurately reflected in the original CFs. Setting CFs to zero, because no risks can be observed, is an approach that has been applied in the assessment of abiotic resource criticality (e.g., Bach et al. (2016) to provide a better reflection of the actual scarcity and facilitate interpretation. In

conclusion, arguments can be made for applying the original CFs and for applying the CFs of the sensitivity analysis. Thus, both approaches should be applied in further case studies to get an even better understanding of the advantages and disadvantages of both approaches and to contribute to a decision, which approach fits the methodological needs best.

Next, a sensitivity analysis is carried out, considering different compositions of fish oil. It is assumed that 1 kg of fish oil consists of one species only, limited to the ten fish species considered in the case study. Fish oil is not produced based on only one species, but the composition of fish oil can vary depending on the caught fish. The sensitivity analysis shows the influence of the chosen fish species on the overall results. As each species has a different oil yield (see Table S1), the amount needed for 1 kg fish oil also differs (see Table S13). The ecopoints are calculated using the original CFs as well as the CFs of the sensitivity analysis.

The ecopoints for 1 kg fish oil based on only one species differ from the results for the fish oil mix. The results are higher for all species for which the CF is above zero. The results for fish oil only based on boar fish (NoS) have the highest result, six times higher than the fish oil mix. This can be explained by the CF of the boar fish (NoS), which is among the highest with 2.2 herring-equivalents (see Table S4). Furthermore, the amount of boar fish (NoS) needed to produce 1 kg fish oil is one of the highest with 4 kg (see Table S13). The result of the capelin (BaS) is the lowest, as its CF is one of the lowest with 0.98 herring

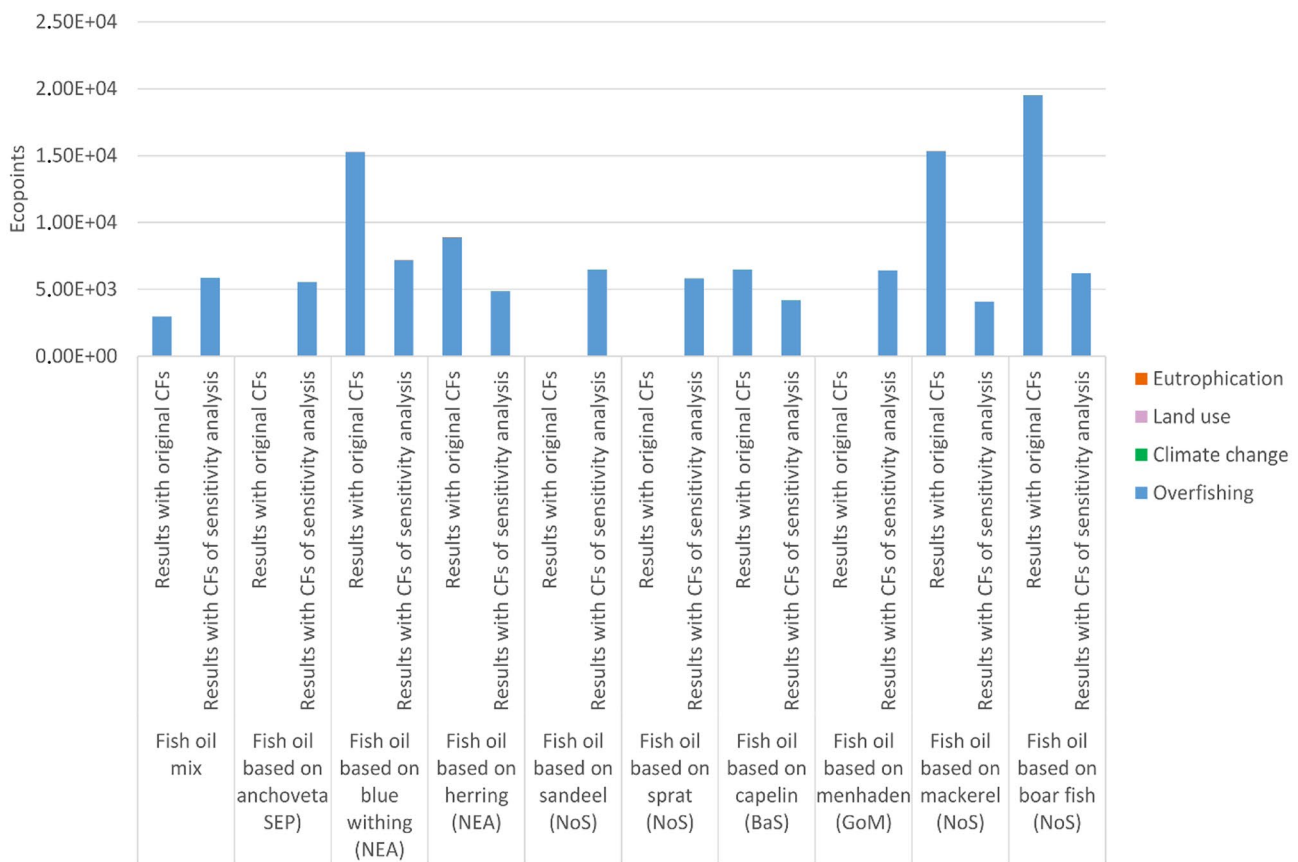


Fig. 7 Comparison of results for 1 kg fish oil mix with fish oil based on one of the considered fish species, applying the original CF and CF of the sensitivity analysis for the regional midpoint approach (consumption)

equivalents. However, its result is still two times higher compared to the fish oil mix. The overall lower result of the fish oil mix can be explained by the influence of the species with a CF of zero like anchoveta (SEP) (Fig. 7). This is different for the results with the CFs of the sensitivity analysis, where some species-specific results are lower and some are higher compared to the fish oil mix. The highest result has the fish oil based on whitefish (NoS), which is 1.4 times higher compared to the fish oil mix. The lowest result has the fish oil based on mackerel (NoS), which has a 0.3-time lower result. The differences of the fish oil results for the CFs of the sensitivity analysis are much lower — deviating between 0.3 times lower and 1.4 times higher — compared to the fish oil results with the original CFs — which deviate between zero and 2.23 times higher. This sensitivity analysis demonstrates that the overall results highly depend on the used fish species and therefore, each fish-based products has to be analyzed individually based on the actually fish species it contains.

5 Conclusions

A contribution to measuring overfishing in LCA is made by introducing five assessment approaches based on the DtT method to account for overfishing from a scarcity point of view and allow for comparing the results with impact assessment results of other categories. Four out of five approaches were applied within a three-level case study to test their applicability and show which factors and parameters influence the results as well as how results should be interpreted. Depending on the goal and scope of the case studies, not all four approaches can be applied as demonstrated in the presented case study. The RMA approaches can be best integrated into LCA, because they allow for a direct comparison of different impact categories. All approaches can be applied to gain a more comprehensive comparison of environmental impacts of different diets as well as aquaculture feed solutions.

Supplementary information The online version contains supplementary material available at <https://doi.org/10.1007/s11367-022-02042-z>.

Acknowledgements We would like to acknowledge the support of Evonik Industries AG and DSM Nutritional Products, parents companies of the joint venture Veramaris, producing omega-3 fatty acids from nature marine algae.

Funding Open Access funding enabled and organized by Projekt DEAL. Financial support was received by Evonik Industries AG.

Data availability The authors declare that all data supporting the findings of this study are available as stated within this published article (and its Supplementary information files).

Declarations

Conflict of interest The authors declare no competing interests.

Open Access This article is licensed under a Creative Commons Attribution 4.0 International License, which permits use, sharing, adaptation, distribution and reproduction in any medium or format, as long as you give appropriate credit to the original author(s) and the source, provide a link to the Creative Commons licence, and indicate if changes were made. The images or other third party material in this article are included in the article's Creative Commons licence, unless indicated otherwise in a credit line to the material. If material is not included in the article's Creative Commons licence and your intended use is not permitted by statutory regulation or exceeds the permitted use, you will need to obtain permission directly from the copyright holder. To view a copy of this licence, visit <http://creativecommons.org/licenses/by/4.0/>.

References

- Ahbe S, Weihofen S, Wellge S (2017) The ecological scarcity method for the European Union - a Volkswagen research initiative: environmental assessments
- Bach V, Berger M, Henßler M et al (2016) Integrated method to assess resource efficiency – ESSENZ. *J Clean Prod* 137:118–130. <https://doi.org/10.1016/j.jclepro.2016.07.077>
- Bach V, Hélias A, Wojciechowski A et al (2020) Development of an approach to assess overfishing in Europe based on the ecological scarcity method. In: Eberle U, Smetana S, Bos U (eds) 12th International Conference on Life Cycle Assessment of Food (LCAFood2020). DIL, Quakenbrück, Germany, pp 181–185
- Bjørn A, Hauschild MZ (2015) Introducing carrying capacity-based normalisation in LCA: framework and development of references at midpoint level. *Int J Life Cycle Assess* 20:1005–1018. <https://doi.org/10.1007/s11367-015-0899-2>
- Bosch H, Wojciechowski A, Binder M, Ziegler F (2019) Life cycle assessment of applying algal oil in salmon aquaculture - challenges for methodology and tool development
- Bossek D, Goermer M, Bach V et al (2021) Life-LCA: the first case study of the life cycle impacts of a human being. *Int J Life Cycle Assess*. <https://doi.org/10.1007/s11367-021-01924-y>
- Büsser S, Frischknecht R, Hayashi K, Kono J (2012) Ecological scarcity Japan. ESU-525 services Ltd., Uster, Switzerland
- Cashion T, Hornborg S, Ziegler F et al (2016) Review and advancement of the marine biotic resource use metric in seafood LCAs: a case study of Norwegian salmon feed. *Int J Life Cycle Assess* 21:1106–1120. <https://doi.org/10.1007/s11367-016-1092-y>
- Coll M, Shannon LJ, Kleisner KM et al (2016) Ecological indicators to capture the effects of fishing on biodiversity and conservation status of marine ecosystems. *Ecol Indic* 60:947–962. <https://doi.org/10.1016/j.ecolind.2015.08.048>
- Emanuelsson A, Ziegler F, Pihl L et al (2014) Accounting for overfishing in life cycle assessment: new impact categories for biotic resource use. *Int J Life Cycle Assess* 19:1156–1168. <https://doi.org/10.1007/s11367-013-0684-z>
- Commission E (2020) Towards more sustainable fishing in the EU: state of play and orientations for 2021. Belgium, Brussels
- European Council (2014) 2030 Climate and energy policy framework
- European Union (2020) Subject: The update of the nationally determined contribution of the European Union and its Member States. In: Update of the NDC of the European Union and its Member States. https://www4.unfccc.int/sites/ndcstaging/PublishedDocuments/EuropeanUnionFirst/EU_NDC_Submission_December2020.pdf (Accessed 11 Feb 2022)
- Food and Agriculture Organization of the United Nations (2020) The state of world fisheries and aquaculture 2020 - sustainability in action. Italy, Rome
- Food and Agriculture Organization of the United Nations (2016) FAO food price indices. <http://www.fao.org/worldfoodsituation/en/> (Accessed 7 Feb 2016)
- Frischknecht R, Büsser KS (2013) Swiss eco-factors 2013 according to the ecological scarcity method. Methodological fundamentals and their application in Switzerland. *Environmental studies* no. 1330
- Frischknecht R, Krebs L, Dinkel F et al (2021) Ökofaktoren Schweiz 2021 gemäss der Methode der ökologischen Knappheit. Bundesamt für Umwelt BAFU. Bern, Switzerland
- Ghamkhar R, Hartleb C, Wu F, Hicks A (2020) Life cycle assessment of a cold weather aquaponic food production system. *J Clean Prod* 244:118767. <https://doi.org/10.1016/j.jclepro.2019.118767>
- Ghosh B, Kar TK (2014) Sustainable use of prey species in a prey-predator system: Jointly determined ecological thresholds and economic trade-offs. *Ecol Modell* 272:49–58. <https://doi.org/10.1016/j.ecolmodel.2013.09.013>
- Grinberg M (2015) Development of the ecological scarcity method - application to Russia and Germany. Technische Universität Berlin
- Hélias A (2019) Data for fish stock assessment obtained from the CMSY algorithm for all Global FAO Datasets. *Data* 4:78. <https://doi.org/10.3390/data4020078>
- Hélias A, Langlois J, Fréon P (2018) Fisheries in life cycle assessment: operational factors for biotic resources depletion. *Fish Fish* 19:951–963. <https://doi.org/10.1111/faf.12299>
- Langlois J, Fréon P, Delgenes J-P et al (2014a) New methods for impact assessment of biotic-resource depletion in life cycle assessment of fisheries: theory and application. *J Clean Prod* 73:63–71. <https://doi.org/10.1016/j.jclepro.2014.01.087>
- Langlois J, Fréon P, Delgenes JP et al (2012) Biotic resources extraction impact assessment in LCA of fisheries. In: *LCA Food*, Saint-Malo, October 2–4, 2012
- Langlois J, Fréon P, Steyer J-P et al (2014b) Sea-use impact category in life cycle assessment: state of the art and perspectives. *Int J Life Cycle Assess* 19:994–1006. <https://doi.org/10.1007/s11367-014-0700-y>
- Lecksiwilai N, Gheewala SH, Silalertruksa T, Mungkalasiri J (2017) LCA of biofuels in Thailand using Thai ecological scarcity method. *J Clean Prod* 142:1183–1191. <https://doi.org/10.1016/j.jclepro.2016.07.054>
- Muhl M, Bach V, Czaplá J, Finkbeiner M (2021) Comparison of science based and policy based distance to target weighting in life cycle assessment using the example of Europe. *J Clean Prod Manusc*
- Muhl M, Berger M, Finkbeiner M (2019) Development of eco-factors for the European Union based on the ecological scarcity method.

- Int J Life Cycle Assess 24:1701–1714. <https://doi.org/10.1007/s11367-018-1577-y>
- Muhl M, Berger M, Finkbeiner M (2020) Distance-to-target weighting in LCA - a matter of perspective. Int J Life Cycle Assess. <https://doi.org/10.1007/s11367-020-01837-2>
- Müller-Wenk R (1978) Die ökologische Buchhaltung: Ein Informations- und Steuerungsinstrument für umweltkonforme Unternehmenspolitik. Campus-Verlag, Frankfurt, Deutschland
- OECD (2004) Key environmental indicators. OECD Environment Directorate, Paris
- Pacoureau N, Rigby CL, Kyne PM et al (2021) Half a century of global decline in oceanic sharks and rays. Nature 589:567–571. <https://doi.org/10.1038/s41586-020-03173-9>
- Pope KL, Pegg MA, Cole NW et al (2016) Fishing for ecosystem services. J Environ Manage 183:408–417. <https://doi.org/10.1016/j.jenvman.2016.04.024>
- Reynolds JD (2008) Conservation of exploited species (Conservation Biology, Band 6)
- Ridoutt BG, Hendrie GA, Noakes M (2017) Dietary strategies to reduce environmental impact: a critical review of the evidence base. Adv Nutr an Int Rev J 8:933–946. <https://doi.org/10.3945/an.117.016691>
- Rogers A, Harborne AR, Brown CJ et al (2015) Anticipative management for coral reef ecosystem services in the 21st century. Glob Chang Biol 21:504–514. <https://doi.org/10.1111/gcb.12725>
- Sala S, Benini L, Mancini L, Pant R (2015) Integrated assessment of environmental impact of Europe in 2010: data sources and extrapolation strategies for calculating normalisation factors. Int J Life Cycle Assess 20:1568–1585. <https://doi.org/10.1007/s11367-015-0958-8>
- Sandin G, Peters GM, Svanström M (2015) Using the planetary boundaries framework for setting impact-reduction targets in LCA contexts. Int J Life Cycle Assess 20:1684–1700. <https://doi.org/10.1007/s11367-015-0984-6>
- Schaefer, M (1954) Some aspects of the dynamics of populations important to the management of the commercial marine fisheries. Inter-American Tropical Tuna Commission Bulletin 2: 23–56
- Schneider L, Bach V, Finkbeiner M (2016) LCA perspectives for resource efficiency assessment. In: Special types of LCA. Springer Berlin/Heidelberg
- Sonderegger T, Dewulf J, Fantke P et al (2017) Towards harmonizing natural resources as an area of protection in life cycle impact assessment. Int J Life Cycle Assess 22:1912–1927. <https://doi.org/10.1007/s11367-017-1297-8>
- Sphera (2021) GaBi Product Sustainability Software
- Stucki M, Keller R, Itten R, Eymann L (2018) Filling the gap of over-fishing in LCIA: eco-factors for global fish resources
- Sustainable Fisheries Partnership (2019) Fish source profiles. <https://www.fishsource.org/> (Accessed 2 Sep 2019)
- Teixeira H, Berg T, Uusitalo L et al (2016) A catalogue of marine biodiversity indicators. Front Mar Sci. <https://doi.org/10.3389/fmars.2016.00207>
- Tuomisto HL, Hodge ID, Riordan P, Macdonald DW (2012) Exploring a safe operating approach to weighting in life cycle impact assessment – a case study of organic, conventional and integrated farming systems. J Clean Prod 37:147–153. <https://doi.org/10.1016/j.jclepro.2012.06.025>
- Tyszler M, Kramer G, Blonk H (2016) Just eating healthier is not enough: studying the environmental impact of different diet scenarios for Dutch women (31–50 years old) by linear programming. Int J Life Cycle Assess 21:701–709. <https://doi.org/10.1007/s11367-015-0981-9>
- United Nations (2016) Sustainable Development Goals. In: Sustainable Development Economic and Social Council Affairs. <https://sustainabledevelopment.un.org/?menu=1300> (Accessed 1 Jan 2016)
- Vargas-Gonzalez M, Witte F, Martz P et al (2019) Operational life cycle impact assessment weighting factors based on planetary boundaries: applied to cosmetic products. Ecol Indic 107:105498. <https://doi.org/10.1016/j.ecolind.2019.105498>
- Winter L, Lehmann A, Finogenova N, Finkbeiner M (2017) Including biodiversity in life cycle assessment – state of the art, gaps and research suggestions. Integr Environ Assess Manag
- Winther U, Ziegler F, Hognes E et al (2009) Carbon footprint and energy use of Norwegian seafood products. Report No SFH80 A096068
- Woods JS, Veltman K, Huijbregts MAJ et al (2016) Towards a meaningful assessment of marine ecological impacts in life cycle assessment (LCA). Environ Int 89–90:48–61. <https://doi.org/10.1016/j.envint.2015.12.033>
- Ziegler F, Emanuelsson A, Eichelsheim JL et al (2011) Extended life cycle assessment of southern pink shrimp products originating in Senegalese artisanal and industrial fisheries for export to Europe. J Ind Ecol 15:527–538. <https://doi.org/10.1111/j.1530-9290.2011.00344.x>
- Ziegler F, Hornborg S, Green BS et al (2016) Expanding the concept of sustainable seafood using life cycle assessment. Fish Fish 17:1073–1093. <https://doi.org/10.1111/faf.12159>
- Ziegler F, Valentinsson D (2008) Environmental life cycle assessment of Norway lobster (*Nephrops norvegicus*) caught along the Swedish west coast by creels and conventional trawls—LCA methodology with case study. Int J Life Cycle Assess 13:487–497. <https://doi.org/10.1007/s11367-008-0024-x>

Publisher's Note Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.