Broadening the perspective on seafood production:

Life cycle thinking and fisheries management

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Till mina älskade barn, Elias och Ella

Mina barn, ni gillar ju att äta fisk och annat gott från havet? Som ni har hört så pratar mamma om att det kanske fiskas för mycket ibland och att det gäller att vi sparar på våra gemensamma resurser? Det gäller ju i allt vi gör. Man ska ju t ex inte kasta maten eller åka för mycket bil eller flyga, för då slösar vi på våra resurser och skadar miljön mer än vad som behövs. Det ska ju finnas fisk att äta även för era barn och barnbarn. Dessutom är ju avgaser från motorer inte så bra, det påverkar ju vår miljö.

Ett av problemet med fiske idag är att de som bestämmer inte har räknat med alla följder från besluten. Elias, du har ju sagt att man kanske inte ska fiska för många dagar, till och med bara en dag om året om fisken inte räcker till. Det låter ju bra, men det där är lite komplicerat. Det finns så många olika människor som tycker olika saker. En del tycker till exempel att man kan använda mindre effektiva redskap istället, så att man kan fortsätta fiska bara man inte tar upp för mycket av den fisk som det inte finns så mycket kvar av.

I min bok har jag försökt att räkna på detta i svenskt fiske. Jag har räknat på hur mycket diesel fiskebåtarna använder när de fiskar på olika sätt, hur mycket avgaser det blir som påverkar vårt klimat, hur mycket av havets botten som påverkas, hur mycket utrotningshotade fiskar man dödar och kastar för att fånga det man vill och hur energin i ekosystemen rubbas från den del av fångsten som kastas tillbaka död i havet igen. Ett bra sätt att räkna på alla dessa saker samtidigt är att använda sig av en metod som heter livscykelanalys, vilket jag har gjort. Då kan man samtidigt titta på hur mycket alla dessa saker påverkar vår miljö och hur mycket resurser vi använder för till exempel ett kilo fisk.

Vad som är viktigt att komma ihåg från den här boken är att de som bestämmer över hur man fiskar är viktiga för att vår fisk på tallriken inte ska ha onödig resursanvändning och miljöpåverkan. Om de använde sig mer av livcykelresultat när de bestämmer, alltså tänka på att minska all form av miljöpåverkan och resursförbrukning med sina beslut, så kan vi bättre skapa fisken som är bäst för oss alla på alla sätt. Livscykelanalys kan därmed kanske vara ett hjälpmedel för att fundera över hur besluten som tas inom fisket förhåller sig till andra saker vi har lovat er, som att inte släppa ut för mycket avgaser som påverkar vårt klimat och er framtid.

Abstract

Decisions made by fisheries managers strongly influence the overall resource use and environmental impacts associated with the seafood product from capture fisheries. These findings come from Life Cycle Assessments (LCA), a method that aims at quantifying all relevant resource use and environmental impacts throughout the life cycle of a product. In this way, important hot spots or improvement potentials can be found. The integrated systems perspective can assist to avoid shifts in impacts between production phases or environmental concerns. LCA is at present a well-established tool to assess environmental impacts of products, but there is no uptake of LCA-based methods or results in fishing policies.

Methods for assessing fisheries-specific impacts within the LCA framework are however incomplete. One part of the research therefore addressed indicators related to pressures on marine ecosystems from discard to be used in seafood LCAs. Swedish fisheries on the west coast were evaluated using the trophic indicators mean trophic level (MTL) and primary production required (PPR). PPR could to some extent reflect properties of ecosystem resource use as PPR from the total catch, including discards, varied considerably between fisheries. Still, it was shown that it is difficult to interpret both indicators in relation to what is known about the ecosystems and the desired properties of the metrics. Complementing metrics of potential pressures on biodiversity are needed. The Swedish IUCN Red List of Threatened Species for fish was evaluated for this purpose. The Red List was found to be coherent with other assessments of vulnerability of fish to exploitation. Different fishing practices also showed different pressures on threatened fish species (aggregated as VEC). VEC together with PPR may thus be used in seafood LCA.

Another part of the research explored LCA-based approaches as integrated decision support to form an overall sustainable fisheries management. Studies comprised of Swedish demersal trawling fleets. In the *Nephrops* fishery, a trade-off was found from promoting species-selective trawls. Local protection of depleted fish stocks comes with an increase in seafloor area swept, fuel use and associated emissions per landed kilo. Even if the overall fuel efficiency of the Swedish demersal trawling fleet has improved between 2002 and 2010, selective trawling required higher fuel use per kilo landing than the equivalent of less selective practices. Improved fuel efficiency was seen from stock rebuilding of the Eastern Baltic cod. However, in another study, the situation of the Eastern Baltic cod fishery was found to have deteriorated in recent years. Selection towards larger size classes has resulted in detrimental ecological consequences, reverberating into poor fish yield and economy. If overall improvements of the present situation are sought for, fisheries management needs to decrease mesh size and effort in the Eastern Baltic cod fishery, as well as include more metrics to assess sustainability.

LCA-based methods can provide integrated decision support to inform various seafood policies, and integrate more objectives than is currently done in a fisheries policy context. To foster an overall sustainable seafood production, fisheries managers however need to acknowledge their role in this development. Altogether, stronger effort cuts and shifts in gear are proposed, while stressing the importance to use LCA-based assessments in order to avoid shifting from one environmental pressure to another.

Populärvetenskaplig sammanfattning

Jag har studerat resursförbrukning och miljöpåverkan av svenska fisk- och skaldjursprodukter, eller mer specifikt vilken roll fiskets förvaltning har i hur stort avtryck produkten gör på miljön. Detta leder till många frågor. Vilken miljöpåverkan innebär fiske – och vilka mått finns det för detta? Hur kan man utforma nya mått som kan inkluderas i bredare utvärderingar där både kunskapen om miljöpåverkan och tillgängliga mätpunkter idag är bristfälliga? Hur ska avvägningen göras mellan olika typer av resursförbrukning och miljöpåverkan, dagens och morgondagens? Behövs systemperspektiv av förvaltningen eller räcker det med att fokusera på att den målart som fisket riktar in sig på är livskraftigt?

Ett sätt att räkna på flera miljöaspekter samtidigt är att använda sig av livscykelanalys. Livcykelanalys studerar resursförbrukning och miljöpåverkan från en produkt eller process. Med denna metodik får man ett brett och integrerat perspektiv, och kan på så sätt undvika att man överför miljöpåverkan från en typ till en annan eller mellan olika delar under produktens livscykel. För fisk- och skaldjursprodukter kan man t ex sammanlagt titta på hur mycket bottenyta, utkastad del av fångsten och bränsle som ett kilo fisk kräver – och hur det kan förändras mellan olika förvaltningsbeslut. Metoden saknar dock fortfarande en del mätpunkter för miljöpåverkan som är unika för fiske. En del av projektet har därför ägnats åt att hitta relevanta ekologiska mätpunkter för den delen av fångsten som kastas (utkast).

Inom livscykelanalys har man tidigare oftast räknat utkast i kilo och diskuterat artsammansättningen. Det är viktigt, men visar inte hur stor skadan är. Ett sätt att räkna på miljöpåverkan från att kasta en del av fångsten är att räkna på hur mycket av primärproduktionen i haven (i form av kol som fixeras av alger under fotosyntes) som har gått åt för alla de arter som kastades, beroende på vilken plats i näringskedjan de har (trofinivå), som ett förfinat mått på slöseri av ekosystemets resurser. Det finns två välkända mätpunkter relaterat till trofinivå som används för att studera ekosystempåverkan från fiske, medeltrofinivån och primärproduktionsbehovet från fångsterna. Syftet med en av mina studier var att studera svenskt fiske på västkusten i ett hundraårsperspektiv med hjälp av dessa två indikatorer för att utvärdera dem som mätpunkter för uthålligt fiske. Resultaten visade att detta sätt att räkna till viss kan del visa på energiflöden i ekosystemet och kan vara ett mått på hur stor del av ekosystemet som rubbats från olika utkastmängd och artsammansättning. Detta är dock en grov skattning och säger ingenting om t ex de verkliga ekosystemeffekterna eller om det finns gott om de arter som kastas (d v s om de är hotade eller inte).

I nästa studie försökte jag därför hitta ett kompletterande mått för eventuella risker att negativt påverka biodiversitet med utkast. Detta gjorde jag genom att först studera den svenska Rödlistan från ArtDatabanken, d v s en bedömning av de relativa riskerna för utrotning för olika arter. Eftersom Rödlistan visade sig stämma väl överens med andra sätt att bedöma om fiskar är känsliga för fisketryck, räknade jag därför sedan också på hur mycket hotade fiskar som man måste kasta för att få upp det man vill ha med olika fiskemetoder. Jag hittade då skillnader mellan olika fiskesätt, det var t ex störst mängder hotade fiskar som kastades när man fiskar havskräfta med trål, men betydligt mindre med selektiv trålning (som sorterar ut fisk) efter räka. Eventuell påverkan på andra känsliga artgrupper inkluderades dock ej på grund av bristfällig data. Trots det kan kvantifieringen av hotade arter tillsammans med primärproduktionsbehovet ge ett bättre mått på påverkan från utkast än enbart kilo fångst i vikt.

Jag tittade även med hjälp av livcykelanalys och de nya mätpunkterna för utkast på trålfisket efter havskräfta på svenska västkusten. Där förordar förvaltningen ett fortsatt fiske efter kräfta i samma omfattning bara man inte fångar torsk samtidigt. Torsken, som fångas tillsammans med havskräftorna, är nämligen på historiskt låga nivåer och skyddas av en återhämtningsplan inom EU. Svenskt fiske efter havskräfta sker nu alltmer genom att man sätter in ett galler i trålen som släpper igenom torsken som tillåter trålfisket efter havskräfta att fortsätta med enbart en nationell begränsning. Denna åtgärd skapar visserligen ett mindre fisketryck på exempelvis torsken, men det man inte tog hänsyn till i beslutsunderlaget var att om man ser till ett systemperspektiv så ökar bränslebehovet och bottenytan som trålas per kilo fångst som fiskarena tar iland.

En av studierna visade dock en minskande trend för bränsleåtgången per kilo som landas inom bottentrålsfisket i Sverige från 2002 till 2010. Avgörande faktorer har till exempel varit högre fångster per ansträngning för Östersjötorsken, vilket påverkar bränsleeffektiviteten. Men, studien visade även att selektiva trålfisken används alltmer och har en generellt högre bränsleförbrukning per kilo fångst som tas iland än mer fångsteffektiva metoder.

Selektion för att minska utkast kan dock ske på olika sätt. I fisket efter torsk från östra beståndet i Östersjön har man kontinuerligt ökat storleken på maskorna i trålarna för att minska utkast av småtorsk. Detta har inneburit att de större fiskarna har fått ett alltför högt fisketryck, och de mindre ett för lågt. Tillväxtpotentialen för torskarna har därför minskat så att det idag knappt finns några stora fiskar kvar. Förvaltningen har ej tagit i beaktan alla mätpunkter för uthålligt fiske, eftersom fisket anses vara uthålligt förvaltat och är miljömärkt – samtidigt som avsaknaden av de stora fiskarna negativt påverkar ekosystemet och industrins ekonomi. Olika förvaltningsscenarier utvärderades därför i ett bredare perspektiv. Resultaten visade att det bästa alternativet för att skapa en bättre utveckling vore att man fiskade mindre storlekar i kombination med mindre ansträngning, och med ett något lägre satt produktionsmål.

Resultat från livcykelanalyser har ofta visat att förvaltningen av fisket är viktig för att fisk- och skaldjursprodukterna inte ska ha onödig resursanvändning och miljöpåverkan. Tyvärr ser beslutsfattarna inte alltid sin roll i optimeringen av produktionen av fisk- och skaldjur från hav till bord. Om de använde sig mer av ett livscykelperspektiv, alltså att försöka minska all form av miljöpåverkan och resursförbrukning från fisket med sina beslut, så skulle fisket kunna få en mycket bättre miljöprestanda – och likaså produkten. Livscykelanalys kan då vara ett hjälpmedel för att skapa integrerade beslutsunderlag som utvärderar de bredare konsekvenserna av förvaltningsåtgärder i förhållande till andra åtaganden, som bevarande av biologisk mångfald, minska växthusgasutsläpp och användningen av fossila bränslen.

One giant leap for a PhD student, a small step for mankind.

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Last, but certainly not least, I would also like to thank my amazing family. Linnéa, my dearest bonus family member, I am so glad to have you around in every sense! Elias och Ella, mina älskade barn, denna bok är till er. Hoppas verkligen att jag blir smartare och gör som ni vill – har mer ledigt och roligt med er! Totte, the love of my life. Thank you for being you and always there for me and our kids (when you are not away ☺), for better or for worse! Love you all!

List of papers

This thesis is based on the following papers, referred to in the text by their roman numerals. The papers are appended at the end of the thesis.

I. **Hornborg, S.**, Belgrano, A., Bartolino, V., Valentinsson, D. & Ziegler, F. (2013) Trophic indicators in fisheries: a call for re-evaluation. *Biology Letters* 9 (1) 20121050 DOI: 10.1098/rsbl.2012.1050

II. **Hornborg, S.**, Svensson, M., Nilsson, P. & Ziegler, F. (2013) By-catch impacts in fisheries: utilizing the IUCN Red List Categories for enhanced product level assessment in seafood LCAs. *Environmental Management* 52(5): 1239-1248 DOI: 10.1007/s00267-013-0096-7

III. **Hornborg, S.**, Nilsson, P., Valentinsson, D. & Ziegler, F. (2012) Integrated environmental assessment of fisheries management: Swedish *Nephrops* trawl fisheries evaluated using a life cycle approach. *Marine Policy* 36(6):1193-201 DOI: 10.1016/j.marpol.2012.02.017

IV. Ziegler, F. & **Hornborg, S.** (2014) Stock size matters more than vessel size: The fuel efficiency of Swedish demersal trawl fisheries 2002–2010. *Marine Policy* 44: 72-81 DOI: 10.1016/j.marpol.2013.06.015

V. Svedäng, H. & **Hornborg, S.** (manuscript) In waiting for a flourishing Baltic cod fishery that never comes: old truths and new perspectives.

VI. Ziegler, F., **Hornborg, S.**, Green, B.S., Eigaard, O.R., Farmery, A., Hammar, L., Hartmann, K., Molander, S., Parker, R., Skontorp Hognes, E., Vázquez-Rowe, I. & Smith, A.D.M. (manuscript) Expanding the concept of sustainable fisheries: Measuring the sustainability of seafood supply chains using a life cycle perspective.

Related papers (not included in thesis):

Svedäng, H. & **Hornborg, S.** (*under revision*) Improving yields by selective fishing induces density dependent growth.

Longo, C., **Hornborg, S.**, Bartolino, V., Tomczak, M. T., Ciannelli, L. & Belgrano, A. (*under revision*) Are trophic indicators a new paradigm shift for marine fisheries management?

Some abbreviations and concepts commonly referred to

By-catch	The part of the catch that is not directly targeted. Consisting of two parts: one that is utilized (landed); one that is discarded at sea.
Discard	The part of the catch in a fishery that is thrown back to sea, most often dead, and is often not reported. This could be non-commercial species, but also juveniles of target species, quota restricted marketable species or marketable species or sizes with lower economic value (high-grading).
CBD	Convention of Biological Diversity; has several specific targets, one important being reduce the rate of biodiversity loss by 2010, set in 2002, with 168 signatures by governments of the world (www.cbd.int, accessed 29th of October 2013).
CFP	Common Fisheries Policy
CPUE	Catch Per Unit Effort
EAF	Ecosystem Approach to Fisheries
F_{MSY}	The fishing mortality rate which corresponds to MSY
GHG	Greenhouse Gases
ICES	International Council for the Exploration of the Sea, scientific community with participants from all states bordering the North Atlantic including the Baltic Sea. Responsible to e.g. scientific advice to setting quotas within the European union.
IUCN	International Union for Conservation of Nature. Administrates the IUCN Red List of Threatened Species. The overarching goal is to "provide information and analyses on the status, trends and threats to species in order to inform and catalyse action for biodiversity conservation" (IUCN 2014), no legal status.
Landings	The part of the catch that is brought to market and recorded in logbooks.
LCA	Life Cycle Assessment. Environmental systems analysis tool which quantifies resource use and environmental impacts associated to a product or process.
LPUE	Landing Per Unit Effort
MSFD	Marine Strategy Framework Directive
MSY	Maximum Sustainable Yield
MTL/MTI	Mean Trophic Level/Marine Trophic Index, indicator to the CBD.
PPR	Primary Production Required (measured in carbon).
RLI	Red List Index, indicator to the CBD.
TAC	Total Allowable Catch, the maximum allowed amount of a certain stock to be landed per year. The concept has in the EU been confounding as it has not referred to catch, but landing, i.e. not including discards. In the new CFP TACs refers to total catches.
TE	Transfer Efficiency
TL	Trophic Level
VEC	Vulnerable, Endangered or Critically Endangered (according to IUCN criteria), proposed as an impact category indicator in seafood LCA (paper II).

"The three main drivers of the modern degradation of the oceans are overexploitation, pollution in all its myriad forms and the rise of carbon dioxide owing to the burning of fossil fuels—the ultimate mega-pollutant of them all."

Extract from The future of the oceans past by Jeremy B. C. Jackson (2010)

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Introduction

Global seafood production from capture fisheries could be seen as troublesome. Landings have stagnated or declined since the 1990s, meanwhile seafood consumption per capita is increasing and is important to human nutrition (FAO 2012). This development could negatively affect sustainable use of fish resources if not properly mitigated by fisheries management. On the positive side, fishing mortality has in recent years decreased in some areas (Worm *et al.* 2009, Cardinale 2011). This may allow recovery and possibly more fish production, especially from improved management of un-assessed fish stocks (Costello *et al.* 2012). It has thus been argued that it is possible to meet the demand for seafood of future generations with better governance of capture fisheries in combination with higher contributions from aquaculture practices that are less dependent of fish as feed (Merino *et al.* 2012). However, on the negative side, collapsed stocks risk to require long-term recovery periods (Hutchings 2000), and current fishing pressure could cause ecosystem overfishing and might therefore require to be considerably decreased (Coll *et al.* 2008, Chassot *et al.* 2010). Altogether, given the constraints of ecosystem capacity to produce fish, immediate increase in production from capture fisheries lies at present arguably mainly in better utilization of what is caught. This involves improvements in the supply chain and less discard. If and when overfished stocks are rebuilt, further fish production may be possible, but in terms of when and how much is left unanswered.

Even if establishing long-term sustainable exploitation levels is top priority, there is still an emerging need to include additional environmental aspects of fisheries. This involves mortality rates for vulnerable species and habitat alteration (Casey & Myers 1998, Watling & Norse 1998, Lewison *et al.* 2004, Puig *et al.* 2012). These aspects are addressed in approaches such as an Ecosystem Approach to Fisheries (EAF, FAO 2003), a framework that include broader ecosystem considerations and is one direction towards sustainable development of fisheries.

There are however some aspects that have so far been paid less attention in fisheries policy and advice. In a fossil fuel sparse and high carbon dioxide world, fuel use development and influence from management decisions should also be of concern, especially with stagnated landings while fishing effort increases (Watson *et al.* 2012). Fuel use per landings has in fact been shown to increase over time in some fisheries (Hospido & Tyedmers 2005, Schau *et al.* 2009). Managers of fisheries do not seem to consider fuel demand to be a problem, as poor profitability caused to some extent by high fuel intensity is mitigated by subsidies (Arnason *et al.* 2008). Even if the fishing fleet is not a major player in a global context, with an estimated requirement of 1.2% of global oil consumption, differences in fuel intensity are vast between fished species (Tyedmers *et al*. 2005). Fuel use development could also be further exacerbated from the rapid increasing contribution of invertebrates to global fisheries, with the predominantly used gears being demersal trawls and dredges (Anderson *et al.* 2011). Besides generally high rates of discard and benthic disturbance, trawling for invertebrates is energy intensive (Ziegler & Valentinsson 2008). In fact, energy intensity of invertebrate fisheries is extremely high compared to other agricultural and aquacultural food production systems (Pelletier *et al.* 2011). Due to the short-term highly profitable catches, fishermen may not see this development as negative; in the longer run, this development may however involve greater ecological as well as socio-economic risk taking (Steneck *et al.* 2011, Howart *et al.* 2013).

Life Cycle Assessment (LCA) is seen as an important tool for sustainability assessment of products (Zagmani *et al.* 2013). LCA's of seafood production systems began in the early 2000s and have since then attracted increasing interest. Seafood LCAs have repeatedly identified management of fisheries as a critical component to overall environmental impacts associated to a seafood product: choices of gears, effort, and quotas are important components in the overall environmental impact of the end product (Thrane 2004, Ziegler & Valentinsson 2008, Driscoll & Tyedmers 2010). Even though managers pay no attention to the fuel efficiency resulting from different management regimes, their decisions do affect fuel efficiency of fisheries. In contrast to lack of interest in fisheries management, there is an increasing interest in accounting for and monitoring greenhouse gas emissions from seafood products, and standards have been initiated in Britain (PAS 2050-02) and Norway (NS 9418).

Nevertheless, fuel intensity and resulting emissions, often the main impacts of standard seafood LCAs, are only two aspects of environmental impacts from fisheries. Methodological development is needed to include more ecological impacts from fishing within the LCA framework (Pelletier *et al.* 2006, Vázquez-Rowe *et al.* 2012a). This is particularly important in order to enable fair LCA based sustainability assessments of food production systems to certification, procurement and not the least, the public debate. As an example, it has been argued that protein from capture fisheries does not require pesticides, fertilizers, land- or water use. These are all important components to agricultural food production. In fact, fisheries could instead represent extremely energy efficient protein production systems, and emit comparatively low amounts of greenhouse gases (Hilborn & Hilborn 2012). However, fisheries are completely different production systems. They depend on natural ecosystems, and have impacts that are unique to fisheries. In order to enable sound product comparisons, there is a need to develop common assessment grounds and expand existing integrated tools.

The studies in this thesis address some potential indicators of ecological effects from fishing that could be useful to add to the existing framework of LCAs, with focus on impacts related to by-catch (in particular discards). In parallel, the potential of utilizing life cycle thinking to obtain an integrated decision support to form an overall more sustainable fisheries management is explored and further discussed.

Aim

There are two overarching aims of the studies in this thesis that are interlinked with each other:

1. To identify, develop and apply indicators of potential ecological pressures from by-catch in fisheries for use in seafood LCA.
2. To explore how a life cycle based approach could be used as a management tool in capture fisheries.

The studies in the first part thus explore potential indicators that could be useful to quantitatively characterize ecosystem pressures from by-catch (**paper I, II, III**).

Part two consists of case studies applying life cycle thinking and the influence on fuel use from different management measures (**paper III, IV, V**) and a review of the role of LCA in relation to other sustainability assessments of fisheries (**paper VI**).

Methodological approach

In short: performing an LCA

Life Cycle Assessment (LCA) aims at quantifying all relevant resource use and environmental impacts linked to the study object, either a product or a process, throughout the life cycle (i.e. from extraction of raw materials to waste or re-cycling). In this way, important hot spots or improvement potentials can be found. LCA thus enables an integrated systems perspective and helps to avoid shifting environmental burdens between production phases or environmental concerns.

LCA consists of four stages, however with an important iterative evaluation of the result from choices made in previous stages:

Goal & scope	Definition of e.g. the aim of the study, system boundaries (e.g. processes and data to include), the study object (functional unit), impacts to be studied and other technical aspects such as allocation procedure, i.e. deciding on how to distribute environmental impacts between multiple outputs.
Inventory	The most time-consuming task, where data required as defined in goal and scope are collected for each step and quantified in relation to the functional unit.
Impact assessment	Resource use and emissions are grouped into impact categories and weighted together based on their relative potential to contribute to impact, called characterization. For example, GHG emissions are weighted according to IPCC standards and measured as kg CO_2 equivalents (Fig. 1).
Interpretation of results	The robustness of the results is tested by e.g. sensitivity analysis, possibly resulting in changes of choices made earlier.

Many processes have multiple products. Strategies for distributing the environmental impact and resource use between the different products, called co-product allocation, have therefore been developed. In fisheries, this applies mainly to two situations: several species being landed together, and in processing into various edible products and non-edible parts. Preferably, according to the ISO standard (ISO 2006a), allocation should if possible be avoided. This could be done by increasing the level of detail (sub-dividing the system) or system expansion (using an alternative production system for one of the co-products). Otherwise, co-product allocation can be based on e.g. the relative mass, energy or protein content of the co-products, or as the last alternative, based on their relative economic value. This order of allocation procedure is not accepted by all practitioners, as there are draw-backs to all approaches (Pelletier & Tyedmers

2011, Svanes *et al.* 2011). Altogether, it is important to remember that these choices affect the results and complicate comparisons.

It is also important to consider that while the systems perspective is useful in including a broad range of impacts and avoid problem-shifting, it still represents a generalized overview of a system and thus involves simplifications. Being an interdisciplinary approach, life cycle impact assessments include several sub-components that themselves are their own research areas. The synthesis thus provides a new and integrated perspective, without going in depth with the details. It is however important that it still provides a relevant perspective for the intended purpose. In other words, an LCA practitioner can provide a broad picture, but only to a certain extent be aware of each topic in detail. In turn, the LCA practitioner is as a result less aware of uncertainties in the modelling procedure, such as potential regional differences in impact pathways, synergetic and accumulative effects.

For further reading on LCA methodology see e.g. Baumann & Tillman (2009).

A note on impact assessment in LCA

Life Cycle Impact Assessment involves grouping of category indicators, such as greenhouse gas (GHG) emissions, of relevance to an impact category (e.g. global warming potential) based on their relative attributes to cause an impact (Fig. 1). These impact categories are called “mid-points” and are expressed as “potentials” (i.e. addressing environmental pressure, problem-oriented). This is the most commonly used framework. It should be noted that there are methods for grouping these impact potentials further towards “endpoint” categories (i.e. addressing environmental impact, damage-oriented). Such an approach involves weighting the different environmental impact categories and form one single score (such as impact on Natural Environment).

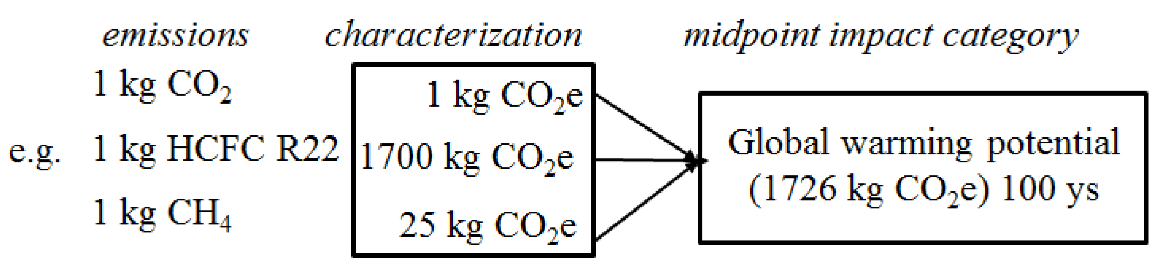

Figure 1. Impact assessment of GHG emissions (midpoint). All emissions causing climate change are related to the impact of carbon dioxide, turned into CO_2-equivalents and weighted together according to their relative radiative forcing into a single score.

The LCA method is still young and improving, and methodological development is needed both in a general context (Finnveden *et al.* 2009) and to address all environmental impacts of relevance in seafood production (Pelletier *et al.* 2007). At present, LCAs encompass a wide range of impact categories such as eutrophication, toxicity, acidification, ozone layer depletion and global warming potential. For seafood products from capture fisheries, many studies have shown dominance of fuel use and derived emissions (Vázquez-Rowe *et al.* 2012a). As they all correlate with the fuel consumption, this suggests that GHG emissions can be used as a proxy for other emissions and less attention can be paid to these other traditional impact categories.

Impact indicators should be quantitative, linked to a functional unit (additive indicator) and fulfil the requirements of the ISO-standard (ISO 2006b):

"*the impact categories, category indicators and characterization models should be internationally accepted, i.e. based on an international agreement or approved by a competent international body*"

Impact assessment in LCA is in particularly complex in terms of assessing impacts that are more complex to quantify, such as impacts on biodiversity (Curran *et al.* 2010). It is impossible in seafood LCAs to comprehensively assess the potential ecosystem effects from removing one kilo of biomass out of an ecosystem (i.e. fishing). Even so, in order for the methodology to fulfil its comprehensive scope, this should in theory be required. Omitting certain environmental aspects, such as potential effects on biodiversity, due to lack of methodology may limit the potential usefulness of the results. In addition, as impacts are intended to be independent of site and time, site-specific or regionalized impacts in the LCA framework has also been limited (Reap *et al.* 2008). Methodological development in this area is currently a hot research topic in LCA. These local impacts are of great importance to the credibility of LCAs of seafood products, as impacts from fishing activities are generally of local concern and most people would say that they are the most important effects of fishing.

Inventoried data may also have a stand-alone importance and could lack methods for assessing impact potentials. In such cases, data could also be presented as quantified results related to the functional unit. There have also been discussions on the benefits of including descriptive indicators (Kruse *et al.* 2009). Such indicators would be required for many socio-economic aspects of sustainability: fair wages, working conditions, etc.

A lot of research has been done in terms of developing ecological indicators for fishing impacts on marine ecosystems outside the context of LCA. Rice (2003) argues that choosing candidate indicators objectively is difficult. For example, questions would relate to which biodiversity metric could be used that is not affected by multiple stressors (as is often the case in coastal zones), not to mention lack of scientific consensus of desired ecosystem status. Rice and Rochet (2005) proposed a step-wise procedure in order to as objectively as possible develop indicators for fisheries management. They suggested that in the initial step, it is important to define intended users and their needs. After that, a suite of candidate indicators can be developed, whereas in the next step, their usefulness should be evaluating based on criteria such as public awareness, theoretical basis and cost in relation to the intended audience. As for LCA results, they are to be communicated to certification, managers, industry as well as the general public. It is therefore most likely that different indices of potential impacts from fisheries are needed for different stakeholders depending on intended purpose of decision support.

LCA has become a regular practice to justify the implementation of environmentally-oriented decisions at cooperative and/or political level (Finnveden *et al.* 2009). Current applications involve e.g. product development, changed sourcing strategies and communication such as in certification schemes and environmental product declarations (**paper VI**). Another intended area of use is for policy-making, such as to follow up on effects of policies adopted, designing new policies or to evaluate the broad environmental effects of alternative future policies.

It should also be noted that there are several different LCA concepts such as life cycle cost (LCC), social life cycle assessment (SLCA), life cycle management (LCM), and life cycle thinking (LCT); the common denominator to all is the systems perspective and having an interdisciplinary approach (Zagmani *et al.* 2013).

Environmental impacts of fisheries

Definition of the research topic

This thesis does not intend to cover all possible ecological impacts from fisheries, nor reviewing all indicators related to ecosystem pressures. A lot of research has been done related to the broad impacts from fishing on the marine environment by experiments, observations and models. To mention the outcome of one study, Fulton *et al.* (2005), it was found that a suite of indicators is required, including four major biological groups: species with fast turnover rates, targeted species, habitat defining species and charismatic/sensitive groups in order to detect fishing impacts.

Another strategy to assess potential environmental impacts from fisheries would involve indicators relating to (Thrane *et al.* 2009):

1) target species or stock (e.g. Hutchings 2000, Jackson *et al.* 2001)
2) by-caught species: commercial, non-commercial and/or threatened (e.g. Casey & Myers 1998, Lewison *et al.* 2004)
3) benthic habitats (e.g. Watling & Norse 1998, Puig *et al.* 2012, van Denderen *et al.* 2013)
4) emissions from fuel use and cooling agents on-board fishing vessels and use of anti-fouling paint (e.g. Ziegler *et al.* 2003, 2013)

Several of these impacts are interlinked with each other and associated with broader and more indirect pressures that affect ecosystem structure and processes. These comprise of ecosystem processes such as potential deficit of food resources to marine mammals and birds (Smith *et al.* 2011), effects on benthic species communities from discards (Bergmann *et al.* 2002a), and trophic interactions (Casini *et al.* 2009).

The aforementioned four categories of potential environmental impacts from fisheries have been addressed in seafood LCAs to various extents. The most obvious pressure from fishing activities is the impact on the targeted stocks, being either on abundance, size structure or the range of a species. However, seafood LCAs have so far poorly covered this impact (Vázquez-Rowe *et al.* 2012a). In terms of potential impacts relating to the availability of these resources to humans, there have been suggestions that impacts from harvesting fish and timber could be modelled on the basis of production capacity, extraction rate and recovery. To in turn assess potential ecosystem damage, making use of the IUCN Red List has been suggested (Lindeijer *et al.* 2002, Pennington *et al.* 2004 and references therein). No seafood LCA study has however so far incorporated these methodological approaches (Vázquez-Rowe *et al.* 2012a). Instead, Primary Production Required (PPR), Mean Trophic Level (MTL) of landings and the Fishing in Balance-index, all related to trophic interactions, have been applied in one attempt to discuss impacts on targeted stocks (Ramos *et al.* 2011). Emanuelsson *et al.* (in press) most recently tried to quantify the distance to optimal exploitation level according to the Maximum Sustainable Yield (MSY) framework as a way to quantify overfishing of target stocks.

As for seafloor impacts, Nilsson and Ziegler (2006) developed a model for seafloor area swept per effort hour deployed and further discussed these results in terms of aggregation of effort, frequency and habitat sensitivity to disturbance. This assessment is data intensive, but the study found that some areas were in a permanently disturbed state due to trawling effort while others were less affected. In **paper III** and **V**, seafloor area swept was included merely as a function of fishing effort in trawl hours. This is a crude measure of impact, both in terms of estimates of the actual area impacted by the gear and the potential disturbance. The approach in **paper III** and **V** does not consider important factors such as aggregation of fishing effort relative to trawl free zones, frequency or recovery time. Such figures on seafloor area impacted are thus difficult to interpret, as the impact from the first time a trawl passes is substantial (Cook *et al.* 2013), whereas the expected impacts in the longer perspective are harder to predict (van Denderen *et al.* 2013). Ellingsen and Aanondson (2006), however, used merely area based metrics to compare production systems for chicken, farmed salmon and wild-caught cod. In this sense, simple land use metrics can provide some interesting insights such as land or sea use requirements from different sources of protein.

The main focus of studies done in this thesis in terms of methodological advancement of LCA has been to find and make use of potential indicators for further refinement of impacts related to by-catch. By-catch has before been regularly quantified in terms of discard ratios in mass, and qualitatively discussed in terms of potential effect on target species (e.g. Ziegler *et al.* 2003, Ziegler & Valentinsson 2008). The selection of potential indicators to study was guided by the scientific literature and has primarily focused on operational indices to be applied in an LCA context, as recommended by the ISO standard (ISO 2006b). As the project progressed, the research area was further narrowed down to study, in particular, discard of fish and commercial invertebrates, mainly due to availability of data.

On by-catch and discard in capture fisheries

By-catch could be defined as the non-targeted part of the catch which could either be landed or discarded at sea (Kelleher 2005). Important to note is that discards could also consist of juveniles of target species, species of less commercial value and quota restricted target species. By-catch is thus not a straightforward sub-set component. Davies *et al.* (2009) coined another definition of by-catch, “un-used or un-managed”. By this definition, roughly 40% of global catches were classified as by-catch. In this sense, by-catch could be seen as less regulated landings.

The part of the catch that is not landed, discards, varies considerably between different fishing practices. The estimated global weighted average is that 8 % of the catch is discarded at sea, however, the range could be 0-98 % of the catch between different fisheries (Kelleher 2005). Reasons behind discards in fisheries are numerous, e.g economic, social, institutional (see e.g.

Feekings *et al.* 2012). It should also be noted that the survival potential of discarded animals varies greatly depending on species, depth, trawl time, water temperature, deck time and more (Suuronen 2005), but is in general low.

By-catch that is discarded could in a sense be separated into two areas of concern: (a) *waste of resources*, it could depending on extent affect the sustainability of the fishery in terms of use of ecosystem production capacity (Coll *et al.* 2008) and (b) as a potential *biodiversity threat* of vulnerable species (Casey & Myers 1998, Hutchings 2000, Lewison *et al.* 2004). This dual approach was in **paper I** and **II** adopted to find indicators of by-catch impacts within the LCA framework.

The resource use perspective: trophic indicators and trophic interactions

Humans have dramatically affected food webs on land, in freshwater and marine ecosystems (Estes *et al.* 2011). Due to a continuous increase in human appropriation of the available primary production of the planet, it has been suggested that policies are needed to slow down this development (Imhoff *et al.* 2004).

Primary Production Required (PPR) is a metric which addresses ecosystem energy flows. It represents an estimate of the amount of carbon required from photosynthesis to produce one kilo of biomass of a species at a certain trophic level (Ryther 1969, Pauly & Christensen 1995). Species at higher trophic levels thus imply higher ecosystem costs. In this sense, PPR could be seen as the currency relative to the total available primary production of an ecosystem, i.e. the carrying capacity (ICES 2005, Swartz *et al.* 2010). The total amount of PPR of fisheries has also been shown to globally exceed levels of sustainable exploitation (Coll *et al.* 2008, Chassot *et al.* 2010, Watson *et al.* 2013).

In terms of by-catch being of concern to resource use, method development thus benefits from being discussed from a trophodynamic perspective. From acknowledging ecosystem energy flows and production related to transfer efficiencies (TE) and trophic levels (TL) of species, ecosystem properties and function are better addressed than from discard ratios in kilos or species count that has been done before in LCA (e.g. Ziegler *et al.* 2003, Ziegler & Valentinsson 2008). PPR has also been used in LCA before to address impacts related to target stocks in capture fisheries (Ramos *et al.* 2011) and in the form of Biotic Resource Use (BRU) on land or in aquaculture systems (e.g. Pelletier *et al.* 2009, Papatryphon *et al.* 2003).

PPR of landings, together with MTL, are the most common trophic indicators in use. MTL was presented in a study by Pauly *et al.* (1998) that concluded that the MTL of global landings was decreasing. They suggested that this was an indication of a sequential depletion of top predators by overexploitation, and that fisheries increasingly had to shift towards lower trophic level species. This metric is addressed in the form of the Marine Trophic Index (MTI) as an indicator within the Convention of Biological Diversity (CBD, 2010 Biodiversity Indicators Partnership).

However, both PPR and MTL have been heavily debated in the scientific community; in particular the MTL concept (see e.g. Baumann 1995, Caddy *et al.* 1998, Branch *et al.* 2010). It should also be noted that several other trophodynamic indicators have been suggested and evaluated in terms of addressing ecosystem impacts of fishing. One problem is that they have in general have been found to be conservative, and respond slowly to changes in fishing pressure (Cury *et al.* 2005).

The aim of **paper I** was to study the trends in PPR and MTL of landings, survey data and total catch data in a well-studied fishing area and in a long historical perspective. This was done in order to analyse the pros and cons of using PPR and MTL for various purposes, one being as an indicator of resource use from discards in LCA. In our study, values for PPR and MTL for landings showed initially an increasing trend, until a breakpoint in the regression identified a decline in MTL commencing before the 1930s, while PPR has declined since the 1990s respectively. The trends correlated poorly with survey data.

The interpretation of the PPR and MTL trends found in **paper I** are however complicated. It was shown that the introduction of a species-selective grid in recent years contributes to low MTL of landings while protecting depleted fish stocks from fishing pressure. In this case, low MTL is the result of an important conservation measure. If a decline in MTL is to be interpreted as a negative signal and a pressure on biodiversity, using total catch data is therefore of vital importance in order to not draw any erroneous conclusions. At the same time, low MTL reflects a change in abundance of top predators induced from overexploitation. Formerly commercially important top predators have dramatically declined in these waters (e.g. Svedäng & Bardon 2003, Cardinale & Svedäng 2004, Cardinale *et al.* 2014). From the trends seen in MTL in **paper I**, it could thus be appropriate to consider the initial increase seen in MTL of landings to be a “pressure” indicator, i.e. increased targeting of higher trophic level species, whereas the decline in MTL of landings with the onset of industrialized fisheries could more be characterized as a “state” indicator, i.e. less predatory fish are available due to sequential depletion. At the end of the time series, low MTL is more of a “response” indicator, as species-selective grids are in place to protect top predators from fishing pressure.

Maybe, this is where the core of the difficulties in interpretation of the MTL trends lies. Depending on data used, it can be an indicator of pressure, response or state. In this sense, Stergiou and Tsikliras (2011) made a point regarding the major challenge to utilize MTL. The “fishing down” theory, i.e. that overexploitation of top predators causes sequential depletion and leaves ecosystem structure altered in shape, could in reality only be falsified if an ecosystem subjected to intense fishing exhibited an increase in biomass and mean length of large predators. This situation is difficult to imagine. However, there is no doubt that a declining trend in the MTL metric will not be detected due to limitations in what could be interpreted from the data used. The importance of not over-aggregating geographical area and use total catch data was also underpinned in Pauly and Palomares (2005), and leads back to how the signal can be interpreted in **paper I** in relation to how the global trend in MTL is interpreted by the CBD.

MTL may however capture ecosystem structure changes as it is. In a study from the Celtic Sea by Pinnegar *et al.* (2002), an initial decline in MTL of landings was shown. This was followed by a decline in survey data, suggesting that there had been changes in the ecosystem structure, not only in fisheries targeting pattern.

Still, for both MTL and PPR, there is a lack of objectives in policy. It is not as obvious what are negative trends in the metrics; it is more straightforward to interpret trends in greenhouse gas emissions. Most likely, ecosystem structure and function were completely different before industrialized fisheries began (see e.g. Jennings & Blanchard 2004). This raises the question: What is the optimum state of PPR by fisheries and MTL of an ecosystem to strive for?

In **paper I** it was also clearly shown that PPR of landings need to pay more attention to fishery-to-fishery specific discards. The proportion of PPR attributed to the landed part varied considerably, 22-88 % of total catch, depending on the targeting pattern. This could strongly affect results of global analyses such as those of Swartz *et al.* (2010) and Coll *et al.* (2008). It could however be argued that as discards are returned to the ecosystem, they could not be considered as resource use by fisheries. The ecosystem can still benefit from the resource in terms of e.g. benthic scavengers (Bergmann *et al.* 2002a). Still, PPR of discard does represents a metric for disturbed energy flow. Discards are of unknown fate and effect (Evans *et al.* 1994), and benefits are not to all species, but could instead induce species community changes such as are seen for birds (Bugoni *et al.* 2010). It should also be noted that attributing PPR to discards was in fact independently suggested by another LCA research group during this project (Vázquez-Rowe *et al.* 2012b).

There are also major challenges with PPR in terms of understanding and defining the impact pathway: Could primary production consistently be seen as limited? Pauly and Christensen (1995) estimated that a range between 20-85% of primary production was required by fisheries in non-tropical shelves. In Chassot *et al.* (2007), a 30% PPR from fisheries was considered to be high. Merely the Swedish landings have required 20-25% of the total primary production in some years (**paper I**), but the additional Danish landings from the same area have been much greater and would significantly add to these figures (Nielsen & Richardson 1996). Meanwhile, increased nutrient loads cause dead zones (Richardson & Heilmann 1995, Diaz & Rosenberg 2008). There have been major changes in fish communities in the Kattegat (Pihl 1994) and overfishing exacerbates the deterioration by eutrophication of important nurseries of seagrass habitats (Baden *et al.* 2012). It could then be speculated that fishing in combination with eutrophication has disturbed the energy flow in the system to the point of multi-functional disturbance. Could the trophic linkages have been affected so that energy is not any longer sufficiently assimilated by available biomass or is too slow to respond? A low PPR in recent years, which from the intended use of the metric should be interpreted as a positive signal, could therefore be confusing, as there is obviously a need to restore the balance and function of the ecosystem.

When modelling PPR and MTL, uncertainty of several parameters highly affects the robustness of the results. There are great uncertainties associated to the trophic level of a certain species, as this is affected by e.g. ontogenetic shifts, area and season. There could also be shifts in the diet in longer perspectives induced by fishing and changes in climate (Heath 2005, Christensen & Richardson 2008). Thus, as diet data are not available for each species on a regular basis, assumptions must be made regarding TL of species. This makes it difficult to grasp the "true" ecosystem effect on MTL in species communities and PPR from fisheries in longer time series. One example of problems with trophic level estimates was shown by Jennings *et al.* (2002). They suggested that changes in size structure was a better predictor of fishing effects than changes in TL as the declining trend found in MTL of demersal fish communities in the North Sea was weak and highly affected by which data were used. In addition, transfer efficiency, i.e. the assumed proportion of prey production taken by predators, is derived from ecosystem modelling and has been found to be affected by a range of factors, such as fishing intensity, size and depth of the ecosystem (Heymans *et al.* 2012). It may also not be suitable to assume constant TE along the food chain. Higher trophic level species have to invest more energy to find food which decrease efficiency, and TE decrease with increasing number of feeding links in the food web (Ryther 1969, Iverson 1990, Baumann 1995). Assuming constant conversion ratios of carbon to wet weight independent of age and TL is not correct either, as higher trophic levels and older ages are characterized by greater respiratory losses (Lindeman 1942).

As a result, shorter food chains (e.g. upwelling zones) have also more robust PPR values than longer food chains (e.g. temperate shelves) due to the modelling procedure. The small uncertainties in TE or TL propagate in the PPR model in longer food chains and have a major effect on uncertainties in the PPR result. Higher trophic levels do not only imply greater PPR values, but also greater uncertainties (Fig. 2). The influence of differences in TE between ecosystems could also call for using different TE values for species caught in different ecosystems, which is not presently done in PPR estimates in LCA.

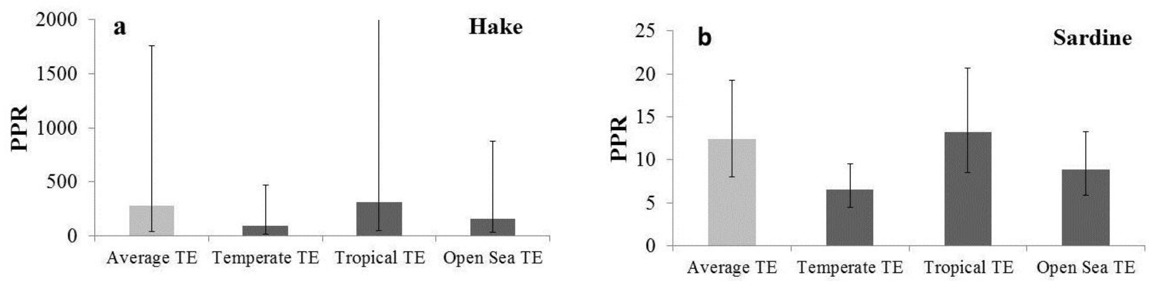

Figure 2. The effect of TE and TL uncertainty on a) hake (*Merluccius merluccius*) and b) sardine (*Sardina pilchardus*). Higher TL, as in the hake case, does not only imply higher PPR values, but also higher uncertainties compared to the sardine case with a lower TL. Data on TL from FishBase, TE from Coll *et al.* (2008).

In terms of applicability of PPR in LCA, consistent assessments are complicated. This includes both comparisons between fisheries from different ecosystems as well as in relation to estimates of terrestrial PPR from crops and livestock. As an impact category in LCA (in the form of Biotic Resource Use), higher values imply greater impact. PPR should thus be low in fisheries. A low PPR from a fishery could however also imply low MTL, which is interpreted as a negative signal in other assessments of fisheries (2010 Biodiversity Indicators Partnership). It is also tempting to use PPR to compare different seafood production systems. The PPR of farmed salmon has been found to have a weighted average (depending on feed composition) of 89 kg C/kg live-weight salmon (Pelletier *et al.* 2009). A large Atlantic cod in a natural ecosystem can have the equivalent of 279 kg C/kg live weight (at TL 4.4 and TE 10%). The question is: Could farmed salmon and a cod from capture fisheries be compared in this sense? One aspect to consider is the great implications of different TE values for the PPR of cod. For the farmed salmon, variability in PPR from different feed formula could also induce shifts in environmental impacts, such as impacts related to land use if agricultural products are utilized instead. It is also important to acknowledge the total biomass removal from fisheries in relation to available production beyond the specific fishery studied. This varies considerably between different ecosystems (Coll *et al.* 2008). In addition, fisheries are to a greater extent dependent on local natural production, which can be exceeded and impaired. Agriculture and aquaculture are man-made systems with external inputs such as feed, fertilizers and pesticides. Altogether, the applicability of PPR is restricted due difficulties in what could be interpreted from the values.

It should also be noted that the preferred human diet from marine fisheries is most often at incomparably higher trophic levels than those on land. Intermediate trophic level species in the marine ecosystem such as herring is the equivalent of bears in terrestrial ecosystems, while tuna has no terrestrial counterpart (Duarte *et al.* 2009). Marine ecosystems can also be different from terrestrial and freshwater systems, as species has been found to be more highly connected in the marine food web than could be expected (Link 2002). In fact, the connectivity between low trophic level species and other components in the food web has been found to be an important predictor of ecosystem impacts from fishing at various trophic levels (Smith *et al.* 2011). This is not accounted for in PPR.

Altogether, comparing merely PPR values between different marine ecosystems, as well as between terrestrial and marine ones, still leaves many questions to be answered related to the full impact of disturbed flows.

Biodiversity threats: extinction risks and threatened fish species

The perception of that fish resources are inexhaustible (Huxley 1883), mainly because of the high fecundity of many fish species relative to other taxa, have been changed as extinction risks for fish have been further understood (Reynolds *et al.* 2005). Other life history traits than high fecundity make fish vulnerable to overexploitation (Sadovy & Cheung 2003), such as late age at maturity (Jennings *et al.* 1998). There is also still much to learn about stock recovery rates. Few depleted fish populations recover rapidly. In a study of 230 stocks by Hutchings and Reynolds (2004) it was found that 15 years after collapse, most stocks exhibited little or no change in abundance, despite reduction of fishing mortality. In fact, it has recently been suggested that recovery rates are (on average) in the same range of that of many terrestrial species (Hutchings *et al.* 2012). This is an important finding, as the perception in fisheries has predominantly been that fish species, based on the fecundity metric, are at low risk of extinction – even when declines of over 80% have been shown for some species (Reynolds *et al.* 2005).

The IUCN Red List Categories and Criteria is considered to be the most widely accepted system for classifying extinction risks of species (IUCN 2014), and the IUCN Red List Index (RLI) is adopted as an indicator within the CBD (Butchart *et al.* 2004, 2010 Biodiversity Indicators Partnership). It is however not straightforward to set universal extinction risks across species and ecosystems. Estimating threat status for commercially exploited fish species has in particular been very hard to reach consensus for, as they are under a management regime that affects their abundance (Mace *et al.* 2008). Still, even if addressing species extinction risks is of importance, ecosystem services, such as production, may diminish faster than species loss and it could be argued that local extirpation should be of greater concern (Schindler *et al.* 2010). In this sense, utilizing the IUCN framework could be seen as a measure which potentially underestimates the impact on biodiversity. Even so, in spite of many fish having been listed as threatened with extinction, there has been no record of a complete extinction of a fish species (Roberts & Hawkins 1999). Threat status could also fail to be valid if the Red List is not satisfactory updated, and concern has been expressed over the credibility of the IUCN Red List if greater efforts are not put in (Rondini *et al.* 2013).

Even though there are several methods to assess extinction risks, decline rate has been the most commonly used criteria for fish (Dulvy *et al.* 2004). Decline rate can be troublesome. According to species abundance distribution, ecosystems are generally composed of a few very abundant species, whereas most of the species are very rare. This results in that it is easier to detect declines of more abundant species than for those that are rarer, as it takes longer time before survey data have the power to detect a true decline (Maxwell & Jennings 2005). Also, historical depletion not accounted in data, makes it likely that the extent of decline is underestimated (Hutchings & Baum 2005). If a species has declined to a stable but historically low level of abundance, the decline rate criteria could also result in a species not being seen as threatened any longer.

By-caught species in multi-species fisheries are at increased threat to be driven towards extinction (Dulvy *et al.* 2003). Discard mortality of threatened fish species is potentially unaccounted for in fisheries data, unless it is a targeted species with discards included in stock assessment. This aspect, together with a prior suggestion of utilizing the IUCN framework to address biodiversity loss in LCA (Lindeijer *et al.* 2002), were behind the aims of **paper II.** The aims were to evaluate the validity of the IUCN categorization and to explore the possibilities of utilizing the amount of threatened fish in discards (i.e. vulnerable, endangered and critically endangered species, abbreviated as VEC) as indicators for the seafood product's impact on sensitive and overexploited fish species. The focus was on fish and not on other species, mainly due to data availability, but also as fish can be useful proxies for ecosystem status due to availability of data (Dulvy *et al.* 2006). It should be noted that MTL is also categorized as a 'biodiversity' indicator to the CBD. Still, due to short-comings of interpretation of trends in both PPR and MTL (**paper I**), there is a need to include an additional metric related to depletion of stocks in sustainability assessments of fisheries.

In terms of robustness of the IUCN Red List Categories and Criteria, **paper II** showed little discrepancies between the scientific advice in fisheries, provided by the International Council for the Exploration of the Sea (ICES), and the Swedish Red List of Threatened Species. Other studies comparing ICES reference points to IUCN criteria have been both positive and negative. Positive, in terms of that it was unlikely that there would be any false alarms from a species being listed as threatened while being considered to within 'safe biological limits' by ICES (Dulvy *et al.* 2005; ICES 2008). Negative, in term of that dynamics of stocks can easily lead to false alarms according to the IUCN criteria while also having a significant risk of overlooking signals which indicate real danger (ICES 2009a,b). There are thus both potentials and drawbacks with applying the IUCN Criteria and Categories. Criteria based on numbers of individuals and geographic range appear to be consistent between IUCN criteria and stock assessments, whereas interpreting the extent of population decline by the different frameworks clash (Rice & Legacè 2007). Results from **paper II** also showed some inconsistency of the Swedish Red List in terms of species status in relation to the status of separate stocks. This emphasizes the drawback of using species rather than stocks as assessment units. One stock can be on the verge to extinction while the species is not threatened on a global scale.

It was also found that the Swedish Red List showed somewhat higher threat status than the global Red List. This finding is not surprising, as there has been no documented global extinction of a marine fish species (Roberts & Hawkins 1999), but several local extirpations, such as the pivotal finding of the disappearance of the common skate (*Dipturus batis*) from the Irish Sea (Brander 1981). Thus, it seems preferable to utilize regional Red Lists when assessing species at threat in LCA.

As for applicability of VEC, there are also limitations. In the studied area of **paper II** (Kattegat and Skagerrak), screening discards for threatened fish species was possible as there were data of species composition, abundance as well as IUCN assessments of all fish species in the Swedish Red List by the Swedish Species Initiative, updated every five years (Gärdenfors 2010). However, in terms of global applicability, data on discard for specific fisheries is limited, and the global assessment of fish by the IUCN Red List is far from complete, even though progress is being made (Collette *et al.* 2013).

Aggregating vulnerable, endangered and critically endangered species into a composite indicator (VEC), as suggested in **paper II,** has both advantages and drawbacks. Different fishing practices exhibit different pressures on threatened fish species. For example, the mixed trawl fishery for Norway lobster (*Nephrops norvegicus*) was found to have the greatest VEC value per landing (**paper II**). In this sense, VEC distinguish different pressures between fishing practices. Still, it does not say anything about the potential effects on an ecosystem scale. Threatened fish in discards could be low per landed volume, but could be great in terms of overall pressure on a particularly vulnerable species. The VEC approach does in addition not distinguish between different degrees of threat status between the species. For communication purposes, several other composite indicators relating to threat status are however suggested, such as the Red List Index (RLI) in the Convention on Biological Diversity (2010 Biodiversity Indicators Partnership).

The interpretation of the VEC indicator is also complicated by the fact that some of the species categorized as VEC could also be target species, and the discarded amount of some species could hence be accounted for in their specific stock assessments. It could thus be argued that if these discards are accounted for in stock assessments, this impact would also be more of an impact on target species. However, as **paper II** showed great coherence between assessments done by fisheries scientists and conservation biologists, great amounts of threatened fish in the discard can doubtfully be seen as negligible. A situation could occur when a threatened species would increase in biomass, which would be a sign of positive development, but would result in greater values for VEC, which is a negative signal. However, even if there is a successful recruitment of a species, a mixed fishery risks catching and subsequently discarding them due to e.g. being below landing size. This would cause great impediment for rebuilding of depleted stocks (Hutchings 2000).

In **paper II**, there was a lack of biological reference points to robustly determine an appropriate fishing mortality for many of the species landed. In this sense, VEC could also be used in LCA to categorize impacts from the landed part of the by-catch (*sensu* Davis *et al.* 2009) “un-managed”, i.e. species caught in mixed fisheries that has no directed fishery and lacks biological reference points). In this case, a hierarchical impact assessment could be done, which would then be considered to assessing the broader concept of by-catch instead of merely discards. If there are sufficient biological reference points for a landed species to be sustainably managed, it may be considered as target stock and be omitted from the VEC-assessment. If reference points are insufficient, a screening of VEC-species in the remaining part of the landings could be done, which then would be considered as by-catch.

Building bridges between conservation and fisheries management

The IUCN Red List Categories and Criteria are intended to identify species at risk of extinction, not assessing conservation priority (Mace *et al.* 2008). However, only a fraction of the threatened fish species in Sweden has direct conservation policies (Table 1). Some species have extra protection in fisheries not seen in the table, such as seasonal bans (for e.g. Atlantic cod, haddock and Pollack). Some of the shark species are protected within the EU. Screening for threatened species (**paper II**) has the potential to identify hot-spots where conservation action may be needed, and could be used to inform and quantitatively follow up on progress related to commitments following the Aichi Biodiversity targets set within the CBD[1]. It could also be of use in certification schemes to overall pave the road towards rebuilding of depleted species and halt biodiversity loss.

Table 1. Swedish Red List status for marine fish in 2005 and 2010, global Red List status and conservation policies ([1]National protection; [2]CITES B; [3]Bonn convention I; [4]Bonn convention II) in place. New species categorized as threatened in 2010 with no threat status in 2005: *Anarhichas lupus* and *Merlangius merlangus*. Species categorized as threatened in 2005, but not in 2010: *Psetta maxima*, *Scyliorhinus canicula* and *Lesueurigobius friesii*.

Scientific name	Common name	Swedish IUCN Red List status (2005)	Swedish IUCN Red List status (2010)	Conservation Policies (2010)	Stock status (ICES Advice 2009)
Dipturus batis	Blue skate	CR	RE	1	Unknown, no targeted fishery and limit by-catch.
Acipenser oxyrinchus	Atlantic sturgeon	RE	RE	1,2,3	No advice.
Lamna nasus	Porbeagle	CR	CR	1,3	Depleted, no targeted fishery.
Cetorhinus maximus	Basking shark	EN	CR	1,2	TAC set at zero.
Squalus acanthias	Picked dogfish	EN	CR	4	Depleted, in danger of collapse.
Anguilla anguilla	European eel	CR	CR	2	Decline at alarming rate, zero impact.
Pollachius pollachius	Pollack	EN	CR		No advice.

[1] The applicability involves several targets set within mainly the Strategic Goal A, “Address the underlying causes of biodiversity loss by mainstreaming biodiversity across government and society” and Strategic Goal B, “Reduce the direct pressures on biodiversity and promote sustainable use”.

Table 1. *continued*

Scientific name	Common name	Swedish IUCN Red List status (2005)	Swedish IUCN Red List status (2010)	Conservation Policies (2010)	Stock status (ICES Advice 2009)
Chimaera monstrosa	Rabbit fish	VU	EN		No advice.
Raja clavata	Thornback ray	VU	EN	1	Unknown, no targeted catches.
Coryphaenoides rupestris	Roundnose grenadier	VU	EN		Constrain catches.
Molva molva	Ling	VU	EN		CPUE reduced, constrain catches.
Gadus morhua	Atlantic cod	EN	EN		4 stocks (SSB_{pa}: 2 reduced; 1 increased risk; 1 undefined)
Melanogrammus aeglefinus	Haddock	NT	EN		Full reproductive capacity (IIIaN).
Anarhichas lupus	Atlantic wolffish	-	EN		No advice.
Hippoglossus hippoglossus	Atlantic halibut	EN	EN		No advice.
Galeorhinus galeus	Tope shark	VU	VU		Unknown, no targeted fishery.
Somniosus microcephalus	Greenland shark	DD	VU		No advice.
Etmopterus spinax	Velvet belly	VU	VU		No advice.
Merlangius merlangus	Whiting	-	VU		No advice.

Fuel intensity as a sustainability indicator

It has been suggested before that energy intensity could act as an indicator for other environmental impacts in LCA (Thrane 2006). This correlation is confirmed in e.g. a study by Ziegler and Valentinsson (2008) on *Nephrops* caught with creels or demersal trawls. Fisheries-specific impacts relatively new to LCA (seafloor area impacted and discard amount) were in line with the fuel use found to be considerably higher for trawled *Nephrops* compared to creel caught ones. Demersal trawling has generally higher discard rates and causes more impact on benthic communities than other fishing methods for the same species, besides being relatively fuel intensive. Passive gears are commonly promoted as Low Impact Fuel Efficient (LIFE) methods (Suuronen *et al.* 2012). An already fuel-intensive gear will also be extra sensitive to poor stock status: a depleted stock needs more fuel per kilo per landing due to lower landing per unit effort (**paper IV**).

There is however a considerable risk of missing out in providing the comprehensive environmental perspective intended with the LCA methodology by not including additional indicators such as VEC. Promoting passive gears as LIFE methods could be nuanced by case-to-case examples of non-universal correlates of lower fuel consumption of passive gears in comparison to demersal trawling (**paper VI**). There could also be substantial by-catches of e.g. marine megafauna in gill-net fisheries (e.g. Lewison *et al.* 2004). In **paper III,** a clear trade-off was identified from using selective grids in the *Nephrops* fishery on the west coast of Sweden: fuel intensity and seafloor area swept is higher while locally depleted fish stocks are protected. Without differentiating impacts on vulnerable fish species imposed from discards as suggested in **paper II**, conventional trawling would have been superior to selective trawling as the fuel intensity and total discard ratio were lower per kilo of landing.

Yet another twist is the fuel intensity of deploying Fish Aggregating Devices (FAD). Using FADs in tuna fisheries has been shown to be more fuel efficient (Tyedmers & Parker 2012). However, FADs also attract and entangle vast amounts of vulnerable species. Entanglement mortality of silky sharks (*Carcharhinus falciformis*) in FADs associated to the purse-seine tuna fleet of the Indian Ocean has been estimated to be 5-10 times higher than that of the known by-catch mortality of the fleet and is of major implication to global mortality estimates of this Near Threatened species (Filmalter *et al.* 2013). It would also be difficult to consider a fuel-efficient tuna fishery as sustainable without acknowledging target stock status. Clearly, this advocates for including energy requirement as one important aspect in an integrated assessment of the sustainability of a fishery.

Fisheries: what is the catch?

Only including fish and commercial invertebrates as discard indicators (**paper II)** is the result of data availability. Even if other discard components could be included in the methodological approach (se discussion below), the metrics used in **paper I** and **II** would still be limited by the availability of total catch data in other fisheries. Merely including any kind of assessment of discard impact on vulnerable species is thus an important inventory task for LCA practitioners to perform.

The discard ratios in *Nephrops* trawling found in **paper III** were based on a snap-shot of data and could be contrasted with other studies of *Nephrops* fisheries. In Kelleher (2005), the estimated weighted discard rate of trawl fisheries targeting *Nephrops* was 43% of the total catch. The report also notes that much of the by-catch is landed, and in the EU some fisheries would not even be economically viable unless other species are landed. Discard rates are also highly variable during a year. In the *Nephrops* fishery on the British north-east coast, it was shown that the discard rate was 52% higher on sunny days than on cloudy days (Catchpole *et al.* 2005). A large variability of discard ratios has also been confirmed for the Swedish *Nephrops* fishery (Feekings *et al.* 2012). Discard ratios at batch level – between boats, fishing trips, seasons or years – are thus highly variable.

Future work should however be done in terms of addressing the discard of non-commercial invertebrates. This part of the catch can be substantial (Bergmann *et al.* 2002b), and has been found to vary in composition in terms of threatened species between different demersal trawling segments in the studied area of **paper III** (Ottoson 2008). These data were however not readily available from observer protocols. Catch of non-commercial invertebrates may however also benefit from being discussed from a benthic disturbance perspective, i.e. a seafloor impact. Benthic impact could be observed from by-catch data, as discarded invertebrates could act as indicators of benthic community disturbance from monitoring species known to be sensitive. In this case, the approach of quantifying VEC (**paper II**) could be extended to cover non-commercial invertebrates and thus function as a proxy of impact on sensitive habitats. Still, for now, monitoring effort regarding discard of non-commercial invertebrates are insufficient, and coverage of the IUCN Red List on marine invertebrates is limited.

Efforts should also be made to include by-catch impacts on other marine animals than fish, such as marine reptiles, birds and mammals (e.g. Lewison *et al.* 2004). In this sense, it could be useful to study what risk-based approaches (e.g. Zhou *et al.* 2012) could bring to seafood LCA. One possibility could be to have separate categories for species complexes in terms of by-catch impact on threatened species, as argued in **paper II**, and link this to the landed volume of the fishery. In either case, future effort would also need to acknowledge the difference between impact on threatened species per landed kilo (which would arguably be low in e.g. pelagic fisheries) in relation to the potential effect on the whole population (which could be substantial). This of course also applies to other LCA impact categories, such as transports being of less concern than the fishing phase to the total outcome for the seafood product, while the transport sector is still of high importance to energy requirements and emissions in society.

Further efforts are also needed to collect and make better use of data on fuel use in fisheries. It is striking that anonymous records on landings per trip in the form of logbooks are available to researchers, and also total catch from observer data, whereas figures on merely the total fuel consumption per year for a vessel are much more restricted. Fuel data are routinely collected within the EU, as a part of the Data Collection Framework, and should in theory be available. However, due to confidentiality, these data are not readily available for research on a boat-to-boat basis. Therefore, an attempt was made in **paper IV** to make use of the aggregated fleet-based data on fuel use provided by the Joint Research Commission (JRC 2014) and study the fuel efficiency of the Swedish demersal trawling fleet between 2002 and 2010. The modelling approach, based on landing per unit effort of a certain gear and vessel size and the total fuel use of vessel size structured fishing segment, was coherent with other estimates of fuel use in the fisheries examined. Results implied an overall improvement of fuel efficiency of the Swedish demersal trawling fleet. Important improvements in fuel efficiency were seen following stock rebuilding of the Eastern Baltic cod. However, species-selective trawls were increasingly utilized, and these were more fuel inefficient than their less selective counterparts.

The modelling procedure in **paper IV** is rather time consuming and it is impossible to estimate uncertainties and possible biases. There are important improvement potentials on a boat-to-boat basis such as vessel design (Basurko *et al.* 2013) that will not be accounted for in this modelling approach. Engine age and condition also affect fuel intensity and emissions (Ziegler & Hansson 2003). Skipper effects on fuel intensity have also been observed in purse seine fisheries (Ruttan & Tyedmers 2007). As active trawling and steaming are the most energy intensive phases, simply reducing speed brings fuel savings (Sala *et al.* 2011). Still, there could be trade-offs in terms of potential loss of landings (Bastardie *et al.* 2010). Simply phrased, there are several technological and behavioural factors of importance to fuel intensity that is not seen in fleet data. However, stock status, target species and gear use may arguably be the most important factors to overall fuel efficiency of a fishery at fleet level (Driscoll & Tyedmers 2010, **paper IV, V** & **VI**).

Applying life cycle thinking in a fisheries management context

Managers of fisheries have tended to focus on single objectives, such as status of the targeted stock. As a result, ad-hoc technical solutions, lacking integrated perspectives, are often in place (Degnbol *et al.* 2006). This section discusses how applying a systems perspective, such as life cycle thinking, could bring an advisory role to seafood management and assessment, with focus on European policy.

The intention is not to provide a detailed account for how fisheries management are organised in Sweden and Europe. Instead, this section aims at integrating the perspectives of fisheries management, product policies and stakeholders in the seafood value chain.

Stakeholders and policies addressing impacts of fishing

Scientists

The main role of science in a fisheries management context may be to provide and refine robust models for advising on appropriate fishing mortality. From an increasing knowledge base and evolving objectives concerning harvesting rate, there has been progress towards more sustainable use of fish stocks (Lassen *et al*. 2014). The focus is still mainly on commercial species. Even so, proper stock assessments are at present data intensive, and less than 20 % of the world's fish landings come from fisheries with formal assessments (Costello *et al.* 2012). A TAC could be seen as a single-score index, including some extent of uncertainty, but is still influenced by a range of other unforeseen or not included uncertainties. In the EU, the TAC has as an example mainly referred to landings, not to total catch (i.e. including possible influence from discards). Also, less attention has been paid to interactions between species. The development of Ecological Risk Assessment (ERA) in Australian fisheries (Hobday *et al.* 2011), however, represents one further progress towards broader ecosystem considerations.

At present, besides advances within natural science research, such as determining extinction risks of marine species (Reynolds *et al.* 2005) and acknowledging trophic complexity (Casini *et al.* 2009), fisheries are intensively studied by several scientific disciplines. One important aspect of management success could be identifying factors that are important for fishermen compliance to rules set by managers (Jagers *et al.* 2012), which requires more interdisciplinary approaches to find sustainable solutions to fisheries. Degnbol *et al.* (2006) argue that disciplinary boundaries tend to narrow the perspectives of fisheries management, creating a tunnel vision, and therefore more integrated perspectives are needed. According to Brander (2010), scientists have a responsibility to clarify the trade-offs between different objectives to policy makers, which then need to be communicated clearly, as well as explore how they may best be satisfied simultaneously. This approach asks for scientist to shift from "what can be said" to "what can be done". This is difficult, as "complex and unpredictable" are the words of scientists, whereas "simple and reliable" are what policymakers require.

Indicators are important in the science-policy interface; however, they can be intensive in terms of data needs. As a result, many indicators are based on general data derived from fishing activities, either landing or survey data. Utilizing data on landings enables studies on different aspects on global scales (such as Pauly *et al.* 1998; Pauly & Christensen 1995). Yet, this approach has also drawbacks such as regional differences on how abundance trends and pressures on biodiversity can be interpreted based on merely data on landings (**paper I**, Pauly *et al.* 2013).

To inform policy, there have been major efforts in testing which indicators that are the most robust to address potential environmental impacts from fisheries, while not being influenced by other factors (see e.g. Rochet & Trenkel 2003, Piet & Jennings 2005, Greenstreet *et al.* 2011). In general, indicators in an Ecosystem Approach to Fisheries (EAF) could be described as "pressures" (technical or ecological, e.g. fishing mortality, discard rate, fleet size), as "response" (technical or institutional, e.g. management actions), and as "state" (ecological, e.g. abundance, size spectra). Most indicator development has been focused on state, but indices of both pressure and response are needed for management (Jennings 2005). The general conclusion has unsurprisingly been that there is a need for a broad range of indicators to fully address ecosystem impacts of fishing activities. This arguably calls for a systems perspective in order to avoid risks to shift environmental problems from one potential environmental impact to another.

Policy makers

Fish resources were for long perceived to be inexhaustible and little regulations were in place. There were however early signs of risks of overexploitation. A petition to the British Parliament was made already in 1376 with concerns expressed regarding the destructiveness of demersal trawling (referenced in Roberts 2007). Policies were gradually introduced, such as in the case of the North Atlantic: restrictions on minimum landing size (MLS) in the 1950s and total allowable catch (TAC) in the 1970s (Halliday & Pinhorn 2002).

Today, it is a time-consuming task to cover all policies that address environmental impacts from fishing. However, they can be grouped into different categories depending on scope and enforcement. The most direct fishing policies are those regulating fishing opportunities, such as the EU Common Fisheries Policy (CFP). The prior regulation from 2002 (EC 2002) stated that it shall ensure:

"exploitation of living aquatic resources that provides sustainable economic, environmental and social conditions"

This was not accomplished since 88% of EU stocks were considered to be overexploited in 2009 (COM 2009), even if the situation has somewhat improved since then (Cardinale 2011). There were also direct references made in the former CFP to adopt an ecosystem approach, thus echoing the commitments set by the CBD (CBD 2000) and the guidelines for Ecosystem Approach to Fisheries (EAF; FAO 2003). The interpretation of what ecosystem approaches implies however varies between stakeholders, as it could to some stakeholders refer to creating

Marine Protected Areas (MPA) or use of selective gears (Hilborn & Hilborn 2012). From this interpretation, some progress has been initiated in different areas, such as establishments of trawl free zones and by-catch mitigation of vulnerable species.

With the newly reformed CFP (EU 2013), all fishing opportunities shall be set in accordance with Maximum Sustainable Yield (MSY), which is generally seen as great progress from the prior policy. This is however a theoretical approach in terms of sustainability as it is based on a single stock perspective whereas fisheries operate in multi-species systems, and may not be the overall most risk-averting strategy (Smith *et al.* 2011).

In the reformed CFP, there are also linkages to the Marine Strategy Framework Directive (MSFD), a new directive aiming at creating a Good Environmental Status (GES) of coastal waters by 2020, with several descriptors and associated indicators related to fishing activities. Marine strategies should (EC 2008b):

"protect and preserve the marine environment, prevent its deterioration or, where practicable, restore marine ecosystems in areas where they have been adversely affected"

There are several other regional commitments and conventions related to fishing policy that focus more on single topics, such as The Bird's Directive and The Habitats Directive. Besides these regional objectives, several global commitments and guidelines with broader scopes are in place such as: UNCLOS Law of the Sea, UN Fish Stocks Agreement, FAO Code of Conduct, Deep Sea Fisheries Guidelines, and more.

Outside the context of more direct fishing policy, several global policies and frameworks aim at addressing the general protection of biodiversity from pressures, such as fisheries, and release of greenhouse gas emissions in order to avoid adverse environmental effects and promote sustainable development. Examples of these are: the IUCN Red List of Threatened Species, the Convention on Biological Diversity (CBD), the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) and the Kyoto Protocol.

To fulfil all objectives set, one must ask: where does the threshold lie for an ecosystem to be defined as deteriorated, restored or adversely affected? If an ecosystem is affected, what are the prioritizations and line of action? And most importantly, what is included in the decision support for managers to be aware of the potential trade-offs – and what are acceptable trade-offs?

In fisheries, a narrow focus by managers can be seen in the current conservation status of many marine species (Dulvy *et al.* 2013) and increasing vulnerability of fishing operations due to fossil fuel dependence (Abernethy *et al.* 2010). Instead, as argued by Salomon *et al.* (2011), objectives need to be clear, integrated policy tools need to be developed and operational to measure trade-offs among all objectives. Butchart *et al.* (2010) also advocate strengthening and integrating impacts on biodiversity loss in policymaking.

It could thus be argued that as there is a broad range of policy commitments in place, it would be utterly important to include existing indicators from a conclusive set of policy objectives in an integrated tool that could inform all policies that have the potential to affect them. Evaluating their operational strengths and weaknesses (e.g. **paper I** and **II**) should therefore be a top priority in order to be cost effective in terms of monitoring needs, not to mention the cost of establishing policy objectives that are not addressed or not even relevant in practice. This could be the case for MTL and RLI. It might be so that as long as enforcement of policy is weak, no objectives are set on desired targets, and there are no operational and integrated tools to address all stressors, policies are but merely ambitious paperwork.

Consumers and certification

From a seafood product perspective, certification as a measure of sustainability has gained great attention in recent years (FAO 2012). The most widely used label, the Marine Stewardship Council (MSC), has now certified around nine per cent of the global landings (Agnew *et al.* 2014). Certification conveys a message that besides stock status, wider ecosystem considerations have been taken regarding the product and management in place (Gutiérrez *et al.* 2012). Even so, there will always be discussions on which and how criteria have been used (e.g. Jacquet *et al.* 2010). Certification schemes are broader than traditional single-stock based approaches in fisheries, yet, there are still missing metrics. Fuel intensity and GHG emissions are not addressed by seafood certification (Thrane *et al.* 2009, besides the Swedish organic certifier KRAV (www.krav.se) and the newly established Aquaculture Stewardship Council (ASC; www.asc-aqua.org). In any case, one important aspect of certification is the improved traceability and assurance. Seafood products have often been found to be mislabelled (Miller & Mariani 2010), whereas certification has proved to have a much higher compliance (MSC 2013).

Consumer demand can be influenced by increased awareness of short-comings of seafood production; it could however also contribute to less sustainable fishing practices being viable due to high market demand, such as the extremely high prices seen for the few remaining Atlantic Bluefin tuna. Levin *et al.* (2010) found that the MTL of seafood recipes increased over time, arguing that the societal preference in relation to sustainable use of the sea must be increasingly addressed. Apostolidis and Stergiou (2012) then argued that, based on the sustainability of on-line recipes, it would be cost-effective to target and educate chefs in order to increase sustainability of consumer patterns. Retailers are also increasingly taking greater responsibility for what they market, with initiatives from several larger companies around the world to source only sustainably produced seafood; the criteria used however vary. There could also be negative sales figures in retail from increasing the information on the sustainability of seafood products. A study using a traffic-light plot at retailers, which indicated green-yellow-red seafood products, showed that overall less seafood was bought (Hallstein & Villas-Boas 2013). There were however no changes in purchase of green or red products, only a decrease in yellow seafood products sold.

If overall sustainable seafood production is sought for, there is also a need to create stronger links between different stakeholders in seafood production chain. From using an LCA perspective, i.e. minimizing resource use in all production phases, important overall improvements including the industry perspective may be achieved. Post-landing aspects, such as filleting yields and product losses, are also important for the sustainability of seafood products as this involves loss of limited resources (**paper VI**). This aspect is not included in certification and is paid relatively little attention to, even if societal interest in sustainable seafood production is on the rise. The decisions taken by fisheries managers are also important from a product perspective, and fishing policy could be influenced from raised awareness of consumers, chefs and retail. However, there will always be a market for less sustainable options as long as there is demand and if they are available, which calls for effective communication. Altogether, LCA results could bring insights to all stakeholders in the product chain and contribute to improvements.

The role of LCA

The unique feature of LCA is that it focuses on products in a life cycle perspective (Finnveden *et al.* 2009). By this scope, and due to the intended comprehensiveness related to use of resources and environmental impacts, it is a useful tool to avoid problem-shifting (e.g. from one type of environmental impact to another or from one production phase to another) which certainly applies to fisheries (**paper VI).**

Addressing policy makers and scientists

The studied indicators (**paper I, II**) are simple and thus have limitations in terms of applicability. A composite indicator such as VEC may be limited in terms of usefulness for managers of fisheries or conservation biologists. For management purposes, use of trophic indicators could instead provide a paradigm shift in fisheries, and they are already implemented to some extent in some fisheries (Longo *et al*. under revision). There are also objectives set within the MSFD related to the food web structure component of Good Environmental Status (GES) (EC 2008b). Still, it is not clear how trophic interactions can be understood in relation to ecosystem functioning and productivity, and how these relationships could be utilized on a quota setting basis. One example could be, as suggested by Smith *et al.* (2011), to set lower reference points to MSY of lower trophic level species in order to allow for higher abundances left in the ecosystem, as they are important prey items to higher trophic level species such as birds.

Navigating trade-offs between multiple objectives in policy is not straightforward, and life cycle thinking could in this sense build bridges. In comparison with more risk based assessments, LCA has the advantage of being a quantitative tool. This makes it easier for managers and politicians to set objectives and measure progress towards the desired state. However, one drawback is coverage of LCA methods of relevance to fisheries in terms of potential ecosystem effects that could be applied in a consistent manner. It might even turn out to be impossible to develop such indicators. Still, from applying a systems perspective on fisheries, results could be used as an integrated decision support. In this sense, LCA based methods can quantify trade-offs between different priority settings of objectives within a policy framework, potentially using metrics proposed within other policies such as the MSFD. Capture fisheries would also benefit from fuel use being taken into consideration in an early step, e.g. when allocating effort and quotas. This is not only beneficial to the environmental profile of the fishery, but could also involve economic benefits to the fishery (Abernethy *et al*. 2010).

The Maximum Sustainable Yield (MSY) concept is strongly influencing European fisheries management, but a strict focus on F_{MSY} might divert the attention from the true sustainability of the fishery. The *Nephrops* stock is as an example fished at F_{MSY} according to the latest ICES advice (ICES 2013), despite the trade-offs shown in **paper III**. Also, the Eastern Baltic cod fishery is fished as F_{MSY} according to ICES. However, Svedäng and Hornborg (*in review*) showed that the combination of fishing effort (F) and size selection (L_c) in place for the Eastern Baltic cod fishery have had dire consequences for both ecosystem functioning and the industry, as it has induced density dependent growth and lower individual growth potential. By acknowledging the present situation in a broader perspective, beyond the F_{MSY} target, such as including seafloor impact, fuel use, greenhouse gas emissions and size structure of the stock, it was found that setting a lower objective of yields and promoting stronger effort cuts and less selectivity would be more economically beneficial to the fishing fleet while being more ecologically risk-averting (**paper V**). Also, major overall improvement potentials could be shown with little compromises in yield. This is arguable a more sustainable development of the fishery, and indicative of that more metrics than F_{MSY} is needed to address sustainability of a fishery.

Even if there are policies covering all environmental impacts of fisheries, no one is at present arguably comprehensive, integrated and strong enough in its enforcement or objective to result in overall sustainable development of fisheries, at least in a European context. In order for different policy decisions to be better integrated, and making broader assessments of fisheries a standard procedure, clearly defined management objectives in a broader perspective than covered by the current framework is needed. So far, fuel use in fisheries has as an example not been considered by fisheries managers. Following the G-20 agreement in Pittsburgh in 2009, to "phase out and rationalize over the medium term inefficient fossil fuel subsidies", this has however started the OECD to initiate further studies of fuel use and the influence of management in fisheries (Martini 2012). In the newly reformed CFP, fuel use considerations are in fact mentioned on several occasions (EU 2013), but it is yet to be seen how this could be incorporated in management. From the upcoming obligation to land all catches, selective fishing practices of both size and species are additionally incentivized, and these management measures would benefit from applying life cycle thinking (**paper III**, **IV** and **V**). In addition, transparency of management decisions in fisheries has also been identified as a "sustainability bottle-neck" of effective management in order to not be sensitive to e.g. harmful subsidies (Mora *et al.* 2009). LCA in this sense represents a tool that is both transparent and integrated, thus allowing decision support identifying trade-offs between e.g. fishing opportunities, local conservation priorities and GHG emissions and could integrate different policies into one framework.

Addressing consumers, retail and certification

There are several applications of LCA as a decision support tool in industry and policy (**paper VI**). For example, LCAs form the basis for eco-labels type III, Environmental Product Declaration (EPD; ISO 2006c). This format conveys non-aggregated information concerning all studied impact categories and is less directed towards consumers. In comparison, eco-labels type I such as MSC, weight all impacts considered into a single metric, i.e. if it can be certified or not, and is in general not based on LCA. However, it may be useful also for certifiers of eco-labels of type I to consider LCA results as a quantitative or qualitative option to follow up their results. This is to some extent done by the Swedish label KRAV (Thane *et al*. 2009).

It has been argued that as threats to biodiversity are facilitated due to complex trade routes, the biodiversity threat of certain products needs to be better integrated into sustainable supply chain certifications and consumer labelling (Lenzen *et al.* 2012). In the LCA related UNEP/SETAC initiative, there is also a call for a consensus regarding the use of environmental impact category indicators in global supply chains, however, not necessarily state-of-the art LCA (Valdivia *et al.* 2013). Altogether, this results in pros and cons of using VEC in seafood LCA. As an impact category in LCA, VEC is not as straightforward as global warming potential, but is still highly relevant. It is therefore yet to be seen how and if VEC could be consistently applied. In theory, VEC could be utilized as information on a single attribute of environmental objectives, such as "dolphin-safe" (Thane *et al.* 2009). In this case, VEC could be utilized as a label "no threatened species caught", given that data are available both in terms of discards as well as threat status. In the case of PPR, trophodynamics is not straightforward in terms of what is a positive or negative trend and is of less relevance to consumer information.

Building bridges between product policies and fisheries management

The European Commission considers Life Cycle Assessments to provide the best currently available framework for assessing the potential environmental impacts of products (EC 2003, EC 2014a). In Europe, there are therefore several initiatives to address sustainability of different products with strong links to LCA methodology, such as the Product Environmental Footprint (PEF) (EC 2014c) and the Integrated Product Policy (IPP) from 2003 (COM 2003). There has also been a policy decision within the EU to decrease environmental impacts of consumption, with LCAs forming the base for evaluation (EC 2008a). LCA is also seen as an important tool to address the implementation of the EU Thematic Strategy on the Sustainable Use of Natural Resources (EC 2014b).

Still, all these policy discussions do seem to apply to all kinds of products except for seafood, since they could be seen to operate apart from the Common Fisheries Policy (CFP). No initiatives have been taken by fisheries managers to apply a life cycle approach to fishing activities and seafood supply chains. It is most peculiar that these two areas are seen as separate entities, given the fact that fisheries management has a strong influence on the overall resource use and impacts of the seafood products (**paper VI**). Acknowledgment of LCA results in fisheries management objectives could therefore be an important link to existing product policies.

Thus, even if LCA based evaluations of fisheries management have shown some important improvement potentials (**paper VI**), LCA in a fisheries management context must still be seen to be at a research level. Its application as a tool for the science–policy interface is yet to be implemented. The synthesis of existing and additional data enables a new perspective. From adding new data, or existing data not formerly included, in combination with new, or borrowed, indicators, life cycle thinking in fisheries may have a potential to contribute to a new and improved management regime. As fisheries make use of both renewable and finite resources, i.e. fossil fuel and fish resources, it could be argued that there is a need for co-management. This is where a systems perspective might prove to be useful.

Promoting the best available technology in the broader perspective

It could be argued that if managers of fisheries would succeed in maintaining stocks at high abundance, this would also result in higher LPUE and thus fuel efficiency. Still, as many stocks are under rebuilding, the path chosen towards stock recovery also influences fuel efficiency, and using different gears for the same species also offers improvement potentials. Identifying trade-offs in the broader perspective, such as fuel inefficiency in the Swedish trawl fisheries for *Nephrops* (**paper III)**, is an example of when a life cycle perspective can quantitatively illustrate problem shifting originating from different prioritizations of objectives. Fuel inefficiency could as an example imply biodiversity threats in distant areas from extraction of fossil fuel resources (Butt *et al.* 2013), not to mention increased contribution to climate change which among other effects feedbacks into the seafood production itself (Branch *et al.* 2013).

The extra seafloor area impacted per kilo landing found in **paper III** is more complex to evaluate. The effort is highly aggregated to certain areas and it is difficult to draw any conclusions on a possible additional benthic community effect. If enough entirely trawl-free zones were established, and knowledge on where exactly fishing takes place and the distribution of benthic habitats, seafloor impact could arguably be compared to land use in agriculture, as agriculture also transforms habitats in order to produce food. Otherwise, comparing farming systems and seafloor disturbance is problematic, as man-made and natural ecosystems are completely different entities. In this particular case study (**paper III**), the species-selective grid fishery is allowed in areas intended to be trawl-free zones. Thus, some of the extra area impacted due to grid trawling would thus not have occurred, if the grid had not been introduced, as the effort restrictions from the EU cod recovery plan would not have allowed continued non-selective trawling. This decision would have diminished overall fuel use and associated emission, besides the seafloor area swept, however with risks of not catching the allocated quota for *Nephrops*.

Instead, all Swedish trawl fisheries targeting *Nephrops* are now obliged to use the species-selective grid. It could however be argued whether a slightly modified business as usual management, i.e. forming a single-species fishery with sustained trawling effort for *Nephrops*, could be part of a sustainable development of Swedish fisheries for the future. There are ecological and socio-economic risks with leaving fishermen dependent on a single species or stock, in ecosystems altered from depletion of fish stocks (Howarth *et al.* 2013). There is as an example a viable creel fishery for *Nephrops* in the area, with less overall impacts compared to demersal trawling (Ziegler & Valentinsson 2008). It would thus be appropriate to investigate to which extent the *Nephrops* quota could be allocated to this fishery instead. Such a transition would however benefit from applying an LCA perspective in order to avoid problem-shifting.

There may also be some interesting hybrids between stock assessment procedures and the broader framework of applying a systems perspective, enabling an integrated assessment of sustainable development in the broader context of a fishery as illustrated in the study of the Eastern Baltic cod fishery (**paper V**). LCA could in this sense be seen as a complement to stock assessments: once a fishing opportunity is discussed, applying a systems perspective could complement with information regarding broader effects beyond target stocks, such as fuel use, seafloor area impacted and quantitative by-catch conflicts.

However, this calls in general for data from fisheries being made more available for researchers, such as trawl effort and fuel use of individual fishing boats (Hinz *et al.* 2013, **paper IV**). With the integrated perspective, effort and quotas could tentatively be distributed to the fishermen that fish in the most overall resource efficient and environmentally responsible manner. This data collection could be of use in post-landing chain perspectives, in the form of a quantified environmental performance to monitor and follow up by certification and seafood suppliers.

A brief note on selectivity

The rationale behind allowing small fish to escape to spawn has been questioned, but is a paradigm in fisheries management to avoid recruitment overfishing. Halliday and Pinhorn (2002) argue that regulation related to selectivity and setting thresholds for minimum landing size (MLS) is more of an ancillary measure that aims to affect exploitation pattern, not rate. Too often regulation to reduce capture of small fish is used as an insufficient substitute instead of the necessary effort cuts. This is clearly shown in **paper V**, when stronger effort cuts and lower selectivity is needed to restore the Eastern Baltic cod productivity that is currently hampered by the selectivity induced density dependent growth (Svedäng & Hornborg *in review*). This would also be a beneficial development in the broader perspective.

It has been argued that selective targeting of species, stocks or sizes could be detrimental to ecosystem structure and function (Garcia *et al.* 2012). Also, as demersal fisheries most often catch a range of species, the optimal mesh size will inevitable vary between species caught in the same haul. As making demersal trawls less effective has trade-offs in terms of fuel efficiency per landing (**paper III)**, it may be argued that selective demersal trawls are merely a short-sighted disguise of overcapacity. From an LCA perspective, stronger effort cuts with a catch-all-land-all-principle would be more beneficial.

Governance towards sustainable development of fisheries: what is it about?

No food production comes without any impact. Important questions related to the sustainability of different food items would relate to what characterizes an impact, how it can be measured and what should be considered by the management system. One important objective that is most often lacking for seafood production from capture fisheries is what the desired state of marine ecosystems is? It has been argued that the simplified ecosystems induced from fishing could be compared to agriculture on land and are necessary to keep up with demand; however, several case studies have clearly shown that these altered marine ecosystems are far more vulnerable, and rather threatening than promoting food security (reviewed in Howarth *et al.* 2013). Instead, in relation to other food production systems, if sustainable development in a broader context is sought for, questions should relate to what an acceptable level of disturbance is in fisheries compared to agriculture – and defining which effects are irreversible.

LCA could synthesize and make more use of already collected data to provide an integrated decision support and link fishing policy with broader objectives, such as halting biodiversity loss and reduce greenhouse gas emissions. Many of the formerly common commercial species in Swedish waters are as an example now considered to be threatened with extinction by the Swedish Red List (**paper II**). Protecting biodiversity one species by another should therefore not be seen as isolated problems, but important tasks for the structure and function of whole ecosystems. However, according to Hilborn and Hilborn (2012), threat status for marine fish is linked to high and continued fishing pressure. They argue that if stocks were protected by managers from depletion, concerns related to biodiversity would be less for fisheries compared to mono-cultures on land. Still, there is an obvious conflict between use of natural resources and impact on threatened species which may require conservation efforts, as illustrated in **paper II** by threatened fish species being discarded. Also, just because a stock is fished in relation to a preferred biological reference point, it does not imply that threats to ecosystem structure and function are non-existing, as illustrated in the Eastern Baltic cod fisheries (**paper V**).

Management will always involve prioritizations of different objectives. Pauly *et al.* (2002) tested future scenarios for fisheries based on different plausible prioritizations of objectives: markets, security, policy, sustainability. They found that many of the investigated fisheries at that time optimized nothing of benefit to society, and it was questioned how long they would be around. Hilborn (2007) however argued that it is therefore mainly a matter of political will, as we have the tools to bring overfishing under control. Progress has also been made in many regions in recent years towards decreasing fishing mortality and rebuilding of stocks (Worm *et al.* 2009).

This also brings on a great debate, where fisheries have split society and scientists into two camps: protectionists on one side and advocates of need for food and employment on the other side. Simply phrased, different stakeholders could arguably be seen to have different priority settings:

- *Fisheries biologists* focus mainly on fishing mortality of single stocks of commercial species, most often from data rich models.
- *Conservation biologists* focus on protection of biodiversity, most often under data-limited circumstances.
- *Managers of fisheries* may put an additional emphasis on profitability of the sector and employment opportunities besides ecological implications.
- To the *fishing industry*, long-term economy and stable conditions for production is a pre-requisite to the sector; however, quotas are set on a year-to-year basis and conflicts are prevailing between single fishermen perspectives versus the fishing community as a whole.
- *Certification schemes* have an emphasis on broader ecological aspects of a certain fishery than the single stock advice, but not all impacts are covered.
- *Non-governmental organizations (NGOs)* address environmental impacts of fisheries, trying to influence policy and human behaviour using methods such as producing consumer guides, lobbying, direct action and more.
- *Consumers* and *retailers* may want to know more about how their products were produced and desire assurance of what is claimed, price is also important.

If better governance of fisheries is sought for, establishing common grounds should therefore be seen as important. In this sense, seafood LCA could build bridges between different stakeholders by providing a scientifically based, standardized and quantified basis for comparison. Carbon footprints of seafood, i.e. a simplified LCA based assessment focusing specifically on greenhouse gas emissions, are not currently addressed by managers of fisheries. It could then be argued that managers consider their role to be outside of the seafood production perspective, in spite of the important outcome of the product based on their decisions. In fact, LCA may not be adopted in a fisheries management context until managers of fisheries acknowledge their role in creating an overall sustainable development of the capture fisheries sector and subsequent seafood supply chains.

Conclusions

The work done in thesis has advanced the methodological framework of seafood LCAs by presenting indicators that could be useful to add to the existing framework of seafood LCAs. More specifically, ecosystem pressures from discards in terms of resource waste (PPR) and biodiversity threat (VEC) have been studied. Still, there is further work to be done to refine the metrics suggested, as well as methodological development to address other areas of fishing impacts such as seafloor disturbance. There is still much to learn about ecosystem properties and function and how to find suitable metrics related to imposed pressures while they are still meaningful for their intended purpose. Even so, for research to be operational in a policy context, simple indices are needed.

Life cycle thinking as an integrated decision support to form an overall sustainable fisheries management has been tested in *Nephrops* trawling on the west coast and demersal trawling for Eastern Baltic cod. Trade-offs were identified in the broader perspectives for both, advocating for using more metrics than fishing mortality on the targeted stock to assess the sustainability of the fishery. In the Eastern Baltic cod fishery, there were environmental and economic benefits of setting a lower objective of yield. Even if overall fuel efficiency in Swedish demersal trawl fisheries has improved in parallel to stock rebuilding, species-selective demersal trawls are increasingly promoted and are more fuel inefficient per landing due to lower catches. Sorting grids in crustacean trawling may be important for protecting vulnerable fish species. Even so, due to trade-offs in the broader perspective, some grid trawling practices could be seen as short-sighted measures to mask over-capacity. From an LCA perspective, stronger effort restrictions and a catch-all-land-all principle, or change of gears, represents improvement potentials.

The benefit of life cycle approaches in comparison to risk based methods is that it is a quantitative tool and applies a systems perspective. Drawbacks involve data intensity and lack of comprehensive assessment methods within the framework. Even so, from a policy perspective, there are several interesting gaps where the integrated perspective may prove to be a valuable link. This could be in the form of hybridization with other assessment methods or from further methodological development within LCA.

Given the importance of fisheries management to the total resource use of the seafood product under its life cycle, improving the fisheries' performance would provide great improvement potentials. Seafood LCA as a fisheries management tool is however still more at a conceptual than operational level. Even if LCA is seen as an important tool from a product perspective, no fisheries management system has yet applied this perspective. It could however be used to evaluate a quota or a total fleet effort, depending on which questions that are asked by managers. This thus requires, among others, that managers of fisheries acknowledge their role in seafood production.

If overall sustainable development of the seafood sector is desired, integration of various environmental objectives also needs to improve. Researchers from various disciplines are at present convincingly well describing the path towards collapse with different lines of reasoning, and there is no lack of policies to address every aspect in isolation. We do have the capacity to overfish, and there are undoubtedly still short-sighted visions allowing biodiversity loss and fuel inefficiencies over production loss. However, managing a broad range of impacts, as well as acknowledging a broader range of policies which address them, involves an integrated decision support. Life cycle thinking could in this sense be useful in order to foster an overall sustainable seafood production.

Altogether, there is a need for management action in order to minimize trade-offs from seafood production, and promote overall sustainable development. This requires, among others, using integrated perspectives on how to promote sustainable use of resources in the broader perspective. Maybe, a new concept that could be called Ecosystem and Life Cycle Approach to Fisheries (ELCAF)!

Future outlook

The most important future work involves more direct collaboration with policy makers in order to identify how LCA best could be operational as a policy tool.

In terms of methodology, further development of the PPR concept is needed and may also involve testing new approaches to trophodynamic indices. This work could include utilizing size based metrics, as these have been shown to better capture fisheries-induced changes in community structures than changes in trophic levels. Doing so, the impact on target species would also be assessed in a broader sense. It would also be most interesting to follow the development on how trophic indicators could be applied in a management context and how they could be used for optimizing and mitigating effects of seafood production from capture fisheries.

It would also be interesting to include comparisons with farmed seafood, given the importance to global seafood production. It would be most interesting to apply the systems perspective to study the trade-offs and shifts of environmental impacts from requiring different amounts of PPR, land- or seafloor use, toxicity from antifouling, etc.

Another interesting research topic to follow up on is if and how LCA methodology could link to ecological risk assessments of fisheries (Hobday *et al.* 2011). In this sense, it would be interesting to develop indicators for encompassing incidental catch of other marine animals, such as mammals and birds, and possibly refinements of seafloor impacts in seafood LCA.

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