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Review

Life cycle assessment of fisheries: A review for fisheries scientists and managers

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ABSTRACT

This review aims to synthesise and discuss current literature applying the Life Cycle Assessment (LCA) framework for the environmental assessment of fisheries. The review introduces and illustrates the LCA framework, and highlights energy use by fishing vessels, among other key factors determining environmental impacts of fisheries operations. Moreover, the review concludes with recommendations on future developments of LCA in the fisheries and seafood sectors.

We reviewed 16 studies on LCA applied to fisheries, with perspectives from a few additional publications on closely related topics. The main Aspects considered in the ad hoc comparison of studies include: scope and system boundaries, functional units, allocation strategies for co-products, conventional and fishery-specific impact categories used, fuel use, impact assessment methods, level of detail in inventories, normalisation of results and sensitivity analyses.

A number of patterns and singularities were detected. Fishery-specific impact categories, despite not being standardised, and fuel use in fishing operations were identified as the main contributors to environmental impacts. Energy efficiency was found to be strongly related to the fishing gear used. Several studies discussed the impacts of antifouling substances and metals use. The need for standardisation of fisheries LCA research is justified and ideas on how to do so and what elements to standardise (fisheries-specific impact categories, inventory details, normalisation references, etc.) are discussed. Finally, fisheries LCA constitute a useful research field when studying the sustainability of seafood and fisheries-based agrifood, and it should likewise contribute to an ecosystem approach to fisheries.

Contents

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1. Introduction

Fisheries represent a primary industry and the starting point of supply chains of local, regional and global relevance. They play a key role in food security due to the rich protein (and often lipid) content of fish: seafood supply chains provided more than half of the world's population with at least 15% of their average animal protein intake as of 2010, and the output of key activities in those supply chains (capture and aquaculture) features a growing trend (SOFIA, 2012). The seafood industry generates over 180 million jobs worldwide, which represents the livelihood of 8% of the world's population (SOFIA, 2010). Moreover, seafood products represented about 10% of total agricultural exports (figure showing a growth trend), while fisheries and aquaculture (including shellfish) provided the world with 142 million tonnes of fish in 2008 (of which almost 20% was used for non-direct human consumption, e.g. for reduction into fishmeal and fish oil; SOFIA, 2010).

Conventional fishery research has, for a long time, focused mostly on individual stock assessment and management. Only in the last decade, in a limited number of countries, has research addressed the ecosystem approach to fisheries (EAF; FAO, 2003; reviews in Fréon et al., 2005a; Garcia and Cochrane, 2005; Plagányi, 2007). The need for understanding and limiting the ecosystem impacts of fisheries is evident in the principles of the EAF (FAO, 2003), and thus research on the environmental impacts of fisheries has expanded. However, it is nowadays mostly limited to on-site effects, including: removal of target species and non-target species, adverse effects on top-predator species populations (e.g. marine birds and mammals), changes in marine food webs and other alterations of ecosystem structure, and cumulative impacts on marine ecosystems related to the destruction of benthic communities and substrates due to certain fishing practices (e.g. bottom trawling). These impacts have been discussed at different levels. For instance, they have been compiled and described in the *FAO guideline for Ecosystem Approach to Fisheries* (FAO, 2003), analysed in great detail in the *Handbook of Fish Biology and Fisheries* (Reynolds and Hart, 2002) and discussed within the context of sustainability (Smith et al., 2010). Nonetheless these direct effects are seldom considered within the context of an integrated life cycle approach. Moreover, the indirect and off-site effects of fishing activities have been largely ignored until only recently.

Environmental impacts resulting indirectly from fishing operations are mostly associated with the extraction and transformation of natural materials and fossil fuels used for the construction, use and maintenance of fishing units. These indirect and often global—or at least large scale impacts—include: emissions related to fuel combustion, release of antifouling substances, use of cooling agents, provision and loss of fishing gear, further transportation, wastewater and waste discharge, release of cleaning agents and refrigerant gases, etc.; as discussed in Ziegler et al. (2003), Thrane (2004a), Hospido and Tyedmers (2005) and Cappell et al. (2007).

Life Cycle Assessment (LCA) is a widespread framework for environmental assessment of food systems, including fisheries. It benefits from an International Organisation for Standardisation (ISO) standard—the ISO 14040 series—and a large body of theoretical and methodological research. LCA is one of the approaches developed to address the increasing concerns regarding environmental impacts inherent in the provision of products and services, and the need to understand and minimise these impacts. LCA allows for comprehensive evaluations to be made on the environmental impacts related to products over their whole life cycle, that is to say, encompassing infrastructure, energy provision, extraction of raw materials, manufacturing (cradle-to-gate), distribution, use and final disposal (cradle-to-grave) (ISO, 2006b). Nonetheless, in practice, all life cycle stages of a product are not always addressed in LCA studies due to data restrictions or to the goal of the study. LCA is thus a tool aimed to, among other purposes, identify opportunities for improving environmental performance and inform decision makers on the environmental performance of products, product systems and even their alternatives (ISO, 2006a). It can moreover assist in selecting environmental performance indicators (e.g. for sustainability assessment) and be used for marketing purposes (ISO, 2006b). Marketing claims based on LCA could reduce the risk of it being perceived as biased, i.e. "green washing" (Horiuchi et al., 2009).

LCA applied to food systems and agricultural production dates at least from the mid 1990s, but has been applied to aquaculture and fisheries research only in the last decade (Fig. 1). Early seafood LCA studies found valuable information on previous research such as energy analyses of fleets and seafood products, for instance as in Tyedmers (2001) and Thrane (2004a). Energy analyses are relevant in relation to fisheries LCA due to the accepted importance of fuel consumption for fleet operations (Tyedmers, 2001) and associated environmental impacts (Thrane, 2004a; Schau et al., 2009; Driscoll and Tyedmers, 2010). Carbon footprint (CF), often considered as a sub-set of LCA (EC/JRC, 2007), is closely associated to fisheries LCA due to the strong impact of fuel consumption on the single impact category considered by CF: global warming. Pioneering studies on LCA and CF applied to fisheries include Eyjólfsdóttir et al. (2003), Ziegler et al. (2003), Thrane (2004a) and Hospido and Tyedmers (2005).

This review mainly aims to illustrate the LCA framework by discussing its application to fisheries research in order to bridge the gap between the conventional fisheries scientist community and the LCA one, and more broadly the Industrial Ecology and environmental management communities. Furthermore, it discusses literature on environmental assessment of fisheries based on LCA and energy analyses of fishing vessels and fleets, in order to identify challenges in fisheries LCA research. This work complements recently published reviews on the use of LCA in fisheries and seafood research, namely Vázquez-Rowe et al. (2012c) and Parker (2012).

2. Material and methods

We reviewed a number of studies, mostly LCAs of fishing vessels and fleets, and identified patterns and discrepancies. The pertinent

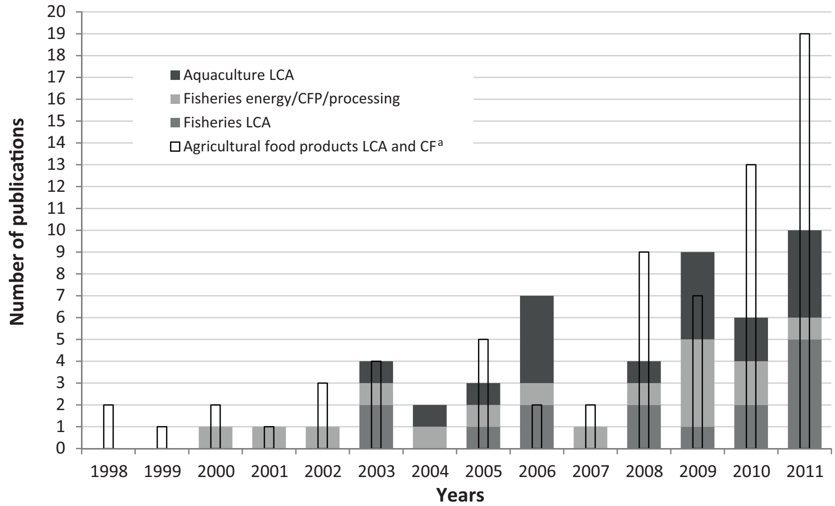

Fig. 1. Histogram of published LCA studies in selected areas from 1998 to 2011. (a) Dominated by dairy and meat products (excluding bio-energy studies). LCA: Life Cycle Assessment; CF: Carbon Footprint.

ISO standard was used as a comparison/analysis structure (ISO, 2006a,b).

The reviewed studies here were found by web searches in environmental assessment and fisheries research journals, and citations in leading fisheries LCA publications. Peer-reviewed literature on fisheries LCA is limited, thus all available studies were included, plus a few additional works focusing on energy aspects and CF of fishing operations: 16 studies on LCA applied to fisheries, two studies focused on energy aspects of national fishing operations (e.g. fuel-per-landed fish ratios) and one CF of a national fishing fleet; as listed in Table 1.

This review focuses on extraction activities and therefore excludes seafood processing (except when it occurs onboard). One of the LCA studies also features extensive energy analysis of various fishing fleets. Additional studies based on the same datasets as these 19 studies are also quoted in various sections of this review. Further studies were identified in the form of master theses, but were excluded to rely almost exclusively on peer-reviewed publications. Two very representative and cited doctoral theses were also included: an energy and ecological footprint analysis (Tyedmers, 2001) and a very detailed LCA (Thrane, 2004a). Theses feature well recognised contribution to fisheries research in a life cycle context and provide supplementary information on primary literature articles by the same authors, also reviewed here. Work in progress by the authors Fréon et al. (in prep.), soon to be submitted for publication, has also been cited in this review. The abovementioned study supports several positions and recommendations expressed in this review, as for instance, the relevance of the construction phase of fishing vessels (often considered as irrelevant in literature) and our contribution to the discussion of co-product allocation in fisheries.

All LCA studies reviewed were dissected using the four LCA stages defined by the ISO standard (Fig. 2): goal and scope definition, life cycle inventory (LCI) analysis, life cycle impact assessment (LCIA) and interpretation (ISO, 2006a). It is worth noting that LCA studies require a critical review process if the results are to be publically disclosed (ISO, 2006b; Klöpffer, 2012). The LCA phases will be explained in more detail and illustrated with examples from fisheries research over the following section, while conclusions drawn on both the state of the art and the future of fisheries LCA are discussed in the last section.

3. Results and discussion

3.1. Goal and scope definition

3.1.1. Goal and scope

The goal and scope definition stage of LCA consists in the design of the study according to its objective. The goal and scope describe a series of methodological decisions made. Such methodological decisions determine assumptions and effort intensity of subsequent stages.

The ISO standard states that goal and scope of LCA studies are to be clearly defined at the beginning, in such a way that they are consistent with the intended application of the study. In reality, both goal and scope are often refined, or even redefined, during the subsequent phases of an LCA, hence the double arrows in Fig. 1. The goal must declare the intended application and audience of the study, while the scope must include the following elements detailed below: the system boundary, the functional unit and its associated reference flow(s) within the system, the allocation strategy, data requirements and other relevant design and implementation decisions (ISO, 2006b).

The goals of reviewed studies were generally clearly stated, and were mainly centred on assessing environmental performance of fisheries, often focusing on the identification of hotspots and/or the comparison of alternative fishing methods, and identifying opportunities to improve that performance. All studies analysed fuel-related impacts, and several also analysed the use of metals.

3.1.2. System boundaries

The system boundaries delimit the studied system by means of including and excluding unit processes. Boundary definition is key to delimit the scope of the study and to be able to compare different LCAs in time or space. The decision on which processes to include within the system boundary should be based on clearly stated and well justified cut-off criteria, including criteria such as mass, energy or environmental significance (ISO, 2006b). Nonetheless, those criteria are not always applied (Suh et al., 2004).

The reviewed studies featured a variety of system boundary definitions. In general terms, four life cycle stages are recognised in fisheries LCA: construction, use, maintenance and end of life (EOL), though stage names vary according to different authors.

Table 1
Major features of reviewed fisheries LCA studies, including some non-LCA complementary studies. Studies are alphabetically ordered. Fuel use data used in (3) is published in Ziegler and Hansson (2003). Fuel use and other data used in (8) are published in Ziegler et al. (2009). Fuel use and other data used in (7) are published in (4). Fuel use data used in (13) was published in (18).

No	Authors	Targeted species	Fishery gear	Fishing region	No. of years	No. of vessels	Capture Construction	Use	Maintenance	EOL	Transport	Processing Construction	Use	Maintenance	EOL	Transport	Functional unit	LCA type, justification and allocation	Sensitivity analysis
1	Tyedmers (2001) [a]	*Codfish, Small pelagic fish, Tuna, Shrimps & prawns, Lobster & crab*	*Trawling purse seining trapping*	*Northeast Atlantic*	*3*	*186*		X	X		X						*N/A*	*N/A*	*N/A*
2	Eyjólfsdóttir et al. (2003)	Cod	Trawling	Northeast Atlantic	1	25		X	X		X		X	X		X	9 kg frozen fillet	ALCA (descriptive), mass	No
3	Ziegler et al. (2003)	Cod	Trawling gillnetting	Northeast Atlantic	1	X		X	X		X		X	X		X	400 g frozen fillet	ALCA (descriptive), economic	No
4	Thrane (2004a) [a]	Codfish (various), Norway lobster, Northern prawn, Shrimp, Herring, Mackerel, Industrial fish (e.g. Tobis)	Trawling purse seining	Northeast Atlantic	1–2	330		X			X		X			X	1 kg frozen fillet	CLCA (stated by author), system expansion	Yes, product substitution, Ecoindicator 99 vs EDIP
5	Hospido and Tyedmers (2005)	Skipjack Yellowfin tuna	Purse seining	Atlantic, Pacific, Indian oceans	10	9	X	X	X								1 t frozen fish	ALCA (descriptive), avoided	Yes, allowable emissions from ships
6	Ellingsen and Aanondsen (2006)	Cod	Trawling purse seining	Northeast Atlantic	1	X		X	X		X		X			X	200 g fillet	ALCA (descriptive), mass & economic	Yes, Ecoindicator 95 vs EDIP
7	Thrane (2006)	Flatfish	Trawling	Northeast Atlantic	1	330		X			X		X			X	1 kg frozen fillet	CLCA (stated by author), system expansion	Yes, product substitution, Ecoindicator 99 vs EDIP
8	Emanuelsson et al. (2008)	Southern pink shrimp	Trawling artisanal trawing	Eastern Central Atlantic	2	19		X	X		X						1 kg frozen packed shrimps	ALCA (descriptive), economic	Yes, 8 different criteria
9	Ziegler and Valentinsson (2008)	Norway lobster	Creeling trawling	Northeast Atlantic	2	19		X			X		X			X	1 kg landed lobster	ALCA (descriptive), economic	Yes, Ecoindicator 99 vs CML
10	Guttormsdóttir (2009)	Cod	Trawling long lining	Northeast Atlantic	3	2		X	X		X		X	X		X	1 kg of frozen light salted fillets	ALCA (descriptive), mass	Yes, elimination of fossil fuels

Table 1 (*Continued*)

No	Authors	Targeted species	Fishery gear	Fishing region	No. of years	No. of vessels	Capture Construction	Use	Maintenance	EOL	Transport	Processing Construction	Use	Maintenance	EOL	Transport	Functional unit	LCA type, justification and allocation	Sensitivity analysis
11	Driscoll and Tyedmers (2010) [a]	*Atlantic herring*	*Trawling purse seining*	*Northwest Atlantic*	*12*	*364*[b]		X									*N/A*	*N/A*	*N/A*
12	Iribarren et al. (2010) [a]	*European hake, Atlantic horse mackerel, European pilchard, Anglerfish, Tuna*	*Trawling long lining purse seining*	*Atlantic, Pacific, Indian oceans*	*1*	*84*		X	X								*1 t landed fish*	*ALCA (descriptive), economic*	*No*
13	Vázquez-Rowe et al. (2010a)	European hake, Atlantic horse mackerel, Atlantic mackerel, Blue whiting	Trawling	Northeast Atlantic	1	24	X	X	X								1 kg landed fish	ALCA (mgmt-policy dimension), not discussed	Data Envelopment Analysis
14	Vázquez-Rowe et al. (2012a)	Common octopus	Trawling	Eastern Central Atlantic	1	8	X	X	X		X						1 t landed fish	ALCA (predictive scenarios), mass & economic	No
15	Ramos et al. (2011)	Atlantic mackerel	Purse Seining	Northeast Atlantic	8	27–45	X	X	X								1 t landed fish	ALCA (descriptive), timeframes	No
16	Svanes et al. (2011a)	Cod	Long Lining	Northeast Atlantic	1	10		X			X		X			X	1 kg product	ALCA (descriptive), mass & economic	Yes, fuel use
17	Vázquez-Rowe et al. (2011)	European hake	Trawling Long Lining	Northeast Atlantic	1	21	X	X	X		X						500 g fillet	ALCA (mgmt-policy dimension), mass	No
18	Vázquez-Rowe et al. (2010b)	Atlantic horse mackerel	Trawling Purse Seining	Northeast Atlantic	1	54	X	X	X								24 kg carton frozen octopus	ALCA (descriptive), mass	No
19	Fréon et al. (in prep.)	Anchoveta	Purse Seining	Southeast Pacific	6	20–400	X	X	X	X							1 t landed fish	ALCA (predictive scenarios), avoided	Yes

[a] Lines in italics are studies which do not present full or exclusively LCA results: (12) is a carbon footprint study, (4) features both LCA and energy analyses, (1) and (11) are energy analyses.
[b] Number of observations refers to number of vessels surveyed, except in (11), where trips are sampled.

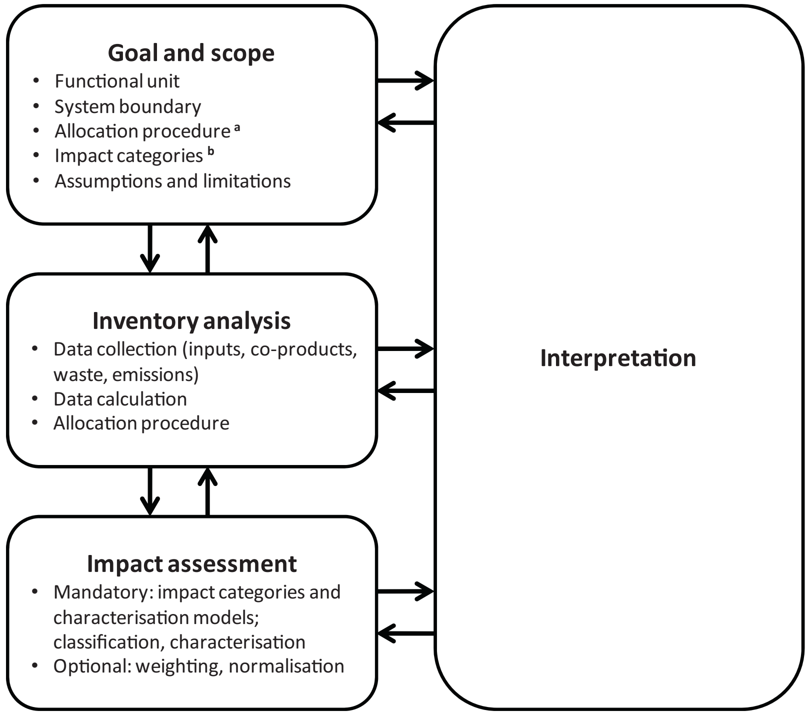

Fig. 2. Stages in LCA (ISO, 2006a,b). (a) In the ISO standard, and in this review, the allocation procedure is introduced in Goal and scope and detailed in Inventory analysis. (b) Impact categories are part of both Goal and scope and Impact assessment. In this review, the discussion on impact categories was carried out in the Impact assessment section.

Most studies encompassed two stages only: the vessel use and maintenance phases of fishing operations (Table 1). A few among the studies included the construction or at least production of materials for construction, end of life phases and pre-fishing activities such as production of diesel and antifouling paints (e.g. Hospido and Tyedmers, 2005; Fréon et al., in prep.). Nonetheless most studies excluded the construction phase (capital goods) deeming its contribution to environmental impacts as negligible (e.g. Ziegler et al., 2003). Most of the studies included the transportation activities related to landing and delivery to places of transformation/processing when necessary, while some also include processing operations clearly separated from the extraction phase (which exceeds the scope of this review). Studies following in full or in part the consequential approach to LCA (see Section 3.1.5) included avoided products and alternative exploitation scenarios (Thrane, 2004a). Others reviewed different management/policy elements such as predictive scenarios (Vázquez-Rowe et al., 2010a, 2011; Fréon et al., in prep.).

The environmental impact of fisheries research, fishery administration (from the fishing companies and the government), surveillance and control, stock assessment, among other non-fishing but complementary fisheries-related activities can be representative in high value fisheries that do not require many fishing vessels. In our opinion, they could, when relevant, be included within the system boundaries of LCA endeavours (at least as a screening) and allocated among seafood products in a coherent way, subject to justification and discussion. Aspects to be considered would be limited to infrastructure (e.g. vessels) and energy consumption (fuel, electricity).

3.1.3. Functional unit

The functional unit (FU) is a numerical representation of the function(s) provided by the studied system. The FU is thus the reference unit that quantifies the performance of a product system and defines a reference flow (measure of the outputs from the system required to fulfil the function defined by the FU) as a systems comparison device (ISO, 2006b). It is thus a representation of the function delivered by the studied system, which can be used to compare it with alternative systems delivering the same function. The functional unit often measures only the primary function of the product system under study. To overcome such limitation, it has been suggested that an FU definition should include not only the magnitude of the service (e.g. 1 kg of product, 1 unit of product) but also temporal and quality constraints (Cooper, 2003). For instance, a partial FU would be "1 kg of Peruvian anchovy", while a comprehensive one would be "1 kg of Peruvian anchovy, with canning quality, landed on a non-El Niño year".

Functional units chosen in the reviewed studies were heterogeneous, ranging from serving or retailing units (e.g. seafood portions) to distribution units (1 kg or 1 tonne of fresh fish, frozen fish or seafood product). Occasionally, packaging material was included in the functional unit (Table 1).

3.1.4. Allocation

Allocation is the process of dividing inputs, outputs and associated impacts among several products (co-products) produced in the same process, or one product supplying several processes (ISO, 2006b). The need to perform such allocation arises in multi-functional systems. In fisheries, the need for allocation arises, for instance, when fishing fleets land by-catch or target multiple species, or when fishing vessels feature both canning and fishmeal factories on board, among other situations. In the Life Cycle Inventory (LCI) phase described later on, allocation strategies used in the reviewed studies, as well as described in other LCA literature, are discussed.

3.1.5. Implications of the attributional and consequential approaches

In LCA literature, there are two main currents or schools of thought regarding the purpose, scope, system boundaries and philosophy of specific studies: Attributional LCA (ALCA) and Consequential LCA (CLCA). The latter has been increasingly used by researchers, yet the approach has not to date been systematised (Zamagni et al., 2012). There is an ongoing debate regarding the pros and cons of each approach, and on when, how and why to perform ALCA or CLCA (Baitz et al., 2012). One of the main conceptual differences between the two approaches is that ALCA describes a given (usually retrospective or present) situation which does not deal with the indirect effects of changing markets, while CLCA attempts to predict future changes of environmental impacts and product flows as indirect consequences of market-mediated choices made within the system boundaries (Weidema, 2003; Brander et al., 2008; Earles and Halog, 2011). In other words, ALCA is descriptive and CLCA is predictive/prospective (Weidema, 2003; Finnveden and Moberg, 2005; Brander et al., 2008; Thomassen et al., 2008; Zamagni et al., 2012).

A very clear feature of CLCA studies is the modelling of substituted systems rather than the actual system under study. An example of the former could be the fishmeal/fish oil process in the Danish LCA Food database (www.lcafood.dk/), which features the substitution of fish oil with rapeseed oil.

Under the consequential and attributional philosophies, different allocation strategies are used. CLCA prioritises system expansion while ALCA commonly applies mass/economic allocation, although system expansion is also applicable within ALCA (see Section 3.2.2). In fisheries context this is illustrated in Thrane (2004a). A simple definition of both approaches, from the perspective of the system delimitation, states that "The consequential approach uses marginal data and avoids co-product allocation by system expansion. The attributional approach uses average or supplier-specific data and treats co-product allocation by applying allocation factors" (Schmidt, 2008).

Some CLCA practitioners defend the use of CLCA in political decision contexts due to its market-based system delimitation. For instance, Thrane (2004a) has argued that the focus of CLCA relies on "hot-spots and improvement potentials regarding production processes rather than environmental consequences of product substitution". In a fisheries context, as illustrated by the reviewed studies, it is almost never clearly indicated to which school (ALCA or CLCA) a specific study belongs to. Moreover, as LCA literature suggests (e.g. Schmidt, 2008; Finnveden et al., 2009; Suh et al., 2010), there is a grey scale between pure attributional and consequential analyses. The reviewed studies were similarly found to occasionally feature elements of both approaches (Table 1), a common case in LCA in general. Those displaying features of the consequential approach addressed substituted products and future exploitation scenarios (Thrane, 2004a), substitutes (Ellingsen and Aanondsen, 2006) or competing present or future technologies (Emanuelsson et al., 2008; Ziegler and Valentinsson, 2008; Guttormsdóttir, 2009).

Seafood LCA studies should clearly state whether consequential elements of analysis are considered. Seafood or agrifood supply chains associated to a fishery influence and could determine systemic (market and policy-based) changes in the fishery, and vice versa. For instance, in the case of the globally important Peruvian *anchoveta* fleet, we observed that the bargaining power of major vertically-integrated fishing/processing companies seems to influence purchase prices of fish landed for reduction by independent fishermen. Moreover, the operational strategy of the fleet seems to be related to the exploitation regime dictated by the government (e.g. introduction of Total Allowable Catch system in 2009), as well as to other economic and policy drivers. Thus, CLCA studies could be used to elaborate scenarios featuring demand and policy changes. We consider that the main criteria to decide whether to carry out an LCA following the attributional or consequential philosophy should depend on the intention of the study (descriptive, predictive/prospective), its intended use of market mechanisms and attention to indirect effects.

3.1.6. Impact categories

Impact categories selected for an LCA study reflect the environmental issues associated to the product system under study, as well as the goal and scope (ISO, 2006b). In the Life Cycle Impact Assessment (LCIA) phase described later on, inventory flows (e.g. methane or nitrogen oxides) are converted using characterisation factors and compiled into LCIA categories (e.g. global warming, eutrophication, acidification) by using sets of rules. See Supplementary Material for details on the LCA impact categories proposed by the major LCIA methods and the distinction between midpoint and endpoint categories.

3.2. Life cycle inventory (LCI)

Life cycle inventories are compiled by collecting data on environmental inputs and outputs belonging to each unit process within the system boundaries. Such data should describe, both quantitatively and qualitatively, material and energy inputs and outputs, as well as releases to air, soil and water (ISO, 2006b). LCI data is compiled and often communicated in relation to the reference flow (e.g. 1 tonne of fish). Following the inventory compilation, allocation of resources and emissions among co-products is performed when necessary.

3.2.1. Inventory

Unit process data describe the inputs and outputs at process level. Today, LCI databases can provide many of the supporting unit processes used by fisheries LCAs. The most commonly used database, *ecoinvent* (www.ecoinvent.org), includes unit processes on energy (electricity, fuels), transportation, building materials, biomass, wood and fibres, metals, chemicals, electronics, mechanical engineering, paper and pulp, plastics, waste treatment and agricultural products (Frischknecht et al., 2007). Such datasets are commonly used by LCA practitioners for background processes to their study system, as well as proxies for processes for which data are not available. There is a trade-off between accuracy of the model and resources invested in its preparation: the use of *ecoinvent* and other third-party LCI databases processes facilitates modelling inputs and outputs, but at the expense of accuracy (given that most unit processes to be modelled display spatial and temporal variation). The reviewed studies featured inventory data collected from fishing fleets, local fishers and local fishing companies, supplies and equipment providers for fishing operations, government statistics, reports and previous publications. Collection of primary data was performed mainly by means of interviews or questionnaires sent to skippers and companies (sample sizes and timeframes are detailed in Table 1). *ecoinvent* was used when other primary or system-specific data were not available, and to populate background processes (e.g. provision of fossil fuels and chemicals).

The number of inventory items included varied amongst the reviewed studies, as well as the detail of their chemical composition (e.g. metal used in engines and onboard installations). Selection of inventory items was inconsistent except for fuel used in fishing vessels, as shown in Table 2. Levels of detail of data collected for LCIs of fisheries appear highly heterogeneous, from narratives included in the reviewed LCA studies, and often briefly documented.

We suggest a more detailed inventory of the construction phase than currently practiced (i.e. different classes of steel, because their relative impacts largely differ) should be performed, unless irrelevant within the chosen goal and scope. This suggestion is based on the fact that the impacts of the construction phase have been found to be important in some studies and reviews, i.e. Svanes et al. (2011a), Vázquez-Rowe et al. (2012c) and Fréon et al. (in

Table 2
Detail level of inventories used in published fisheries LCA studies. X = considered, blank space = excluded, N/A = not applicable.

		Studies																		
Category	Inventory items	Tyedmers (2001) [a]	Eyjólfsdóttir et al. (2003)	Ziegler et al. (2003)	Thrane (2004a) [a]	Hospido and Tyedmers (2005)	Ellingsen and Aanondsen (2006)	Thrane (2006)	Emanuelsson et al. (2008)	Ziegler and Valentinsson (2008)	Guttormsdóttir (2009)	Driscoll and Tyedmers (2010) [a]	Iribarren et al. (2010) [a]	Vázquez-Rowe et al. (2010a)	Vázquez-Rowe et al. (2010b)	Ramos et al. (2011)	Svanes et al. (2011a)	Vázquez-Rowe et al. (2011)	Vázquez-Rowe et al. (2012a)	Fréon et al. (in prep.)
Fishing unit	Steel					X								X	X	X		X	X	X
	Engine					X														X
	Wood					N/A									X					X
Operations	Diesel	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
	Antifouling and paint		X	X	X	X	X	X	X	X	X		X	X	X	X	X	X	X	X
	Lubricating oil					X								X	X	X		X	X	X
	Refrigerants		X		X			X	X, X[b]		X		X [c]		X	X	X		X	N/A
	Ice		N/A	X	X	N/A		X	Xb		N/A			X	X	X	X[b]	X	N/A	N/A
	Grid energy		X[b]	N/A	X[b]	X		X[b]	X[b]	X[b]			X	N/A	N/A		X[b]	X[b]	X[b]	N/A
	Packaging		X[b]	X	X[b]	N/A		X[b]	X[b]	X[b]	X[b]			N/A	N/A		X[b]	X[b]	X[b]	N/A
	Bait		N/A	N/A	N/A	N/A	N/A	N/A	N/A	X	N/A			N/A	N/A		X	X	N/A	N/A
Gear	Net		X	X	X			X		X	X		X	X	X	X	X	X	X	X
	Hooks and lines		N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A		X	N/A	N/A	N/A	X		N/A	N/A

[a] Lines in italics are studies which do not present full or exclusively LCA results: (12) is a carbon footprint study, (4) features both LCA and energy analyses, (1) and (11) are energy analyses.
[b] Materials used for a separate processing stage, after landing.
[c] Iribarren et al. (2010) features a follow-up study (Iribarren et al., 2011) which includes refrigerants in the inventory.

prep.). Furthermore, certain behaviours leading to further emissions to water could be considered when relevant, depending upon the impact categories that are in focus. For instance, items such as solid waste, wastewater and used lubricating oil wasted at sea should be considered in detail when the fishery under study is associated with fishing grounds where they could accumulate or reach the shore, particularly in countries/regions where such waste is common practice. They should also be considered when relevant impact categories, such as eco-toxicity, are accounted for. Most of the reviewed studies are far too reliant on the use of third-party LCI database without considering to what extent such processes accurately represent the conditions of the supply chains they are modelling.

The exploitation status of the stocks (target and non-target species) and the type of marine ecosystem impacted (e.g. levels of biodiversity, productivity, global scarcity) could be indicated when available and trustable, in order to qualitatively or quantitatively weight the impact of species removal.

3.2.2. *Allocation strategies*

The selection of an allocation strategy, is one of the most difficult and controversial methodological aspects of LCA studies, and often greatly influences the results (Weidema, 2000; Guinée et al., 2001; Ayer et al., 2007; Suh et al., 2010; Peacock et al., 2011; Pelletier and Tyedmers, 2011; Svanes et al., 2011b). Allocation problems have been discussed and contextualised in detail by several authors (e.g. Weidema, 2000; Ekvall and Finnveden, 2001; Curran, 2007; Reap et al., 2008). The approach for allocation recommended by the ISO standard (ISO, 2006b) suggests a hierarchy of steps to address allocation problems, in the following order:

1. Avoidance of allocation when possible by means of a) subdivision: dividing the multifunction process into independent sub-processes that can be assigned to individual co-products, or b) system expansion: the product system is "expanded" to include the functions associated to the co-products, that is to say, the system boundaries are expanded to include the whole subsystem of co-products;
2. Allocation based on a physical relationship (e.g. mass or energy content); and
3. Allocation based on other non-physical relationship (e.g. economic value).

A common, yet non-strictly ISO interpretation of the system expansion approach is known as "substitution", and consists in modelling the processes associated to the avoided production of co-products, considering them as alternatives to other products on the global market. The system expansion/substitution approaches can be very complex, its application is not shared by many attributional analysts, and have been profusely discussed in LCA literature (e.g. Weidema, 2000; Ekvall and Finnveden, 2001; Suh et al., 2010; Weidema and Schmidt, 2010). Moreover, certain authors consider avoiding allocation by means of system expansion, or allocating based on a physical relationship, is always possible and thus reject the use of economic allocation. Nonetheless economic allocation is widely practiced in many fields of LCA application.

In the context of fisheries, the strongest influence of allocation arises from landed by-catch, not necessarily targeted by separate fisheries and thus unsuitable for allocation avoidance; and secondary co-products (by-products) from seafood processing (review in Ayer et al., 2007). The study of multi-species fisheries also poses important allocation challenges (Schau et al., 2009).

Subdivision is rarely attempted in fisheries LCA literature, because processes for multi-species fisheries, by-catch and seafood by-products often cannot be isolated and fully accounted for (Ayer et al., 2007; Schau et al., 2009). In the other hand, system expansion/substitution would be always possible, since fisheries are commonly destined for delivering protein and thus their products can always, in theory, be substituted by another fishery or non-fishery animal protein source. A notable example is the analysis in Thrane (2004a), where the fuel consumption per landed kg of fish of several (or most) Danish fishing operations was calculated and contrasted by applying mass allocation, economic allocation and system expansion. Each by-catch species was addressed separately by assessing additional fleets targeting these species (also landing by-catch) and summarising their contribution to landings of each species (target and by-catch).

A number of approaches have been suggested when subdivision/system expansion is not possible or impractical:

- Ayer et al. (2007) propose gross energy content for LCA at different stages of seafood (and in general food) products —including all food co-products—, suggesting that such an approach more realistically reflects flows of food co-products occurring within and outside production systems. See Pelletier and Tyedmers (2011) for more details.
- Suh et al. (2010) suggest allocation problems in general to be handled as numerical problems under an input–output economic approach, specifically the supply-use framework. It would be challenging to apply this approach to fisheries, because it requires often unavailable data at country or regional scales. None of the reviewed studies applies input–output analysis in combination with LCA, a novel research field often aiming to broaden and deepen LCA (Suh et al., 2004; Finnveden et al., 2009; Jeswani et al., 2010).
- Svanes et al. (2011b) describe a hybrid allocation approach combining mass and economic allocation, and the use of global functional units where all products are included within the same FU. Hospido and Tyedmers (2005) made use of a global functional unit by considering various target species within their FU. Nonetheless this is practical only if the proportion of co-products is constant over time and space and when the goal of the study is to improve environmental performance of the fishing stage in general. This global functional unit can be understood as system expansion.
- Schau and Fet (2008) propose the use of quality-corrected functional units (QCFUs), for food products, including seafood. A QCFU incorporates in the definition of the FU nutritional features of the product (i.e. yield, lipids, protein and carbohydrates, the basis for gross energy content computation). The author suggests QCFUs can be used as a basis for allocation, or may even overcome the need for co-product allocation at all.

Pelletier and Tyedmers (2011) understand LCA as a bio-physical accounting framework, and therefore state it should rely on bio-physically-driven relationships, not market ones. Therefore they suggest market information should be avoided in life cycle modelling, due to its sourcing on the current neoclassical economic system, which patently fails to account for the value of ecosystem services and limits to growth as opposed to the continuous (eco-efficient or not) growth paradigm. Instead they defend the use of bio-physical drivers such as gross energy content for addressing issues such as allocation in seafood systems, based on the assumption that the ultimate driver behind food production is the provision of food energy. In contrast, a recent publication (Ardente and Cellura, 2012) revisits economic allocation (the last alternative according to the ISO standard) as a very suitable approach in several situations. Both perspectives nonetheless conclude that there is no "best" allocation method or allocation decision rule, but the allocation procedure/strategy has to be established on a case-by-case basis.

Amongst the reviewed, few studies discussed allocation challenges and addressed the selection/development of the best allocation strategy for the studied system. Occasionally, allocation between targeted catches and by-catch was not necessary due to the nature of the targeted fish stock (e.g. Hospido and Tyedmers, 2005; Fréon et al., in prep.). Further allocations beyond catch and co-products are not explicitly mentioned in the reviewed studies.

Given that subdivision and system expansion (the recommended allocation avoidance approaches according to the ISO standard) are not always practical, we stand for the contrasting application of at least two allocation methods in LCA studies, as practiced in many of the reviewed studies and promoted by ISO 14040 (e.g. Thrane, 2004a; Ellingsen and Aanondsen, 2006; Vázquez-Rowe et al., 2010b; Svanes et al., 2011a). The choice of the allocation methods should be aligned with the goal and scope of the study and data availability.

We suggest an approach for the specific case of multi-species finfish fisheries, where three main situations can be identified: (1) one or several high-value target species and one or several edible by-catch species of lower commercial value; (2) one or several high-value target species and one or several non-edible by-catch species; and (3) one or several abundant low-value target species and one or several high-value target or non-target species. In our opinion, regarding cases (1) and (2), if the low-value species are discarded at sea, obviously the direct environmental impacts associated to their mortality may simply be fully attributed to the landed species, using a mortality rate of discard lower than 100% when necessary in order to reflect the proportion of discard survival (if relevant). If these low-value species are mainly used for direct human consumption, we suggest using a mass allocation if the energy-content (and/or protein content) of all species are equivalent. If not, an energy or protein content-based allocation should be preferred, as economic allocation could underestimate the impact of the by-catch species compared to the target one. The situation is more complex if the low-value species are aimed at reduction into fishmeal and fish oil on land, and even more if reduction occurs onboard as possibly in case (2) or (3), because of increased complexity for subdivision. In both cases mass allocation is not appropriate because environmental impacts differ largely according to the fate of the fish. Subdivision is not always possible because disentangling the processes related to each species appears not always practical (common sub-processes, detailed data required).

According to Pelletier and Tyedmers (2011) and Ardente and Cellura (2012), practical issues should guide the choice between alternative methods (system expansion, economic or energy-content allocation), a recommendation that seems consensual in LCA practice (EC/JRC, 2010) and shared by us. Moreover, consistency with methodological principles and the internal consistency of the resulting model and model outputs should guide the choice between alternative methods. Nonetheless, according to the ISO standard, system expansion would be preferable—when possible—to allocation (ISO, 2006b: Section 3.2.2), despite the fact that resource or data constraints might render following such a path impractical in particular cases.

3.3. Life cycle impact assessment (LCIA)

The LCIA phase (optional according to the ISO standard) consists of classifying and assigning characterisation factors to the LCI results, for the selected impact categories (ISO, 2006b). In such a way, the diverse LCI results can be more easily expressed as a reduced number of environmental indicators.

LCIA methods are usually applied by means of dedicated LCA software. However, it is equally possible to implement LCIA methods in a self-made spreadsheet or even by means of proprietary scripts. All the reviewed studies used SimaPro (http://www.

pre-sustainability.com/content/simapro-lca-software), the most widely used LCA software application. LCIA methods used where CML (Center of Environmental Science of Leiden University; Guinée et al., 2001), EDIP (Environmental Design of Industrial Products, DK LCA Center; Wenzel et al., 1997) and Ecoindicator 99 (PRé Consultants; Hischier et al., 2010; Huppes and van Oers, 2011).

In the three following subsections we first indicate the impact categories most commonly used in the reviewed studies, their classification (assignation) and characterisation, and finally two optional steps: normalisation and weighting.

3.3.1. Impact categories in LCIA

The reviewed studies focused on typical LCA impact categories: global warming, acidification, eutrophication, ozone layer depletion and aquatic/marine/terrestrial eco-toxicity; and dealt mainly with indirect/off-site impacts (see Table 3 and Supplementary Material). Only a few of the studies addressed CED, at various levels of detail, and identified fuel used for fishing operations as the larger contributor to energy consumption in fishing operations. On this regard, Thrane (2004a, 2006) and Schau et al. (2009) analysed energy consumption as a function of both fish species and fishing gear.

A vast majority of the reviewed studies also discussed some fishery-specific impacts aimed to account for direct/on-site impacts, namely discards, by-catch and seafloor disturbance impacts (see Supplementary Material). These specific impacts were commonly assessed outside the LCA methodology, and often in a qualitative way. A notable exception is Emanuelsson et al. (2008) and related approaches proposed by Ziegler et al. (2009, 2011) and Vázquez-Rowe et al. (2012b), which quantitatively analyse discard data. Other impacts were occasionally addressed: undersized catch, idle and ghost fishing gear, and marine pollution.

Species removal and seafloor impacts were sporadically accounted for in the reviewed studies, under novel impact categories related to food systems (e.g. Biotic Resource Use) and fisheries-specific (e.g. seafloor disturbance indices). These effects can be detailed as: removal of target and non-target species, unintended mortality of non-removed species, physical damage to habitat (in particular for benthic habitats), alteration of trophic dynamics and reduction in genetic diversity; all those elements not being directly accounted for in a specific LCA indicator.

It seems particularly difficult to account for some of these effects, thus various initiatives have addressed such need in fisheries, aquaculture (not considered in this study) and environmental assessment literature:

- Biotic Resource Use (BRU), based on the carbon content of crop inputs and trophic levels/transfer efficiencies of fish inputs (Pauly and Christensen, 1995), is widely applied and seems a good candidate for standardisation, as proposed by Libralato et al. (2008) and Langlois et al. (2011, 2012), although in different ways. The BRU concept and its equation for exploited fish resources, BRU = catches/9(̂Trophic Level − 1), are widely accepted, yet they rely on fundamental assumptions that might be challenged by fish scientists: a 9:1 ratio of fish wet weight to carbon and a 10% transfer efficiency per trophic level. For instance compiled estimates of transfer efficiency by type of ecosystems show variations ranging from 5 to 14 (Libralato et al., 2008), which are likely to reflect mainly fish species variability. Additionally, BRU is extremely sensitive to the estimation of the species trophic level, which varies with ontogeny.
- Efforts to quantify BRU include estimates of the Primary Production (PP) appropriated by the harvested biomass. According to various authors, this quantity is called PP required (PPR), net PP (NPP) or net PP used (NPPU) (Pauly and Christensen, 1995; Cappell et al., 2007; Pelletier et al., 2007; Hornborg et al., 2012) although it is not always clear if net of gross PP is used. This impact category allows comparing diverse food systems, including terrestrial ones. A recent publication proposes a specific discard assessment indicator, the Global Discard Index (also based on PPR), to be included in fisheries LCAs (Vázquez-Rowe et al., 2012b). Another recent publication proposes the combined use of two differentiated discard indicators in LCA, namely, appropriation of PPR and the potential discard impacts on vulnerable, endangered and critically endangered (VEC) species (Hornborg et al., 2012).
- Another approach based on PP has been suggested, and consists in considering not only the PPR to produce the harvested species but also the depletion in secondary production downstream of the trophic flow, with respect to the unfished state, using it as a proxy for quantifying ecosystems effects of fishing (Libralato et al., 2008). Such an approach encompasses both ecosystem properties and features of fishing activities (trophic level of catches and PPR).
- Langlois et al. (2011, 2012) suggest going further in a broader use of PP appropriation within a framework of a sea-use impact category, similar to land use. They suggest using the three-dimensional approach proposed by Mila i Canals et al. (2007) to account for time (occupation and restoration), space and a quality index reflecting transformation by usage and including a possible permanent or irreversible impact. The authors proposed a typology of marine activity and suggested regrouping under sea use at least the following three ones: artificial structures, biotic resource extraction, shipping lanes. Some additional marine activities such as seafloor destruction (in particular by trawling) or change of habitat surface or volume could also be accounted for using the same index through avoided or added (artificial reef) biomass (Langlois et al., submitted).
- A recent consultation report (Emanuelsson et al., 2012), produced in the context of an LCA-related project under the EU's Seventh Framework Programme for Research (FP7), proposes a new midpoint impact indicator to quantify depletion of exploited fish stocks: the Wasted Potential Yield (WPY). This indicator utilises current stock assessment data to predict future yields by means of a surplus yield production function. The WPY is the difference in future yields between the consequences of current exploitation levels and alternative exploitation levels defined by the maximum sustainable yield (MSY) approach.
- Alteration of trophic dynamics has also been addressed in various publications and identified with the "fishing down marine food webs" situation as measured by the catches' mean trophic level (Pauly et al., 1998). Pauly et al. (2000) proposed the Fisheries-in-Balance (FiB) index to represent such situations. Although these indicators are standardised, they cannot be used within the LCA framework because they are not fishery-specific and nearly all marine ecosystems are exploited by more than one fishery.
- Spatialised indicators of fishing pressure were proposed by different authors. For instance, Linnane et al. (2000) summarised various bottom trawling impact studies and Nilsson and Ziegler (2007) proposed a spatialised seafloor impact (i.e. damage to benthos) methodology based on the number of time per year a given area was likely to be swept by a trawl. They combined this value with the recoverability of the habitat to estimate impact on seafloor. In another example, Fréon et al. (2005b) proposed a mean ratio of fished area and area of distribution by species, exploited fraction of the ecosystem surface area, mean bottom depth of catches, and mean distance of catches from the coast. Hornborg et al. (2012) used seafloor disturbance data from bottom trawling to assess impacts of discard for VEC fish species.
- To date, there is no accepted/standardised method to assess target and non-target species removal. It has been argued that

Table 3
Impact categories used in published fisheries LCA studies. Excludes non-LCA/CFP studies and Fréon et al. (in prep.), which applies ReCiPe (hybrid method featuring 18 midpoint indicators plus three endpoint indicators).

Impact assessment method	Ecoindicator 99 (endpoint)									CML 2000/2001 (midpoint)											EDIP 97 (endpoint)									Additional impact categories
Impact categories	FF	A/E	E	CC	RI	RO	M	OL	C	GWP	AP	EP	POFP	ODP	HTP	FETP	METP	TETP	CED	ADP	GW	OD	A	NE	OF	ETWC	ETWA	ETSC	Total	
Eyjólfsdóttir et al. (2003)	X	X	X	X	X			X																					6	By-catch, discards, seafloor disturbance
Ziegler et al. (2003)										X	X	X	X				X												5	By-catch, discards, seafloor disturbance
Hospido and Tyedmers (2005)										X	X	X	X	X	X		X												7	
Ellingsen and Aanondsen (2006)	X	X	X	X	X				X																				6	Feeding efficiency for non-fishery products, land use vs. seafloor disturbance
Thrane (2004a), Thrane (2006)																			X		X	X	X	X	X	X	X	X	8	Catch, discards, by-catch
Emanuelsson et al. (2008)										X	X	X	X	X	X		X	X	X										9	By-catch, discards, under-sized seafloor disturbance
Ziegler and Valentinsson (2008)										X	X	X	X				X			X									6	Discards, seafloor disturbance
Guttormsdóttir (2009)	X	X	X	X	X	X	X	X	X												X								10	By-catch, discards, seafloor disturbance (qualitative)
Iribarren et al. (2010)										X																			1	
Vázquez-Rowe et al. (2010a)										X	X	X	X	X						X									6	Discards
Vázquez-Rowe et al. (2010b)										X	X	X	X	X	X	X	X			X									9	Discards
Ramos et al. (2011)										X	X	X		X			X			X									6	Discards, Fisheries-in-Balance
Svanes et al. (2011a)										X	X	X	X	X					X										6	
Vázquez-Rowe et al. (2012b)										X	X	X	X	X			X			X									7	Discards
Vázquez-Rowe et al. (2011)										X	X	X		X			X			X									6	Discards, seafloor impact
Total	3	3	3	3	3	1	1	2	2	11	10	10	8	8	3	1	8	1	3	6	2	1	1	1	1	1	1	1		

FF: Fossil Fuels; A/E: Acidification/Eutrophication; E: Ecotoxicity; CC: Climate Change; RI: Resp. Inorganics; RO: Resp. Organics; M: Minerals; OL: Ozone Layer; C: Carcinogens; GWP: Global Warming Potential; AP: Acidification Potential; EP: Eutrophication Potential; POFP: Photo-oxidant Formation Potential; ODP: Ozone Depletion Potential; HTP: Human Toxicity Potential; FETP: Freshwater Aquatic Ecotoxicity Potential; METP: Marine Aquatic Ecotoxicity Potential; TETP: Terrestrial Ecotoxicity Potential; CED: Cumulative Energy Demand; ADP: Abiotic Depletion Potential; GW: Global Warming; OD: Ozone Depletion; A: Acidification; NE: Nutrient Enrichment; OF: Ozone Formation; ETWC: Ecological Toxicity Water Toxic; ETWA: Ecological Toxicity Water Acute; ETSC: Ecological Toxicity Soil Chronic.

species removal, together with seafloor impacts, should not necessarily be included in quantitative LCAs for hot-spot identification (Thrane, 2006). However, we assert that when the goal of the study is providing data for environmental protection, those categories must be considered, ideally also quantitatively, by means of existing or new approaches. In contrast to most of above-reviewed indicator related to species removal, the used primary production can be assigned to a given fishery and can allow comparison with other activities such as aquaculture and agriculture.

- Regarding alteration of marine ecosystems, Cappell et al. (2007, p. 24) states that "Although biotic resource use is a recognised LCA impact category, seafood LCA research has largely failed to take into account impacts such as direct impacts to targeted stocks, by-catch of target and non-target species, loss of genetic diversity, alteration of trophic dynamics, and disturbance and displacement of benthic communities"; a view shared at large by the fisheries LCA community including us.

In conclusion, we believe fisheries-specific impact categories addressing seafloor disturbance, sea use and species removal should be used in fisheries LCA, and when possible standardised towards comparability of studies. Moreover, we assert it is necessary to apply seafood-relevant/specific impact categories such as the above-mentioned in order to allow for whole supply chain analyses and comparisons. If done so, specific impacts could be followed, in an additive fashion, along whole supply chains (e.g. BRU of various products along a seafood supply chain could be contrasted/combined, namely, fish, fishmeal, crop inputs to feeds, feed formulations, and final aquaculture products).

3.3.2. Classification and characterisation

Once the LCIA method and/or list of impact categories have been chosen, it is mandatory to associate LCI results to specific major impact categories, process known as classification (ISO, 2006b), although we believe "assignation" would be a less confusing terminology. The next step is characterisation, which consists in expressing LCI results in a reduced set of common units, which can be aggregated into impact categories (ISO, 2006b). In other words, impacts associated to LCI results are aggregated into categories. Characterisation factors are used for such aggregation, and are usually included in LCIA methods as constituencies of characterisation models.

Characterisation factors used in the reviewed studies identified major contributions from fuel production and use, besides direct specific impacts due to target and non-target species removal. GWP was the main impact indicator affected, mostly due to fuel use. Other inventory items identified as contributing to impacts are maintenance activities and substances (antifouling, refrigerants, lubricants, cleaners, etc.), and fishing gear use. The maintenance stage was found to have a small contribution to impacts, although it was often insufficiently inventoried. Most of the studies found the fishing phase to be the main contributor to impacts in the seafood lifecycle. Bottom trawling was identified as having a higher impact than other fishing methods in terms of associated emissions (GWP) and certain fishery-specific impact categories (e.g. seafloor impacts).

The reviewed studies featured fisheries-specific impact categories such as species removal and seafloor impacts. Various studies calculated the impacts related to the removal of target and non-target species (by-catch and discard), as shown in the Supplementary Material. Several studies calculated the seafloor area disturbed, some of them by means of the seafloor impact index methodology proposed by Nilsson and Ziegler (2007). The impact of trawling and other bottom gear is discussed in detail in Eyjólfsdóttir et al. (2003), Thrane (2004a), Ziegler and Valentinsson (2008), Guttormsdóttir (2009) and Vázquez-Rowe et al. (2012a).

Emissions to air and water were calculated based on fuel use data and accepted ratios for substance losses, for instance, two thirds of antifouling paint lost to the marine environment, as applied by Hospido and Tyedmers (2005).

We observed that antifouling paints in use contain toxic components (i.e. biocides) that are not considered because data is not available in the currently used databases and/or characterisation is not considered in current LCIA methods. Besides, persistent pollutants (e.g. metals) get very high characterisation factors in toxicity models used by LCIA methods used in the reviewed studies. A way to overcome such limitations could be the utilisation when applicable of the United Nations Environment Programme/Society of Environmental Toxicology and Chemistry (UNEP/SETAC) USEtox toxicity model (Rosenbaum et al., 2008), which claims to be a consensus model. USEtox, for instance, models exposure and impacts only in coastal waters, treating deep sea as a sink; and features characterisation factors for some basic antifouling substances used. It must nonetheless be noted that toxicity models in general feature great intrinsic uncertainty.

3.3.3. Normalisation and weighting

Normalisation can be understood as the scaling of non-comparable category indicators (e.g. GWP and Eutrophication Potential) towards the same reference as to render them comparable and better understand the relative magnitude of each one (ISO, 2006b). It is an optional and controversial step in LCA, carried out by means of dividing indicators results by a selected reference value known as normalisation factor (ISO, 2006b). LCIA methods such as ReCiPe (Sleeswijk et al., 2008) feature normalisation references like: European and global normalisation factors based on reference year 2000, considered as a follow up (and improvement) to Huijbregts et al. (2003)'s 1990/1995-based normalisation factors; and characterisation factors updated from Guinée et al. (2001). Normalisation factors are simply the total sum of the characterised flows at the corresponding scale. As a result one estimates the share of the modelled results in a European or worldwide total. Normalisation only highlights the most important impact dimensions if one assumes that all impact categories are of equal importance; a view that few endorse. Weighting is another optional step in LCA, which consists in deciding—on the base of subjective value choices—the relative importance of impact categories, characterised and normalised (occasionally with regards to an aggregated single score).

Normalisation was carried out in only two of the reviewed studies: Thrane (2006) applied normalisation references for Danish economic activities (Thrane (2004a) used earlier Danish normalisation references), while Hospido and Tyedmers (2005) used total global emissions for baseline years 1990/1995 like normalisation references, as defined in Huijbregts et al. (2003). The reason for this limited use of normalisation is, in our opinion, linked to frequent criticisms of this approach. This in particular regards to the referent regional or global systems used for scaling (e.g. featuring localisation in terms of regions and impact categories), which are often poorly estimated leading to uncertainty (Sleeswijk et al., 2008). Lack of emission data and/or characterisation factors leading to bias (Heijungs et al., 2007) may be another reason, alongside overall congruency issues (Norris, 2001).

Because normalisation is useful in highlighting the most important environmental impact dimensions of the fishing activities, we suggest that when normalisation is performed, to apply global resource consumption and emission rates in order to show the specificity of fisheries, as in Hospido and Tyedmers (2005), but to present only semi-quantitative results such as an indication of which factors have the most impact (with additional attention in the cases where toxicity impact categories are taken into consideration).

In addition to a normalisation based on reference global data on all types of human activities, the fishery and seafood research community could keep contrasting individual case studies to global data on this sector for the major impacting factors: fuel consumption ratios (as completed in Tyedmers (2001) and Table 4 in this review) and extracted species indicators, although in this later case proper generic indices still need to be agreed upon. Impact categories formalising fuel consumption in LCA are abiotic resource depletion (of fossil fuels) and Cumulative Energy Demand (energy equivalence of fossil fuels consumed) (VDI, 1997; van Oers et al., 2002).

3.4. Interpretation

The final stage of LCA, interpretation, consists of extracting conclusions based on the inventory analysis and impact assessment, in such a way that results of the LCA can be presented and used for decision-making (ISO, 2006b). It includes identification of key issues derived from the LCI and/or LCIA stages (given that LCIA is an optional phase); an evaluation of completeness, sensitivity and consistency; and the statement of conclusions, limitations and recommendations (ISO, 2006b).

3.4.1. Key findings

The reviewed studies focused mainly on European operations, mostly in the Atlantic Ocean and North Sea, but occasionally in African waters and the other oceans waters, during the last decade (2001–2011).

The most common pattern found in the studies is the fact that fishing operations (the vessel use phase, including but not limited to fuel use) are the main contributor to environmental impacts during the extraction phase as previously found in other reviews (e.g. Pelletier et al., 2007).

Construction and EOL phases were generally roughly considered (i.e. limiting the inventory to steel weight of hull or hull plus engine) or directly omitted from analyses, due to the extended assertion that those phases generate negligible impacts in comparison to use and maintenance. The few studies dealing with construction, too partially in most cases, indicate the dominant importance of metals on environmental impacts related to toxicity to humans and the environment, but also to the metal depletion indicator recently implemented in the ReCiPe method (Goedkoop et al., 2009). The only work addressing EOL of vessels shows a non-negligible effect on freshwater and marine water toxicity (Fréon et al., in prep.).

All studies reviewed deal with emissions to air due to combustion of fossil fuels during the use phase. Some of the studies discussed the impacts due to vessel maintenance, emphasising the use of antifouling and refrigerants and assigning great importance to these substances as contributors to impacts. Copper was the most often mentioned antifouling component, although other conventional components such as lead, zinc, tributyltin (TBT), xylene and various phenyl-, pyridine- or ethyl-derived—or new ones like dibutyltin oxide, "sea nine" 211e (4,5-dichloro-2-n-octyl-4-isothiazolin-3-one)—are often used with or without copper and also present a high level of toxicity. However, most of those elements are either not considered in the LCA, not described in used databases or not characterised in used LCIA methods (except for zinc, tin and copper ions, as well as TBT, but not complex molecules). Ideally, the USEtox consensus method should be enlarged to include marine eco-toxicity, and thus enriched with characterisation factors for substances included in antifouling paints and other waterborne emissions such as bilge oil or (used) lubricating oil. Many of those waterborne emissions are also missing from other toxicity models.

Table 4
Fuel use by targeted species aggregation vs. fish gear. Fuel consumption has been standardised to kg fuel per tonne of landed fish. Marine fuel density used for calculations: 0.832 kg/L. Studies reference numbers are linked to Table 1 This table is expanded and complemented in the Supplementary Material.

Species aggregation	# Studies	Artisanal trawling Fuel use average (kg/t)	# Studies	Creeling/trapping Fuel use average (kg/t)	# Studies	Gill-netting Fuel use average (kg/t)	# Studies	Long lining Fuel use average (kg/t)	# Studies	Purse seining Fuel use average (kg/t)	# Studies	Trawling Fuel use average (kg/t)	Total # Studies	Fuel use average (kg/t)	Fuel use standard deviation
Cephalopods											(14)	1736	1	1736	N/A
Codfish					(3)	283	(10) (16)	270			(1) (2) (3) (4) (6) (10)	666	7	536	315
Ground fish											(7) (12)	1518	2	1518	1455
Hake							(12) (17)	1428			(17)	2104	2	1653	409
Lobster & crab			(1) (9)	1052							(1) (4) (9)	3852	3	2732	2882
Mackerel									(4) (15) (18)	86	(4) (12) (18)	298	4	192	183
Shrimps & prawns	(8)	524									(1) (4) (8)	1056	3	950	698
Small pelagic fish									(1) (4) (6) (11) (12) (19)	75	(4) (11)	99	6	83	51
Tuna									(1) (5) (12)	708			3	708	641
# Papers/average	1	524	2	1052	1	283	4	849	9	290	13	1416			

It is not always clear which substances are present in the studied fisheries although nowadays all antifouling paints use several toxic substances with or without addition of copper derivates (Yebra et al., 2004). Eyjólfsdóttir et al. (2003) mentioned that in the case of Icelandic fisheries the non-use of TBT reduces considerably the environmental impacts of antifouling use. This early discontinuation of TBT use is exceptional, because a 2001 ban on the use of this agent (and organotins in general) by the International Maritime Organisation just entered into force in 2008. Consequently, TBT and other organotins are currently banned in most European and American countries, as well as in a few Asian countries (Sonak et al., 2009). Nowadays, substitute agents—as well as polymer coatings, biocides, etc.—are used (IMO, 2002), but as mentioned, many of those substances are not characterised in most environmental databases and LCIA methods and thus unfortunately omitted from studies.

Impacts resulting from the use of lubricating oil and refrigerating agents, ice production and net production, use and loss were considered in various studies (Eyjólfsdóttir et al., 2003; Ramos et al., 2011; Vázquez-Rowe et al., 2010a,b, 2011, 2012a), as detailed in Table 2. Those inventory items were generally found to have a minor (even negligible) contribution to impacts, especially when compared with impacts derived from fuel consumption.

3.4.2. Fuel use

Fuel use was found to be the greatest single source of most environmental burdens among all inputs to fishing activities. This item was assessed in all reviewed studies, but in general terms it is not clear which fuel-burning activities are included (i.e. only fishing trips plus on-board processing or also tests, relocations between fishing ports or areas, etc.).

Many of the studies emphasised that energy efficiency and other environmental impacts depend, among other factors, on the fishing gear used, a conclusion confirmed by Thrane (2004a), Vázquez-Rowe et al. (2011) and by this review (Table 4 and Supplementary Material). From the reviewed studies it can thus be generalised that energy efficiency in relation to fuel use is strongly dependent on the fishing gear utilised. Generally, trawling methods (despite the differences between bottom and mid-water trawling) were the most energy-intensive among those listed when related to landed kg of fresh fish equivalents. Ultimately, fuel consumption per functional unit of a given fleet operating on a given ecosystem is subject to complex factors often unaccounted for (Thrane, 2004b). Among them, natural abundance of the resource, stock status, spatiotemporal variability of catchability (level of aggregation, depth, distance from the coast, etc.) management regime, skill level of the vessel crew (the “skipper effect”, Vázquez-Rowe and Tyedmers, 2012), proportion of by-catch or hull technology. Additionally, emissions from marine fuel combustion depend on the quality of the fuel itself and the condition and technology of the engine, yet those factors have been overlooked in the reviewed studies.

All of the studies discussed energy use of fisheries expressed as quantities of fuel consumed per landed mass of fish at different stages of transformation and using different units of mass or volume. We standardised energy efficiency data from all studies as a ratio of kg of fuel used per tonne of landed fresh unprocessed fish, and thus render comparison possible. Onboard processing losses and energy used for processing were considered when applicable. Additionally, non-LCA studies where included (Tyedmers, 2001; additional fuel use calculations in Thrane, 2004a; Driscoll and Tyedmers, 2010) in order to extend the energy use in the data set. Figures offered by Thrane (2004a) are calculated using system expansion including cross-calculation of all main by-catches. Nonetheless, the study also presents fuel figures calculated using mass allocation and thus results are found closer to other studies, for instance, Tyedmers (2001). Despite the availability of these system expansion calculations, the mass-allocated figures were used in this review to make comparisons with other studies feasible (Table 4 and Supplementary Material). Differences arising from the allocation method used can be important. For instance, trawling of Norway lobster (aggregated into the Lobster and crab category) consumes 3 214 kg fuel/tonne landed according to mass allocation and 16,762 kg fuel/tonne landed according to systems expansion (Thrane, 2004a).

Fuel use should be disaggregated as far as possible regarding the specific activities involved (e.g. on-board processing). Non-fishing, fuel-consuming activities can be non-negligible and thus it should be clear whether they are accounted for. For instance, we have found that in countries like Peru, fishing vessels are often seasonally relocated between North and South, over a very long coastline. Moreover, we assert that in multi-species fisheries there is a need of allocating those activities between landed catches of different species.

3.4.3. Sensitivity and uncertainty analyses

Sensitivity analysis consists in the evaluation of the impacts of changes in data and methodological choices over LCIA results, while uncertainty analysis is the evaluation of the impacts of the propagation of data- and assumptions-related uncertainties over LCIA results (ISO, 2006b). Sensitivity and uncertainty analyses should be performed in order to better reflect the accuracy of LCI and LCA studies.

The ISO standard and Guinée et al. (2001) recommend sensitivity analyses to be carried out when several choices seem applicable by contrasting allocation methods. In practice often mass vs. economic allocations are contrasted, despite the fact that, according to the ISO standard, economic allocation should be the least preferred alternative. However, a sensitivity analysis could be carried out, for example, among criteria such as energy content and nutritional value. In fisheries LCA, the selection of allocation strategy and accounting of fuel consumption are the main causes for large variation in results, and thus sensitivity analysis is very relevant in such a context. As discussed for instance in Thrane (2004a) and EC/JRC (2010), there are several sources of uncertainty in LCA (methodological, inventory data and characterisation factors) that need to be evaluated quantitatively via uncertainty analyses, which typically are performed by means of random sampling methods (e.g. Monte Carlo simulations).

Several of the reviewed studies performed sensitivity analyses. Nonetheless, sensitivity results are communicated in very diverse fashions, ranging from a simple statement to several paragraphs of discussion. The analyses themselves have been carried out based on various criteria, including: contrasting impact assessment methods or allocation methods, modifying allowable emissions, simulating different volumes of key substances (i.e. fossil fuels) and varying several operational factors.

Data uncertainty was mentioned in some of the reviewed studies, but explanations on how uncertainties were dealt with are superficial. One single study, a doctoral thesis (Thrane, 2004a); discussed in great detail data and methodological uncertainty as well as their effects on LCA results. Moreover, only one additional study (Fréon et al., in prep.) accounts for variability and uncertainty during the LCI, by means of Monte Carlo simulations.

The ISO standard, as well as guidelines such as Guinée et al. (2001) and the ILCD Handbook (EC/JRC, 2010); offer criteria for sensitivity analysis. We detail those recommendations by suggesting it should preferentially be related to important (i.e. >5% of individual contribution to impacts) items, especially when high uncertainty is associated to underlying data and assumptions. Results from such analyses should be presented in such a way that scenarios can be outlined based on important variations of critical items. A key example in fisheries would be data on fuel use, catches and discards, the last two being relevant for computing fisheries-specific impact categories. Furthermore, in the LCIA stage, extreme values should be investigated, as discussed in Thrane (2004a).

3.4.4. Fishery-specific methodological concerns

To date there is no agreement regarding methodological choices for carrying out and presenting LCAs of fisheries, which makes it difficult to compare studies (Ayer et al., 2007; Svanes et al., 2011b). However, the studies in this review have been screened for patterns and singularities in an attempt to characterise the state-of-the-art of LCA applied to fishing activities and to contribute to the ongoing discussion on sensitive issues of LCA in general (i.e. allocation, impact categories, normalisation, sensitivity and uncertainty analyses).

In fisheries LCA, a number of methodological issues arise, beyond the issues inherent to the state of the art of LCA. Those issues include the lack of standardised and widely applicable fishery-specific impact categories and how to deal with important technological, spatial and even temporal variability in fishing operations, especially with regards to fuel use.

3.5. LCA in the context of sustainability assessment methods

Discussion of socio-economic issues has been minimal in the context of fisheries LCA literature (Pelletier et al., 2007). Since LCA alone has focused on environmental impacts of production systems, as well as on resource depletion, other life cycle methods, namely Social LCA (SLCA) and Life Cycle Costing (LCC), have been developed as necessary complements for capturing trade-offs between environmental, social and economic interest along the life cycle of production systems (Dreyer et al., 2006; Guinée et al., 2011).

LCA, SLCA and LCC are philosophically related tools within the larger framework of Life Cycle Sustainability Assessment (LCSA) (Klöpffer, 2008; Klöpffer and Ciroth, 2011; Swarr et al., 2011; Valdivia et al., 2011).

A comprehensive review of approaches for SLCA has been compiled by Jørgensen et al. (2008) and a recent guideline attempts to pioneer the standardisation of SLCA practice (Andrews et al., 2009). LCC is, on the other hand, a mature approach aimed to assess all costs associated to the life cycle of a (product) system (Huppes et al., 2004). No dedicated and comprehensive standard exists to date for LCC (other than guidelines and sector-specific standards—e.g. ISO 15686-5 for the construction sector—), but it predates on a rich body of literature and accepted accounting/costing techniques. See Supplementary Material for a list of standards and guidelines for life cycle methods.

Table 5
Methods for environmental assessment in the context of fisheries (partially based on Loiseau et al., 2012).

Method	Local/global	Mono/multicriteria	Qualitative/quantitative	Real/potential impacts	Life cycle thinking	Strategy	Comment
CF	Global	Mono	Quantitative	Potential	++	Bottom-up	Useful in combination with or as a preliminary step of LCA.
EAF	Local	Multi	Mostly quantitative	Real	−−	Bottom-up/Top-down	This approach encompasses many methods dealing with ecological, environmental and socio-economic issues.
EF	Global	Mono	Quantitative	Potential	+	Bottom-up/Top-down	Useful in combination with or as a preliminary step of LCA.
Exergy/Emergy	Global	Mono	Quantitative	Potential	++	Bottom-up/Top-down	Focus on energy. Relevant for fisheries.
(HE)RA	Local	Mono or Multi	Qualitative	Real	−	Bottom-up	Not adapted to fisheries, except in special circumstances (possibly stock or fishery collapses due extreme events such as large-scale oil spill, tsunami, strong El Niño events)
Input–Output Analysis	Local or Global	Multi	Quantitative	Potential	++	Top-down	Useful for extending and completing LCA to better quantify flows of material and energy.
LCA	Global	Multi	Quantitative	Potential	++	Bottom-up	Can be considered as one of the various methods included in EAF.
LIA	Local	Multi	Quantitative	Real	−−	Top-down	Could be applied to contamination impacts, seabed disturbance, etc. Can be considered as one of the various methods included in EAF.
MFA	Local	Multi	Quantitative	Potential	+	Top-down	Useful as a preliminary step of LCA to better quantify flows of material and energy.
Specific EAF methods	Local	Mostly Mono	Mostly quantitative	Real	−−	Bottom-up/Top-down	These methods, included in EAF, aim at the evaluation of exploited stock (or whole marine ecosystem) status through population dynamics models, trophic models, bio-economical models, operational management procedure, management strategy evaluation, etc.

CF: Carbon Footprint; EAF: Ecosystem Approach to Fisheries; EF: Ecological Footprint; (HE)RA: (Human and Environmental) Risk Assessment; LIA: Local Impact Assessment; MFA: Material Flow Analysis.

LCSA has also been applied to fisheries research, for instance, precursor works such as Kruse et al. (2008) attempted to apply in a seafood context recent developments in SLCA (although this approach is still under development).

Beyond life cycle methods, a great variety of system analysis tools have been developed, focusing on diverse types of impacts and dimensions of sustainability (natural resources, environmental, social, economic impacts), and spanning different spatial scales/levels of study (micro, macro, meso), as described in great detail, for instance, in Finnveden and Moberg (2005) and Jeswani et al. (2010). Some of those methods could complement fisheries LCAs for a wider, more holistic study. Examples include the combination of LCA and data envelopment analysis (Vázquez-Rowe et al., 2010a), the use of geographical information systems (GIS) data for computing certain impact categories (Ziegler and Valentinsson, 2008) and the computation of the FiB index in the context of an LCA study (Ramos et al., 2011). See Table 5 for a list of environmental methods and Supplementary Material for sustainability methods.

4. Conclusions and perspectives

Future LCA studies of fisheries will hopefully continue contributing to a) mapping the environmental performance of fisheries worldwide (most studies to date focus on the Northern Atlantic and other few fishing areas), showing increasing attention to aspects that have been neglected so far; and b) enriching LCA studies on supply chains based on or strongly connected to fisheries (e.g. aquaculture, animal husbandry, etc.).

We advocate not only for more strictly following the ISO norms (for the sake of increased consistency and comparability) but, beyond this, for specific standardisation of fisheries LCA practice towards an accepted fisheries/seafood LCA framework. Such framework would address, when possible, boundaries setting, impact categories and characterisation, normalisation references, allocation strategies and sensitivity analyses, presentation of results, etc. These suggestions are in line with an on-going project developing a carbon footprint standard for the fisheries industry, by the British Standards Institution (BSI, 2012).

Despite the fact that existing fisheries LCA studies are difficult to contrast due to a general lack of detail and standardisation, valuable conclusions can be mined from available literature, concluding that fuel consumption, use of antifouling paints and associated release of substances are key contributors to environmental impacts as measured by conventional LCAs. Such findings can easily be translated into operational recommendations to improve environmental performance of fisheries, within the framework of the ecosystem approach to fisheries and, in the future, certification and labelling of fisheries.

Nonetheless, target and non-target species removal and other fisheries-specific impact categories, such as sea use and seafloor disturbance, are not included in most quantitative LCAs to date. Furthermore, the stage of fishing unit construction, and the lesser contributing stage of EOL, are often neglected. Another pressing need, not specific to fisheries LCA, is to use data that actually reflects the specifics of the supply chains of concern, instead of an over-reliance on often unrepresentative data from third-party commercial LCI databases. The treatment of these issues, perceivable as weak points in fisheries LCA research, should be included in the abovementioned standardisation of LCA practice for fisheries and seafood research. Thus, future LCAs would ideally include (a) key inventory data and detailed explanations of energy input per mass of landed fish; (b) inclusion of the whole life cycle of vessels, namely construction, use, maintenance and, to a lesser extent EOL with focus on use of fuel, metals and toxic products release; (c) justification of allocation strategies applied; (d) when necessary use proper data for most impacting processes, instead of LCI databases, or modify/adapt the later; (e) sensitivity and uncertainty analyses (focusing for instance on seafood-specific allocation criteria such as energy content, content of protein and lipids, etc.); (f) inclusion of fisheries-specific impact categories, detailed and explained; and (g) generalised normalisation references presented semi-quantitatively.

We advocate for the consensual elaboration of a Product Category Rule (PCR) for Fisheries and Seafood LCIs. A PCR is the set of guidelines, requirements and specific rules for communicating LCA results, under the form of an Environmental Product Declaration (EPD), of a family of products fulfilling equivalent functions (known as product category) (ISO, 2006c; Schau and Fet, 2008). Such a PCR for fisheries and seafood would include a standard format with optional sections according to the type of fishery and the purpose of the study. It would demand a number of observations, inventory items and other methodological details to be clearly communicated. Such standardisation may result from a workshop gathering LCA practitioners and fisheries scientists under the auspices of an international LCA entity such as the UNEP/SETAC Life Cycle Initiative.

Future, ideally standardised fisheries LCAs, should contribute to better decision making on fisheries management and seafood consumption. The decrease of environmental impacts produced by marine fisheries depends not only on technical improvement aimed at reducing adverse effects of construction, use, maintenance and EOL of fishing units, but specially on the management of the fishing sector in order to decrease fishing effort on overexploited stocks and limit fishing and processing overcapacity. For instance, we believe that some of the driving factors of fuel use per landed catch, namely the selection of fishing gear and the size of fishing units, depend on design/management decisions that should be addressed by fisheries policy and management.

There is also a need for a comprehensive assessment of environmental (and sustainability) impacts of fisheries in the context of whole supply chains, as well as for standardised tools, approaches and methods to do so. The LCSA framework seems promising, and once it reaches maturity, life cycle comprehensive sustainability assessment of seafood supply chains will be more accessible. In the meantime, the inclusion of brief discussion on socio-economic issues in future fisheries LCA studies would be advisable to render them more valuable for decision makers, fishing and seafood companies, as well as for social and economic researchers.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.fishres.2013.01.006.

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1 Fuel use by targeted species aggregation vs. fish gear

	Artisanal trawling		Creeling/ trapping		Gillnetting		Long lining		Purse seining		Trawling				
Species aggregation	**# Papers**	**Fuel use average (kg/t)**	**# Papers**	**Fuel use average (kg/t)**	**# Papers**	**Fuel use average (kg/t)**	**# Papers**	**Fuel use average (kg/t)**	**# Papers**	**Fuel use average (kg/t)**	**# Papers**	**Fuel use average (kg/t)**	**Total # papers**	**Fuel use average (kg/t)**	**Fuel use standard deviation**
											(14)	1,736	**1**		
Cephalopods												**1,736**	**1**	**1,736**	**N/A**
					(3)	283	(16)	241			(6)	470			
							(10)	300			(4)	391			
											(3)	1,165			
											(2)	632			
											(10)	915			
											(1)	424			
Codfish						**283**		**270**				**666**	**7**	**536**	**315**
											(12)	2,547			
											(7)	489			
Ground fish												**1,518**	**2**	**1,518**	**1,455**
							(12)	1,551			(17)	2,104			
							(17)	1,305							
Hake								**1,428**				**2,104**	**2**	**1,653**	**409**
			(9)	1,830							(4)	3,214			
			(1)	275							(9)	7,488			
											(1)	853			

Lobster & crab				**1,052**								**3,852**	**3**	**2,732**	**2,882**
									(15)	15	(12)	316			
									(4)	67	(4)	83			
									(18)	176	(18)	496			
Mackerel										**86**		**298**	**4**	**192**	**183**
	(8)	524									(8)	2,163			
											(4)	449			
											(4)	849			
											(1)	764			
Shrimps & prawns		**524**										**1,056**	**3**	**950**	**698**
									(6)	70	(4)	125			
									(12)	175	(4)	83			
									(4)	116	(11)	90			
									(19)	19					
									(1)	52					
									(11)	17					
Small pelagic fish										**75**		**99**	**6**	**83**	**51**
									(5)	363					
									(12)	313					
									(1)	1,448					
Tuna										**708**			**3**	**708**	**641**
# studies/average	**1**	**524**	**2**	**1,052**	**1**	**283**	**4**	**849**	**9**	**290**	**13**	**1,416**			

Fuel consumption has been standardised to kg fuel per t of landed fish. Marine fuel density used for calculations: 0.832 kg/l. Studies reference numbers as follows: (1) Tyedmers (2001), (2) Eyjólfsdóttir et al. (2003), (3) Ziegler et al. (2003), (4) Thrane (2004a), (5) Hospido & Tyedmers (2005), (6) Ellingsen &

Aanondsen (2006), (7) Thrane (2006), (8) Emanuelsson et al. (2008), (9) Ziegler & Valentinsson (2008), (10) Guttormsdóttir (2009), (11) Driscoll & Tyedmers (2010), (12) Iribarren et al. (2010), (13) Vázquez-Rowe et al. (2010a), (14) Vázquez-Rowe et al. (2012a), (15) Ramos et al. (2011), (16) Svanes et al. (2011a), (17) Vázquez-Rowe et al. (2011), (18) Vázquez-Rowe et al. (2010b), (19) Fréon et al. (in prep.).

2 Fuel use by targeted species aggregation vs. ecosystem type

	Coastal pelagic		Estuary		Hard shelf		Hard slope		Offshore pelagic		Soft shell				
Species aggregation	**# Papers**	**Fuel use average (kg/t)**	**# Papers**	**Fuel use average (kg/t)**	**# Papers**	**Fuel use average (kg/t)**	**# Papers**	**Fuel use average (kg/t)**	**# Papers**	**Fuel use average (kg/t)**	**# Papers**	**Fuel use average (kg/t)**	**Total # papers**	**Fuel use average (kg/t)**	**Fuel use standard deviation**
					(14)	1,736									
Cephalopods						**1,736**							**1**	**1,736**	**N/A**
					(6)	470									
					(16)	241									
					(4)	391									
					(3)	1,165									
					(3)	283									
					(2)	632									
					(10)	915									
					(10)	300									
					(1)	424									
Codfish						**536**							**7**	**536**	**315**
							(12)	2,547			(7)	489			
Ground fish								**2,547**				**489**	**2**	**1,518**	**1,455**
					(12)	1,551									

			(17) 1,305						
			(17) 2,104						
Hake			**1,653**				**2**	**1,653**	**409**
			(4) 3,214			(1) 275			
			(9) 7,488						
			(9) 1,830						
			(1) 853						
Lobster & crab			**3,346**			**275**	**3**	**2,732**	**2,882**
	(12) 316								
	(15) 15								
	(4) 67								
	(4) 83								
	(18) 496								
	(18) 176								
Mackerel	**192**						**4**	**192**	**183**
		(8) 524				(8) 2,163			
						(4) 449			
						(4) 849			
						(1) 764			
Shrimps & prawns		**524**				**1,056**	**3**	**950**	**698**
	(6) 70								
	(12) 175								
	(4) 125								
	(4) 116								
	(4) 83								
	(19) 19								

	(1)	52													
	(11)	17													
	(11)	90													
Small pelagic fish		**83**											**6**	**83**	**51**
									(5)	363					
									(12)	313					
									(1)	1,448					
Tuna										**708**			**3**	**708**	**641**
# studies/average	**8**	**138**	**1**	**524**	**11**	**1,818**	**1**	**2,547**	**3**	**708**	**4**	**607**			

Fuel consumption has been standardised to kg fuel per t of landed fish. Marine fuel density used for calculations: 0.832 kg/l. Studies reference numbers as follows: (1) Tyedmers (2001), (2) Eyjólfsdóttir et al. (2003), (3) Ziegler et al. (2003), (4) Thrane (2004a), (5) Hospido & Tyedmers (2005), (6) Ellingsen & Aanondsen (2006), (7) Thrane (2006), (8) Emanuelsson et al. (2008), (9) Ziegler & Valentinsson (2008), (10) Guttormsdóttir (2009), (11) Driscoll & Tyedmers (2010), (12) Iribarren et al. (2010), (13) Vázquez-Rowe et al. (2010a), (14) Vázquez-Rowe et al. (2012a), (15) Ramos et al. (2011), (16) Svanes et al. (2011a), (17) Vázquez-Rowe et al. (2011), (18) Vázquez-Rowe et al. (2010b), (19) Fréon et al. (in prep.).

3 Publications on Life Cycle Assessment applied to food systems (agriculture and seafood)

Year	Fisheries: energy, Carbon Footprint, processing	Fisheries: LCA	Aquaculture: LCA	Agricultural food products: LCA and Carbon Footprint
1998				(Andersson et al., 1998) (Cederbeg, 1998)
1999				(Andersson & Ohlsson, 1999)
2000	(Tyedmers, 2000)[a]			(Andersson, 2000) (Cederberg & Mattsson, 2000)
2001	(1)			(Haas et al., 2001)

2002	(Ziegler, 2002)			(Berlin, 2002) (Cederberg, 2002)* (Eide, 2002)
2003	(Ziegler & Hansson, 2003)	(3) (2)	(Silvenius & Grönroos, 2003)	(Cederberg & Stadig, 2003) (De Boer, 2003) (Heller & Keoleian, 2003) (Hospido et al., 2003)
2004	(4)[a]		(Papatryphon et al., 2004)	
2005	(Hospido, 2005)[a]	(5)	(Mungkung, 2005)[a]	(Anton, 2005) (Casey & Holden, 2005) (Nunez et al., 2005) (Sanjuan et al., 2005) (Strid Eriksson et al., 2005)
2006	(Hospido et al., 2006)	(6) (7)	(6) (Aubin et al., 2006) (Grönroos et al., 2006) (Mungkung, 2006)	(Casey & Holden, 2006) (Ramírez et al., 2006)
2007	(Ziegler, 2007)			(Dalgaard, 2007)[a] (Ogino et al., 2007)
2008	(Thrane, 2008)	(8) (9)	(Ramírez et al., 2008)a	(Avraamides & Fatta, 2008) (Dalgaard et al., 2008) (Lovett et al., 2008) (Nemecek, 2008a) (Nemecek, 2008b) (Nemecek, 2008c) (Pelletier, 2008) (Thomassen et al., 2008a) (Thomassen et al., 2008b)
2009	(11) (Schau et al., 2009) (Winther et al., 2009) (Thrane et al., 2009)	(10)	(Ayer & Tyedmers, 2009) (Pelletier et al., 2009) (Sun, 2009)a (d'Orbcastel et al., 2009)	(Blengini & Busto, 2009) (Cederberg, 2009) (Coltro et al., 2009) (Davis et al., 2009) (Edwards-Jones et al., 2009) (Lehuger et al., 2009) (van der Werf et al., 2009)

2010	(12) (Fulton, 2010)[a]	(13) (18)	(Iribarren et al., 2010a) (Iribarren et al., 2010b)	(Beauchemin et al., 2010) (Biswas et al., 2010) (Drastig et al., 2010) (Knudsen et al., 2010) (Ledgard, 2010) (Muñoz et al., 2010) (Nilsson et al., 2010) (Pelletier et al., 2010a) (Pelletier et al., 2010b) (Peters et al., 2010) (Röös et al., 2010) (Rotz et al., 2010) (Schmidt, 2010)
2011	(Iribarren et al., 2011)	(14) (15) (16) (17) (Svanes et al 2011b)	(Phong et al., 2011) (Cao et al., 2011) (Henriksson et al. 2011) (Bosma et al. 2011)	(Flysjö et al., 2011) (Freitas de Alvarenga et al., 2011) (Williams & Wikström, 2011) (Crosson et al., 2011) (Lesschen et al., 2011) (Hagemann et al., 2011) (Browne et al., 2011) (Yan et al., 2011) (Chauhan et al., 2011) (Bartl et al., 2011) (O'Brien et al., 2011) (Nemecek et al., 2011) (Gerber et al., 2011) (Beauchemin et al., 2011) (Cerutti et al., 2011) (Cooper et al., 2011) (Karakaya & Özilgen, 2011) (Parent & Lavallée, 2011) (Freitas et al. 2011)

[a] Thesis. Studies reference numbers as follows: (1) Tyedmers (2001), (2) Eyjólfsdóttir et al. (2003), (3) Ziegler et al. (2003), (4) Thrane (2004a), (5) Hospido & Tyedmers (2005), (6) Ellingsen & Aanondsen (2006), (7) Thrane (2006), (8) Emanuelsson et al. (2008), (9) Ziegler & Valentinsson (2008), (10)

Guttormsdóttir (2009), (11) Driscoll & Tyedmers (2010), (12) Iribarren et al. (2010), (13) Vázquez-Rowe et al. (2010a), (14) Vázquez-Rowe et al. (2012a), (15) Ramos et al. (2011), (16) Svanes et al. (2011a), (17) Vázquez-Rowe et al. (2011), (18) Vázquez-Rowe et al. (2010b).

4 Comparison of current Life Cycle Impact Assessment methods

Based on Rosenbaum et al. (2008), van Zelm et al. (2009), ILCD (2010) and Hischier et al. (2010).

Major methods → **Criteria ↓**	**CML 2001** **CML 2002**	**Eco-indicator 99**	**EDIP 97** **EDIP 2003**	**ReCiPe**
Background publication	Guinée et al. (2001a,b) Guinée et al. (2002)	Goedkoop and Spriensma (2000a,b)	Wenzel et al. (1997) Hauschild and Potting (2005)	Goedkoop et al. (2009)
Origin	Netherlands: Centre of Environmental Science - Leiden University (CML)	Netherlands: Pré Consultants	Denmark: Technical University of Denmark, Danish Environmental Protection Agency **EDIP 2003 is an alternative to EDIP 97, not an update**	Netherlands: National Institute for Public Health and the Environment (RIVM), Radboud University, CML, PRé Consultants, CE Delft **This method integrates CML 2002 and Eco-indicator 99**
Regional validity	Global (except for acidification and photo-oxidant formation: Europe)	Global for climate change, ozone depletion and resources; Europe for other categories	EDIP 97: Global EDIP 2003: Europe	Global for climate change, ozone depletion and resources; Europe for other categories
Midpoint impact categories	Acidification potential Climate change Eutrophication potential Freshwater aquatic ecotoxicity Human toxicity Land use Marine aquatic ecotoxicity Photochemical oxidation Resources Stratospheric ozone depletion	Carcinogenics Climate change Ionising radiation Ozone layer depletion Respiratory effects Stored carcinogenics Stored ionising radiation Acidification and eutrophication Ecotoxicity Land occupation	Acidification Ecotoxicity (acute) Ecotoxicity (chronic) Global warming Human toxicity Land filling Non-renewable resources Nutrient enrichment Photochemical ozone formation Renewable resources	Climate change (IPCC 2007 factors) Ozone depletion Terrestrial acidification Freshwater eutrophication Marine eutrophication Human toxicity Photochemical oxidant formation Particulate matter formation Terrestrial ecotoxicity

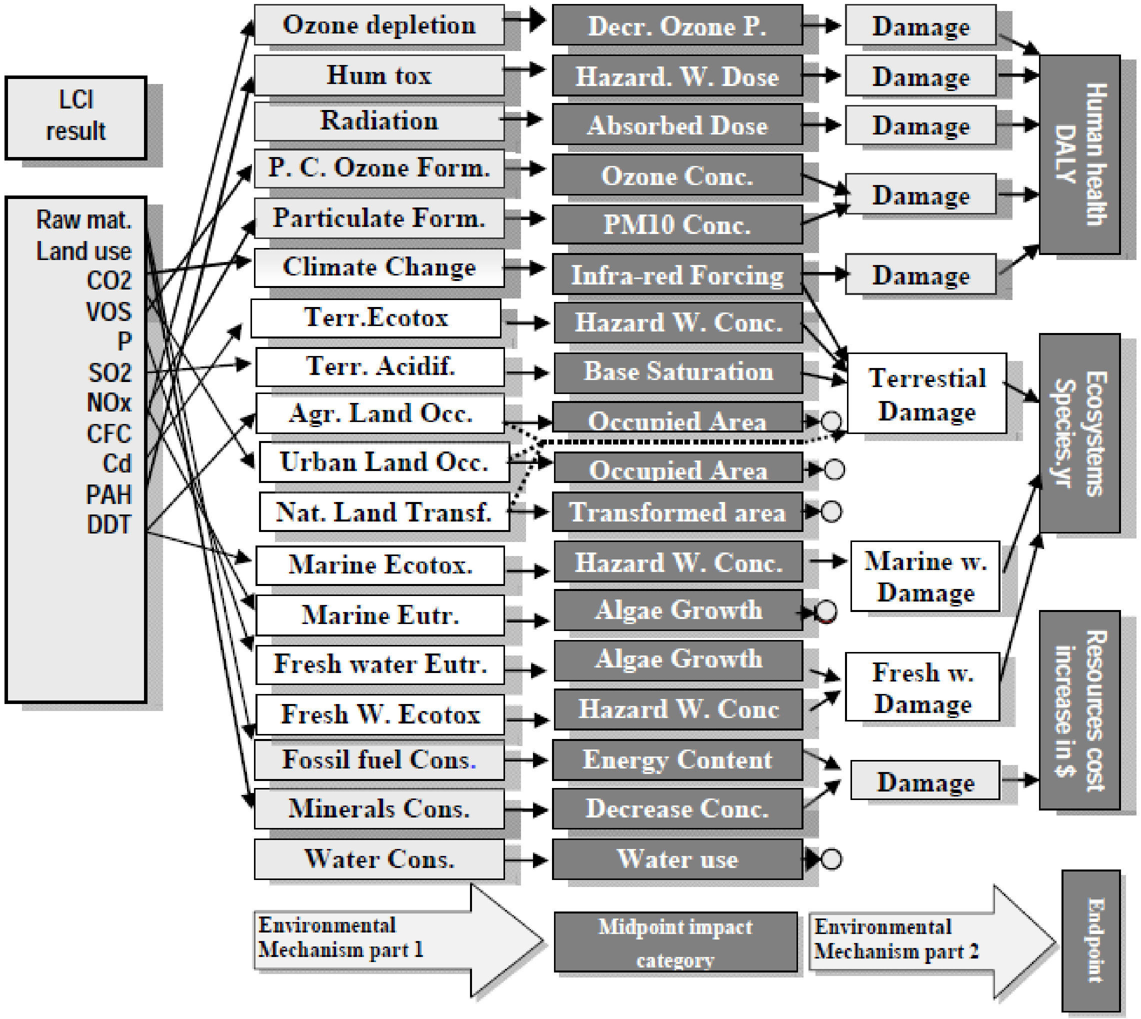
	Terrestrial ecotoxicity Freshwater sediment ecotoxicity Malodours air Marine sediment ecotoxicity Ionising radiation	Stored ecotoxicity Fossil fuels Mineral extraction	Stratospheric ozone depletion Stored ecotoxicity Stored human toxicity Stored nutrient enrichment	Freshwater ecotoxicity Marine ecotoxicity Ionising radiation Agricultural land occupation Urban land occupation Natural land transformation Water depletion Metal depletion Fossil depletion
Endpoint impact categories		Human health Ecosystem quality Resources		Human health Ecosystem Resources
Remarks on implementation in ecoinvent v2.2	Multiple characterisation methods implemented. Normalisation factors not implemented. Explicit handling of long-term emissions.	Three weighting sets (cultural perspectives) included: Hierarchist, Individualist and Egalitarian. Normalisation and weighting implemented for each perspective. Explicit handling of long-term emissions.	Spatially differentiated characterisation models implemented in EDIP 2003, for 40+ European regions. Normalisation and weighting factors not implemented. Explicit handling of long-term emissions.	Three weighting sets (cultural perspectives) included: Hierarchist, Individualist and Egalitarian. Normalisation and weighting implemented for each perspective (except for land transformation and fresh water depletion). Explicit handling of long-term emissions.

Single issue methods → Criteria ↓	Cumulative Energy Demand (CED)	Ecological footprint	IPCC 2007	USEtox	USES-LCA 2.0
Background publication	VDI (1997)	Wackernagel et al. (2005); Huijbregts et al. (2006)	Fourth Assessment Report (IPCC 2007)	Rosenbaum et al. (2008); Hauschild et al. (2008)	van Zelm et al. (2009)
Issue	Energy	Land use	GWP	Toxicity (3000 substances)	Toxicity

Units	MJ	Ha	t CO2eq	Human: CTUh, increase in morbidity in the total human population per unit mass of a chemical emitted (cases per kg) Other: CTUe, potentially affected fraction of species (PAF) integrated over time and volume per unit mass of a chemical emitted (PAF m3 day kg^{-1})	Human: DALY, life years lost or disabled by diseases, which are influenced by impacts. Other: species.yr, potentially disappeared fraction of species over area per year.
Definition	Determination of the primary energy use along the life cycle of a product.	Determination of the sum of time integrated direct land occupation and indirect land occupation, related to nuclear energy use and to CO2 emissions from fossil energy use and clinker production.	Characterisation of different gaseous emissions according to their global warming potential and the aggregation of different emissions in the impact category climate change.	Characterisation of human and ecotoxicological impacts. USEtox is a scientific consensus model based upon a list of previous widely used toxicity models: CalTOX, IMPACT 2002, USES-LCA, BETR, EDIP, WATSON, and EcoSense.	Characterisation of human and ecotoxicological impacts. Implemented in the ReCiPe LCIA method, but not standalone in ecoinvent.
Impact categories	Non-renewable resources (fossil, nuclear, primary forest) Renewable resources (biomass, wind, solar, geothermal, water)	Carbon dioxide, fossil Nuclear (uranium, in ground) Land occupation (arable, construction site, dump site, forest, industrial area, industrial area, benthos, pasture and meadow, permanent crop, sea and	Climate change (GWP 100a, 20a, 500a)	Human toxicity, cancer Human toxicity, non-cancer Ecotoxicity	Extra features, compared to USEtox: Endpoint characterization factors are calculated. Seawater and terrestrial ecotoxicity are also addressed. Various scenario assumptions can be tested

		ocean, unknown)			by changing settings.

Life Cycle Impact Assessment methods implement midpoint and endpoint indicators. Midpoint indicators refer to the environmental mechanisms used to represent potentials impacts (problems) associated to the emission or extraction of substances (e.g. climate change, ozone depletion), while endpoints refer to effective impacts (damages) occurring at the level of “areas of protection” (e.g. human health) (Bare, 2000; Finnveden et al., 2009). Midpoint indicators are considered as more certain, while endpoints are considered as more concise and thus more suitable for informing decision-making (Bare, 2000). The mechanism by which midpoints are consolidated into endpoints in the ReCiPe method, generalisable for other methods, is depicted in the following figure.



Source: Goedkoop et al. (2009)

5 Standards and guidelines for life cycle methods

Life Cycle methods	ISO standards	Other standards and guidelines
Carbon Footprint	ISO 14067 (draft)	British Standards Institution: PAS 2050:2011 (BSi, 2011) World Business Council for Sustainable Development: Greenhouse Gas Protocol guidelines (WBCSD, 2000) International Panel for Climate Change: 2006 IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2006) United Nations Framework Convention on Climate Change: Clean Development Mechanism methodologies (CDM Methodologies, http://cdm.unfccc.int/methodologies/index.html) and tools
Ecological Footprint		Global Footprint Network: GFN (2009)
Life Cycle Accounting and Reporting		Global Reporting Initiative: Sustainability Reporting Framework (GRI, 2006) United Nations Conference on Trade and Development (UNCTAD) and Intergovernmental Working Group of Experts on International Standards of Accounting and Reporting (ISAR): guidelines on corporate responsibility reporting and eco-efficiency (UNCTAD, 2004; UNCTAD/ISAR, 2006, 2008) World Business Council for Sustainable Development: Corporate, value chain and life cycle accounting and reporting standard (WBCSD 2000, 2011a,b)
Life Cycle Assessment	ISO 14040 ISO 14044	Guinée et al. (2001) International Reference Life Cycle Data System: ILCD (2010)
Life Cycle Costing		Society of Environmental Toxicology and Chemistry (SETAC): Swarr et al. (2011)
Material Flow Analysis		Brunner and Rechberger (2003)
Social Life Cycle Assessment		United Nations Environment Programme/SETAC Life Cycle Initiative: UNEP/SETAC (2009)
Water Footprint	ISO 14046 (draft)	Water Footprint Network: Hoekstra et al. (2011)

6 Non-exhaustive taxonomy of sustainability assessment tools and methodologies

Based on Finnveden and Moberg (2005), Haberl et al. (2004b), Hoekstra et al. (2011), Jeswani et al. (2010), Ness et al. (2007), Schepelmann et al. (2009), Štreimikienė et al. (2009), Tukker et al. (2006) and Tyedmers (2000).

Procedural frameworks	Focus/Level	EN	EC	SO
Environmental Impact Assessment (EIA): Multi-tool framework aimed to explicitly consider environmental and social impacts associated to new project developments. Often required by legislation in public projects.	Micro (project)	X		X

Multi-Criteria Decision Analysis (MCDA): Collection of decision support methods aimed to compare alternatives based on a set of decision criteria. Suitable for conflicting decision situations.	Micro, meso, macro (project, policy)	X	X	X
Strategic Environmental Assessment (SEA): Multi-tool framework similar to EIA but oriented to evaluate policy instruments, often in situations of high uncertainty.	Meso, macro (policy)	X		X
Sustainability Assessment (SA): Umbrella term encompassing different methods and tools aiming to comprehensive sustainability assessment. Often benefiting of life cycle methods.	Macro, micro (policy, project)	X	X	X
Analytical frameworks	**Focus/Level**	**EN**	**EC**	**SO**
Cost–Benefit Analysis (CBA): Analysis tool for the assessment of costs and benefits, expressed in terms of money, of projects or activities (often government projects). Used to compare alternatives. Includes the costs associated to environmental and social impacts.	Micro, meso, macro (project, policy)		X	
Eco-Efficiency (EE) Analysis: Concept aligned with the growing environmental concerns of the economic sectors, which can be defined as a management philosophy encouraging business to search for more environmentally-sound alternatives producing similar economic benefits.	Micro (product, service)	X	X	
Energy/Exergy Analysis (EA): Group of methods aimed to account for energy flows occurring in the studied system, usually a process or product system. Exergy refers to energy of certain quality (useful to produce work). *Energy Return On Investment (EROI):* A ratio of industrial energy embedded in a product vs. the energetic content of the product, representing energy efficiency. A variation of EROI, Edible Protein EROI, is used to compare energy efficiency of food production systems.	Micro (process, product, service)	X		
Environmental (Extended) Input-Output Analysis (E(E)IOA): Extension of the established Input Output Analysis (IOA) methodology to include environmental impact data in a sector-wise economic assessment. The conventional IOA monetary datasets are either extended with environmental impact coefficients or replaced with biophysical based datasets. *Hybrid LCA:* combination of IOA/EIOA with LCA usually aimed to provide data for the cradle-to-gate portion (basic industries providing raw materials).	Meso, macro (policy, product, service)	X		
Life Cycle Assessment (LCA): Life-cycle tool aimed to account for the environmental impacts, expressed in a number of impact categories, associated to the provision of a good or service over its whole life cycle. Various existing “footprints” are related to LCA, but focusing on single issues/indicator categories: • *Carbon Footprint (CFP)*: Can be considered as a sub-set of LCA focusing on global warming potential. • *Ecological Footprint (EFP)*: Accounts for the land use associated to the provision of a product. EF can be complemented with Human appropriation of net primary production (HANPP), which studies the proportion of original primary production that remains on a space-specifically defined land area given specific land use practices. • *Water Footprint*: Accounts for the freshwater resource appropriation (including fresh, rain and polluted water	Micro (process, product, service) Macro, Meso (footprints)	X		

volumes affected) associated to the provision of a product, in a spatiotemporally explicit fashion.				
Life Cycle Costing (LCC): Life-cycle tool aimed to account for all the costs associated to the provision of a good or service. Proposed as a complement to LCA.	Micro (product, service)		X	
Local Impact Assessment: Environmental analysis approach for local impacts of activities.	Micro, macro (local impacts)	X		
Material Flow Assessment/Analysis/Accounting (MFA): Systematic accounting of flows and stocks of materials and energy occurring within an economic system, often a whole region or country. *Substance Flow Analysis (SFA):* MFA-type assessment focusing on the fate of specific substances, at the regional or national level.	Macro (policy, plan)	X		
Material Input per Service Unit (MIPS): Estimation of the environmental pressure associated to products and services expressed as a life cycle-wise ratio of natural resources consumption to benefit provided.	Micro (product, service)	X		
Risk Analysis/Assessment (RA): Assessment toolset aimed to environmental, health and safety-related risks associated to projects or product systems (chemicals, hazardous substances, and industrial facilities).	Micro (project, chemicals)	X		
Social Life Cycle Assessment (SLCA): Life-cycle tool aimed to account for all the social impacts associated to the provision of a good or service. Proposed as a complement to LCA.	Micro (product)			X
Total Cost of Ownership (TCO): Can be considered as a limited type of LCC focused on the product user and addressing only the use phase. *Total Cost Accounting (TCA)*: Equivalent to LCC, focusing on less tangible, hidden and liability costs.	Micro (product, service)		X	
Sustainability dimensions: EN - Environmental, EC - Economic, SO - Social.				

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