DELIVERABLE FRONTPAGE

Project number:	243827 FP7-ENV-2009-1
Project acronym:	LC-IMPACT
Project title:	Development and application of environmental Life Cycle Impact assessment Methods for improved sustainability Characterisation of Technologies.
Deliverable number:	D 1.8
Deliverable name:	Recommended assessment framework, method and characterisation factors for marine resource use impacts: phase 3 (report and model + factors)
Version:	1
WP number:	1
Lead beneficiary:	SIK
Nature:	R [1]
Dissemination level:	PP[2]
Delivery date Annex I:	33 (month number)
Actual delivery date:	24/04/2013
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Comments:	#

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List of abbreviations

B: Total biomass of a fish stock
B_{lim}: The threshold level of spawning stock biomass size above which recruitment is not impaired, the lower limit for a stock to be within "safe biological limits"
B_{MSY}: Size of spawning stock at MSY
B_{pa}: Threshold level of spawning stock biomass size according to the precautionary approach, higher than B_{lim}, lower than B_{MSY}.
CF: Characterization Factor
CR: Critically Endangered (Red List category)
EN: Endangered (Red List category)
F_{MSY}: Fishing mortality at MSY
F_{pa}: Fishing mortality according to the precautionary approach, lower than Flim, higher than F_{MSY}.
F_{lim}: Fishing mortality over which recruitment is impaired, often termed as the upper "safe biological limit".
FAO : Food and Agriculture Organization
IPCC : Intergovernmental Panel on Climate Change
ISO: International Standards Organization
LC: Least concern (Red List category)
LCA: Life Cycle Assessment
LCIA: Life Cycle Impact Assessment
LCI: Life Cycle Inventory
LPY: Lost Potential Yield
MSY: Maximum Sustainable Yield, the highest long-term sustainable yield from a stock
NE: Not evaluated (Red List category)
NT: Near threatened (Red List category)
NPP: Net Primary Productivity
TL: Trophic level
UNEP : United Nation Environmental Program
VU: Vulnerable (Red List category)

Definitions

B-Overfishedness: Current biomass compare to target biomass at MSY.
Biomass (B): Estimate of the live-weight of the total mature part of a stock, in a surplus production model this equals SSB the spawning stock biomass
By-catch: The part of the catch that is not directly targeted for. This could be commercial species that are landed, often species with little data available for and thus less managment opportunities. By-catches could also consist of other species that fish, or fish species of no or little commercial interest or at sizes below legal landing size, which then would be thrown back to sea as discards.
Carrying capacity (K): when biomass of a species has reached a limit when it is restricted to increase due to factors such as e.g. competition of resources.
Demersal fishery: Fishery targeting demersal species (i.e. living close to or in the seafloor as opposed to pelagic species who live higher in the water mass) often in connection to the seafloor) conducted near bottom of the sea, often in contact with the seafloor.
Discard: The part of the by-catch that is thrown overboard at sea (non-target species, juveniles or over-quota target species)
F-Overfishing: Current fishing mortality compared to target fishing mortality at MSY
Fishing mortality (F): A proportion of stock harvested by fisheries each year
Landing: The portion of the fish catch brought to the market
LPY: Lost Potential Yield is our suggested impact category for quantifying overfishing, it is a projection-based model based on current relative to optimal levels of fishing mortality and spawning stock biomass.
Maximum Sustainable Yield (MSY): The theoretical largest sustainable yield possible to take out from biological system for a long time
Quota: Politically determined maximum amount of fish that can be harvested from a stock per year, preferably based on a scientific basis
Pelagic fishery: Fishery conducted in the pelagic, i.e. the free water column
Red List: A categorization of species according to different need of conservation priorities depending on the relative threat of extinction. This could be done either at a global scale (by the IUCN) or regionally by national authorities
Red List Index: Indicator for monitoring global biodiversity trends suggested as a measure of progress towards conservation goals, recognized by the Convention of Biological Diversity (CBD). **Replenishment (R):** Re-growth of a population
Spawning: Reproduction of fish
Spawning stock biomass (SSB): Biomass of fish in a stock that have reached maturity, i.e. are reproducing
Stock: Geographically and genetically limited population of a species, e.g. North Sea haddock, Eastern Baltic cod.
Stock assessment: Scientific assessment of the size and composition of fish stocks. Many types of data can provide the basis for a stock assessment which itself is the basis for scientific advice regarding quotas and other limitations of a fishery.

Surplus Production Model: Dynamic biomass model capturing the mechanisms of density dependency and logistic growth
TAC: Total Allowable Catch, i.e. total quota that is politically determined each year
Target stock/species: One or several stocks/species that are the main targets of a fishery, e.g. cod or groundfish (cod/haddock) or herring
TL: Trophic leveof a fish, i.e. its position on the food web
Yield (Y): Total annual landing from a stock
VEC: Sum of weight or number of individuals of fish in the discard per kilo of landing that belongs to the threat categories Vulnerable, Endangered or Critically Endangered according to the Red List.

Recommended assessment framework, method and characterisation factors for marine resource use impacts: phase 3 (report and model + factors)

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Executive Summary

Seafood LCAs have up until now excluded assessment of impacts on target fish stocks, or dealt with them in a qualitative way, and mainly addressed by-catches in kilo per landing. These biological impacts of fishing are certainly the most direct impacts related to fisheries, and are the focus of fisheries management and certification schemes. The aim of the research undertaken within LC-IMPACT related to marine resource use has been to go from qualitative to quantitative assessment of biological impacts of fishing. One important goal has been to explain why biotic impact assessment is not static over time and how to calculate updated characterization factors for the specific stock and time period relevant in a study. Utilizing the same data sources and methodology provided, anyone should be able to apply the methods presented in other situations.

To quantify overfishing, we compare current level of spawning stock biomass and fishing mortality with optimum levels, i.e. the present distance to Maximum Sustainable Yield (MSY), which is the current management goal for EU fisheries. Three midpoint impact categories to account for overfishing in LCA are defined: *lost potential yield* (LPY), *overfishing through fishing mortality* (OF) and *overfishedness of biomass* (OB). OF reflects the ongoing overfishing while OB represents the present stock status in relation to stock size required for maximum sustainable yield. The complementary categories OF and OB can be used either to interpret LPY results, or as a simpler choice when the necessary input parameters are not available. These methods for target stock impact assessment only concern the direct impact on the stock. The wider ecosystem effects of fishing remain to be covered by additional midpoint or endpoint categories.

The same methodology could be applied to by-catch if data was available. However, the methods suggested for target stocks require availability of data from stock assessments, which only exist for the most important target species. Most by-catch species (some of which are landed and some discarded at sea) are data deficient in this respect. Two alternative approaches were therefore evaluated. The first one is primary production required (PPR), i.e. the amount of carbon needed to produce a kilo of a species at a certain trophic level, previously used in LCAs of aquaculture. Applying PPR to by-catch conveys important information on the composition of this part of the catch. However, interpretation of PPR values is not completely straightforward: It is e.g. difficult to interpret PPR as being

related to limited resources in a eutrophied coastal area, and comparing values between different ecosystems and over time. In addition, it does not provide any information of the sensitivity of the fishing impact in terms of effect on species' abundances. Therefore, the IUCN Red List of Threatened Species is suggested as a complement to assess by-catch. This approach distinguishes between fishing pressure on sensitive (i.e. threatened) species compared to more abundant species. Both by-catch methods request detailed landing and discard data, which is still rarely available. However, the two methods combined illustrate well the by-catch impact of a fishery.

Altogether, the three new mid-point indicators complement traditional fisheries LCA to more comprehensively quantify relevant environmental impacts of fisheries, or in aquaculture using marine feed inputs such as fish meal and oil. While requiring additional data inventory from non-typical sources for LCA practitioners as well as new approaches such as calculation of characterization factors specific for the fisheries under study, they can make seafood LCAs more relevant and avoid sub-optimisation of supply chains. LCA could, used in this way, prove to be a useful tool in fisheries management, by providing a methodology to bench-mark the environmental performance of fisheries (and its products), which is known to be highly influenced by the way fisheries are managed. In this way, the methods developed here could extend the use of LCA to new areas and the methods could also be used for other purposes than LCA, e.g. as indicators to fulfill national and international regulations.

1. Marine resource use: Target stock use or quantifying overfishing

Accounting for Overfishing in Life Cycle Assessment:

New impact categories for biotic resource use

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Note: This work has been submitted for publication to the International Journal of LCA

Abstract

Purpose Overfishing is a relevant issue to include in all life cycle assessments (LCAs) involving wild caught fish, as overfishing of fish stocks clearly targets the LCA safeguard objects of natural resources and natural ecosystems. Yet no robust method for assessing overfishing has been available until now. Therefore, we propose *lost potential yield* (LPY) as a midpoint impact category to quantify overfishing, primarily reflecting the impact on biotic resource availability, but also to act as a proxy for ecosystem impacts within each stock.
Materials and methods LPY represents average lost catches owing to ongoing overfishing, assessed by simplistic biomass projections covering different fishing mortality scenarios. It is based on the maximum sustainable yield concept and complemented by two alternative methods, overfishing though fishing mortality (OF) and overfishedness of biomass (OB), that are less data-demanding.
Results and discussion Characterization factors are provided for 31 European commercial fish stocks in 2010 representing 74% of European and 7% of global landings. However, large spatial and temporal variations were observed, requiring novel approaches for the LCA practitioner. The methodology is considered compliant with the International Reference Life Cycle Data System (ILCD) standard in most relevant aspects, although harmonization through normalization and endpoint characterization is only briefly discussed.
Conclusion With the LPY midpoint methodology, LCA provides a more complete tool for assessing the environmental impacts of seafood products.

Keywords Overfishing, Life cycle impact assessment, Seafood life cycle assessment, Maximum sustainable yield, Lost potential yield

1.1. Introduction

Today, over 80% of the world's fish stocks are considered fully exploited or overexploited (FAO 2012). Global marine fish catches have stabilized around 80 million tonnes annually since the early 1990s (FAO 2012), although the effort spent to catch fish has steadily increased after the catches started to peak (Anticamara et al. 2011), and the fishing fleets have expanded toward deeper and more remote fishing locations (Swartz et al. 2010). This problem of overfishing of fish stocks, which are spatially or temporally separated in their reproduction and depend on their own stock size and structure for growth, has been widely acknowledged in scientific press (Pauly et al. 2002; Worm et al. 2009; Froese and Proelß 2010).

The present extinction rate and loss of biodiversity have been identified as humanity's most severe passing of the planetary boundaries (Rockström et al. 2009), and the Millennium Ecosystem Assessment established overfishing as the main driver of biodiversity loss, as opposed to habitat change for most terrestrial systems (MEA 2005). Thus the commercial harvesting of a few stocks indirectly affects the entire ecosystem. Overfishing directly limits a biotic resource that currently accounts for 17% of the animal protein intake worldwide, with high nutritional and economic values that are crucial for many low-income and food-deficient countries (FAO 2012).

1.1.1 Life Cycle Assessment

Increased knowledge about environmental threats has raised the demand for sustainable seafood and increased the incentives to improve products and production processes (Thrane et al. 2009). Life cycle assessment (LCA) is here a useful, acknowledged, and standardized method to assess potential environmental impacts over a product life cycle from cradle to grave (ISO 2006a, 2006b). The European Commission has concluded that LCA provides the best framework for describing the environmental impacts of products and services currently available (EC 2003). One of the benefits is the ability to compare products and impacts in a quantitative way, either by potential impacts in terms of midpoint impact categories or by potential damage as endpoint categories. Endpoint categories have higher model uncertainty but also higher explanatory value, and they target three defined areas of protection (AoPs): natural ecosystems, natural resources, and human health (Finnveden et al. 2009; ILCD 2010b). According to the International Reference Life Cycle Data System (ILCD) standard for best practice in LCA, an interpretation of the ISO standards, it is mandatory to check and address the interpretation of damage pathways towards these AoPs, to support the choice of suitable impact categories. If no such method exists, it should be developed and included, or it should be clearly stated in the Goal and Scope definition of the LCA, that is does not account for all relevant flows (ISO 2006a, 2006b; ILCD 2010a).

1.1.2 State of the art in seafood LCA

The theory behind biotic resource use in LCA was outlined in the 1990s and reviewed by the Society of Environmental Toxicology and Chemistry (SETAC), which led to a conclusion of a two-fold impact pathway separating resource and ecosystem damage (Haes et al. 2002). It also forecasted that more sub-impact categories would be developed to tackle the heterogeneity of impact pathways, under the broad impact category of "biotic resource use"

(Haes et al. 2002); note that a primary production-based impact category has been proposed under the same name (Papatryphon et al. 2004).

Since the 1990s, more than one hundred seafood production systems have been described with LCA, including both fisheries and aquaculture systems, the latter often depending on feed inputs from capture fisheries (Parker 2012), and a rapid increase in seafood LCAs has been recorded (Avadí and Fréon 2013). Yet none of the original methods (Haes et al. 2002) have been used in published seafood LCA case studies (Pelletier et al. 2007; Parker 2012; Vázquez-Rowe et al. 2012a; Avadí and Fréon 2013), possibly owing to lack of applicability.

The lack of methodology to assess impacts on target stocks has limited the scope of seafood LCAs, and this limited scope has been concluded to significantly impair the value for LCA as a management tool (Pelletier et al. 2007). Yet a wide range of seafood-specific impact categories have been proposed and presented, mainly regarding bycatches and discard (Ziegler et al. 2003; Emanuelsson 2008; Ziegler et al. 2011; Vázquez-Rowe et al. 2012b). Discards have also been recently characterized based on the primary production required and the frequency of threatened species (Hornborg et al. 2012). Some methodology for seafloor disturbance area (Ziegler et al. 2003) and a number of methods for assessing specific aquaculture impacts (Ford et al. 2012) have also been presented, but a methodology to include target stock overfishing in LCAs does still not exist.

1.1.3 Lost yields

Maximum sustainable yield (MSY), which is the theoretical maximum annual landing (or yield) that can be harvested from a wild population for an infinite amount of years, has been the backbone of fisheries science since the beginning of the 19th century, outlined even before mathematical models were available to support it and typically comparing landings with MSY (Punt and Smith 2001). In economic terms, global fishery systems are currently far from optimized, leaving many fisheries with low profitability as a result of low stock size and overcapacity (FAO 2012). If stocks were restored to larger biomasses and after that exploited sustainably, global profits have been estimated to increase by US$50 billion annually, which represents more than half of the value of current landings (FAO 2008).

MSY management is now reinstated as a goal of the European Union, which has agreed to restore all stocks to levels capable of producing maximum sustainable yield (EC 2006). However, the MSY is not a fixed goal, but rather is defined by its regulating components: (1) optimal fishing mortality F_{MSY} (a proportion of the stock harvested) and (2) optimal biomass size B_{MSY} (Froese and Proelß 2010; ICES 2012a).

With increased fishing pressure over time, the fishing mortality F will increase, and the biomass will decline from a pristine, i.e. unfished, condition (see the dotted line in Fig. 1.1). Note that in Europe, the biomass in B_{MSY} typically refers to the spawning stock biomass SSB_{MSY}, i.e. the reproducing part of the stock (Froese and Proelß 2010).

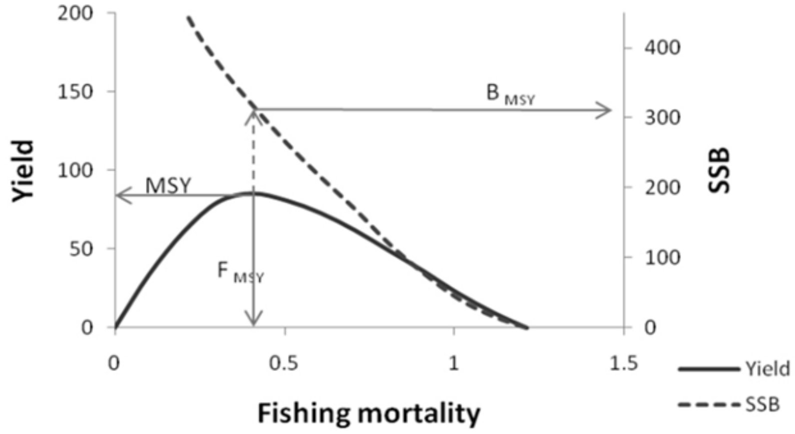

Fig. 1.1 MSY reference points and the relationship between fishing mortality, yield, and biomass (i.e. spawning stock biomass, SSB). The maximum sustainable yield (MSY) indicates the level of fishing mortality (F_{MSY}) resulting in a long-term biomass of B_{MSY} that could support the MSY. Reproduced with kind permission of ICES 2012

This conceptual model implies increased long-term yields with increasing fishing mortality (the solid line in Fig. 1.1) until F = F_{MSY}, after which the biomass and long-term yield will start to decrease as a result of overfishing, owing to density-dependent mechanisms. After continuous exploitation at F_{MSY}, the biomass B will fluctuate around B_{MSY}, enabling long-term average yields at MSY (Schaefer 1954; ICES 2012a), a well-established model (Jennings et al. 2001) with mathematical features tempting for biotic resource characterization in LCA.

1.1.4 Aim

The aim of this study was to develop quantitative methods to include overfishing in seafood LCAs based on the MSY framework. We suggested three midpoint impact categories for use under different conditions, for each of them providing characterization factors (CFs) for 31 European fish stocks representing 74% of all commercial landings in the region. We show that the suggested impact categories efficiently capture the mechanism of overfishing for these stocks, and indicate large spatial and temporal differences.

1.2. Methods

We defined three midpoint impact categories to account for single-stock overfishing in LCA: *lost potential yield* (LPY)[3] and two complementary categories, *overfishing through fishing mortality* (OF) and *overfishedness of biomass* (OB). The complementary categories may be used either for interpretation of the LPY results, or as a simpler choice when neither updated characterization factors nor input parameters are available.

In this context, we defined a stock to be fished too hard in relation to MSY, resulting in ongoing overfishing, if the rate F of exploitation exceeds F_{MSY}. This exploitation rate should be distinguished from the state of the stock, saying that if the biomass is found below the B_{MSY}, then the stock should be considered as overfished in relation to MSY. We found this terminology most suitable for LCA purposes, since it relates to the present target for fishery management (F_{MSY}) and to optimal resource levels for biotic resource implementation in LCA (B_{MSY}, and indirectly MSY).

1.2.1 Main characterization model (LPY)

The main characterization function was based on *the difference in average annual yield between a projected optimal MSY scenario and a scenario based on current fishing management.* The projection is regulated by fishing mortality F, which includes and aggregates not only (1) reported landings, but also (2) discards of juveniles (3), and assessment of (3) underreports (4) and illegal catches, when 2-4 is found relevant by the International Council for the Exploration of the Sea (ICES). However, this projection is not intended to forecast the future, since for example, a constant F is highly unlikely, but only to quantify present impacts and enable comparisons of biotic resource use among seafood products originating from different fish stocks and years.

The theoretical optimal (MSY) scenario was defined by setting F = 0 until B reaches B_{MSY} and then harvesting at F_{MSY}. The difference between the projection sums of the optimal (Y_{opt}) and current yield (Y) scenarios is then divided by the sum of current yields; see equation 1.1.

$$CF_{x,y,T} = \frac{\sum_T Y_{opt} - \sum_T Y}{\sum_T Y}$$

Equation 1.1

The characterization factors (CF) generated from equation 1 represents mass units of lost yield per current yield, from stock x during year y, averaged over a time period T. Each CF was calculated from two time series of projected biomass (current and optimal), multiplied by the annual average fishing mortality, but since ICES communicates the instantaneous fishing mortality[4] measured on a log scale, the F had to be transformed into F_{annual}; see equation 1.2.

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[3] Previously presented as wasted potential yield (WPY) in conference presentations and EU reports.

[4] *Instantaneous fishing mortality* (F_{inst}) is the F used and communicated most frequently in fisheries management, e.g. the one given in ICES advice, although it is less intuitive (measured on a log scale) than the *annual fishing mortality* (F_{annual}, the proportion killed each year), which was our input data into the projection function. For example, an instantaneous fishing mortality of 0.5, 1, and 1.5 (very high) corresponds to an annual fishing mortality of 39%, 63%, and 78% respectively of the spawning stock biomass of that stock harvested each year by the fishery.

$$Y_t \approx \hat{F}_{annual,t} B_t = (1 - \exp(-\hat{F}_{inst,t})) * B_t$$

Equation 1.2

The biomass time series B_T were established by inputs of F and B, specific for each stock and year, and F_{MSY} and B_{MSY}, specific for each stock; see section 2.3 on input data. All inputs were inserted into a year-discrete Schaefer surplus production function (Schaefer 1954), which projects next year's biomass from the previous year's biomass by adding growth and subtracting annual yield; see equation 1.3.

$$B_{t+1} = B_t + 2\hat{F}_{MSY} B_t \left(1 - \frac{B_t}{2B_{MSY}}\right) - \hat{F}_t B_t$$

Equation 1.3.

The intrinsic growth rate (r) is substituted by $2*F_{MSY}$ and the carrying capacity (K) by $2*B_{MSY}$, which follows from the assumption of logistic growth (Schaefer 1954). We also used a five year moving average of B to establish an initial B_t to better comply with the idea of B_{MSY} as a long-term goal around which B should fluctuate, in line with previous recommendations to cope with variability in seafood LCAs (Ramos et al. 2011). All other biomasses are iteratively generated from this value in the free statistical software "R" (R 2012), see the code in supplementary material S3.

We chose 30 years as the default time perspective; see the sensitivity analysis in section 2.4. To avoid undesired effects, two logic rules were applied to the iteratively derived CFs:

Logic rule 1: If a positive LPY value describes an underexploited stock ($F<F_{MSY}$) and $B>B_{MSY}$), it should not be considered as lost yield, but rather as a buffer that enables initial exploitation higher than F in a long-term harvest plan. The "lost yield" in such cases is multiplied by (-1) and represents a potential future yield.

Logic rule 2: If a false-negative CF is found due to long break-even times, a value from a more conservative (larger) T = [10 20 30 100 500] should be used or the CF should be excluded from the dataset.

1.2.2 Complementary characterization models (OF and OB)

We define *overfishing through fishing mortality* (OF) as a midpoint impact category based on the F/F_{MSY} ratio, but for LCA purposes, the characterization model has been expressed as $F/F_{MSY}-1$, so that the optimum case ($F = F_{MSY}$) equals no impact. The twin category, *overfishedness of biomass (OB),* describes the present biomass state B in relation to B_{MSY}, which for LCA purposes was modeled as $B_{MSY}/B-1$, i.e. likewise adjusted so that the impact is zero when $B = B_{MSY}$, but also inverted so that higher values mean higher impact.

1.2.3 Input data

The input data on fishing mortality, landings, and spawning stock biomass were retrieved from stock assessments[5] regarding the years 1995 to 2010 that had been conducted by ICES. Data included F_{MSY} values for 31 major European stocks available in the public ICES “Stock Summary/Standard Graph” Database (ICES 2012b). In addition, we used corresponding B_{MSY} values from Froese and Proelß (2010). All input data are provided in supplementary material S2.

1.2.4 Sensitivity & robustness analysis

To evaluate the model choice uncertainties, two major sensitivity analyses were performed: first, the dependency on the time period T was tested from 0 to 500 (approximating infinity), and second, the F_{MSY} values were replaced by the Froese and Proelß (2010) dataset. To further ensure the robustness of the LPY results, we also verified it by trends in the more simplistic OF and OB impact categories, i.e. F/F_{MSY} and B/B_{MSY} ratios throughout the time series, and we discussed qualitatively the input variability via uncertainty ranges of F, B, F_{MSY}, and B_{MSY}. General trends in temporal and spatial variability were analyzed by comparisons of coefficient of variations, but these comparisons were only done for cod, haddock, herring, and sole, which had more than four stocks in the dataset.

1.3 Results

The LPY characterization factors varied considerably between species, and even more between stocks within species; see Fig. 1.2 and Table 1.1. Three out of 31 stocks had negative LPY values, indicating underexploitation, while the remaining stocks exhibited varying degrees of overexploitation. For full names corresponding to the stocks’ geographic IDs, see supplementary material S1.

[5] A cautious LCA practitioner might notice that both parameters B (SSB) and F will vary (slightly) retrospectively for each new stock assessment, since more data are fitted to the assessment time series, increasing the model’s accuracy. For example, B regarding 2010 assessed in 2011 will be slightly updated in the 2012 assessment.

1.3.1 Main results LPY Europe 2010

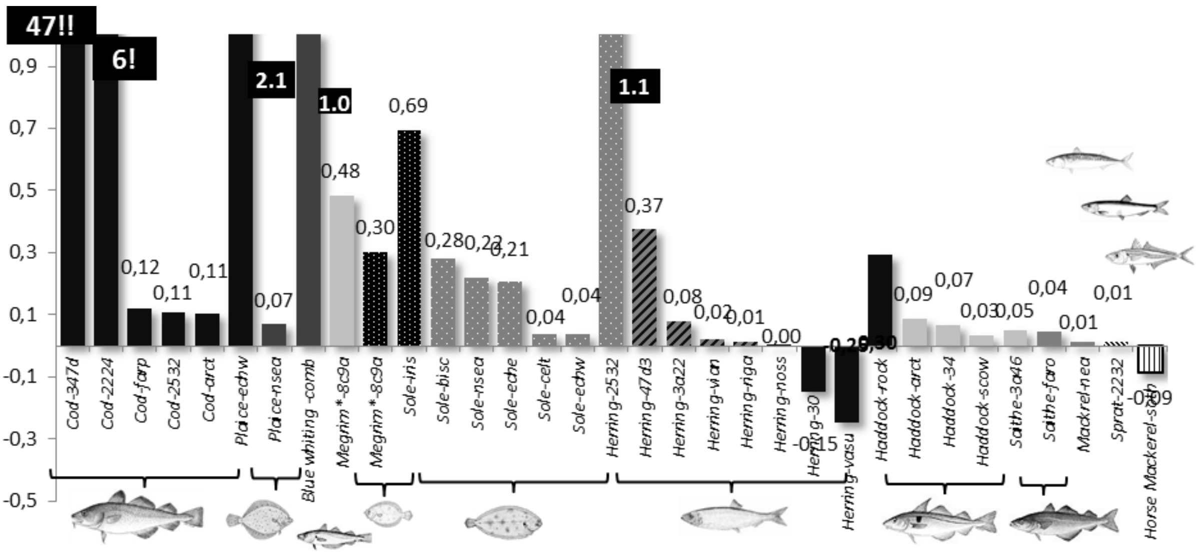


Fig. 1.2 LPY characterization factors of fish stocks in Europe 2010, sorted by highest average value for each species, and then per stock. Note the extreme values outside the scale marked with black boxes (with exclamation signs to highlight the magnitude), and the negative values describing underexploitation. Illustrations: FAO.

Generally stocks of Atlantic cod (*Gadus morhua*) were found to have the highest LPY values, although the ranking here was mostly driven by two stocks that were in extremely poor condition in 2010: cod in the North Sea and the Skagerrak (cod-3472) and Western Baltic cod (cod-2224). Plaice (*Pleuronectes platessa*) was positioned at the second-worst ranking, but consisted of two stocks in highly different condition: a high potential impact for plaice in the Western Channel (echw), and the North Sea plaice (nsea), which was close to the median LPY value. The five species closest to optimum MSY levels were all pelagic lower trophic–level species of typically smaller body size. Southern horse mackerel (*Trachurus trachurus*) and two stocks of herring (*Clupea harengus*) were actually found to be underexploited in 2010 with negative LPY values: herring in the Bothnian part of the Baltic Sea (her-30) and Icelandic summer-spawning herring (her-vasu); see Table 1.1.

Table 1.1 Characterization factors of lost potential yield (LPY) for European fish stocks in 2010, based on a 20-, 30-, and 100-year time perspective. CFs with LPY indicating underexploitation are highlighted in dark grey, and CFs corrected for false negatives due to short time perspectives are highlighted in light grey. See supplementary material S1 for full stock names and supplementary material S4 for additional time perspectives (10 and 500 years)

Species	ICES Stock id	$LPY_{20\ years}$ (short)	$LPY_{30\ years}$ (recommended)	$LPY_{100\ years}$ (long)	B/B_{MSY}	F/F_{MSY}
Cod	cod-2224	4.33	**5.93**	10.61	***6%***	***2.3***
Cod	cod-2532	0.17	**0.11**	0.04	***14%***	***0.8***
Cod	cod-347d	16.1	**47.2**	263.3	***2%***	***3.6***
cod	cod-arct	0.13	**0.11**	0.07	***21%***	***0.7***
cod	cod-farp	0.16	**0.12**	0.08	***26%***	***1.3***
haddock	had-34	0.08	**0.07**	0.05	***45%***	***0.8***
haddock	had-arct	0.10	**0.09**	0.07	***86%***	***0.7***
haddock	had-rock	0.32	**0.30**	0.27	***40%***	***0.5***
haddock	had-scow	0.05	**0.03**	0.01	***35%***	***1.0***
herring	her-2532-gor	0.69	**1.05**	1.74	***22%***	***2.0***
herring	her-30	-0.17	**-0.15**	-0.10	***200%***	***0.7***
herring	her-3a22	0.10	**0.08**	0.05	***32%***	***1.2***
herring	her-47d3	0.40	**0.37**	0.34	***71%***	***0.5***
herring	her-noss	0.00	**0.00**	0.00	***141%***	***1.1***
herring	her-riga	0.00	**0.01**	0.03	***81%***	***1.2***
herring	her-vasu	-0.27	**-0.25**	-0.21	***100%***	***0.6***
herring	her-vian	0.03	**0.02**	0.01	***49%***	***1.1***
horse mackerel	hom-soth	-0.10	**-0.09**	-0.05	***139%***	***0.8***
mackerel	mac-nea	0.01	**0.01**	0.01	***91%***	***1.2***
megrim*	mgb-8c9a	0.13	**0.30**	0.84	***72%***	***1.9***
	mgw-8c9a	0.53	**0.48**	0.42	***22%***	***0.4***
plaice	ple-echw	1.31	**2.10**	6.60	***22%***	***2.4***
plaice	ple-nsea	0.11	**0.07**	0.02	***26%***	***1.0***
saithe	sai-3a46	0.05	**0.05**	0.05	***50%***	***1.3***
saithe	sai-faro	0.02	**0.04**	0.08	***72%***	***1.4***
sole	sol-bisc	0.30	**0.28**	0.23	***26%***	***1.5***
sole	sol-celt	0.05	**0.04**	0.03	***61%***	***0.8***
sole	sol-eche	0.18	**0.21**	0.24	***43%***	***1.6***
sole	sol-echw	0.05	**0.04**	0.01	***44%***	***0.9***
sole	sol-iris	0.53	**0.69**	0.64	***19%***	***1.7***
sole	sol-nsea	0.18	**0.22**	0.25	***40%***	***1.5***
sprat	spr-2232	0.01	**0.01**	0.01	***111%***	***1.2***
blue whiting	whb-comb	1.01	**1.01**	1.01	***101%***	***1.0***

1.3.2 Temporal variation and influence of OB and OF

The temporal variation in characterization factors can be exemplified by the historical development of OB and OF regarding three stocks: Eastern Baltic cod improving from high ongoing overfishing ($F \gg F_{MSY}$), North Sea plaice following a similar but less dramatic change, and Baltic Bothnian herring having negative lost yields during the whole period; see Fig. 1.3. The consequential variation in temporal LPY scores can be seen in Table 1.2.

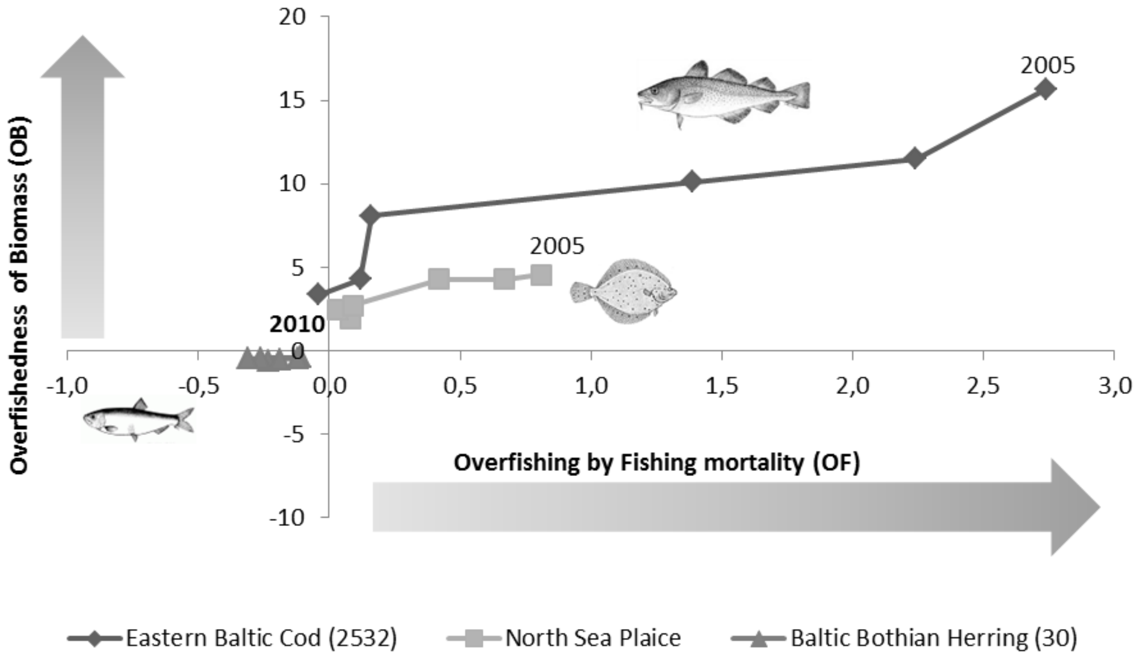


Fig. 1.3 Examples of temporal variation in characterization factors of OB and OF for Eastern Baltic cod, North Sea plaice, and Baltic Bothnian herring between 2005 and 2010. Fish illustrations used by kind permission of FAO 2012.

In Table 1.2, three sets of typical contributions to LPY by OF and OB can be seen, the data indicating a larger variance between stocks than over time. Similar trends are also found in the larger dataset, where the typical coefficient of variation was notably higher between stocks of the same species (cod, haddock, sole, and herring with more than four stocks per species) than between years for each stock.

Table 1.2 Examples of annual variation in OF, OB, and LPY characterization factors for Eastern Baltic cod, North Sea plaice, and Baltic Bothnian herring between 2005 and 2010.

	Eastern Baltic Cod (2532)			North Sea Plaice			Baltic Bothnian Herring (30)		
	OF	OB	LPY	OF	OB	LPY	OF	OB	LPY
2010	-0.04	3.35	**0.1**	0.1	1.9	**0.1**	-0.2	-0.6	**-0.1**
2009	0.12	4.26	**0.1**	0.0	2.4	**0.1**	-0.2	-0.5	**-0.1**
2008	0.16	8.09	**0.1**	0.1	2.7	**0.1**	-0.1	-0.4	**-0.1**
2007	1.39	10.11	**2.5**	0.4	4.3	**0.2**	-0.1	-0.4	**-0.1**
2006	2.24	11.50	**9.5**	0.7	4.3	**0.4**	-0.3	-0.4	**-0.2**
2005	2.74	15.67	**14.4**	0.8	4.6	**0.5**	-0.3	-0.5	**-0.2**

1.3.3 Sensitivity analysis

When all time perspectives up to 500 years (approximately infinity) were tested, three groups of stocks could be observed: (a) the constantly increasing, (b) the stabilizing, and (c) the stabilizing false positives (underexploited). In fact, all LPY trends are by definition stabilizing over time (owing to a constant proportion of the stock being harvested) but at different rates, see matrix of plots with the increased time perspectives on the x-axis against time in Fig. 4.

the time perspective from one to 500 years was tested (in analogy with IPPC global warming scenarios), three groups of stocks could be observed: a) the constantly increasing, b) the stabilizing and c) the stabilizing false positives which were sign adjusted. In fact all LPY developments are by definition stabilizing (constant proportion of the stock harvested) but at different rates during the first 500 year, see plots in Fig. 4. The iterative characterization function behind LPY can be displayed by plots with the increased times perspective on the x-axis.

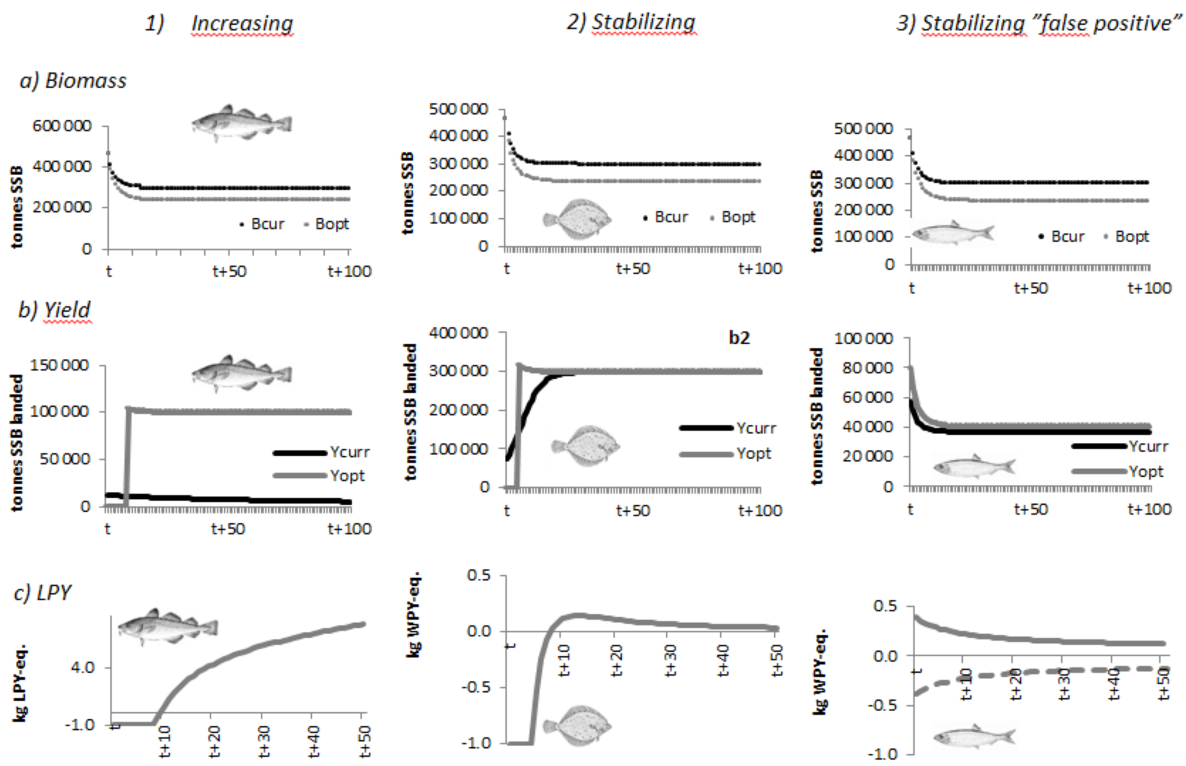


Fig. 1.4 Influence of increased time perspective T (all x-axis), plotted against the long-term development in (a) biomass, (b) yield, and (c) LPY. Three typical patterns are displayed: (1) constantly increasing, (2) stabilizing, and (3) stabilizing "false positive." Fish illustrations used by kind permission of FAO 2012.

The worst stock exploitation rate (OF) and status (OB) results in a projection (LPY) that increases over a longer time perspective, as illustrated by Western Baltic cod in 2010 (cod-2224); see (a) in Fig. 4. Most projections, however, render a quicker stabilizing pattern—such as for North Sea plaice 2010 (ple-nse) (b) or for Bothnian Baltic herring (her-30) (c)—that decreases over time but is sign-adjusted according to logic rule 1; see the dotted and filled line (sign adjusted) in Fig. 1. 4 column 3.

1.4 Discussion

The present study provides midpoint characterization factors (CF) for LPY, OF, and OB regarding 31 European commercial stocks in 2010, introducing essential aspects of overfishing into the LCA framework. However, the methodology could also be used beyond LCAs as a quick ranking index for comparison of fish stock status. Characterization through either one offers various possibilities to account for target stock impacts even if all data to calculate LPY are not available.

The variation in all sets of CFs was considerable between species, but even larger between stocks within each species. Therefore, when overfishing is to be included in LCA, the stock is the necessary spatial resolution just as it is in fisheries management. The temporal analysis showed that LPY, OF, and OB values also varied substantially over time, indicating that changes in stock status resulting from natural variation and/or management actions are well reflected in all three CFs. The large spatial and temporal variation requires novel approaches for LCA, such as dynamic CFs that need to be updated each year for the stocks under study, to ensure representativity and accuracy. This updating could be done either by each LCA practitioner when needed or in a database.

1.4.1 Verification

The LPY methodology involves both model and input uncertainties on top of the natural stock variability, i.e. the biological reality for fisheries science and management. While the joint uncertainty is hard to assess, some aspects can be verified: (1) the LPY can be compared with the more robust components OF and OB, (2) the development over time in all impact categories can be compared with qualitative descriptions of stock status, and (3) the LPY can be compared with other assessments of lost yields.

Fishing mortality (F) is the prime indicator used to regulate fisheries (EC 2008a), and the F/F_{MSY} ratio is widely used to evaluate MSY management (Gutiérrez et al. 2012). The parameter B_{MSY} is more uncertain, and there is considerable debate about how to calculate and apply it in practice (Agnew et al. 2013). However, Froese and Proelß (2010) provided uncertainty ranges for F_{MSY} and B_{MSY} for all stocks included in this study, showing a narrower uncertainty range of F_{MSY} than of B_{MSY}, based on the average of three modeling approaches. For a few ICES stocks for which uncertainty ranges are provided in the assessment, such as Western Baltic cod, F also has a narrower uncertainty range than B (ICES 2011). As a consequence, the iterative characterization model of LPY magnifies this uncertainty and therefore is less robust than OB and OF, the latter being the most robust alternative.

ICES provides F_{MSY} values for an increasing proportion of the stocks assessed, but currently no information is given on B_{MSY}. Instead, a lower biomass level B_{LIM} (also called $B_{TRIGGER}$) is used as a limit above which the biomass is allowed to fluctuate, a strategy to counter the natural variation in biomass (ICES 2012a). For LCA purposes, a five-year moving average of B related to B_{MSY} is more suitable as an optimal point associated with potential damage to the AoP of natural resources; see the overview of reference points in Fig. 1.5.

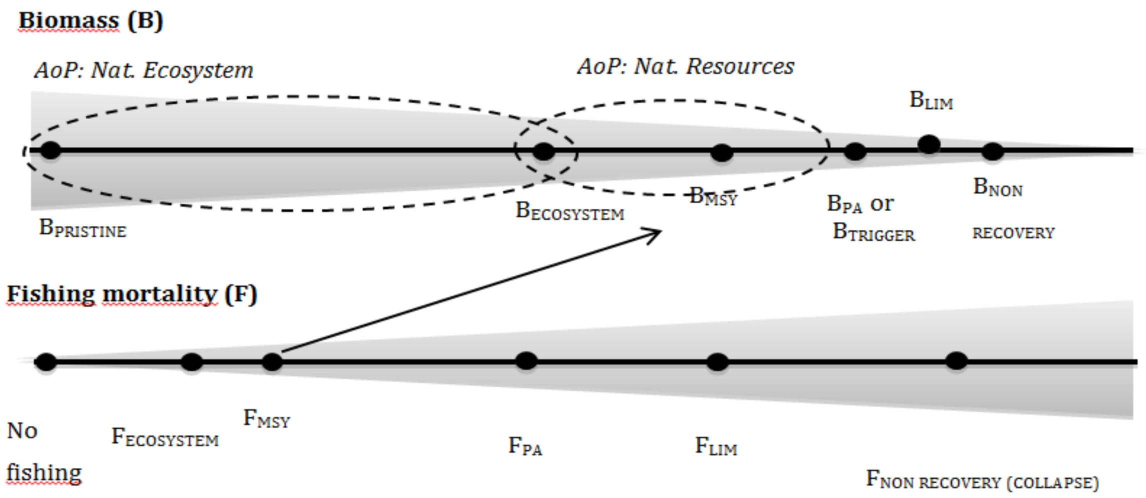


Fig. 1.5 Schematic overview of reference points for biomass (B) and fishing mortality (F); note that the relative distance between the reference points varies between stocks. Each fishing mortality has a corresponding long-term biomass (see example F_{MSY} and B_{MSY} marked with an arrow) positioned at a different length from the LCA areas of protection (AoPs), which are marked with dotted circles.

Another way of verifying, or at least illustrating, the LPY method would be to follow certified fisheries over time to see if certification of a previously overfished stock correlated with a drop in LPY and if suspension of certification correlates with an increase. The Eastern Baltic cod stock was certified by both the Marine Stewardship Council and Sweden's KRAV in 2010 (KRAV 2010; MSC 2013a), which correlates well with a large drop of LPY in 2009 due to a reduction in fishing mortality. The Portuguese sardine fishery, on the other hand, had its MSC certification suspended in 2010 (more recently, the suspension was lifted), mainly because of low recruitment (MSC 2013b), but still the effect on F and SSB could be seen in LPY values.

The concept of accounting for lost yield due to current fishing practice is not novel outside the LCA communities, and it is typically based on landings (L) related to MSY values (FAO 2008; Froese and Proelß 2010). However, such a comparison based on L and MSY has three major drawbacks. First, the values with landing do not take into account the development of stocks, i.e. a heavily overfished stock could still have L = MSY, but the stock would be at high risk of collapse. Second, a moving average of landings could partly solve this problem, but the MSY does not directly correspond to the goal of fisheries management, which is F_{MSY} and in some cases B_{MSY}. Third, the total fishing mortality is a much more accurate parameter than only landings, since it includes illegal catches and discarded juveniles in line with the LCA concept of aggregating environmental flows. Lately, the use of landing data to assess the condition of fish stocks has also been questioned (Hilborn and Branch 2013). Therefore, we think that the LPY is a more accurate characterization model than an MSY/L-based model, although the latter could represent a value for comparison or even a last choice, rather than not quantifying at all, if neither LPY nor OF or OB values could be established.

In Europe, the fishing mortality has been reduced for many heavily exploited gadoid stocks during the past decade, but the biomass of many stocks has not yet been restored ($B<B_{MSY}$) (Kraak et al. 2013). The LPY values led to lower values of lost yield than, for example, MSY/L-1, since LPY responds rapidly to reduced fishing mortality, which is a desired property for a stock status indicator used to follow up on the Marine Strategy Framework Directive (EC 2008b). This could also be an attractive feature for fisheries certification, since, for example, the MSC can approve stocks being above B_{LIM} but below B_{MSY} in cases where the stocks are moving toward B_{MSY} (Gutiérrez et al. 2012).

1.4.2 Model choice uncertainty

The LPY values will generally increase with a higher number of iterations, but the ranking between stocks remains essentially the same; thus for decision support and quick ranking of stocks, we find the model robust with respect to the time perspective T. Strictly, however, the time perspective T should be interpreted as an iteration number to compare stocks during an assessed year, rather than as a forecast, and the number chosen is of less importance as long as the same number is used throughout the LCA study. We chose 30 years as the default projection time based on a minimization of the deviation toward MSY/L-1, which also minimized the number of corrections resulting from the second logic rule, i.e. long enough time horizons for the initial fishery closure to be more beneficial than the current practice. In fact, all stocks are past the break-even point, except one example in the 30-year time perspective and two examples in the 20-year time perspective - however, with F and B very close to reference values and LPY close to zero.

The F_{MSY} consensus values suggested by Froese and Proelß (2010) deviate from the 2012 F_{MSY} target values stated by ICES, with notable consequences for seven stocks that deviate more than 50%. In this study, we chose the ICES F_{MSY} dataset as the default, since it is supported by a large international body. However, the availability of this independent dataset, (Froese and Proelß 2010), as well as the uncertainty ranges provided in this source, is beneficial for the methodology as a whole.

The characterization function was based on a Schaefer surplus production approach (Schaefer 1954), which fulfilled the criteria of simplicity and accuracy in our study. The FAO and the World Bank used both a Schaefer and a Fox model in an assessment of lost yields (FAO 2008), but using the Fox model in our context would result in more complicated calculations and a less conservative response to high fishing effort (FAO 2008).

1.4.3 Completeness of scope

We have assessed the methodology as compliant in most essential aspects according to the ILCD standard (ILCD 2010a), even though some aspects of biological variation and complexity are previously untested in the LCA framework. The provided impact categories are at midpoint level, yet it is important to check how well they cover the relevant damage to the three defined areas of protection (AoPs) (Haes et al. 2002; ILCD 2010a). The LPY is measured in mass units of lost unspecified roundweight of fish, with a clear impact pathway toward the natural resources AoP. However, the category indicator results are not related to a reference unit, e.g. like greenhouse gas emissions, which are related to CO_2 equivalents—since no static reference stock exists. For example, if LPY were measured in "North Sea Cod 2010 equivalents," this would introduce and add the uncertainty and biological variability of the reference stock to all other CFs in the dataset. Modern stock assessment is based on time series fitting (ICES 2012a), meaning that the previous years data will be updated and improved in each forthcoming annual assessment, making a reference stock impractical.

In terms of damage to the natural environment AoP, LPY indirectly indicates a clear impact pathway but with practical restrictions, since comparison between species or stocks would be like "comparing apples with pears." At this point, we did not find it meaningful to quantify the difference between species or stocks in terms of ecosystem damage, since each stock plays a different role in the ecosystem. A larger LPY value for a single stock will, however, always indicate larger damage to the ecosystem, and in the future, different weight sets of ecological relevance could be applied. Finally, it is worth noting that the AoPs are not precise targets in fisheries owing to the associated natural variation and uncertainties, thus separated means but with possibly overlapping ranges; see the schematic overview in Fig. 1.5.

Without presenting any normalization scores in this publication, we note that all of the suggested impact categories are in theory possible to normalize for European waters, since approximately 74% of the European landings are covered by the 31 included stocks (representing 7% of global marine capture fisheries). However, future inclusions of fully depleted stocks where B_{MSY}/B approaches infinity will prove a challenge.

1.4.4 Applicability

The LPY methodology could be used by any fishery expert or LCA practitioner with a basic understanding of biological systems, although a step-wise guide including other fishery-specific impact categories and seafood-specific LCA considerations would facilitate this process. The final score in lost unspecified fish biomass per landing could function as a quick index to assess and rank fisheries or fish products, but probably would be most useful at an expert level, such as producer organizations, fishery managers, LCA practitioners, and labeling organizations. Especially via labeling organizations and producers, the methodology could benefit the final consumer and in the long run contribute to more sustainable seafood use.

The use of the LPY methodology is an important step forward in biotic resource use in LCA, but raises questions of how a static framework can cope with biological variation. Our proposal implies the introduction of "dynamic impact categories", designed to be annually updated for optimal spatial and temporal resolution, and based on the same indictors frequently used in fisheries science. The practitioner has to collect F, B, F_{MSY}, and B_{MSY} values from, for example, ICES, to calculate the relevant characterization factors as input data for an R script/spreadsheet software (see supplementary material S3) or to retrieve them from an annually updated database. The results should mainly be used as a quick index complemented by more extensive qualitative descriptions for decision support, since the strength of the methodology lies in the relative comparison of potential impacts. Depending on data availability, the practitioner may choose any combination of the fishery-specific impact categories LPY, OF, or OB for seafood products containing either wild caught fish or farmed fish that rely on marine feed inputs.

A central limitation of LPY is that it only can be used on stocks for which the required input data are available, which in practice means only the most important commercial stocks. However, these are also the stocks most likely to be assessed in LCA studies. For other fish stocks that are affected by a fishery, either as target or bycatch species, other complementary methods will be required. Examples are the recently developed fishery-specific methods described in section 1.2.

If it is not possible to provide any form of quantitative information, qualitative descriptions of relevant biological considerations should at least be provided, to avoid wrong conclusions or even "greenwashing." For example, a seafood product could have low greenhouse gas emissions, but still be overfished (like Portuguese sardines after certification was suspended). However, it is important to note that the LPY (and/or OF and OB) only provides a quick index based on single-stock assessment. Multispecies interactions, age structure, and recruitment of juveniles are all issues that could be describe separately and qualitatively regarding the target stock status. At present, no guidelines exist for biotic impact assessment, which might result in double counting and use of non-ISO or non-ILCD–compliant methodology. Thus there is an urgent need for explicit guidelines to deal with biological uncertainty, which could lead to a boost of LCAs used to describe, optimize, and facilitate the path toward sustainable fisheries.

1.5 Conclusions

- Overfishing can be quantified in terms of lost potential yield (LPY), a midpoint impact category comparing the outcome of current vs. target fisheries management.

- Stock and year are the optimal resolution in seafood LCAs when LPY is to be used, which means that a characterization factor per stock need to be updated every year for best spatial and temporal resolution.

- The additional impact categories of overfishing through fishing mortality (OF) and overfishedness of biomass (OB) are presented as simpler alternatives, suitable when fewer data are available, or to facilitate interpretation of the LPY.

- Characterization factors for 31 European fish stocks in 2010 are provided as a proof of concept for LPY, OF, and OB, as well as a methodology to update "dynamic" characterization factors in upcoming years.

- Seafood LCAs using any of the three approaches presented represent a more powerful complementary tool for the fishing industry in benchmarking and product development, for seafood certification programs, and for fisheries management.

References

Agnew DJ, Gutiérrez NL, Stern-Pirlot A, Smith ADM, Zimmermann C, Sainsbury K (2013) Rebuttal to Froese and Proelß "Evaluation and legal assessment of certified seafood". Mar Policy 38:551-553

Anticamara JA, Watson R, Gelchu A, Pauly D (2011) Global fishing effort (1950-2010): trends, gaps, and implications. Fish Res 107 (1-3):131-136

Avadí A, Fréon P (2013) Life cycle assessment of fisheries: a review for fisheries scientists and managers. Fish Res 143 (0):21-38

EC (2003) COM 302 Integrated product policy - building on environmental life-cycle thinking. Comisssion of the European Communities, Brussels

EC (2006) COM 360 Implementing sustainability in EU fisheries through maximum sustainable yield. Communication from the Commission to the Council and the European Parliament, Brussels

EC (2008a) Council regulation No 99/2008 concerning the establishment of a community framework for the collection, management and use of data in the fisheries sector and support for scientific advice regarding the Common Fisheries Policy. Council Regulation (EC), Brussels

EC (2008b) Directive 2008/56/EC - establishing a framework for community action in the field of marine environmental policy (Marine strategy framework directive). Official Journal of the European Union, Brussels

Emanuelsson A (2008) Bycatch and discard in Senegalese artisanal and industrial fisheries for southern pink shrimp (Penaeus notialis). vol 774. SIK, Gothenburg, Sweden

FAO (2008) The sunken billions - the economic justification for fisheries reform. World Bank and United Nations Food and Agriculture Organization, Washington DC

FAO (2012) The state of world fisheries and aquaculture. United Nations Food and Agriculture Organization, Rome

Finnveden G, Hauschild MZ, Ekvall T, Guinée J, Heijungs R, Hellweg S, Koehler A, Pennington D, Suh S (2009) Recent developments in life cycle assessment. Environ Manage 91 (1):1-21

Ford JS, Pelletier NL, Ziegler F, Scholz AJ, Tyedmers PH, Sonesson U, Kruse SA, Silverman H (2012) Proposed local ecological impact categories and indicators for life cycle assessment of aquaculture: a salmon aquaculture case study. Journal of Industrial Ecology 16 (2):254-265

Froese R, Proelß A (2010) Rebuilding fish stocks no later than 2015: will Europe meet the deadline? Fish Fish 11 (2):194-202

Gutiérrez NL, Valencia SR, Branch TA, Agnew DJ, Baum JK, Bianchi PL, Cornejo-Donoso J, Costello C, Defeo O, Essington TE, Hilborn R, Hoggarth DD, Larsen AE, Ninnes C, Sainsbury K, Selden RL, Sistla S, Smith ADM, Stern-Pirlot A, Teck SJ, Thorson JT, Williams NE (2012) Eco-label conveys reliable information on fish stock health to seafood consumers. Plos One 7 (8):e43765

Haes HAUd, Finnveden G, Goedkoop M, Hofstetter P, Jolliet O, Klöpffer W, Krewit W, Lindeijer E, Muller-Wenk R, Olsen SI, Pennington DW, Potting J, Steen B (2002) Life cycle impact assessement: striving towards best practice. Society of Environmental Toxiology and Chemistry (SETAC), Brussels

Hilborn R, Branch TA (2013) Fisheries: Does catch reflect abundance? No, it is misleading. Nature 494 (7437):303+305-306

Hornborg S, Nilsson P, Valentinsson D, Ziegler F (2012) Integrated environmental assessment of fisheries management: Swedish nephrops trawl fisheries evaluated using a life cycle approach. Mar Policy 36 (6):1193-1201

ICES (2011) Report of the Baltic Fisheries Assessment Working Group (WGBFAS) : ANNEX: WGBFAS Cod in 22-24, Copenhagen

ICES (2012a) ICES Advice 2012 Book 1 - Introduction, overviews and special requests. The International Council for Exploration of the Seas, Copenhagen

ICES (2012b) Stock summary database, at ICES website. The International Council for Exploration of the Seas. Accessed 1 December 2012 2012

ILCD (2010a) ILCD Handbook - General guide to life cycle assessments - Detailed guidance. First edition edn. Joint Research Centre of the European Commission, Luxenburg

ILCD (2010b) ILCD Handbook - Handbook LCIA models and indicators. First editition., First edition edn. Joint Research Centre of the European Commission, Luxenburg

ISO (2006a) ISO14040 Environmental management - Life cycle assessment - Principals and framework. International Organization for Standardization, Geneva, Switzerland

ISO (2006b) ISO14044 Environmental management - Life cycle assessment - Requirements and guidelines. International Organization for Standardization, Geneva, Switzerland

Jennings S, Kaiser M, Reynolds JD (2001) Marine fisheries ecology. Wiley-Blackwell, Oxford

Kraak SBM, Bailey N, Cardinale M, Darby C, De Oliveira JAA, Eero M, Graham N, Holmes S, Jakobsen T, Kempf A, Kirkegaard E, Powell J, Scott RD, Simmonds EJ, Ulrich C, Vanhee W, Vinther M (2013) Lessons for fisheries management from the EU cod recovery plan. Mar Policy 37:200-213

KRAV (2010) Godkännade av kustnära fiske med bur, krok och garn på torsk i Östersjön. KRAV. http://krav.se/Upload/327/Beslut_torsk_ostersjon_2010-12.pdf. Accessed 2013-03-27

MEA (2005) Ecosystems and human well-being: Biodiversity synthesis. Millennium Ecosystem Assessment (MEA), Washington DC

MSC (2013a) Fiskbranschens Sweden eastern Baltic cod. Marine Stewardship Council. http://www.msc.org/track-a-fishery/fisheries-in-the-program/certified/north-east-atlantic/sweden-eastern-baltic-cod. Accessed 2013-03-27

MSC (2013b) Portugal sardine purse seine. http://www.msc.org/track-a-fishery/fisheries-in-the-program/certified/north-east-atlantic/portugal-sardine-purse-seine. Accessed 2013-03-27

Papatryphon E, Petit J, Kaushik SJ, van der Werf HMG (2004) Environmental impact assessment of salmonid feeds using life cycle assessment (LCA). Ambio 33 (6):316-323

Parker R (2012) Review of life cycle assessments research on products derived from fisheries and aquaculture: a review for Seafish as part of the collective action to adress greenhouse gas emissions in seafood. Sea Fish Industry Authority, Edinburgh,UK

Pauly D, Christensen V, Guenette S, Pitcher TJ, Sumaila UR, Walters CJ, Watson R, Zeller D (2002) Towards sustainability in world fisheries. Nature 418 (6898):689-695

Pelletier N, Ayer N, Tyedmers P, Kruse S, Flysjo A, Robillard G, Ziegler F, Scholz A, Sonesson U (2007) Impact categories for life cycle assessment research of seafood production systems: Review and prospectus. Intl J Life Cycle Assess 12 (6):414-421

Punt A, Smith ADM (2001) The Gospel of maximum sustainable yield in fisheries management: birth, crucifixion and reincarnation. In: Reynolds JD, Mace GM, Redford KH, Robinsson HG (eds) Conservation of exploited species. Cambridge University Press, Cambridge UK, pp 41-66

R (2012) R: A language and environment for statistical computing. R Development Core Team. http://www.r-project.org/

Ramos S, Vázquez-Rowe I, Artetxe I, Moreira M, Feijoo G, Zufía J (2011) Environmental assessment of the Atlantic mackerel (*Scomber scombrus*) season in the Basque country. Increasing the timeline delimitation in fishery LCA studies. Int J Life Cycle Assess 16 (7):599-610

Rockström J, Steffen W, Noone K, Persson A, Chapin FS, Lambin EF, Lenton TM, Scheffer M, Folke C, Schellnhuber HJ, Nykvist B, de Wit CA, Hughes T, van der Leeuw S, Rodhe H, Sorlin S, Snyder PK, Costanza R, Svedin U, Falkenmark M, Karlberg L, Corell RW, Fabry VJ, Hansen J, Walker B, Liverman D, Richardson K, Crutzen P, Foley JA (2009) A safe operating space for humanity. Nature 461 (7263):472-475

Schaefer M (1954) Some aspects of the dynamics of populations important to the management of the commercial Marine fisheries. Bulletin of the Inter-American Tropical Tuna Commission 1(2): 27-56

Swartz W, Sala E, Tracey S, Watson R, Pauly D (2010) The spatial expansion and ecological footprint of fisheries (1950 to present). Plos One 5 (12):e15143

Thrane M, Ziegler F, Sonesson U (2009) Eco-labelling of wild-caught seafood products. J Clean Prod 17 (3):416-423

Vázquez-Rowe I, Hospido A, Moreira MT, Feijoo G (2012a) Best practices in life cycle assessment implementation in fisheries. Improving and broadening environmental assessment for seafood production systems. Trends Food Sci Tech 28 (2):116-131

Vázquez-Rowe I, Moreira M, Feijoo G (2012b) Inclusion of discard assessment indicators in fisheries life cycle assessment studies. Expanding the use of fishery-specific impact categories. Int J Life Cycle Assess 17 (5):535-549

Worm B, Hilborn R, Baum JK, Branch TA, Collie JS, Costello C, Fogarty MJ, Fulton EA, Hutchings JA, Jennings S, Jensen OP, Lotze HK, Mace PM, McClanahan TR, Minto C, Palumbi SR, Parma AM, Ricard D, Rosenberg AA, Watson R, Zeller D (2009) Rebuilding global fisheries. Science 325 (5940):578-585

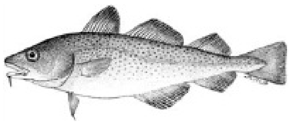
Ziegler F, Emanuelsson A, Eichelsheim JL, Flysjö A, Ndiaye V, Thrane M (2011) Extended life cycle assessment of southern pink shrimp products originating in Senegalese artisanal and industrial fisheries for export to Europe. J Ind Ecol 15 (4):527-538

Ziegler F, Nilsson P, Mattsson B, Walther Y (2003) Life cycle assessment of frozen cod fillets including fishery-specific environmental impacts. Int J Life Cycle Assess 8 (1):39-47

S1 Species and stock list

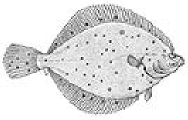
Common name, *Scientific name*, *Family, Order*

Stock id	ICES name & spatial definition



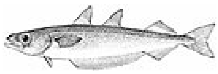
Atlantic cod, *Gadus morhua*, *Gadidae, Gadiformes*

Stock id	ICES name & spatial definition
cod-347d	Cod in Sub-area IV, Division VIId & Division IIIa (Skagerrak)
cod-2224	Cod in Sub-divisions 22 to 24
cod-farp	Faroe Plateau cod (Sub-division Vb1)
cod-2532	Cod in Sub-divisions 25 to 32
cod-arct	North-East Arctic cod (Sub-areas I and II)



Plaice, *Pleuronectes platessa*, *Pleuronectidae, Pleuronectiformes*

Stock id	ICES name & spatial definition
ple-echw	Plaice in Division VIIe (Western Channel)
ple-nsea	Plaice Sub-area IV (North Sea)



Blue whiting, *Micromesistius poutassou*, *Gadidae, Gadiformes*

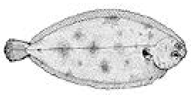
Stock id	ICES name & spatial definition
whb-comb	Blue whiting combined stock (Sub-areas I-IX, XII & XIV)

Megrim, *Lepidorhombus whiffiagonis, Scophthalmidae, Pleuronectiformes*

Stock id	ICES name & spatial definition
mgw-8c9a	Megrim (L. whiffiagonis) in Divisions VIIIc and Ixa

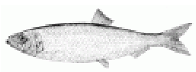
Four-spot megrim, *Lepidorhombus boscii*, *Scophthalmidae, Pleuronectiformes*

Stock id	ICES name & spatial definition
mgb-8c9a	Megrim (L. boscii) in Divisions VIIIc and Ixa



Common sole, *Solea solea*, *Soleidae, Pleuronectiformes*

Stock id	ICES name & spatial definition
sol-iris	Sole in Division VIIa (Irish Sea)
sol-bisc	Sole in Divisions VIIIa,b (Bay of Biscay)
sol-nsea	Sole in Sub-area IV (North Sea)
sol-eche	Sole in Division VIId (Eastern Channel)
sol-celt	Sole in Divisions VIIf and g (Celtic Sea)
sol-echw	Sole in Division VIIe (Western Channel)

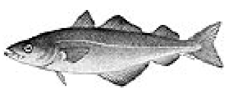


Atlantic herring, *Clupea harengus*, *Clupeidae, Clupeiformes*

Stock id	ICES name & spatial definition
her-2532-gor	Herring in Sub-divisions 25 to 29 and 32 minus Gulf of Riga
her-47d3	Herring in Sub-area IV, Divisions VIId & IIIa (autumn-spawners)
her-3a22	Herring in Sub-divisions 22-24 and Division IIIa (spring-spawners)
her-vian	Herring in Division VIa (North)
her-riga	Herring in the Gulf of Riga
her-noss	Norwegian spring-spawning herring
her-30	Herring in Sub-division 30, Bothnian Sea
her-vasu	Icelandic summer-spawning herring (Division Va)

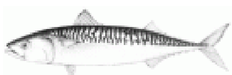
Haddock, *Melanogrammus aeglefinus*, *Gadidae, Gadiformes*

Stock id	ICES name & spatial definition
had-rock	Haddock in Division VIb (Rockall)
had-arct	North-East Arctic haddock (Sub-areas I and II)
had-34	Haddock in Sub-area IV (North Sea) and Division IIIa
had-scow	Haddock in Division VIa (West of Scotland)



Saithe, *Pollachius virens*, *Gadidae, Gadiformes*

Stock id	ICES name & spatial definition
sai-3a46	Saithe in Sub-area IV, Division IIIa (Skagerrak) & Sub-area VI
sai-faro	Faroe saithe (Division Vb)



Atlantic mackerel, *Scomber scombrus*, *Scombridae, Scombriformes*

Stock id	ICES name & spatial definition
mac-nea	Mackerel (combined Southern, Western & N.Sea spawn.comp.)

European sprat, *Sprattus sprattus*, *Clupeidae, Clupeiformes*

spr-2232 Sprat in Sub-divisions 22 to 32

Horse mackerel, *Trachurus trachurus,* *Carangidae, Perciformes*

hom-soth Southern horse mackerel (Divisions Ixa)

S2 Input data WMY, OB and OF

Source:	**ICES Stock summary database regarding 2010 (Retrieved 2012)**						**Froese & Proelß 2010** – "Rebuilding Fish Stocks No Later Than 2015: Will Europe Meet the Deadline?" Supporting information		
Type:	Year specific 2010						Valid to 2015		
id \input parameter	SSB	SSB5	L	L5	MeanF (Instantaneous)	Fmsy ICES	Fmsy_cons	Bmsy_cons (SSBmsy)	MSY_cons
cod-2224 2010	25 642	27 681	14 120	19 248	0.58	0.25	0.27	328 920	83 634
cod-2532 2010	232 139	140 463	50 277	51 465	0.25	0.30	0.19	707 155	255 735
cod-347d 2010	52 733	42 433	69 286	61 209	0.68	0.19	0.24	1 975 535	373 543
cod-arct 2010	1 134 247	844 934	609 983	524 422	0.29	0.40	0.20	3 302 330	837 049
cod-farp 2010	31 404	23 521	12 737	9 740	0.41	0.32	0.20	74 165	22 267
had-34 2010	182 559	225 958	39 640	48 477	0.23	0.30	0.25	342 308	259 119
had-arct 2010	361 519	244 334	249 334	183 936	0.25	0.35	0.31	204 448	127 387
had-rock 2010	17 109	17 919	3 710	4 710	0.15	0.30	0.21	35 001	11 037
had-scow 2010	15 868	23 058	4 824	8 441	0.29	0.30	0.31	47 195	22 745
her-2532-gor 2010	535 120	523 927	136 706	124 678	0.32	0.16	0.18	1 989 603	372 837
her-30 2010	617 784	462 160	71 726	69 965	0.13	0.19	0.13	188 585	51 579
her-3a22 2010	95 152	128 578	42 214	67 501	0.30	0.25	0.26	329 079	116 470
her-47d3 2010	1 301 092	1 342 014	187 611	307 001	0.12	0.25	0.23	1 195 945	529 790
her-noss 2010	9 176 000	8 610 400	1 457 014	1 385 199	0.16	0.15	0.11	3 613 027	1 515 458
her-riga 2010	76 800	78 354	30 174	31 768	0.43	0.35	0.24	74 361	30 927
her-vasu 2010	386 000	600 000	44 000	106 000	0.13	0.22	0.18	447 073	126 943
her-vian 2010	61 649	82 147	19 877	22 280	0.27	0.25	0.16	129 158	59 344
hom-soth 2010	241 400	264 904	27 217	24 524	0.09	0.11	0.23	154 612	32 721
mac-nea 2010	2 992 033	2 763 579	869 451	653 845	0.26	0.22	0.23	2 512 443	676 655
mgb-8c9a 2010	4 797	4 741	1 297	1 121	0.34	0.18	0.17	5 502	1 302
mgw-8c9a 2010	717	716	83	133	0.08	0.17	0.16	2 332	644
ple-echw 2010	2 629	1 992	1 227	1 217	0.45	0.19	0.24	6 723	1 883
ple-nsea 2010	460 700	345 040	106 500	102 080	0.24	0.25	0.21	1 135 716	162 123
sai-3a46 2010	213 500	256 820	102 500	112 880	0.38	0.30	0.21	415 148	156 804
sai-faro 2010	110 606	112 055	43 959	57 354	0.38	0.28	0.19	129 092	41 624
sol-bisc 2010	11 765	12 083	3 966	4 214	0.39	0.26	0.20	42 523	7 107
sol-celt 2010	3 869	3 461	862	872	0.26	0.31	0.19	5 180	989
sol-eche 2010	10 224	11 354	4 391	4 774	0.45	0.29	0.26	22 097	4 496
sol-echw 2010	2 760	2 600	688	867	0.25	0.27	0.15	5 345	1 051
sol-iris 2010	1 218	1 402	275	399	0.27	0.16	0.21	5 799	1 494
sol-nsea 2010	35 200	29 980	12 600	13 580	0.34	0.22	0.26	57 430	18 742
spr-2232 2010	891 000	895 800	342 000	374 000	0.41	0.35	0.31	607 746	388 386
whb-comb 2010	3 043 490	4 464 718	539 539	1 198 975	0.18	0.18	0.15	2 734 177	1 344 398

S3 R-code

Core code in R:

```
#REVISTED WASTED MAXIMUM YIELD MODEL (18 DECEMBER 2012)

#FmsyICES + Transformed Instantaneous Fishing mortality


#LOAD INPUT DATA

rm(list=ls())

setwd("C:/Users/AndreasEm/R/R-WPC/Input data")

#Matched input data F,SSB per year + Fmsy,Bmsy all years

data    <- read.delim("data_matched2012_ICESfmsy_selected3.txt")

attach(data)


#MODEL PARAMETERS

year = c(2010:2005)           #Target evaluation year for WMY

timeframe =c(10,30)#,30,100,500)      #Fix 30 for plot

B_years = 3                #Moving average SSB


#START STOCK LOOPS

projections = max(timeframe)

id <- levels(FishStock)


for (y in 1:length(year)) {

  for (s in 1:length(id))   {


    #LOAD YEAR & STOCK SPECIC INPUT DATA

        D       <- subset(data, FishStock == id[s] & Year == year[y])


        #CONSTRUCT 5YEAR MOVING AVERAGE SSB

        B = B0 <-   mean(      SSB[    Year        <=     year[y]

                              & Year       >    year[y] - B_years

                              & FishStock      ==   id[s]]

                     )

        Bmsy        <-  D$BmsyCons
```

```
#TRANSFORM INSTANTANEOUS FISHING MORTALITY TO ANNUAL FISHING MORTALITY
F      <-  1-exp(-D$MeanF)
   Fmsy       <-  1-exp(-D$Fmsy_ICES)  #ICES2005
#Fmsy   <-  1-exp(-D$FmsyCons)  #FROESE, secondary dataset


# STEP 1:GENREATE CURRENT BIOMASS [Schaeffer annual discrete surplus production]

   for (i in 1:projections)
   B[i+1] <-   B[i] + 2*Fmsy*B[i] * (1 - B[i]/2/Bmsy) - F*B[i]

Bcurr <- B


# STEP 2:  GENREATE OPTIMAL BIOMASS  [Schaeffer annual discrete surplus production]

#2a: Time t: (B from SSB5, not projected)

   B      <-  B0
   Fopt    <-     ifelse(B >= Bmsy, Fmsy, 0)


#2b: Time t+1 to t+n: (Bt projected from Bt-1; Fopt defined from Bt)

   for (i in 1:(projections)) {

      B[i+1] <- B[i]+2*Fmsy*B[i]*(1-B[i]/2/Bmsy)-Fopt[i]*B[i]

      Fopt[i+1]  <-  ifelse( B[i+1] >= Bmsy,   Fmsy,                            0)

   }
Bopt <- B
```

```
    # STEP 3:  MAIN MECHANISM (EQ1)

    Ycurr   <- Bcurr * F

    Yopt    <- Bopt * Fopt

    WMY         <- 0
    for (k in 1:projections+1)
        WMY[k]      <-  sum(Yopt[1:k])/sum(Ycurr[1:k])-1

# STEP 4: CONSTRUCT RESULT MATRIX

    F_Fmsy       <- round( D$MeanF/Fmsy, digits = 2)
    B_Bmsy       <- round( D$SSB/Bmsy, digits = 2)
    L_MSY        <- round( D$Landings/D$MSYcons, digits=2)
    loggtime     <- as.character( Sys.time())
    B_mod        <- paste("SSB ", B_years,"y", collapse="")
    SSB5        <- B0

    projection.data   <- t(c(WMY, Bopt, Bcurr, Fopt, F))       #raw     data   output,   all
timeseries values

        result <- data.frame( id[s],     #stock id
                    substring(id[s], 1,3), #species
                    year[y],
                    t(WMY[timeframe+1]),
                    SSB5,
                                F_Fmsy-1,    1/B_Bmsy-1,   1/L_MSY-1,
                                F_Fmsy,    B_Bmsy,      L_MSY,
                                loggtime,
                    projection.data
                    )

    if(s==1)                result.allyears <- result
```

```
      else                    result.allyears <- rbind(result.allyears, result)

  }


 if(y==1)              result.allyears.allstocks <- result.allyears

  else                        result.allyears.allstocks <- rbind(result.allyears.allstocks, result.allyears)

}


# SAVE AS EXCEL SPREADSHEET


  colnames(result.allyears.allstocks) <- c(    "FishStock",
                              "Species",
                           "Year",
                           paste(timeframe,"years"),
                           paste(c(B_mod),"y.avrg.SSB"),
                           "OF (F/Fmsy-1)", "OB (Bmsy/B-1)","REF(MSY/L-1)",
                           "F/Fmsy","B/Bmsy","L/MSY",
                                    "Logg date/time",
                           paste("WMY          t+",0:projections+1),           paste("B.opt t+",0:projections+1),
                              paste("B.cur t+",0:projections+1),
                              paste("Fopt t+",0:projections+1), "1-exp(-F)")


filename = paste("C:/Users/AndreasEm/R/R-WPC/Results/Results", gsub(":", "_",loggtime),".csv", sep="",dec = ".")
 write.csv2(result.allyears.allstocks, file = filename, row.names=FALSE)
```

S4 Main results 2010 including extra data

Stock id	MSY/L-1	10y	20y	30y	100y	500y	B/B_{MSY}	F/F_{MSY}
cod-2224 2010	4.92	0.63	4.33	5.93	10.61	23.92	6%	2.3
cod-2532 2010	4.09	0.28	0.17	0.11	0.04	0.03	14%	0.8
cod-347d 2010	4.39	16.1	16.1	47.2	263.3	1499.3	2%	3.6
cod-farp 2010	0.75	0.17	0.16	0.12	0.08	0.06	26%	1.3
cod-arct 2010	0.37	0.23	0.13	0.11	0.07	0.07	21%	0.7
cod-farp 2010	0.75	0.17	0.16	0.12	0.08	0.06	26%	1.3
had-34 2010	5.54	0.12	0.08	0.07	0.05	0.04	45%	0.8
had-arct 2010	-0.49	0.12	0.10	0.09	0.07	0.07	86%	0.7
had-rock 2010	1.97	0.38	0.32	0.30	0.27	0.26	40%	0.5
had-scow 2010	3.71	0.07	0.05	0.03	0.01	0.00	35%	1.0
her-2532-gor 2010	1.73	0.69	0.69	1.05	1.74	2.15	22%	2.0
her-30 2010	-0.28	-0.22	-0.17	-0.15	-0.10	-0.09	200%	0.7
her-3a22 2010	1.76	0.03	0.10	0.08	0.05	0.04	32%	1.2
her-47d3 2010	1.82	0.46	0.40	0.37	0.34	0.33	71%	0.5
her-noss 2010	0.04	0.00	0.00	0.00	0.00	0.00	141%	1.1
her-riga 2010	0.02	0.00	0.00	0.01	0.03	0.03	81%	1.2
her-vasu 2010	1.89	-0.34	-0.27	-0.25	-0.21	-0.20	100%	0.6
her-vian 2010	1.99	0.02	0.03	0.02	0.01	0.00	49%	1.1
hom-soth 2010	0.20	-0.14	-0.10	-0.09	-0.05	-0.03	139%	0.8
mac-nea 2010	-0.22	0.01	0.01	0.01	0.01	0.02	91%	1.2
mgb-8c9a 2010	0.00	0.13	0.13	0.30	0.84	1.26	72%	1.9
mgw-8c9a 2010	6.76	0.17	0.53	0.48	0.42	0.40	22%	0.4
ple-echw 2010	0.53	0.07	1.31	2.10	6.60	34.66	22%	2.4
ple-nsea 2010	0.52	0.12	0.11	0.07	0.02	0.00	26%	1.0
sai-3a46 2010	0.53	0.00	0.05	0.05	0.05	0.05	50%	1.3
sai-faro 2010	-0.05	0.02	0.02	0.04	0.08	0.09	72%	1.4
sol-bisc 2010	0.79	0.15	0.30	0.28	0.23	0.21	26%	1.5
sol-celt 2010	0.15	0.07	0.05	0.04	0.03	0.02	61%	0.8
sol-eche 2010	0.02	0.07	0.18	0.21	0.24	0.25	43%	1.6
sol-echw 2010	0.53	0.07	0.05	0.04	0.01	0.01	44%	0.9
sol-iris 2010	4.43	0.53	0.53	0.69	0.64	0.57	19%	1.7
sol-nsea 2010	0.49	0.00	0.18	0.22	0.25	0.26	40%	1.5
spr-2232 2010	0.14	0.01	0.01	0.01	0.01	0.02	111%	1.2
whb-comb 2010	1.49	0.00	1.01	1.01	1.01	1.01	101%	1.0

The typical (median) values of wasted maximum yield for European stocks in the 2010 (T=30) dataset is around 0.1kg per kg, although for 5 out of 34 where above 1 with one extreme value for North sea cod (cod-347d) approaching almost 50 kg wasted per every 1 kg landed.

2. Marine resource use: Primary production required as a proxy for by-catch impacts

Trophic indicators in fisheries: a call for re-evaluation

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Note: This chapter has been published in the journal Biology Letters (2013, 9:20121050) as a publication with the same title and author list as this chapter

Abstract

Mean Trophic Level (MTL) of landings and Primary Production Required (PPR) by fisheries are increasingly used in the assessment of sustainability in fisheries. However, in their present form, MTL and PPR are prone to misinterpretation. We show that it is important to account for actual catch data, define an appropriate historical and spatial domain, and carefully consider the effects of fisheries management, based on results from a case study of Swedish fisheries during the last century.

Keywords: discard; fisheries; management; MTL; PPR; seafood

2.1 Introduction

Understanding trophic interactions and how fisheries affect them is essential for management of fisheries [1,2,3]. Indicators related to a species' position in the food web are accordingly used in several ways in attempts to quantify sustainability of fisheries.
One of these indicators is the mean trophic level of landings (MTL), intended to represent, and account for, the phenomenon of "fishing down the food web" [4]. If fishing pressure depletes top predators more severely than low-trophic-level species, a decline in the MTL of fisheries landings can be expected. This concept has been adopted by the Convention of Biological Diversity (CBD) (e.g., to be "ready for global use" [5]).

Factors determining the trophic composition of landings, however, are complex and difficult to disentangle. For example, decreased MTL can be caused by an increase in the contribution of low-trophic-level species to landings rather by depletion of top predators [6]. Moreover, top predators are not the main driver behind fishing revenues as often assumed previously [7], and there is evidence that low-trophic-level species collapse more frequently than do top predators [8]. In fact, the negative trend in global MTL observed at the end of the 1990s is no longer supported by the last two decades of data [9]. Given the unresolved complexities in the factors and mechanisms that determine MTL of fishery landings at different scales [10], further evaluation of MTL, as an indicator of sustainable fishery exploitation, is needed.

The Primary Production Required (PPR) by fisheries is an indicator that is closely related to MTL. PPR represents an estimate of the carbon utilised by photosynthesis to produce one kilo of biomass in the population of a species at a certain trophic level [11]. The present global rate of biomass removal by fisheries in terms of PPR is thought to exceed the limits required for long-term sustainable marine ecosystem production [12]. Lower PPR values would accordingly be associated with lower ecosystem costs. PPR has been suggested as a common currency or ecological footprint that enables comparison of the ecological cost of fishing over time or between ecosystems [13,14]. PPR is also increasingly applied in environmental systems analysis of seafood production; in this case it serves as a measure of biological resource use from aquaculture or fisheries [e.g. 15,16].

We have studied the patterns of MTL and PPR using data that represents over a century of Swedish fishing in the Kattegat and Skagerrak and related these results to other sources of information on fisheries development and ecosystem dynamics for this area. Our intent is to evaluate their strengths and weaknesses in detecting trends related to fishing pressure on ecosystem functioning and on their potential use as indicators of sustainability for fisheries and their management.

2.2 Methods

In this study, we used Swedish landing data from the International Council for Exploration of the Sea (ICES; area IIIa, years 1903-2010). Because data from IIIa were pooled with IIIc or IVb+c in 1932-1933 and 1962-1974, these years were excluded from consideration. Estimates of trophic levels (TL) were obtained from FishBase and SeaAroundUs, and assumed to be invariant over time (see [17] for constraints on these assumptions). Landings with insufficient species-specific information were excluded (on average 2 % of the total biomass). For details see Supporting Information 1.
Primary Production Required was estimated as in [11], by assuming a conservative 9:1 conversion ratio of wet weight to carbon:

$$\text{PPR}=\sum_i \left(\frac{Y_i}{9}\right) \times \left(\frac{1}{TE}\right)^{(TL_i-1)}$$

where Y_i is the yield for species *i* (measured as landings) with trophic level TL_i, and transfer efficiency TE, (assumed to be 14 % in this study, as it is higher than the standard 10% TE in northerly regions [12]). TE was assumed to be constant during the time period of this study.

The Mean Trophic Level (MTL) was estimated as

$$\mathrm{MTL} = \frac{\sum_i (Y_i \cdot TL_i)}{\sum Y_i}$$

for each year, where TL is the trophic level and the Y yield from species *i*.
Fishery-independent MTL was calculated using catch-per-unit-effort data from the International Bottom Trawl Survey (IBTS, quarter 1, 1979-2010; Supporting Information 2). Our study also included analysis of the PPR and MTL of actual catch data (landings and discard) from pre-separated fishing segments for one year (2009), including all fish and commercial invertebrates (for details see Supporting information 3). Estimates of primary production for 1985-2010 (PP, mean mC*m^2*year) were provided by the Swedish Meteorological and Hydrological Institute (SMHI).

2.3 Results

A progressive increase in the total quantity of landings is seen until the end of the 1990s, followed by a sharp decline (figure 2.1*a*). Initially, gadoids contributed more to the landings volume, but at the end of the time covered by these data, shrimp and small pelagic fishes were more dominant (Supporting Information 4). A breakpoint regression analysis revealed an increasing trend in MTL prior to the 1930s, followed by a decreasing trend (figure 2.1*b*). Trends in MTL and volume of landings exhibit independent patterns during the entire period, with both variables declining in recent years (Supporting Information 5).

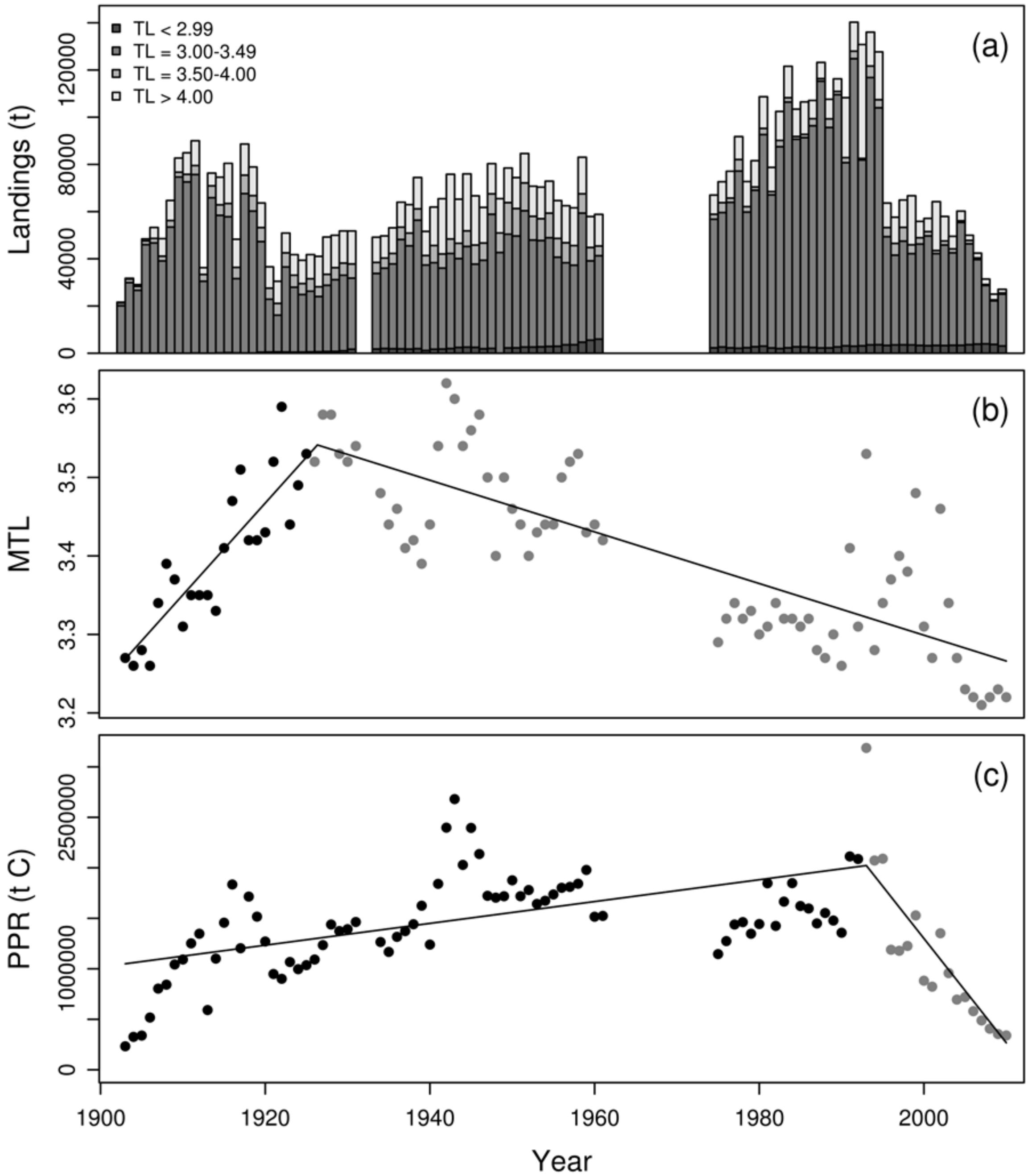

Fig. 2.1. (a) Landings by Swedish fisheries in mass per TL (ICES area IIIa), data for 1962–1974 are lacking. The species dominating TL, less than 2.99 were invertebrates; TL, 3–3.49 was dominated by sprat and herring; for TL, 3.5–4 it was mackerel; and for TL more than 4 gadoids dominated. The large drop in landings in 1996 is related to changes in quota access at the time Sweden joined the European Union. (b) Trends observed in MTL, and (c) trends in PPR are for the same set of data as shown in (a).

PPR followed a pattern similar to that exhibited by MTL, but the breakpoint was more sensitive to the number of iterations in the analysis, finally stabilizing in the 1990s (figure 2.1*c)*. Combining the two, it can be seen that, over the past two decades, landings exhibit both low MTL and PPR relative to the total available ecosystem production (%PPR) (figure 2.2*a*), with occasional peaks due to increased landings of blue whiting (*Micromesistius poutassou*).

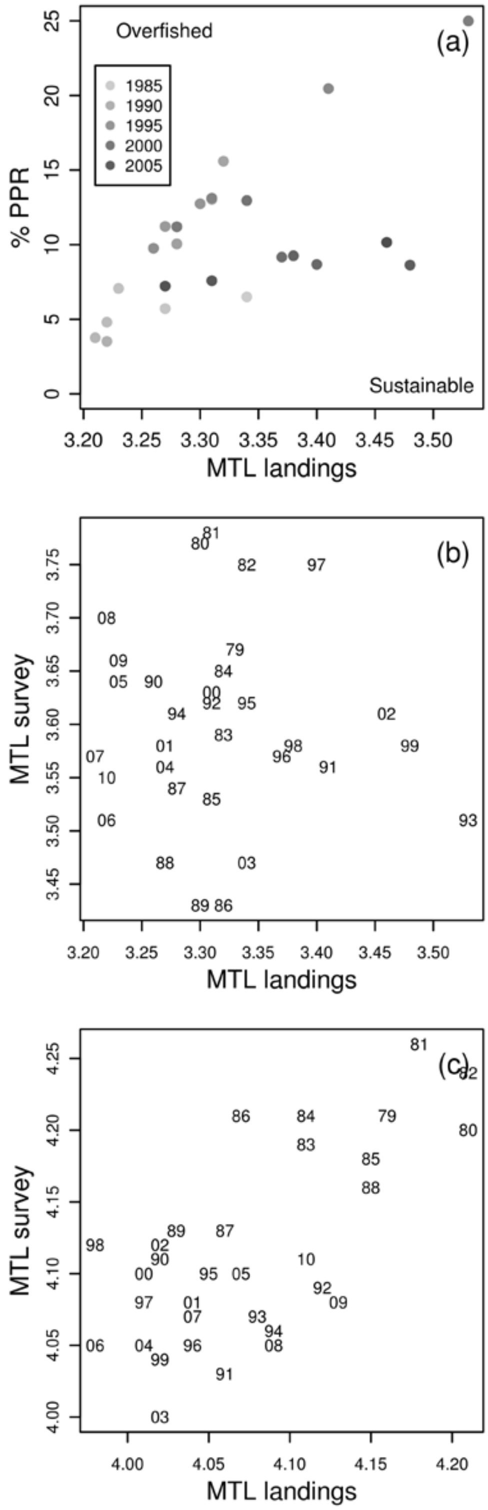

Fig. 2.2. In (a) landings are expressed as % PPR out of total ecosystem production plotted against MTL, format adopted from [2]. In (b), MTL based on survey data (1979–2010) are compared with landings (fish and commercial crustaceans) and in (c) MTL for species with a TL more than 3.25 are compared with landings using the same data as shown in (b).

We found no correlation between the MTL of landings and that of survey data for fish and commercial crustacean species (r^2=0.001, p<0.843; figure 2.2*b*). However, with only species above trophic level 3.25 in the sample, there was a weak correlation (r^2=0.448, p-value<0.001; figure 2.2*c*). This is explained on the grounds that herring, the major contributor to catches is removed when calculating MTL for species above 3.25.

On a more detailed scale than represented by figure 1 and 2, landings represented a highly variable part of the actual catch PPR (i.e., including discards) depending on fishing segment, where the PPR from landings ranged between 22 and 83% of the total catch PPR of the fishery (Supporting Information 3). Likewise, MTL also differed between the total catch and the landed portion, ranging between 2.50 and 4.18 for the total catch) (Supporting Information 3).

Overall, measures of the CPUE showed a positive trend for all species in the survey data (including non-commercial species) (r^2=0.400, p<0.001) (Supporting Information 2).

2.4 Discussion

The major fishing pattern behind the trend in MTL within the areas of the Kattegat and Skagerrak is a reduction in the contribution of stocks of large predator fish to landings, consistent with earlier reports [18,19]. Taken together, the observed peak of small pelagics in the 1990s and the decreased contribution from top predators in recent years favour a "fishing through" scenario [6] in which lower trophic levels are increasingly exploited. However, this trend itself does not necessarily indicate that fishing practices at present are unsustainable. In part, this is because recent decreases in landing MTL are highly influenced by management efforts aiming at protecting and rebuilding gadoid stocks [20,21]. Furthermore, there was no correlation between landings and survey data MTL (unless the low-trophic-level species are ignored). The trend in landing MTL appears to be, at most, a weak measure of the ecosystem state and pressures on biodiversity in the area. From this, we conclude that inferences concerning global fishing mortality and abundance trends in top predators using relative patterns in aggregate MTL are difficult to interpret without the consideration of actual total catch data (including discards or from survey data).
A clear conclusion from our work is that PPR estimates based on data restricted to landings are inadequate and possibly misleading. Including PPR of discards is in fact essential to enable evaluations of the ecological costs from different fishing practices [20]. Nevertheless, before making detailed comparisons between regions, further refinement of appropriate values for transfer efficiencies might be needed [22].

It is additionally clear that care must be taken in interpreting data involving estimated PPR. For example, declining PPR from fisheries can be erroneously interpreted as a fishery with decreasing costs to the ecosystem. However, low PPR values in the Kattegat and Skagerrak may very well involve commercial landings at an all-time low because of commercial stocks that are severely depleted, which cannot be interpreted as advantageous. Regarding other metrics, survey data in our study do not indicate a lower overall production in the area. Such indications of good health can also be misleading when considered alone, as other measures could easily indicate serious ecosystem-level problems caused by the synergistic effects of overfishing and eutrophication [23-26]. Therefore, before any fisheries indices related to trophic interactions can be interpreted properly, it is essential to have a much better understanding of which factors contribute to local fish production [27], and how fishing affects the dynamics of energy within ecosystems [3].

Taken together, complex systems involve complex sets of interactions, and it is most likely that a large suite of indicators is needed to assess the sustainability of fisheries; single trends in PPR or MTL could send conflicting messages. Decreases in MTL (interpreted as a negative signal by the Convention of Biological Diversity) can contribute to decreasing PPR (considered as a necessary transition towards more sustainable fisheries [12,14] and as a positive factor in the environmental systems analysis of seafood [15]).

To conclude, there are serious constraints on what conclusions can be drawn from information on trends and levels of estimated MTL and PPR. As one significant step toward improvement, we strongly recommend using actual catch data (including discards/surveys), ecologically-sound spatial resolution to account for obvious differences among different regions (e.g. fisheries management systems and ecosystem functioning) and take into account temporal factors, especially the influence attributable to the duration of fisheries exploitation.

References

3 Casini, M., Hjelm, J., Molinero, J. C., Lövgren, J., Cardinale, M., Bartolino, V., Belgrano, A. & Kornilovs, G. 2009 Trophic cascades promote threshold-like shifts in pelagic marine ecosystems. *Proc. Natl. Acad. Sci. USA* **106**, 197-202.

4 Conti, L. & Scardi, M. 2010 Fisheries yield and primary productivity in large marine ecosystems. *Mar. Ecol. Progr. Ser.* **410**, 233-244.

5 Frank, K. T., Petrie, B., Fisher, J. A. D. & Leggett, W. C. 2011 Transient dynamics of an altered large marine ecosystem. *Nature* **477**, 86-89.

6 Pauly, D., Christensen, V., Dalsgaard, J., Froese, R. & Torres, F. Jr. 1998 Fishing down marine food webs. *Science* **279**, 860-863.

7 2010 Biodiversity Indicators Partnership, 2010. Biodiversity indicators and the 2010 Target: Experiences and lessons learnt from the 2010 Biodiversity Indicators Partnership. Secretariat of the Convention on Biological Diversity, Montreal, Canada. Technical Series No. 53, 196 pages.

8 Essington T.E., Beaudreau A.H. & Wiedenmann J. (2006). Fishing through marine food webs. Proc. Natl Acad. Sci. U.S.A., 103, 3171-3175.

9 Sethi S.A., Branch T.A. & Watson R. (2010). Fishery development patterns are driven by profit but not trophic level. Proc. Natl Acad. Sci. U.S.A., 107, 12163-12167.

10 Pinsky M.L., Jensen O.P., Ricard D. & Palumbi S.R. (2011). Unexpected patterns of fisheries collapse in the world's oceans. Proc. Natl Acad. Sci. U.S.A., 108, 8317-8322.

11 Branch, T. A., Watson, R., Fulton, E. A., Jennings, S., McGilliard, C. R., Pablico, G. T., Ricard, D. & Tracey, S. R. 2010 The trophic fingerprint of marine fisheries. *Nature* **468***,* 431-435.

12 Stergiou, K. I. & Tsikliras, A. C. 2011 Fishing down, fishing through and fishing up: fundamental process versus technical details. *Mar. Ecol. Progr. Ser.* **441**, 295-301.

13 Pauly, D. & Christensen, V. 1995 Primary production required to sustain global fisheries. *Nature* **374***,* 255-257.

14 Coll, M., Libralato, S., Tudela, S., Palomera, I. & Pranovi, F. 2008 Ecosystem overfishing in the ocean. *Plos One* **3**, e3881.

15 ICES 2005. Ecosystems effects of fishing: impacts, metrics, and management strategies. ICES Cooperative Research Report, No. 272, 177 pp.

16 Swartz, W., Sala, E., Tracey, S., Watson, R. & Pauly, D. 2010 The spatial expansion and ecological footprint of fisheries (1950 to Present). *Plos One* **5***,* e15143.

17 Pelletier, N., Tyedmers, P., Sonesson, U., Scholz, A., Ziegler, F., Flysjo, A., Kruse, S., Cancino, B. & Silverman, H. 2009 Not all salmon are created equal: Life cycle assessment (LCA) of global salmon farming systems. *Environ. Sci. Technol.* **43***,* 8730-8736.

18 Vázquez-Rowe, I., Moreira, M. & Feijoo, G. 2012 Inclusion of discard assessment indicators in fisheries life cycle assessment studies. Expanding the use of fishery-specific impact categories. *Int. J. Life Cycle Assess.* **1-15**. doi 10.1007/s11367-012-0395-x.

19 Jennings, S., Greenstreet, S. P. R., Hill, L., Piet, G. J., Pinnegar, J. K. & Warr, K. J. 2002 Long-term trends in the trophic structure of the North Sea fish community: evidence from stable-isotope analysis, size-spectra and community metrics. *Mar. Biol.* **141***,*1085-1097.

20 Cardinale, M., Hagberg, J., Svedang, H., Bartolino, V., Gedamke, T., Hjelm, J., Börjesson, P. & Norén, F. 2010 Fishing through time: population dynamics of plaice (Pleuronectes platessa) in the Kattegat-Skagerrak over a century. *Popul Ecol*. **52**, 251-262.

21 Bartolino V, Cardinale M, Svedäng H, Linderholm, H. W., Casini, M. & Grimwall, A. 2012 Historical spatiotemporal dynamics of eastern North Sea cod. *Can. J. Fish. Aquat. Sci.* **69**, 833-841.

22 Hornborg, S., Nilsson, P., Valentinsson, D. & Ziegler, F. 2012 Integrated environmental assessment of fisheries management: Swedish *Nephrops* trawl fisheries evaluated with a Life Cycle approach. *Mar. Pol.* **36**, 1193-1201*.*

23 Catchpole, T. L., Tidda, A. N., Kella, Kella, Revilla, A.S. & Dunlinb, G. 2007 The potential for new *Nephrops* trawl designs to positively effect North Sea stocks of cod, haddock and whiting. *Fish. Res.* **86,** 262-267.

24 Baumann, M. 1995 A comment on transfer efficiencies. *Fish. Oceanogr.* **4**, 264-266.

25 Pihl, L. 1994 Changes in the diet of demersal fish due to eutrophication-induced hypoxia in the Kattegat, Sweden. *Can. J. Fish. Aquat. Sci.* **51**, 321-336.

26 Diaz, R. J. & Rosenberg, R. 2008 Spreading dead zones and consequences for marine ecosystems. *Science* **321**, 926-929.

27 Baden, S., Emanuelsson, A., Pihl, L., Svensson, C. J. & Åberg, P. 2012 Shift in seagrass food web structure over decades is linked to overfishing. *Mar. Ecol. Progr. Ser.* **451**, 61-73.

28 Nielsen, E. & Richardson, K. 1996. Can changes in the fisheries yield in the Kattegat (1953-1992) be linked to changes in primary production? *ICES J. Mar. Sci.* **53**, 988-994.

29 Hunt, G. L. Jr. & McKinnel, S. 2006. Interplay between top-down, bottom-up, and wasp-waist control in marine ecosystems. *Prog. Oceanogr.* **68**, 115-124.

3. Marine resource use: Assessing by-catch impacts in fisheries LCAs

By-catch impacts in fisheries: utilizing the IUCN Red List Categories for enhanced product level assessment in seafood LCAs

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Note: This work has been accepted for publication in Environmental Management

Keywords: conservation, discards, fisheries management, IUCN Red List, LCA, seafood

Abstract

Overexploitation of fish stocks causes concern not only to fisheries managers and conservation biologists, but also engages seafood consumers, and more integrated product perspectives would be useful. This could be provided Life Cycle Assessment (LCA); however, further complements of present LCA methodology are needed to assess seafood production, one being by-catch impacts. We studied the scientific rationale behind using the IUCN Red List of Threatened Species^{TM} for assessment of impacts relating to fish species' vulnerability. For this purpose, the current Red List status of marine fish in Sweden was compared to the advice given in fisheries as well as key life history traits known to indicate sensitivity to high fishing pressure. Further, we quantified the amount of threatened fish (Vulnerable, Endangered, or Critically Endangered) that was discarded in demersal trawl fisheries on the Swedish west coast. The results showed that not only did the national Red List of marine fish have a high consistency with advice given in fisheries and indices of vulnerability, the different fishing practices studied were also found to have vastly different amounts of threatened fish discarded per kilo landing. The suggested approach is therefore promising as a carrier of aggregated information on the extent to which seafood production interferes with conservation priorities, in particular for species lacking adequate stock assessment. To enable extensive product comparisons it is important with increased coverage of fish species by the global IUCN Red List, and to reconsider the appropriate assessment unit (species or stocks) in order to avoid false alarms.

3.1. Introduction

The old perception of the inexhaustible sea, legendarily expressed by Huxley (1883), has increasingly been altered towards the acknowledgement that modern fisheries have caused depletion of many commercial fish stocks (Hutchings 2000; Christensen et al. 2003; Myers and Worm 2003). For targeted species, it has been shown that certain characteristics contributes to extra susceptible to overfishing: extremely limited geographical distribution (Sadovy de Mitcheson et al. 2012), highly migratory (Collette et al. 2011), or late maturity, large final size, and long life span (Jennings et al. 1998; Roberts and Hawkins 1999). Several species have also been locally extirpated from being caught as by-catch (Brander 1981; Casey and Myers 1998; Dulvy et al. 2000), when insufficient monitoring effort results in inadequate data for assessing fishing mortality (Johannes 1998). The definition of what could be considered as by-catch fish species is not always straightforward, but if defined as un-used or unmanaged (i.e. including discards), over 40 % of global catches could be considered to be by-caught (Davies et al. 2009). It has also been estimated that 80 % of global landings lack proper stock assessment (Costello et al., 2012). Altogether, this makes the risk of further depletion due to overexploitation still of great concern.

With better understanding of critical factors that determine fishing mortality, as well as acknowledging the broader effects from fisheries on the ecosystem structure and function, fisheries management is evolving by various approaches. One emerging concept is Ecosystem-Based Fisheries Management (EBFM), an integrated framework that requires managers to focus not only on target stocks, but also, among other aspects, on conservation of marine biodiversity (Pikitch et al. 2004). By implementing this approach, the risks of depletion may decrease; however, EBFM is still far from current practice.

Outside of fisheries management, there are several other assessments available of the vulnerability of species to fishing pressure. The work by the International Union for Conservation of Nature (IUCN) Red List of Threatened Species™ commenced with the aim to "*identify and document those species most in need of conservation attention if global extinction rates are to be reduced*", and has over time expanded its remit to also monitor trends in global levels of biodiversity loss (IUCN 2012). The assessments are arduous and costly; still, these are rarely considered in fisheries management, even if fisheries play a key role for abundance levels of single fish species. This is due to various reasons, one being that conservation biology and fisheries management take on rather different perspectives, i.e. conservation versus maximum sustainable yield (MSY). The Red List provides no information on which level of catch that is a sustainable outtake, whether a size limit would be effective, etc., but only identifies species to be at different levels of conservation need. As a result, fisheries managers focus on data-rich assessments of commercial species; conservation biologists often operate with data-scarce assessments of rare and declining species (Mace and Hudson 1999). Noteworthy, the target for biomass B_{MSY}, i.e. a 50% reduction of un-fished biomass that in theory is considered to produce MSY, is also a level of decline qualifying a species to be threatened with extinction, i.e. the Red List Category 'Vulnerable' according to the IUCN Red List A-criterion (Reynolds et al. 2005; IUCN 2010). Exacerbating this possible clash in perception, the IUCN Red List framework has been mistrusted (Possingham et al. 2002). This is often due to prevailing false perceptions that the assessments are made based on "expert opinions", instead of acknowledging the now operational global standards and strict procedures (Rodrigues et al. 2006; IUCN Standards and Petitions Subcommittee 2010).

Parallel to discussions on how to improve fisheries management systems and mitigate possible conflicts with conservation biologists, growing consumer awareness and global policies call for improved transparency of the environmental impacts attributed to seafood production. In our global society it is also increasingly important to evaluate and communicate differences in impacts between consumer products, as the visualizing of cause-effect chain of threat to biodiversity caused by demand of different product is lost along global distribution routes (Lenzen et al. 2012). There are initiatives at policy level, such as the COP 10 decisions adopted in Nagoya, which identified a need for a product perspective on their relative conflict with conservation, and stated that the COP "*Invites… the development of methods to promote science-based information on biodiversity in consumer and producer decisions*" (Decision X/44: 11,12). In terms of consumer initiatives, recent years have shown a growing interest in easy-to-grasp information such as eco-labeling (FAO 2010), with e.g., the Marine Stewardship Council (MSC) label increasingly found in seafood displays (Thrane et al. 2009). Eco-labeling is, in general, recommended to be based on quantitative and systematic approaches. One form of eco-label, Type III Environmental Product Declarations Type III (ISO 2006a), is even required to be based on a full product level environmental systems analysis called Life Cycle Assessment (LCA). LCA has been identified as a "prerequisite for sustainability assessments" (Klöpffer 2003), and provides a broad and integrated product perspective (ISO 2006bc). Still, methodological advancement is imperative to better inform labeling of seafood products (Pelletier and Tyedmers 2008) and the present coverage of the LCA framework with regard to seafood production systems is developing by various approaches (Vázquez-Rowe et al. 2012a; Ford et al. 2012; Kruse et al. 2009). In the eco-labeling schemes, various ways of assessing impacts related to un-wanted catches have been developed, such as e.g. the concept of Endangered, Threatened or Protected (ETP) species in the MSC certification (a label not based on LCA). Still, the definition of what is needed to qualify as an ETP species is unfortunately not clearly stated (MSC 2011). From this lack of standardization and methodology, product level approaches to seafood accounting for differences in fishing impacts on vulnerable species risks to be insufficient.

Altogether, there is a societal need for a robust framework to assess impacts of fishing on vulnerable species from the perspectives of managers, practitioners and consumers. One way forward is to use an integrated product level approach such as LCA, with quantitative science-based, yet easily comprehensible, indicators (ISO 2006c). This study therefore focuses on advancement of one area of assessment, by-catch of sensitive species; more specifically the un-used part i.e. discards, to complement the present LCA framework. The general impact assessment methodology in traditional LCAs is independent of time and space, yet discards could be characterized as being a proximate ecological concern. This is an area in general not adequately covered in traditional LCAs (Reap et al. 2008), but important attempts have been made (Ford et al. 2012). In previous seafood LCAs, discards have at best been assessed in terms of live weight (in kilo discard per landing, possible separated by species composition), live weight relative a global discard rate (GDI) or in terms of primary production required from discards (Vázquez-Rowe et al. 2012b). Estimating the primary production required is an important advancement; still, it does not convey information on the discarded species sensitivity to impact, an area identified to be yet to develop in LCA (Pelletier et al. 2006).

The overarching goal is to discuss the potentials of utilizing the Red List Categories in order to provide at quantitative indicator to enhance product-related information covering discard impacts from seafood. The aims of our study are therefore two. Firstly, to critically examine the current Red List classification of marine fish species found in Sweden. Secondly, to explore the prospects of assessing discard impacts of fisheries in LCA by utilizing the IUCN Red List in a case study of demersal trawling in Sweden.

3.2. Methods

We studied the science rationales behind the classification of marine fish species by the Swedish Red List (for regional assessment details see Gärdenfors et al. 2010). This was done by comparing identified the conservation status with other assessments of vulnerability, such as fisheries advice (that are based on the same data sets as the Red List assessments) and life history traits known to indicate sensitivity to fishing pressure (Table 3.1). Species with life history traits that characterize extra sensitivity to high fishing pressure should be more likely to be at increased level of threat from exploitation relative to other species. A correlation was therefore done between Red List status and mean life history traits of the fish species belonging to the each IUCN Category. If higher values for the studied traits are found to correlate with fish species having higher threat status, this may suggest accuracy of the categorization made by the Red List. For literature references and values for life history traits used in this study (age at maturity A_{mat}, maximum age A_{max} and maximum length L_{max}), see Online resource 1).

The other approach to test for scientific validity was to compare Red List status with the advice of fisheries managers. In this study, advice in fisheries is considered to be an accurate perception of the current status of the stocks. If spawning stock biomass (SSB) was not at full reproductive capacity (SSB_{pa}) according to the International Council for Exploration of the Seas (ICES) in 2009, the year for the latest Swedish Red List assessment, this was considered to be consistent with having a threat status (either VU, EN, CR), i.e. “hit” (ICES 2009ab; for detailed information see Table S3 in Supporting Information). The term “miss” implies failure of the Swedish Red List to include a species being at risk, which for stocks implies to not be at full reproductive capacity. “False alarm” is a species identified as Threatened according to the Red List (VU, EN, CR) with a biomass above SSB_{pa} (Dulvy et al. 2005).

In the second part of this study, we focused on the applicability of using this form of screening of catches. The amount of threatened fish (according to the Red List) that was discarded per kilo landed (suggested abbreviation “VEC”, stands for Vulnerable, Endangered or Critically Endangered) was quantified in terms of mass (kilo) and individuals (number), from demersal trawl segments on the Swedish west coast (Skagerrak and Kattegat area). We also studied the discards in terms of Red List Index (RLI); this is a dimensionless indicator used as a measure of progress towards conservation goals, recognized by the CBD as suitable for monitoring trends in global biodiversity or species complexes (Biodiversity Indicators Partnership 2010; Bubb et al. 2009; Butchart et al. 2005). The RLI of discard compositions from pre-defined fishing segments was analyzed relative to an ecosystem state of RLI (i.e. the RLI for all marine fish in the area according to the Swedish Red List). This was done in order to examine whether the different fishing segments catch what is out there (RLI would be the same for all discard compositions) or if different degrees of impact on threatened fish species could be seen between the various fishing practices due to targeting patterns (RLI would differ). RLI is estimated as follows (from Bubb et al. 2009):

$$RLI_t = 1 - \frac{\sum_s W_{c(t,s)}}{W_{EX} \times \mathrm{N}}$$

where $W_{c(t,s)}$ is the weighting of category *c* for species *s* at time *t*, W_{EX} is the weighting for Extinct, and *N is* the number of assessed species excluding Data Deficient (DD) species in the time period, and those considered to be Extinct in the initial assessment year. In regional assessments W_{RE}, the weighting for Regionally Extinct, is used instead of W_{EX}.

The RLI weighting W is defined by IUCN as LC=0, NT=1, VU=2, EN=3, CR=4 and RE=5. A RLI value of 1 implies that all species are of Least Concern, a value of zero is equivalent to all species having gone extinct. By this approach, monitoring of rates of extinction over time is enabled. In this study, the same weights were used for ranking values in order to test for a possible relationship between species' life history traits and Red List categorization by a randomization test for correlation, calculated with the software package "Resampling" (Howell 2007).

Data

We obtained actual catch data (including discards) on a haul basis from pre-defined fishing segments (Fig. 1) for one year (2009). These were sampled by trained scientific observers in accordance with the Data Collection Framework (DCF), and include all fish and commercial invertebrates (EC Council Regulation 199/2008). Non-commercial invertebrates are not sufficiently monitored in the current observer programs to be included in this assessment, but it should be noted that it has been found that the proportions of threatened non-commercial invertebrates according to the Red List vary between different trawling segments in the area (Ottosson 2008). The same constraint relating to data deficiency applies for unintentional by-catch of threatened marine mammals and birds.

The discard survival is assumed to be zero. In general, the mortality rate of discarded fish species depends on a number of factors: species, gear, trawling haul length, depth, temperature, water salinity, handling time, and more (Suuronen, 2005). Due to these uncertainties, applying a general percentage of survival is hazardous. In this study it is however most likely that survival is zero, as the discarded threatened fish species were mainly gadoids, which have little chance of survival after having been trawl caught.

Table 3.1. Assessment methods used to evaluate robustness of current IUCN classifications of Swedish marine fish species; references for each trait are studies supporting the choice of parameter.

Assessment method	Comment
ICES advice for 2010	State of spawning stock biomass in relation to precautionary limits. If impaired, the stock is considered to be in need of conservation actions.
Global IUCN Red List of Threatened Species	Conservation status for the species globally.
L_{max}	Maximum length (Jennings et al. 1998; Piet and Jennings 2005; Greenstreet and Rogers 2006)
A_{mat}	Age at maturity (Jennings et al. 1998; Robert and Hawkins 1999)
A_{max}	Maximum life span (Roberts and Hawkins 1999)

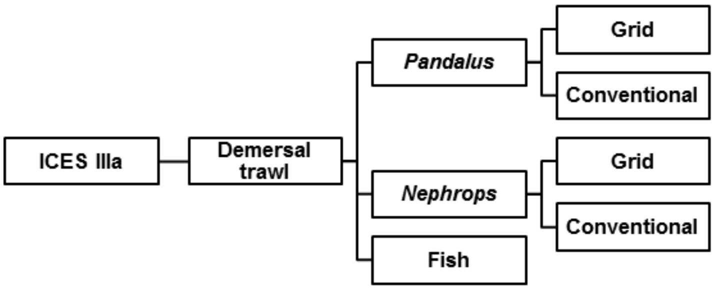


Fig. 3.1. Demersal trawl segments in ICES area IIIa for which specific discard sampling was carried out and used in this study. Grid implies species selective trawling. The numbers of hauls were: Northern prawn *Pandalus borealis* (L) without grid n=25, with grid n=18; Norway lobster *Nephrops norvegicus* (L) without grid n=49, with grid n=44; and mixed demersal fish n=31.

The discard survival is assumed to be zero, which in this case should be fairly accurate as the threatened fish species that were found to be discarded in this study were mainly gadoids, which have little chance of survival after having been trawl caught. In general, the mortality rate of discarded fish species depends on various factors: species, gear, trawling haul length, depth, temperature, water salinity, handling time, and more (Suuronen, 2005). Due to these uncertainties, applying a general percentage of survival is hazardous.

3.3. Results

Validity of Red List Categories and Criteria for fish in Sweden

The Red List classification of the studied fish species in terms of threat status was in general found to be coherent with fisheries advice at the time of assessment (Table 3.2). Differences between stocks would be one factor that could affect coherency relative to the threat status of the species. The Swedish Red List assessment showed consistency in relation to the global Red List assessment with regard to five species, whereas seven species had a higher level of threat status regionally and six species were Not Evaluated (NE) globally (Table 3.3). Major differences were found in the area in terms of coverage of species assessed by ICES compared to the Red List (Table 3.4). While nearly 50 % of the marine fish species occurring in the waters around Sweden were assessed by the national Red List, ICES provided advice for merely 14 %. This originates in part from not all species being of interest to fisheries; still, many important commercial species also lacked biological reference points for management purposes due to data deficiency. This hindered evaluation with regard to the Red List status for some of the species (listed as uncertain in Table 3.2).

Table 3.2. Red List compared to ICES advice. The abbreviations are Critically Endangered (CR), Endangered (EN), Vulnerable (VU), Near Threatened (NT), Least Concern (LC) and Not Applicable (NA). 'Hit' implies coherent fisheries management advice and Red List status; 'miss' is a failure by the Red List to identify species at risk; 'false alarm' is a fish listed as threatened, yet considered to be sustainably exploited. For details see Online resource Table 3.6.2.

IUCN	IUCN/ICES compatibility	Species/stocks with ICES advice (2009)	Comment
CR	4 hits	4 species	
EN	5 hits, 2 false alarms	4 species, 7 stocks	False alarms: Haddock[a] and cod[b] (Eastern Baltic, stock 25-32).
VU	1 uncertain	1 species	
NT	-	-	
LC	3 hits, 2 misses and 14 uncertain	13 species, 19 stocks	Misses: Herring[c] (stock IIIa, autumn spawners) and salmon[d].
NA	-	8 species	

[a] According to the Swedish Board of Fisheries, the local stocks in the Kattegat and Skagerrak are depleted (Fiskeriverket 2010), thus supporting regional threat status.

[b] Stock status has improved in recent years, now considered to be sustainably fished and open for KRAV-and MSC certification. Still, the mean weight of larger cod has sharply declined in recent years and the former three spawning grounds have decreased to only one.

[c] Increased risk on SSB level. However, naturally fluctuating species are prone to have difficulties in abundance trends for Red List assessments.

[d] SSB considered to be low.

Table 3.3. Marine fish species on the Swedish Red List of Threatened Species in 2010 (assessment done in 2009). The categories are Regionally Extinct (RE), Critically Endangered (CR), Endangered (EN), Vulnerable (VU). Definitions of criteria are found in IUCN Standards and Petitions Subcommittee (2010).

Scientific name	Common name	Swedish IUCN Red List	Global IUCN Red List*
Dipturus batis	Blue skate	RE	CR
Acipenser oxyrinchus	Atlantic sturgeon	RE	CR
Lamna nasus	Porbeagle	CR	VU (CR)
Cetorhinus maximus	Basking shark	CR	EN
Squalus acanthias	Picked dogfish	CR	VU (CR)
Anguilla anguilla	European eel	CR	CR
Pollachius pollachius	Pollack	CR	NE
Chimaera monstrosa	Rabbit fish	EN	NT
Raja clavata	Thornback ray	EN	NT
Coryphaenoides rupestris	Roundnose grenadier	EN	NE
Molva molva	Ling	EN	NE
Gadus morhua	Atlantic cod	EN	VU

Melanogrammus aeglefinus	Haddock	EN	VU
Anarhichas lupus	Atlantic wolffish	EN	NE
Hippoglossus hippoglossus	Atlantic halibut	EN	EN
Galeorhinus galeus	Tope shark	VU	VU
Somniosus microcephalus	Greenland shark	VU	NT
Etmopterus spinax	Velvet belly	VU	NE
Merlangius merlangus	Whiting	VU	NE

*In brackets: conservation status in the studied area.

Table 3.4. Number of fish species or stocks in the studied area and the proportion assessed by IUCN or ICES.

Assessment method	Number of species (stocks)	Proportion of marine fish occurring in the area (214 species)
National IUCN Red List	106	49%
Global IUCN Red List	9	4%
ICES*	31(40)	14%

There was a strong correlation found between all three life history traits that indicate sensitivity to fishing and Red List category (A_{mat}: r=-0.75, n=73 P<0.0001; A_{max}: r=-0.65, n=75, P<0.0001; and L_{max}: r=-0.61, n=94 P<0.0001). This implies that fish species in this area that have later onset of maturity, longer life span, and larger final size are at a progressively higher level of threat status compared to short-lived, early-maturing and small species (Fig. 3.2).

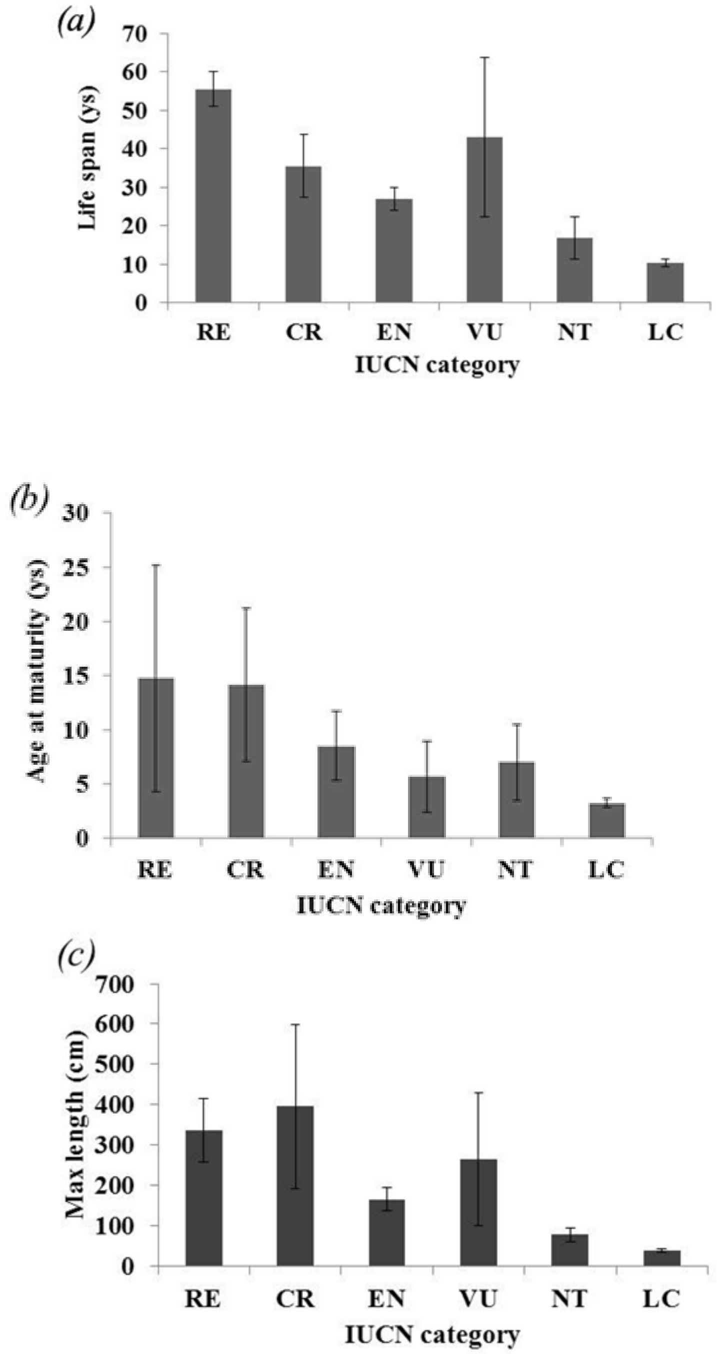
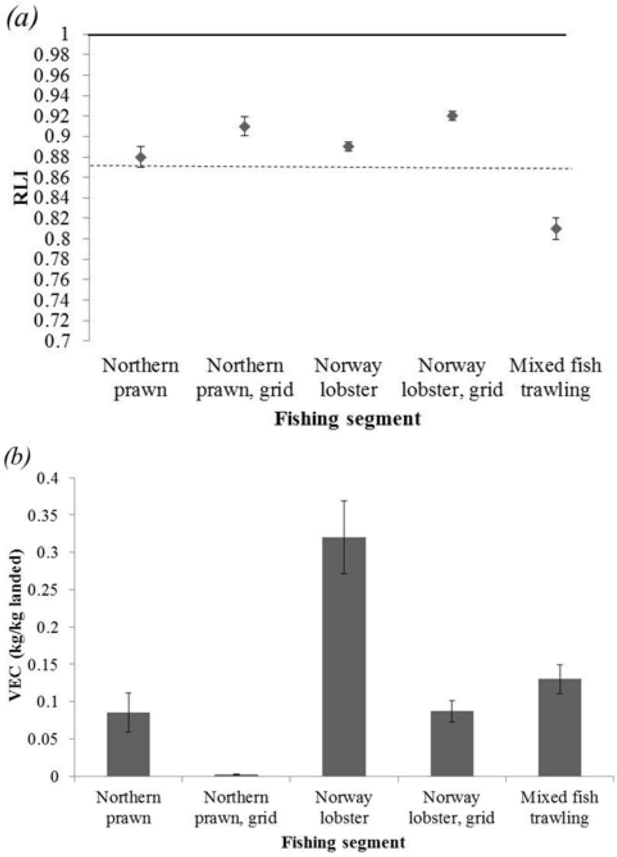


Fig. 3.2. Mean values (with standard errors) for life history traits of Swedish marine fish species found at different levels of conservation status nationally, with a) maximum life span b) age at maturity; c) maximum length.

Applicability of the approach

As the RLI of discards varied between fishing segments and relative the ecosystem RLI, it is clear that targeting patterns affect by-catch of threatened fish species (Fig. 3.3a). The highest occurrence of threatened fish species in the discard was found in the mixed fish trawl segment. In terms of amount of threatened fish discarded relative to landings (VEC), the conventional trawl fishery targeting Norway lobster *Nephrops norvegicus* (L.) had the highest amount discarded (3000 individuals or 320 kg per landed ton), whereas species-selective trawling for northern shrimp *Pandalus borealis* (Krøyer, 1838) showed the lowest amount of Threatened fish discarded (300 individuals or 85 kg per ton landed) among the studied trawl fisheries (Fig. 3.3bc). Haddock *Melanogrammus aeglefinus* (L.), whiting *Merlangius merlangus* (L.), and cod *Gadus morhua* (L.) were in general the most abundant threatened fish species in both mass and numbers; in the *P. borealis* fishery, rabbit fish *Chimaera monstrosa* (L.) was also abundant.



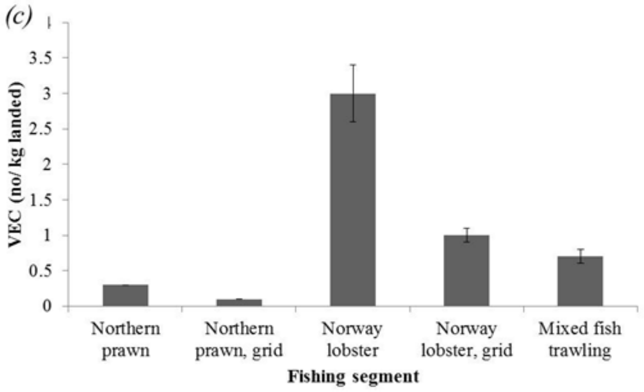


Fig. 3.3 a) The Red List Index (RLI) with standard errors for discards in different fishing segments. The overall RLI value for all marine fish species in the area (year 2010) was 0.87 (dotted line) and the optimum RLI value is 1 (bold line): mixed demersal trawling discard the highest number of threatened fish species; b) VEC for the various fishing segments, in threatened fish in kilo per kilo of total landings (with standard errors): the trawling for Norway lobster has the highest discard rate of threatened fish; and c) as in Fig b, but VEC quantified in individuals instead of weight per kilo landing: Norway lobster trawling has the highest impact.

3.4. Discussion

To study the amount of Vulnerable, Endangered or Critically Endangered (VEC) fish discarded per landed kilo of seafood is a new and promising quantitative approach for assessing differences in un-wanted catches of sensitive species from product level. By complementing impact assessments in seafood LCAs with VEC, the assessment is broadened in terms of covering local ecological sensitivity of different fishing practices (for methodological details, see Online resource 3.6.4). As seafood consumption per capita is on the rise (FAO 2012), sustainable development of the sector is imperative. Catch of threatened fish species adds to stock depletion and impedes the rebuilding of fisheries, and is in direct conflict with the Convention on Biological Diversity (CBD, Target 12). Seafood guides and eco-labels make some recommendation of the purchase of seafood on display, i.e. the landed part, but little information is available of the 'unseen' discards. In addition, landed fish species which lack proper management plans could also benefit from increased acknowledgement of the Red List classification system; VEC may be used as a general by-catch impact assessment of data deficient species. Integrated assessments of seafood production including catch of threatened species, such as VEC, should therefore be highly prioritized and has a step-wise benefit; it enables enhanced communication, which in the end hopefully leads to enforcement of stricter discard management policies.

Results also show that we can no longer dismiss the fact that many fish species are now classified as threatened according to the IUCN Red List Categories and Criteria. On land, occurrence of threatened species in an area can impede development plans and induce active management measures. Still, both targeted and non-targeted fish species with a Red List threat status are subject to continued fishing pressure (Colette et al. 2011; Sadovy de

Mitcheson et al. 2012). The public lack of equally acknowledging threat status between different species complexes (i.e. by-catch of birds and mammals normally gets more attention than fish) might come from prevailing popular beliefs that fish stocks have in general greater recovery potentials. However, the failure to consider and protect marine fish species as is done with more charismatic species should be brought to greater attention, as the recovery potential of marine fish (especially for elasmobranchs), on average, recently has been suggested to be in the same range, or less, as that of terrestrial mammals (Hutchings et al. 2012).

In this study, conventional trawling for *N. norvegicus* had the greatest impact on VEC, both in terms of mass and numbers, which is in line with previous findings on general discard rates in fisheries (Kelleher 2005). These results are therefore promising, as the intent of the suggested approach of utilizing the Red List framework is to provide a representative and quantitative measure of discard impacts. In this case, having a considerable VEC value is due to the fact that a higher number of depleted, thus likely to be threatened, fish species are discarded, in combination with the fact that this particular fishery has a higher discard proportion relative to landing quantity.

In terms of RLI, mixed demersal fish trawling was shown to be of the greatest concern. This indicates a higher occurrence of different fish species with a Red List threat status in the discards. Still, RLI does not take volume into account; it is merely a species count. Even if RLI has been suggested as an indicator of biodiversity targets (Brooks and Kennedy 2004), this makes the prospects less favorable of using RLI from a product perspective, i.e. in LCA. The implications of using RLI compared to VEC can be seen in species-selective trawling for *N. norvegicus,* namely a fishery that has the lowest RLI, but a rather high VEC. This is the result from the fishery being characterized by great discard quantities of juvenile whiting (VU) in combination with a generally higher discard rate relative to "cleaner" fish trawling. Applying RLI would therefore fail to include information on the amount of threatened fish species discarded, as it only represents the presence of a threatened species, which in terms of conservation efforts is highly relevant information. From this, including weight or the number of individuals of VEC fish species discarded per landing would probably be the most illustrative and easily interpreted measure to assess the sensitivity of discard impacts of seafood.

There are uncertainties connected to the proposed method to assess discard impact. One of these is the robustness of the current IUCN framework to capture threat status. In this study it was shown to be satisfactory, further supported by prior studies (Dulvy et al. 2005; ICES 2008b). There have however been doubts on whether the assessment by the IUCN is appropriate for actively regulated stocks, where it could falsely lead to false alarms as well as missing signals that indicate risk (ICES 2009cd). Based on results in this study, the discrepancies may be the result of the assessment unit; the Red List uses species level while it would be more accurate to study stock level as fisheries management do. The second uncertainty is related to which kind of data that is going to be available to support the methodology proposed in this paper. Data on species composition in discards is most often scarce, but ideally, collection of discard data should be mandatory in a seafood LCA; this would also be of considerable benefit to both conservation and fisheries managers.

Another uncertainty relates to coverage of marine species by the IUCN Red List in general. For improved assessments of discards from seafood products, increased coverage of species by the IUCN Red List is essential. Species groups known to be extra sensitive to fishing pressure have been given priority in terms of assessment, and the global IUCN Red List currently covers e.g. all cartilaginous fishes (Hoffman et al. 2010). The assessment of marine species by the IUCN Red List is highly prioritized, with currently one-quarter of marine fish assessed, and recent initiatives intend to complete assessments within five years (Collette et al. 2013). This work is certainly promising for an equal basis for product comparisons in the future. There are also a few initiatives of assessing threat status on a national level; these have proven to be fairly consistent with global assessments (Gärdenfors 2001). In our study, the national Red List reported somewhat higher levels of threat status than the global Red List. This is not surprising as fish stocks, rather than entire fish species, have been, and still are, under the most severe risk (Reynolds et al. 2005). This finding is further supported by the fact that some of the possible misses/false alarms of the Red List in this study originated from differences in status between stocks. Being an impact of proximate ecological concern, it is most likely that the higher resolution, the better, i.e. calling for national Red Lists to be useful from a LCA perspective.

The present attribution of discard impacts in seafood LCAs by using the specific discard weight relative to a global average or the equivalent for resource appropriation in terms of primary production required (PPR; Vázquez-Rowe et al. 2012a), or as before, in terms of total discard weight and possible a species lists (Pelletier et al. 2006) is of relevance but could lead to erroneous comparisons. The PPR approach has additional values compared to a mere weight perspective, as differences in ecosystem energetics are better accounted for by differentiating between the trophic levels; still, it lacks the dimension of vulnerability. As a consequence, deep-sea fisheries could inappropriately appear less impacting because low PPR values or discard weight do not convey information regarding different species composition in terms of their relative sensitivity to impact. In addition, the discarded amount could also be great in terms of weight or PPR, but with relatively low impact on threatened fish species, whereas another fishery has a lower discard rate but a higher amount of threatened species (Hornborg et al. 2012). Hence, the amount of discard, especially in kilos, does not provide a good picture of the impact of discarding. Altogether, this calls for using complementing indices for discard impacts, depending on the level of data detail that is available: total weight (in kilo), resource waste (PPR) and sensitivity (VEC) per kilo seafood product. These quantities all convey different messages. Still, even if the VEC approach includes sensitive fish species, it fails to provide information regarding invertebrates, birds and marine mammals due to insufficient data. Further work should be directed towards this issue, and possibly, there could be different categories of VEC (fish, mammals, birds, etc.). Still, in terms of in particular invertebrates, improved monitoring and understanding of invertebrate abundances and discard rates must be prioritized to more fully assess the discard impact caused by a seafood product (Ottosson, 2008). This is especially urgent in relation to deep-sea trawling, where fishing activities are likely to degrade existing remnants of sensitive ecosystems dominated by invertebrates (Roberts et al. 2006). In terms of deep-sea fish species, the global Red List unfortunately does not sufficiently cover them either (Devine et al. 2006), and the VEC approach would therefore still fail to represent conflicts between deep-sea fisheries and conservation interests.

In this paper, we have shown that Red List threat status, that to a great extent rely on abundance trends, also correlated with advice given in fisheries and life history traits sensitive to fishing. Even if it based on earlier findings could be suggested that life history traits alone could be an indicator for a fish species' response to exploitation (Jennings et al.

1998), it has also been shown that fast-growing species could also have a higher probability of collapse than slow-growing species, even if the recovery time of slow-growing species is longer (Pinsky et al. 2011). The higher risk of collapse of fast-growing species is related to management allowing higher fishing pressure on these species. It should be noted that by-catch species rarely have target fishing mortalities, which would be needed in order to be concerned with fast-growing species, and results from this study support the finding that slow-growing species are more of concern in the studied area. Still, VEC has an important dual approach; it accounts both for abundance trends and correlates with life history traits that indicate sensitivity to fishing pressure, thus being a carrier of aggregated information.

Last, it could well be so that as the aim of the Red List is to provide an alarm signal when a decline is observed for an unmanaged species, this suggests that targeted stocks could be considered to be under a management framework, hence of less concern (Mace and Hudson, 1999). It is however remarkable to find that in the studied area, only a minor fraction (14 %) of the fish species were at all considered by ICES in 2009, and many species with a quota did not have defined reference points for setting an appropriate fishing mortality. The Swedish Red List, on the other hand, covered nearly half of the occurring fish species, i.e. all species known to reproduce within the Swedish EEZ. The considerations by ICES have increased since 2009, and in the 2012 advice, great effort has been made to include information on data-limited stocks (ICES 2012). Still, utilizing VEC for assessing discard impacts related to seafood products in terms of vulnerability is promising; it could possibly also be extended to be used as a general by-catch assessment of data deficient species in LCA, or be of relevance to precautionary fisheries advice in in data-limited situations and other policy areas that relate to pressures on marine ecosystems.

Online Resources

Online Resource 1. Literature used for estimates of life history traits and the mean values used in this study.

Online resource 2. ICES advice compared to the Swedish IUCN Red List status of marine fishes in Sweden.

Online Resource 3. Technical details on the VEC methodology in relation to the ISO standard (ISO 2006c).

References

Biodiversity Indicators Partnership (2010) *Biodiversity indicators and the 2010 Target: Experiences and lessons learnt from the 2010 Biodiversity Indicators Partnership*. Secretariat of the Convention on Biological Diversity, Montreal, Canada. Technical Series No. 53, 196 pages.

Brander K (1981) Disappearance of common skate *Raia batis* from Irish Sea. Nature 290: 48–49.

Brooks T, Kennedy E (2004) Conservation biology: Biodiversity barometers. Nature 431: 1046–1047.
Bubb PJ, Butchart SHM, Collen B, Dublin H, Kapos V, Pollock C, Stuart SN, Vié J-C (2009) *IUCN Red List Index – Guidance for National and Regional Use.* Gland, Switzerland.
Butchart SHM, Stattersfield AJ, Baillie J, Bennun LA, Stuart SN, Akçakaya HR, Hilton-Taylor C, Mace GM (2005) Using Red List Indices to measure progress towards the 2010 target and beyond. Philos Trans R Soc B 360: 255–268.
Casey JM, Myers RA (1998) Near extinction of a large, widely distributed fish. Science 281: 690–692.
Christensen V, Guenette S, Heymans J, Walters C, Watson R, Zeller D, Pauly D (2003) Hundred-year decline of North Atlantic predatory fishes. Fish Fish 4: 1-24.
Collette BB, Carpenter KE, Polidoro BA, Juan-Jordá MJ, Boustany A, Die DJ, Elfes C, Fox W, Graves J, Harrison LR et al. (2011) High value and long life—Double jeopardy for tunas and billfishes. Science 333: 291–292.
Collette BB, Polidoro, B, Carpenter K (2013) Corrected numbers for fish on Red List. Nature 495: 47
CBD, Target 12: *"By 2020 the extinction of known threatened species has been prevented and their conservation status, particularly of those most in decline, has been improved and sustained."* Aichi Biodiversity Targets, available at http://www.cbd.int/sp/targets/
Costello C, Ovando D, Hilborn R, Gaines SD, Deschenes O, Lester SE (2012) Status and solutions for the world's unassessed fisheries. Science 338: 517-520.
Davies RWD, Cripps SJ, Nickson A, Porter G (2009) Defining and estimating global marine fisheries bycatch. Mar Policy 33: 661–672.
Devine JA, Baker KD, Haedrich RL (2006) Fisheries: Deep-sea fishes qualify as endangered. Nature 439: 29.
Dulvy NK, Metcalfe JD, Glanville J, Pawson MG, Reynolds JD (2000) Fishery stability, local extinctions and shifts in community structure in skates. Conserv Biol 14: 283–293.
Dulvy NK, Jennings S, Goodwin NB, Grant A, Reynolds JD (2005) Comparison of threat and exploitation status in North-East Atlantic marine populations. J Appl Ecol 42: 883–891.
FAO (2012) *State of the World's Fisheries and Aquaculture*. FAO Fisheries and Aquaculture Department, Food and Agriculture Organization of the United Nations, Rome.
Fiskeriverket (2010) *Fiskbestånd och miljö i hav och sötvatten- Resursöversikt 2010* (in Swedish).
Ford JS, Pelletier NL, Ziegler F, Scholz AJ, Tyedmers PH, Sonesson U, Kruse SA, Silverman H (2012) Proposed local ecological impact categories and indicators for life cycle assessment of aquaculture. J Ind Ecol 16: 254-265.
Greenstreet SPR, Rogers SI (2006) Indicators of the health of the North Sea fish community: Identifying reference levels for an ecosystem approach to management. ICES J Mar Sci 63: 573–593.
Gärdenfors U (2001) Classifying threatened species at national versus global levels. Trends Ecol Evol 16: 511–516.
Gärdenfors U (ed.) (2010) *Rödlistade Arter i Sverige 2010 – The 2010 Red List of Swedish Species.* ArtDatabanken, SLU, Uppsala (In Swedish).
Hoffmann M, Hilton-Taylor C, Angulo A, Böhm M, Brooks TM, Butchart SHM, Carpenter KE, Chanson J, Collen B, Cox NA et al. (2010) The impact of conservation on the status of the world's vertebrates. Science 330: 1503–1509.

Hornborg S, Nilsson P, Valentinsson D, Ziegler F (2012) Integrated environmental assessment of fisheries management: Swedish *Nephrops* trawl fisheries evaluated using a Life Cycle approach. Mar Policy 36: 1193–1201.
Howell DC (2007) University of Vermont, www.uvm.edu/~dhowell/StatPages/Resampling/Resampling.html.
Hutchings J (2000) Collapse and recovery of marine fishes. Nature 406: 882–885.
Hutchings JA, Myers RA, García VB, Lucifora LO, Kuparinen A (2012) Life-history correlates of extinction risk and recovery potential. Ecol Appl 22: 1061-1067.
Huxley T H (1883) Inaugural Meeting of the Fishery Congress. W. Clowes and sons, limited. 22 p.
ICES (2008a) *Report of the ICES Advisory Committee 2008.* ICES Advice, 2008. Book 6, 326 pp.
ICES (2008b). Report of the Working Group on Fish Ecology (WGFE), 3-7 March 2008, ICES, Copenhagen, Denmark. ICES CM 2008/LRC:04.
ICES (2009a) *Report of the ICES Advisory Committee 2009.* ICES Advice, 2009. Book 6, 236 pp.
ICES (2009b) *Report of the ICES Advisory Committee* 2009. ICES Advice, 2009. Book 8, 132 pp.
ICES (2009c) *Report of the Workshop on analytical methods for evaluation of extinction risk of stocks in poor condition (WKPOOR1)*, 18-20 May 2009, Copenhagen, Denmark. ICES CM 2009\ACOM:29.
ICES (2009d) *Workshop for the Exploration of the Dynamics of Fish Socks in Poor Conditions (WKPOOR2)*, 24-27 August 2009, Bergen, Norway. ICES CM 2009\ACOM:49.
ICES (2012) *ICES' Implementation of RGLIFE advice on Data Limited Stocks (DLS)*. ICES Report, 2012. ICES Advisory Committee. ICES CM 2012, ACOM:68. Draft report. 26 pp.
ISO (2006a) Environmental labels and declarations–Type III environmental declarations–Principles and procedures. ISO 14025:2006.
ISO (2006b) Environmental management–Life cycle assessment–Principles and framework. ISO 14040:2006(E). International Organization of Standardization, Geneva.
ISO (2006c) Environmental management–Life cycle assessment–Requirements and guidelines. ISO 14044:2006(E). International Organization of Standardization, Geneva.
IUCN Standards and Petitions Subcommittee (2010). *Guidelines for Using the IUCN Red List Categories and Criteria.* Version 8.1. Prepared by the Standards and Petitions Subcommittee in March 2010. http://intranet.iucn.org/webfiles/doc/SSC/RedList/RedListGuidelines.pdf.
IUCN (2011) *IUCN Red List of Threatened Species*. Version 2011.2. <www.iucnredlist.org>. Downloaded on 31 March 2012.
Jennings S, Reynolds JD, Mills SC (1998) Life history correlates of responses to fisheries exploitation. Philos Trans R Soc B 265: 333–339.
Johannes RE (1998) The case for data-less marine resource management: examples from tropical nearshore finfisheries. Trends Ecol Evol 13: 243–246.
Kelleher K (2005) Discards in world's marine fisheries, un update. FAO. Fisheries Technical Paper (No 470): 131 pp.
Klöpffer W (2003) Life-cycle based methods for sustainable product development. Int J Life Cycle Assess 8: 157-159
Kruse SA, Flysjö A, Kasperczyk N, Scholz AJ (2009) Socioeconomic indicators as complement to life cycle assessment- an application to salmon production systems. Int J Life Cycle Assess 14: 8-18.

Lenzen M, Moran D, Kanemoto K, Foran B, Lobefaro L, Geschke A (2012) International trade drives biodiversity threats in developing nations. Nature 486: 109–112.

Mace GH, Hudson EJ (1999) Attitudes toward sustainability and extinction. Conserv Biol 13: 242–246.

Myers RA, Worm B (2003) Rapid worldwide depletion of predatory fish communities. Nature 423: 280-283.

MSC (2011) *Amendments to the Fisheries Assessment Methodology.* TAB D-032 v1. Date of issue 19th of January 2011, Effective date 7th of February 2011

Ottosson L (2008) *By-catches of non-commercial invertebrate taxa in Skagerrak and Kattegat, generated by demersal otter trawling.* Master Thesis in Marine Ecology, Department of Marine Ecology, Gothenburg University, Sweden. Contribution no. 374.

Pelletier NL, Ayer N, Tyedmers P, Kruse S, Flysjo A, Robillard G, Ziegler F, Scholz A, Sonesson U (2006) Impact categories for life cycle assessment research of seafood production systems: Review and prospectus. Int J Life Cycle Assess 12: 414–421.

Pelletier N, Tyedmers P (2008) Life Cycle considerations for improving sustainability assessments in seafood awareness campaigns. Environ Manage 42: 918-931

Piet GJ, Jennings S (2005) Response of potential fish community indicators to fishing. ICES J Mar Sci 62: 214–225.

Pikitch EK, Santora C, Babcock EA, Bakun A, Bonfil R, Conover DO, Dayton P, Doukakis P, Fluharty D, Heneman B et al. (2004) Ecosystem-based fishery management. Science 305: 346–347.

Pinsky ML, Jensen OP, Ricard D, Palumbi SR (2011) Unexpected patterns of fisheries collapse in the world's oceans. Proc. Natl. Acad. Sci. U. S. A. 108:8317-8322.

Possingham HP, Andelman SJ, Burgman MA, Medellin RA, Master LL and Keith DA (2002) Limits to the use of threatened species lists. Trends Ecol Evol 17: 503–505.

Reap J, Roman F, Duncan S, Bras B (2008) A survey of unresolved problems in life cycle assessment, Part 2: Impact assessment and interpretation. Int J Life Cycle Assess 13: 374-388

Reynolds JD, Dulvy NK, Goodwin NB, Hutchings JA (2005) Biology of extinction risk in marine fishes. Philos. Trans. R. Soc. B 272: 2337–2344.

Roberts CM, Hawkins JP (1999) Extinction risk in the sea. Trends Ecol Evol 14: 241–246.

Roberts JM, Wheeler AJ, Freiwald A (2006) Reefs of the deep: The biology and geology of cold-water coral ecosystems. Science 312: 543–547.

Rodrigues ASL, Pilgrim JD, Lamoreux JF, Hoffmann M, Brooks TM (2006) The value of the IUCN Red List for conservation. Trends Ecol Evol 21: 71–76.

Sadovy de Mitcheson Y, Craig MT, Bertoncini AA, Carpenter KE, Cheung WWL, Choat JH, Cornish AS, Fennessy ST, Ferreira BP, Heemstra PC et al. (2012) Fishing groupers towards extinction: a global assessment of threats and extinction risks in a billion dollar fishery. Fish FishDOI: 10.1111/j.1467-2979.2011.00455.x

Suuronen P (2005) Mortality of fish escaping trawl gears. FAO Fisheries Technical Paper. No. 478. Rome, FAO. 2005. 72p.

Thrane M, Ziegler F, Sonesson U (2009) Eco-labelling of wild-caught seafood products. J Clean Prod 17: 416–423.

Vázquez-Rowe I, Moreira MT, Feijoo G (2012a) Inclusion of discard assessment indicators in fisheries life cycle assessment studies. Expanding the use of fishery-specific impact categories. Int J Life Cycle Assess 17: 535–549.

Vázquez-Rowe I, Hospido A, Moreira MT, Feijoo G (2012b) Best practices in life cycle assessment implementation in fisheries. Improving and broadening environmental assessment for seafood production systems. Trends Food Sci Technol. 28:116-131.

3.6 Online resources to Chapter 3

Table 3.6.1: Literature used for obtaining values for life history traits.

Name	Full source
Curry-Lindahl, K.	*Våra fiskar. Havs- och sötvattensfiskar i Norden och övriga Europa*. P.A. Norstedts & Söners Förlag. Stockholm. 1985
Fishbase	http://www.fishbase.se
IUCN-redlist	http://www.iucnredlist.org
Kottelat, M. & Freyhof, J.	*Handbook of European Freshwater Fishes.* Kottelat, Cornol, Switzerland and Freyhof, Berlin, Germany. 2007
Pethon, P.	*Aschehougs store fiskebok. Norges fisker i farger.* H. Aschehoug & Co. (W. Nygaard) A/S. 4 uppl. 1998
Wheeler	*The Fishes of the British Isles and North-West Europe*. Macmillan and Co Ltd, London. 1969

Table 3.6.2: Life history traits derived from mean values from literature found in Table 3.6.1.

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IUCN	Scientific name	Authority	A_{mat}	A_{max}	L_{max}
RE	*Acipenser oxyrinchus*	Mitchill, 1815	17.5	60	417
RE	*Dipturus batis*	Linnaeus, 1758	12	51	258
CR	*Cetorhinus maximus*	Gunnerus, 1765	16	50	1184
CR	*Lamna nasus*	Bonnaterre, 1788	9.5	38	384
CR	*Pollachius pollachius*	Linnaeus, 1758		14	130
CR	*Squalus acanthias*	Linnaeus, 1758	11.5	56	130
CR	*Anguilla anguilla*	Linnaeus, 1758	19.5	19.5	154
EN	*Anarhichas lupus*	Linnaeus, 1758	9	25	129

EN	*Chimaera monstrosa*	Linnaeus, 1758	15.5	29	140
EN	*Hippoglossus hippoglossus*	Linnaeus, 1758	9	40	360
EN	*Raja clavata*	Linnaeus, 1758	7.5	21.5	112
EN	*Melanogrammus aeglefinus*	Linnaeus, 1758	6	16	111
EN	*Molva molva*	Linnaeus, 1758	6.5	25	196
EN	*Coryphaenoides rupestris*	Gunnerus, 1765			114
EN	*Gadus morhua*	Linnaeus, 1758	6	32.5	170
VU	*Etmopterus spinax*	Linnaeus, 1758	4.5	9.5	65
VU	*Galeorhinus galeus*	Linnaeus, 1758	10	47.5	181
VU	*Somniosus microcephalus*	Schneider, 1801		100	750
VU	*Merlangius merlangus*	Linnaeus, 1758	2.5	15	67
NT	*Petromyzon marinus*	Linnaeus, 1758	7.5	7.5	104
NT	*Sebastes viviparus*	Krøyer, 1845		35	46
NT	*Cyclopterus lumpus*	Linnaeus, 1758	4	13	62
NT	*Zoarces viviparus*	Linnaeus, 1758	2	11.5	50
NT	*Dipturus linteus*	Fries, 1838			133
LC	*Labrus bergylta*	Ascanius, 1767	8	25	57
LC	*Microstomus kitt*	Walbaum, 1792	4.5	17	67
LC	*Pomatoschistus pictus*	Malm, 1865	1.5	1.5	8
LC	*Zeugopterus punctatus*	Bloch, 1787			29
LC	*Labrus mixtus*	Linnaeus, 1758	7.5	20	37
LC	*Ciliata mustela*	Linnaeus, 1758	1		29
LC	*Trachinus draco*	Linnaeus, 1758	4.5	15	45.6
LC	*Callionymus maculatus*	Rafinesque, 1810	1.5		15.2
LC	*Crystallogobius linearis*	Düben, 1845	1	1	5
LC	*Trisopterus minutus*	Linnaeus, 1758	2	5	30.6
LC	*Pollachius virens*	Linnaeus, 1758	5.5	27.5	124
LC	*Centrolabrus exoletus*	Linnaeus, 1758		8	17.4
LC	*Argentina silus*	Ascanius, 1775	11.5	27.5	66.3

LC	*Ammodytes marinus*	Raitt, 1934	2	7.5	24
LC	*Triglopsis quadricornis*	Linnaeus, 1758	4	13.5	43
LC	*Aphia minuta*	Risso, 1810	1	1	6
LC	*Amblyraja radiata*	Donovan, 1808	7.5	18	90
LC	*Chelidonichthys gurnardus*	Linnaeus, 1758			50
LC	*Nerophis lumbriciformis*	Jenyns, 1835	2		17
LC	*Ammodytes tobianus*	Linnaeus, 1758	1.5	7	20
LC	*Salmo salar*	Linnaeus, 1758	6	11.5	141
LC	*Thorogobius ephippiatus*	Lowe, 1839	3.5	6.5	13
LC	*Hippoglossoides platessoides*	Fabricius, 1780	2.5	19.5	47
LC	*Pomatoschistus microps*	Krøyer, 1838	1.5	2.5	7
LC	*Scomber scombrus*	Linnaeus, 1758	3.5	17.5	67
LC	*Nerophis ophidion*	Linnaeus, 1758	1.5	3.5	30
LC	*Syngnathus rostellatus*	Nilsson, 1855	1		17
LC	*Belone belone*	Linnaeus, 1761			93
LC	*Taurulus bubalis*	Euphrasen, 1786			19
LC	*Raniceps raninus*	Linnaeus, 1758			29
LC	*Psetta maxima*	Linnaeus, 1758	3.5	18	100
LC	*Myxine glutinosa*	Linnaeus, 1758	6	9	61
LC	*Callionymus lyra*	Linnaeus, 1758		6.5	30
LC	*Liparis liparis*	Linnaeus, 1766		0	17
LC	*Pleuronectes platessa*	Linnaeus, 1758			97
LC	*Glyptocephalus cynoglossus*	Linnaeus, 1758	5	19	58
LC	*Myoxocephalus scorpius*	Linnaeus, 1758	3	9	48
LC	*Pleuronectes limanda*	Linnaeus, 1758	3.5	14	40
LC	*Pomatoschistus minutus*	Pallas, 1770	1.5	1.5	10
LC	*Clupea harengus*	Linnaeus, 1758	2.5	11	44
LC	*Argentina sphyraena*	Linnaeus, 1758	3.5	15.5	28
LC	*Gobiusculus flavescens*	Fabricius, 1779	1.5	2	6

LC	*Sprattus sprattus*	Linnaeus, 1758			18
LC	*Platichthys flesus*	Linnaeus, 1758	4	18	54
LC	*Agonus cataphractus*	Linnaeus, 1758	1.5	3	20
LC	*Symphodus melops*	Linnaeus, 1758	2.5	10	26
LC	*Lycenchelys sarsii*	Collett, 1871			20
LC	*Scophthalmus rhombus*	Linnaeus, 1758	3.5	17.5	72
LC	*Scyliorhinus canicula*	Linnaeus, 1758	7	13	93
LC	*Pungitius pungitius*	Linnaeus, 1758	1	2.5	8
LC	*Buglossidium luteum*	Risso, 1810	3	13	14
LC	*Phrynorhombus norvegicus*	Günther, 1862			13
LC	*Lumpenus lampretaeformis*	Walbaum, 1792			47
LC	*Lesueurigobius friesii*	Malm, 1874	2.5	11	34
LC	*Ctenolabrus rupestris*	Linnaeus, 1758	2.5	8	19
LC	*Gasterosteus aculeatus*	Linnaeus, 1758	1.5	3	11
LC	*Entelurus aequoraeus*	Linnaeus, 1758	2	0	61
LC	*Syngnathus acus*	Linnaeus, 1758	1	3.5	49
LC	*Gobius niger*	Linnaeus, 1758	1.5	4	17
LC	*Pholis gunnellus*	Linnaeus, 1758	3	5.5	26
LC	*Chelon labrosus*	Risso, 1827	6	22.5	73
LC	*Hyperoplus lanceolatus*	Le Sauvage, 1824	2.5	5	36
LC	*Leptoclinus maculatus*	Fries, 1838			19
LC	*Solea solea*	Linnaeus, 1758	3.5	20	64
LC	*Arnoglossus laterna*	Walbaum, 1792			24
LC	*Liparis montagui*	Donovan, 1804			8
LC	*Syngnathus typhle*	Linnaeus, 1758	1	3.5	33
LC	*Chirolophis ascanii*	Walbaum, 1792			26
LC	*Spinachia spinachia*	Linnaeus, 1758	1	2	20
LC	*Salmo trutta*	Linnaeus, 1758	5.5	13	118

Table 3.6.3: Threat level according to the Swedish IUCN Red List of marine fishes in comparison to ICES advice (for details see ICES 2008, 2009ab). The categories are Regionally Extinct (RE), Critically Endangered (CR), Endangered (EN), Vulnerable (VU), Least Concern (LC), Data Deficient (DD) and Not Applicable (NA). Hit implies coherent fisheries management advice and presence on the Red List; miss is a failure by the Red List to identify species at risk, false alarm is the red-listing of a fish considered to be sustainably exploited.

Species	Stock	IUCN	Spawning biomass in relation to precautionary limits	Fishing mortality in relation to precautionary limits	Fishing mortality in relation to high long-term yield	Fishing mortality in relation to agreed target F	Hit	Miss	False alarm	Comparison not possible
Argentines		NA	No expansions of fisheries							NA
Basking shark		CR	0	0	0	0	1			
Blue ling		NA	No directed fisheries							NA
Blue whiting		NA	Full reproductive capacity	Harvested sustainably	Overfished	Above				NA
Brill	22-32	LC	Unknown	Unknown	Unknown	No target				1
Cod	22-24	EN	Increased risk	Undefined	Over-exploited	Above	1			
	25-32	EN	Undefined	Sustainable	Appropriate	Below			1	
	Kattegat	EN	Reduced reproductive capacity	Unknown	Unknown	Unknown	1			
	Skagerrak	EN	Reduced reproductive capacity	Increased risk	Overfished	Above	1			
Dab		LC	Unknown	Unknown	Unknown	No target				1
European eel		CR	0	0	0	0	1			
Flounder	1	LC	Unknown	Unknown	Unknown	No target				1
Greater forkbeard		NA	No expansions							NA
Haddock	IIIaN	EN	Full reproductive capacity	Harvested sustainably	Appropriate	Below			1	
Hake	IIIa	NA	Full reproductive capacity	Harvested sustainably	Overfished	Appropriate				NA
Herring	IIIa (spr), 22-24	LC	Undefined	Undefined	Overfished					1
	IIIa (aut)	LC	Increased risk	Harvested sustainably	Overfished	Above		1		
	25-29	LC	Undefined	Increased risk	Over-exploited	No target				1
	30	LC	Undefined	Sustainable	Appropriate	No target				1
	31	LC	No assessment	-	-	-				1
Horse mackerel	NS	NA	Unknown	Unknown	Unknown	-				NA
Ling	1	EN	CPUE reduced level				1			
Mackerel	NS	LC	Full reproductive capacity	Increased risk	Overfished	Above	1			
Norway	NS	NA	Full	Undefined	Undefined					NA

pout			reproductive capacity						
Plaice	Baltic	LC	Unknown	Unknown	Unknown	No target			1
	IIIa	LC	Unknown	Unknown	Unknown	No target			1
Porbeagle	1	CR	0	0	0	0	1		
Roundnose grenadier	IIIa	EN	No expansion				1		
Salmon	1	LC	Low			No target		1	
Saithe	IIIa	LC	Full reproductive capacity	Harvested sustainably	Appropriate	Appropriate	1		
Sandeel	IIIa	LC	No assessment	-	-	-			1
Sea trout	1	LC	-			No target			1
Sole	IIIa	LC	Full reproductive capacity	Harvested sustainably	Appropriate		1		
Sprat	22-32	LC	Undefined	At risk	Over-exploited	No target			1
	IIIa	LC	No advice for TAC			-			1
Spurdog	1	CR	0	0	0	0	1		
Turbot	22-32	LC	Unknown	Unknown	Unknown	No target			1
Tusk		NA	CPUE reduced level						NA
Whiting	IIIa	VU	Unknown	Unknown	Over-exploited	No target			1

Online resource 3.6.4 Applicability of VEC relative to mandatory LCIA elements according to ISO 14044:2006.

4.4 Life cycle impact assessment (LCIA)

4.4.2 Mandatory elements for LCIA

4.4.2.2 Selection of impact categories, category indicators and characterization models

4.4.2.2.1

According to the ISO requirements, related information and sources shall be referenced when applying new impact categories, category indicators or characterization models. For VEC, this implies referring to the assessments made by the IUCN Red List Categories and Criteria (www.iucnredlist.org) or, if applicable, national assessments. It is also vital to include a species list from the discard assessment for transparent results.

As the selection of impact categories shall reflect a comprehensive set of environmental issues related to the product system, including assessments of discard impacts in terms of potential effect on Threatened fish species (i.e. VEC) is essential in seafood LCAs.

4.4.2.2.2 Necessary components of the LCIA related to VEC

Environmental mechanism:

Term	Example
Impact category	Threatened fish discard impact potential (VEC)
LCI results	Amount of threatened fish discarded per functional unit
Characterization model	A quantification of the amount of fish species listed as Threatened by the IUCN Red List Categories and Criteria that are directly impacted from being discarded. National Red Lists assessments, if available, should be prioritized over the global Red List assessments, as a higher assessment resolution gives a more robust result.
Characterization factor	Quantification of the amount of fish that is categorised as Vulnerable (VU), Endangered (EN) or Critically Endangered

	(CR) according to the IUCN Red List (VEC/kg or individual of fish species discarded). VU, EN, CR-species =weight or number x 1 Other species =weight or number x 0
Category indicator result	Kilo or individuals of VU, EN and CR fish species discarded per FU.
Category endpoint	Marine ecosystems
Environmental relevance	Seafood production that from discard practices affects threatened fish species is a proxy for possible irreversible depletion of fish species. This impedes effective rebuilding of fish stocks and deteriorates ecosystem structure and function.

4.4.2.2.3 The VEC approach in relation to further recommendations for impact categories, category indicators and characterization models:

a) The IUCN Red List of Threatened Species™ is an internationally recognized approach that categorizes species in terms of conservation status. Using their assessments as characterization model may therefore be considered as robust.

b) one of the outputs in fishing operations is discards, which could consist of Threatened species. This method is therefore covering one of the known impacts of seafood production systems.

c) value-choices and assumptions are avoided by utilizing a common and standardized framework (IUCN Red List) for assessing all fish species that are discarded.

d) discards in seafood production is a local impact with several environmental mechanisms. In order to minimize risks of double-counting, it is important to state in the goal and scope which of the impacts that will be included in the study (discard weight, primary production requirements and/or Threatened species).

e) the characterization model utilizing the IUCN Red List Categories and Criteria is scientifically and technically valid, and by assessing discards in terms of VEC, the environmental mechanism is direct.

f) it has been identified in the related publication to this document that the current assessment unit for the Red List (i.e. species), is less appropriate than assessing fish at stock level. Utilizing national Red Lists in general gives a higher level of threat than global assessments.

g) the category indicators are highly environmentally relevant, as the occurrence of threatened fish species is directly linked to overexploitation caused by fishing.

4.4.2.2.4 The environmental relevance of VEC in terms of endpoint

a) the ability of the category indicators (VU, EN, CR) to reflect the consequences of the LCI results on the category endpoint is, in theory, related to potentially disappeared fraction of species (PDF) and damage to AoP natural Environment and/or AoP Natural Resources. This needs to be further elaborated on in the future.

b) in an LCA of capture fisheries utilizing VEC, spatial and temporal aspects should be discussed, if known. Ideally, the fishing operation is directly assessed, which in this case directly links the landed part to the discarded part. The reversibility of this impact is depending on discard mortality (which should be discussed) and possible recent trends in population estimates for VEC species identified.

4.4.2.3 Classification

During classification, the assignment of LCI results to VEC

a) is not exclusive to one impact category, as Biotic Resource Use in terms of Primary Production Required also uses fish species lists from discards derived from LCI.
b) if discards impacts are assessed in PPR, VEC and weight in kilos, these different impacts are based on the same LCI results. It could be seen as these different impacts are parallel mechanisms, all related to a combined impact on the endpoint marine ecosystem structure and function. PPR and weight of discards may be impacts more related to resource use, at an ecosystem level, whereas VEC is to a greater extent concerning sensitivity of the different species that are directly impacted, at a species level. Loosing key stone species could also affect the functioning of the ecosystem, which also make the approaches intertwined.

4.4.2.4 Characterization

The characterization model makes a distinction between different fisheries discarding different species, based on their relative level of threat. As the LCI result is fish species in weight or numbers, this gives a numerical result. The Red List assessment behind species threat status is well documented, including the assumptions used (www.iucnredlist.org).

Discard data is most often scarce, and if there are no records available at a species level, this should be stated in the goal and scope. Ideally, due to the importance of this impact, this collection of data should be performed during LCI, if there are no data available.

Geographical region could affect the accuracy of the impact category VEC, as coverage of Red List assessments of fish species differ between areas. Time of assessment could also vary. These possible influences on results should be tested or at least discussed in a sensitivity analysis.

4.4.2.5 Resulting data after characterization

Discard outputs of seafood production systems that are not covered by VEC or PPR should, if available, be included as LCI results. These outputs would comprise of by-catches of marine mammals, birds, reptiles and non-commercial invertebrates.