The Fish We Feed Fish: An Evaluation Of Primary Production Requirements And Greenhouse Gas Emissions Of Reduction Fisheries In The Age Of Sustainable Boundaries

by

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"It is almost as though we use our military to fight the animals in the ocean. We are gradually winning this war to exterminate them. And to see this destruction happen, for nothing really – for no reason – that is a bit frustrating. Strangely enough, these effects are all reversible, all the animals that have disappeared would reappear, all the animals that were small would grow, all the relationships that you can't see any more would re-establish themselves, and the system would re-emerge. So that's one thing to be optimistic about. The oceans, much more so than the land, are reversible…"

- Daniel Pauly

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Abstract

Human society has placed substantial demands on the limited natural environment through increasing resource use and waste production. One of the greatest demands is the increasing appetite for animal-based food products. Thus, it is necessary to quantify relative merits and demerits of the relative and cumulative extent of impacts of food production sectors. This thesis evaluates the global scale of fishmeal and fish oil products from dedicated reduction fisheries towards two areas: global climate change and appropriation of primary production. To evaluate the impacts of these products, advancement to the current method for assessing primary production required in life cycle assessment was necessary. This thesis demonstrates that reduction fisheries' products are not equally impactful, and that greater consideration should be given to their use from both an efficiency and total impacts perspective. This work is contextualized with a focus on humanity's safe operating space.

List Of Abbreviations Used:

BAT- Biophysical Accounting Tool

BRU- biotic resource use

CO_2-e – carbon dioxide equivalents

DHC- direct human consumption (in contrast to reduction fisheries)

EFA- Ecological Footprint Analysis

EROI- Energy Return On Investment

FAO- Food and Agricultural Organization, subdivision of the United Nations

FMFO- fishmeal and fish oil

HANPP- human appropriation of net primary production

L- trophic level

LCA- Life cycle assessment

LME- Large marine ecosystem

MFA- Material Flow Analysis

NPPR- Net primary production required

PPR- Primary production required

SFA- Substance Flow Analysis

T- transfer efficiency

ISO- International Organization for Standardization

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Chapter 1. Introduction

Human society has placed substantial demands on the limited natural environment through increasing resource use and waste production (Millennium Ecosystem Assessment, 2005; Rockström et al., 2009). One of the greatest demands is the increasing appetite of developing and developed countries for animal-based food products (Alexandratos & Bruinsma, 2012). This demand led to the limits of natural pasture land being replaced by limits of arable agricultural land to provide feed for these animals (Foley et al., 2011), and in marine environments with wild capture fisheries being substituted by a growth in aquaculture production (Goldburg & Naylor, 2005). The environmental impacts of animal food production are manifold, and their pressure is being exerted at global and local levels (Cederberg, Hedenus, Wirsenius, & Sonesson, 2013; Foley et al., 2011; Pelletier & Tyedmers, 2010a). These environmental impacts can be examined in the important sub-systems of food production, such as reduction fisheries. Reduction fisheries operate at the intersection of capture fisheries and livestock, including aquaculture, production. Although there is diversity in species targeted, these fisheries typically target small pelagic finfish species. Upon landing, these species are reduced, or processed, into fishmeal and fish oil. Historically these products have found markets in a variety of sectors but increasingly are used by the aquaculture industry (Klinger et al., 2013). As this sub-sector has an important role in global capture fisheries and livestock production, it is important to understand its own environmental impacts.

Global society continues to engage to minimize environmental impacts at all scales. Through this process, a number of quantification strategies have arisen under the broad title of biophysical accounting tools. To understand and potentially minimize humanity's environmental impacts, quantification of these impacts and the implications of alternative production systems is a necessary first step. Only a few biophysical accounting tools include measures of our dependency on living (i.e. biotic) resources, to quantify human reliance on natural ecosystems. Two techniques that can be used to understand this dimension are ecological footprint analysis (EFA) and life cycle assessment (LCA). Both of these have previously been used to account for environmental and ecological impacts of agriculture (Antón, Castells, & Montero, 2007; Braband, Geier, & Köpke, 2003; Deutsch & Folke, 2005; Haas, Wetterich, & Köpke, 2001; Jansson, Folke, & Rockström, 1999), livestock (Cederberg et al., 2013; Pelletier, Lammers, Stender, & Pirog, 2010; Pelletier, 2008; Thomassen, Dalgaard, Heijungs, & Boer, 2008), and fisheries and aquaculture (Folke, Kautsky, Berg, Jansson, & Troell, 1998; Papatryphon, Petit, Kaushik, & van der Werf, 2004; Pelletier et al., 2009; Tyedmers, 2000; Ziegler, Nilsson, & Walther, 2003) production systems. However, their current application of accounting methods for understanding human impacts on living resources derived from capture fisheries is disparately applied, underdeveloped conceptually, and limited in its application.

1.1 Aim

Thus, this thesis is focused towards improving our understanding of the biophysical implications of fishmeal and oil (FMFO) production and use. These products were evaluated to discern relative differences in their abiotic and biotic impacts, and the total extent of these impacts. However, the current method for evaluating biotic impacts was not specific enough to the scale of this analysis, and so I refined the current method used in LCA. In addition, I evaluate the extent of these impacts of this sub-sector to operationalized sustainable limits for the relevant biotic and abiotic impacts. These aims will be achieved through two applications.

First, I review the current method for evaluating biotic resource dependency of marine products within LCA, and I refine the use of this metric. The refined method proposed is then applied to a commonly studied sector, Atlantic salmon (*Salmo salar*) feeds, to test the refined method against the conventionally applied method through an LCA framework. Second, I evaluate the biotic and abiotic impacts of fishmeal and oil products from a broad range of dedicated reduction fisheries that together represent approximately 50% of total global 2012 reduction fisheries. Both analyses consider the sustainable limits of this appropriation within large marine ecosystems, while the research on global reduction fisheries also addresses the scale of the FMFO sector's contribution towards humanity’s sustainable limits of climate change. The global overview is applicable to all consumers of fishmeal and fish oil products and a novel application to reduction fisheries globally with respect to sustainable boundaries.

1.2 Fisheries And Aquaculture

Food production systems are inherent to human survival, and the acquisition of fish protein specifically plays a vital role in the livelihoods and food security of over a billion people world-wide (FAO, 2014a). Historically, fish protein from capture fisheries has made up the majority of that consumed. However, overfishing has led to 90% of marine fish stocks being fully exploited or in a state of overexploitation (FAO, 2014a). The current state of fish stocks means they have been reduced below their optimal levels for maximum economic gain and maximum sustainable yield (Kelleher, Willmann, & Arnason, 2009; Worm et al., 2009). To continue to meet the growing demand for fish protein, aquaculture has risen as a dominant sector over the past 50 years and now accounts for almost half of all fish for human consumption (FAO, 2014a).

Many aquaculture systems that were previously unfed and relied on natural foraging in a restricted area have shifted to be at least partially supplemented by the addition of agricultural, livestock, and fisheries inputs. Currently, almost 70% of aquaculture’s production is fed aquaculture, which often uses wild fish inputs for feed for the culture species (FAO, 2014a; Tacon & Metian, 2009). The culture of carnivorous fish is of most concern for their often high levels of marine inputs (Naylor & Burke, 2005), which totaled 50% of the feed as a global average for salmonids in 2006 (Tacon & Metian, 2008). However, there is a current trend to minimize the amount of marine inputs into feeds as the sector continues to grow and competition for these high quality protein and fat sources grow increasingly expensive (Tacon & Metian, 2008). Despite various levels of FMFO inclusion in feeds across species, aquaculture broadly remains a net supplier of fish protein for human consumption but concerns have continued to be expressed regarding the use of fish for feed (Fréon et al., 2013; Naylor et al., 2000, 2009; Wijkström, 2009, 2012).

Reduction fisheries are those that catch mainly small pelagic fish like sardines, although a substantial diversity does exist in target-species trophic level, habitat, and includes non-fish species such as krill, and converts them into fishmeal and fish oil (FMFO) for the feed production industry. The fed aquaculture industry thus relies heavily on the production of FMFO from reduction fisheries, which is a large extractive industry that landed 16.3 million tonnes of round fish in 2012 (Figure 1; FAO, 2014). The characteristics of the fish commonly caught by reduction fisheries, including their typical short-life history and high fertility rates (Alder, Campbell, Karpouzi, Kaschner, & Pauly, 2008), make them likely more resilient than many species that are commonly targeted for direct human consumption, but not immune from collapse as has been seen (Myers, Hutchings, & Barrowman, 1997). They are often, but not exclusively, lower trophic level species, which thus require less net primary production than higher trophic level species because of the loss of energy up the food chain. However, their position often in the middle of food webs causes special concern regarding their overharvest for marine mammals and sea birds (Cury et al., 2011; Smith et al., 2011). In addition, reduction fisheries contribute to multiple biotic impacts such as decreased population of the target species, and by-catch (Alder et al., 2008; Naylor et al., 2009). These current concerns regarding reduction fisheries relate broadly to biotic impacts caused by their removal from ecosystems. These individual impacts can result to cumulative effects on ecosystems and their constituent parts.

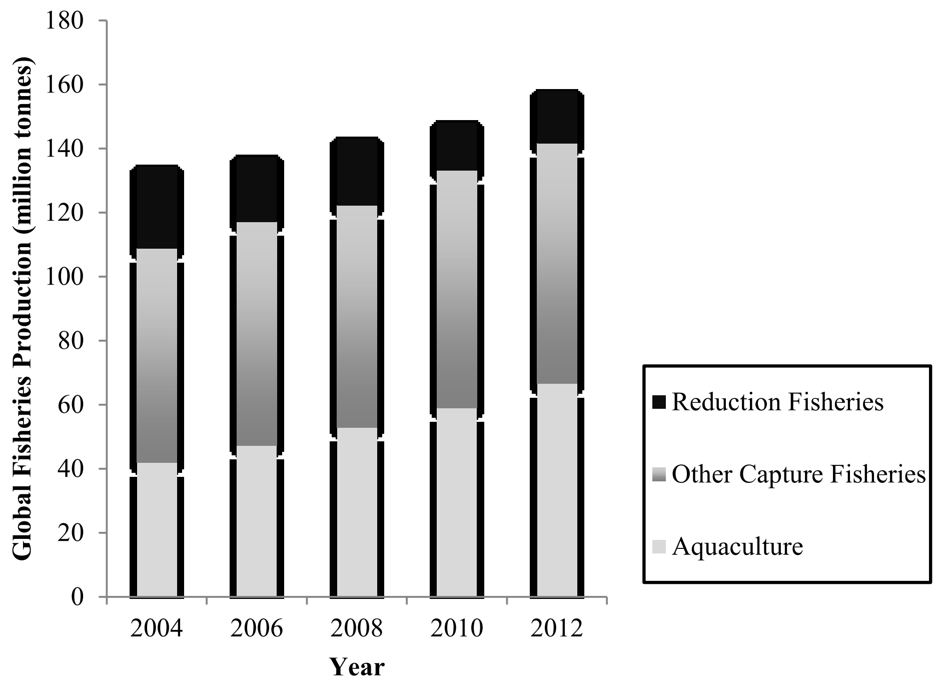


Figure 1. Global fisheries production separated by reduction fisheries, other capture fisheries, and aquaculture.
Adapted from data from (FAO, 2006, 2008, 2010, 2012, 2014a).

While biotic concerns have dominated the previous focus of fisheries work, there has arisen a body of work devoted to other biophysical implications, especially focused on abiotic impacts resulting from fisheries, and food production more broadly. These concerns include fuel and energy use and their resulting acidifying and greenhouse gas emissions, as well as photochemical ozone creation and eutrophication. The abiotic and biotic impacts of reduction fisheries are not fully known, but many methods have been used to attempt to quantify the impacts of these industries.

1.3 Biophysical Accounting Tools

Biophysical accounting tools were originally developed to quantify resource, energy and waste concerns of human economic activity (Baumann & Tillman, 2004). This underlying purpose still underpins these tools today, but they has also been broadened to attempt to account for environmental impacts at micro and macro scales. The fundamental purpose of these tools is to produce information for a better understanding of the material and energy flows that underpin economic activity, but are often not directly visible. The current economic system often ignores this information, which creates a disparity between market signals and environmental benefits (Farley, 2012; Kosoy & Corbera, 2010; Pelletier & Tyedmers, 2011; Wilson, 2013). These tools are separate from 'holistic' sustainability or sustainable development assessment methods, which often involve measures of human wellbeing and socioeconomic development (e.g. genuine progress index and the human development index; Ness, Urbel-Piirsalu, Anderberg, & Olsson, 2007; Singh, Murty, Gupta, & Dikshit, 2012). Additionally, biophysical accounting tools are not in, and of themselves, systematic decisions tools that give an end result of a preferable alternative (e.g. multi-criterion decision analysis), but aim to elucidate biophysical realities of existing systems and economies (Ness et al., 2007). All biophysical accounting tools describe current systems and their attributes (Robèrt et al., 2002). However, these tools can be used to model potential future use of resources or environmental impacts based on potential changes (Pelletier & Tyedmers, 2010a; Weidema, 2003). There are various examples of biophysical accounting tools that will be addressed below and I will compare them by their scale, their motivation, their area of focus or concern, and their worldview (Figure 2).

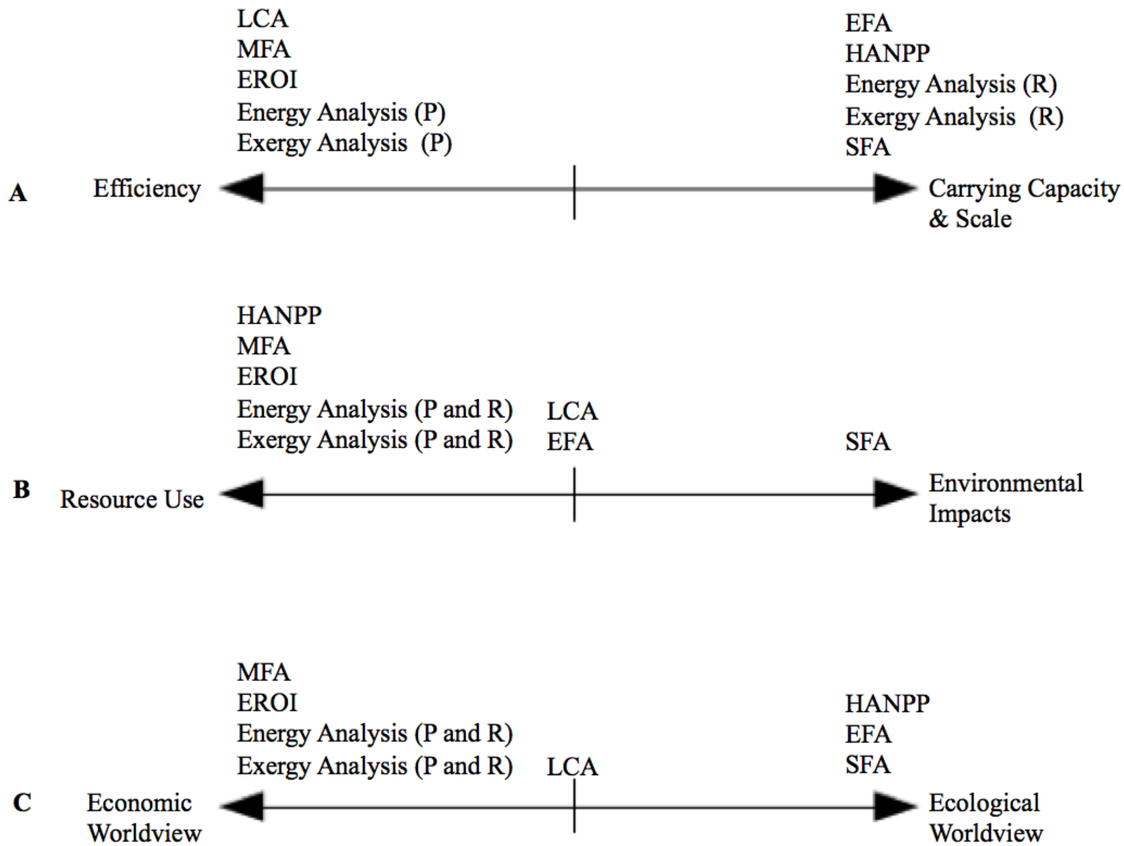


Figure 2. Common Assessment methods and their relative position on three spectrums of: A) motivation as compared by carrying capacity or efficiency; B) concern as compared by resource use or environmental impacts; and C) worldview or conceptual framework as compared by economic or ecological
LCA: Life Cycle Assessment; EFA: Ecological Footprint Analysis; EROI: Energy Return On Investment; HANPP: Human Appropriation of Net Primary Production; MFA: Material Flow Analysis; SFA: Substance Flow Analysis. P: Product Level Analysis; and R: Regional Level Analysis (for energy and exergy analysis).

Biophysical accounting tools vary in their focus on micro or macro level concerns. This can also be described as whether they focus at the product level or at a broader level of the region, nation or global scale (Ness et al., 2007). Life cycle assessment focuses solely on individual products or projects. Alternatively, many other types of material and energy analysis (material flow analysis [MFA], EFA, energy and exergy analysis, etc.) can be focused at either this scale or a broader regional scale. Other tools like human appropriation of net primary production (HANPP) have only been performed at regional and global scales, in addition to the suite of energy and material analyses more commonly performed at this scale (substance flow analysis [SFA], and energy analysis). The scale at which the analysis is performed often is an indication of the major motivation of the analysis.

The two major motivations I have categorized are: i) an efficiency or eco-efficiency motivation; and ii) a carrying capacity or scale-focused motivation. Efficiency is a broad concept of achieving more useful outputs with less or the same inputs, often focused on economic gain, but increasingly on resource, energy and waste concerns. Eco-efficiency is the specific version that seeks to reduce the material and energy intensity to perform the same task and thus results in environmental benefits of reduced wastes produced, or reduced material and energy required. In contrast, scale is the relation of humanity's use of resources or environmental impacts to a societal level or in relation to the assimilative capacity of the natural environment. This is a common distinction made in ecological economics as two ways to understand humanity's impact on the environment and how to address it (Costanza, Cumberland, Daly, Goodland, & Norgaard, 1997; Huesemann & Huesemann, 2008). Many have argued that eco-efficiency will solve the environmental challenges of today (Hawken, Lovins, & Lovins, 1999; Simon, 1995). Others have argued against this efficiency paradigm through the 'rebound effect' that often occurs with efficiency gains that promote further use of resources and thus increased environmental impacts (Daly, 2013; Huesemann & Huesemann, 2008; Jänicke & Lindemann, 2010; Lawn, 2001). In addition to the rebound effect, focusing solely on increasing eco-efficiency does not necessarily reduce environmental impacts to sustainable levels, as the current scale of the system may already be unsustainable in a broader focus. Although these tools have a clear focus on either efficiency gains (e.g. LCA, MFA and EROI) or total impact through scale and carrying capacity (e.g. EFA, HANPP, and SFA), their focus does not necessarily align with their respective motivation. The motivation with which these analyses are performed can be because of biophysical concerns, such as SFA focused on the emissions of chemicals to the environment and their effects on ecosystems and human health. While these tools differ on their motivation, they also clearly differ on the main subject they attempt to quantify.

The major concern addressed using biophysical accounting tools is material and energy flows from both biotic and abiotic sources. A renewed interest in material and energy concerns came out of the energy crises of the 1970s (Baumann & Tillman, 2004), although there is a greater history of interest in accounting for energy and land use concerns (Røpke, 2004). Reductions in either material or energy use would increase efficiency to potentially reduce economic or environmental costs. The earlier forms of analysis including energy analysis and MFA focus solely on resource use, while some developers have included quantifying relevant environmental impacts resulting from these material and energy flows. The biophysical accounting tools that include environmental impacts do so in addition to other foci, such as resource concerns in LCA (Baumann & Tillman, 2004). Along with motivation, these areas of concern can be traced back to the worldview these tools arose from and reflect in their practice.

The worldview or conceptual framework from which these tools arise can be understood or traced by understanding the perspective their developers and promoters regarding the relationship between the environment and the economy. An economic worldview is in line with current mainstream economics, and for the purposes of this delineation supports the view that the environment is a subsystem of the economy in that the environment produces raw materials and absorbs waste from the economy (Edwards-Jones, Davies, & Hussain, 2000). An ecological worldview comes out of ecology and has been adopted in ecological economics, as a critique of mainstream economics, that recognizes that the environment supports all economic activity by providing basic resources and assimilating wastes, and this environment is not able to be replaced by human technology (Edwards-Jones et al., 2000; Farley, 2012). Under this framework, many have developed biophysical accounting tools that are often rooted in ecology (Edwards-Jones et al., 2000). These tools including EFA and HANPP are rooted in the bioproductive capacity of the Earth (Haberl, 1997; Krausmann et al., 2013; Vitousek, Ehrlich, Ehrlich, & Matson, 1986), and for EFA its ability to assimilate wastes (Wackernagel & Rees, 1996;

Wackernagel & Yount, 1998). Many other biophysical tools do not incorporate the fact that material and energy flows originate from the natural environment, and these tools generally reflect the users' view that human activity is separate from the environment (Røpke, 2004).

These different assessment techniques thus represent different attempts to quantify the biophysical underpinnings of human economic activity, from the individual product to the global scale. Biophysical accounting tools can be used to understand efficiency and carrying capacity concerns of different systems. The work of this thesis draws upon the methods of LCA and EFA to understand reduction fisheries in both these areas, and I will now address these methods specifically.

Life cycle assessment emerged for the analysis of materials and energy over the life-cycle of a product. It began as a measure of efficiency and to quantify resource use, but grew into accounting for processes' potential impacts on the natural environment and human health. This method is now used to quantify many environmental impacts over the complete life cycle of a product or process from raw material extraction to end of life disposal (i.e. from 'cradle-to-grave'), although, in practice, it is often used with different downstream boundaries (i.e. from 'cradle-to-mill' focusing on raw material production or 'cradle-to-gate' focusing on a finished product before use and disposal) depending on the focus and concern of the study. Life cycle assessment can also quantify resource dependence of products to demonstrate the often hidden resource intensity of certain products (Baumann & Tillman, 2004). This is particularly useful because it takes a holistic view of the sources of resource depletion and environmental impacts of good or service provisioning, and shifts the scope from solely local issues that are often the focus of popular environmental concerns.

Life cycle assessment is used to quantify resource depletion and environmental impacts through aggregating relevant inputs and waste streams, respectively, in impact categories (ISO, 2006). Impact categories are used to sum the sources of a specified environmental impact into a common unit called an indicator (ISO, 2006). For example, the impact category of global warming potential is expressed in kilograms of CO_2-e, where e stands for equivalents, and all greenhouse gases are re-expressed to their common unit of CO_2-e by established equivalencies. While abiotic impacts like eutrophication (with an indicator of PO_4-e), acidification (with an indicator of SO_2-e), and global warming potential (CO_2-e) are common impact categories used in food production LCAs, the quantification of biotic impacts is less developed and applied.

Previous LCA research has attempted to quantify relative biotic concerns of different production systems. Agriculture LCAs have often focused on impacts to biodiversity through land-use and changes to land-use especially when considering land recently converted to agriculture from grassland or forest (Lindeijer, Müller-Wenk, & Steen, 2002; Schenck, 2001). Fisheries and aquaculture LCAs have focused on a wide diversity of impacts including by-catch and seafloor impacted by fishing gear (detailed below). However, the most commonly applied indicator, although still recently emergent, of the use of biotic resources in seafood LCAs is the primary production required (PPR) measure with a common unit of kilograms of carbon obtained through primary production (i.e. photosynthesis; Avadí & Fréon, 2013; Pelletier et al., 2007). Almost all living organisms obtain their energy through photosynthesis at some level, and the transfer of that energy up trophic pyramids demonstrates the dependence on primary production of living organisms. Animal food production, including aquaculture, utilizes biotic resources (e.g. agricultural products, wild fish, etc.) in transforming feed into food for human consumption.

The amount of primary production required can be broadly seen as a measure of the eco-efficiency of food production systems. This measure is useful as products derived from organisms that naturally or functionally (e.g. based on what they are fed by humans) feed on higher food-chain items require greater PPR per unit of production because of losses of energy in trophic relationships (Pauly & Christensen, 1995; Slobodkin, 2001). Thus this measure can be used to quantify the comparative biotic energy required for different food products (Tyedmers, 2000). This measure has increasingly been integrated into LCAs of food production systems (Hornborg, Nilsson, Valentinsson, & Ziegler, 2012; Papatryphon et al., 2004; Pelletier & Tyedmers, 2007), but its current use is diverse. The strengths and weakness of current practice and its potential refinement is an important aspect of this thesis, and is detailed in Chapter 2.

The method of estimating PPR in LCA was adopted from EFA, but in EFA this appropriation is expressed as an area measure (Folke et al., 1998; Larsson, Folke, & Kautsky, 1994). EFA has been used to demonstrate the vast amount of human appropriation of the biocapacity of the planet's ecosystems that are not directly visible; resource use of cities and their required productive area has often been a key example (Wackernagel & Rees, 1996). It has also been applied to terrestrial food production systems (Deutsch & Folke, 2005; Wackernagel et al., 2002), and to the growing sector of aquaculture including its wild marine inputs (Folke et al., 1998; Kautsky, Berg, Folke, Larsson, & Troell, 1997; Larsson et al., 1994; Parker & Tyedmers, 2012b; Tyedmers, 2000). While it is often applied at a holistic scale, this thesis only focuses on the subsections that address greenhouse gas emissions (carbon footprint), and the marine ecosystem area's primary production appropriated through fishing (marine footprint). While this is not a full application of the ecological footprint concept, it fits firmly within the EFA framework in measuring human impact in relation to bioproductive and waste assimilation capacity.

As previously mentioned, biophysical accounting tools can focus on two issues of human environmental impact: eco-efficiency and impact relative to total scale or carry capacity. An appropriate scale for human activity has recently been expressed as planetary boundaries or limits for specific environmental impacts that if society exceeds will likely lead to large-scale ecological damage (Rockström et al., 2009). Human society must work towards both eco-efficiency and total scale limit concerns, and this discussion is important to how this thesis addresses biotic resources. LCA is focused towards eco-efficiency while ecological footprint is focused towards scale, however both can inform the other perspective through expanding or minimizing the scope of the study.

1.4 Assessing Impacts On Living Resources

Outside of biophysical accounting tools, there have been many methods developed to understand negative effects on living resources often due to harvesting. Most pertinent to this thesis is the body of work established as fisheries science and management for the purpose of monitoring and utilizing fish populations for human benefit. Fish stocks have often been the common measurement for monitoring a single population over a specific geographical area (Iversen, 1996). Measuring fish in this way led to the development and increased use of single stock assessment techniques to measure dynamic populations for the purpose of establishing annual catch amounts and total allowable catch (TAC; Iversen, 1996).

The benefit of single stock assessments and management is the ability to monitor and influence the state of the population for maximizing human benefit. Traditionally, fisheries managers attempt to maximize harvestable fish biomass by maintaining total mortality (e.g. the combination of natural and fishery-induced mortality) at a level that maximizes population growth. Human benefit can thus be maximized depending on the focus of maximum fish biomass yields (maximum sustainable yield [MSY]) or maximum economic gain (maximum economic yield [MEY]; Royce, 1996). These different assessment and evaluation methods balance the need for the health of the fish stock and the desire for human benefit from it (Iversen, 1996; Kelleher et al., 2009; Royce, 1996). These methods reliance on limited (e.g. inferring population from catches) and uncertain (e.g. unknown natural growth and mortality rates) data have been central challenges to their effective application to fisheries management (Larkin, 1977; Walters & Maguire, 1996).

Single stock assessments came under criticism for not considering the full suite of ecosystem effects and the role of different species within ecosystems (Pauly et al., 2003). The, as yet evolving, theory and practice of ecosystem-based management (EBM) has arisen in response. Under EBM, effort is focused on managing ecosystems as a whole because fisheries affect not only the target population but potentially disrupt food web interactions through altering predator and prey populations (Dickey-Collas et al., 2013; Royce, 1996). Along with this management approach, a suite of indicators has arisen from fisheries science to complement this practice. Two of these indicators, mean trophic level (MTL; Branch et al., 2010; Hornborg, Belgrano, Bartolino, Valentinsson, & Ziegler, 2013; Pauly, Christensen, Dalsgaard, Froese, & Torres, 1998; Pauly & Christensen, 2000) and the fishing-in-balance index (FiB; Christensen, 2000; Pauly & Watson, 2005; Walters, Christensen, & Pauly, 1997) are focused on the trophic level of species harvested from the ecosystem.

The MTL measure has demonstrated a global trend of progressively fishing lower trophic level species as higher trophic level species are exhausted (i.e. 'fishing down the food web'; Pauly et al., 1998). Fishing down the food web would eventually lead to declining global landings (Pauly et al., 1998; Pauly & Christensen, 2000). However, intentionally fishing at lower trophic levels could yield more available fish biomass for human use and would result in a lower MTL that is not reflective of declines in high trophic level species (Essington et al., 2006). The fishing down the food web hypothesis has been demonstrated in various examples (Pauly & Watson, 2005; Pauly et al., 2001). However, there have also been intentional decisions to fish lower trophic level species for other benefits (i.e. an additional fishery for 'fishing through the food web'; Essington et al., 2006), and global catch MTL is highly influenced by annual variation of catch (e.g. high variation in landings of Peruvian anchoveta [*Engraulis ringens*]) and uncertain data parameters (e.g. trophic levels; Branch et al., 2010). Furthermore, it is unclear if MTL effectively addresses the impacts of fisheries on fished populations (Branch et al., 2010). The FiB index is meant to refine the intent of the MTL measure by estimating whether the decrease in MTL is occurring at a rate that properly considers the increase in available biomass harvestable by fishing at a lower trophic level (Pauly & Watson, 2005). This has been applied to contextualize fisheries relative to a base year and document their expansion or possible collapse (Cury et al., 2005).

Another coarse ecosystem-based indicator is the PPR measure discussed above. As previously mentioned, the calculation of PPR attempts to equate the harvest of these biotic resources to the base level of photosynthetic activity required to produce them. This measure equalizes the catch of different species to an equal unit of primary production and attempts to quantify pressure on the source ecosystem based on less biomass energy being available for other purposes. These measures of MTL, FiB, and

PPR are focused on impact on the whole ecosystem and represent a shift from single stock assessments only. However, analyses of fisheries have also been broadened to account for other issues of human concern and the natural environment.

Various forms of energy analyses have been applied to fisheries to understand the economic and energy costs of fisheries. EROI has been applied to compare gross energy inputs to food energy outputs of fisheries as a measure of efficiency (Tyedmers, Watson, & Pauly, 2005). Studies of varying fuel use intensity (FUI) have demonstrated a large range of fuel inputs among capture fisheries that yield substitutable products of fish for human consumption (Parker & Tyedmers, 2014; Tyedmers et al., 2005). Lastly, a few studies of energy inputs into fisheries have been performed over the years (Mitchell & Cleveland, 1993; Parker & Tyedmers, 2014; Schau, Ellingsen, Endal, & Aanondsen, 2009; Tyedmers et al., 2005; Tyedmers, 2001; Watanabe & Okubo, 1989), and fisheries have more recently become adopted as an area of research in LCA.

Early LCAs of fisheries focused on common species fished in the North Atlantic (Eyjólfsdóttir, Yngvadóttir, Jónsdóttir, & Skúladóttir, 2003; Thrane, 2004; Ziegler et al., 2003). Immediately, these analyses recognized the need for fisheries-specific LCA indicators to account for abiotic and biotic impacts specific to fisheries (Eyjólfsdóttir et al., 2003; Thrane, 2004; Ziegler et al., 2003). Conventional LCA practice only accounted for material and energy flows and generally did not include impacts of a biotic nature because of the history of research on chemical processes and packaging (Baumann & Tillman, 2004). These fishery specific indicators have been adopted and formulated to account for the unique impacts of fishing in a holistic way including the impacts of trawling (seafloor swept; Ziegler et al., 2003), discards (Global Discard Index; Vázquez-Rowe, Moreira, & Feijoo, 2012), primary production required (PPR, and other terms of biotic resource use and net primary production; Hornborg et al., 2012; Parker & Tyedmers, 2012; Vázquez-Rowe et al., 2012; Vázquez-Rowe, Villanueva-Rey, Hospido, Moreira, & Feijoo, 2014), the population status of by-catch (vulnerable, endangered, and critically endangered status of by-catch; Hornborg, Svensson, Nilsson, & Ziegler, 2013), marine toxicity from anti-fouling paint (Hospido & Tyedmers, 2005), and the wasted potential yield from overharvest (WPY; Emanuelsson, Ziegler, Pihl, Sköld, & Sonesson, 2014). Many of these indicators borrow from previously developed methods (e.g. WPY and PPR), or express previous human concerns related to fisheries and biodiversity (e.g. by-catch and impact on endangered species). This suite of indicators has never been comprehensively applied together, and is consistently being added to and adapted for improved quantification of the impacts of fisheries in LCA. These indicators attempt to make LCA a more holistic quantification of biotic and abiotic impacts of fisheries.

I have attempted to separate out the major focus or concern of these various techniques in Figure 3 into four categories: abiotic, human, stock, and ecosystem. The boxes of overlap between each area of focus show that these are useful in quantifying multiple items, or bridge the gap between these two concerns. For example, MSY is a measure that attempts to maximize the human benefit from fish stocks, while recognizing the need for this stock to maintain its own healthy population. Assessing or quantifying MSY thus bridges the concern for healthy stocks with human benefit and interest. The new LCA indicators, discussed above, were intended to quantify more abiotic and biotic concerns in the LCA framework. This thesis is rooted in this crossover space of improving LCA to better account for biotic impacts within its already robust framework.

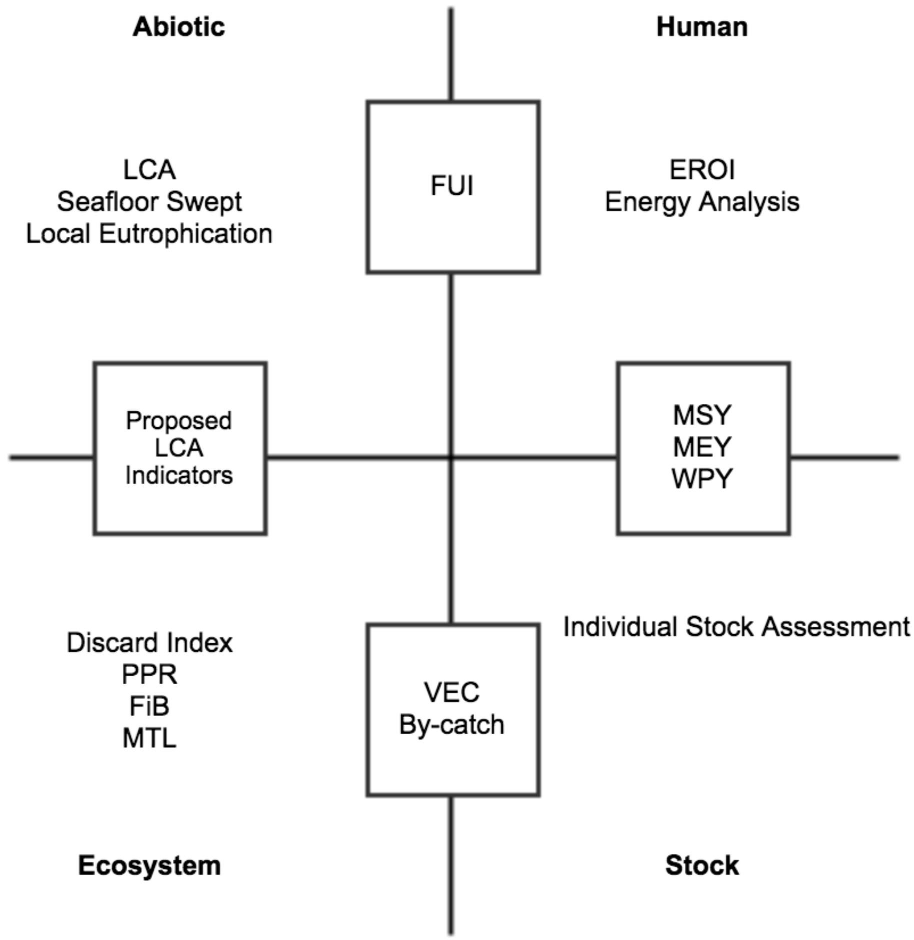


Figure 3. The focus/foci or concern(s) of different assessment techniques applied to fisheries and aquaculture including recently developed LCA indicators. The boxes of overlap between each area of focus show that these are useful in quantifying multiple items, or bridge the gap between these two concerns.
EROI: Energy Return On Investment; FiB: Fisheries-in-Balance; FUI: Fuel Use Intensity; LCA: Life Cycle Assessment; MEY: Maximum Economic Yield; MSY: Maximum Sustainable Yield; Proposed LCA Indicators include seafloor swept, local eutrophication, discard index, and VEC Status of By-catch; WPY: Wasted Potential Yield, VEC By-catch: Vulnerable, Endangered and Critically Endangered IUCN status of By-catch.

1.5 Objectives

The research that follows will improve understanding of biophysical implications of FMFO production from dedicated reduction fisheries. This includes the evaluation of wild marine inputs for their biotic and abiotic impacts to demonstrate relative efficiencies among the diverse set of marine feed inputs for aquaculture production. These impacts will then be contextualized for this sector for energetic appropriation from marine ecosystems and their contribution to global carbon emissions. To better evaluate the biotic impacts of reduction fisheries, the method for estimating primary production required had to be refined based on current weaknesses of the method (Chapter 2). The objectives are thus:

- To clarify the current use of the primary production required metric in LCA, and refine it to be more context specific to yield more accurate results in LCAs of fisheries systems; and
- To evaluate the primary production required and greenhouse gas emissions of FMFO products from dedicated reduction fisheries with respect to relative and absolute impacts.

1.6 Significance

Environmental sustainability must be a goal of global human society in the pursuit of sustainable development. To pursue environmental sustainability we must be able to evaluate the impacts of our society and the products we use everyday. Therefore, understanding human society's resource use and environmental impacts for food production is a necessity. Both EFA and LCA can be used in determining the resource intensity, efficiency and scale of human appropriation of biotic resources, as well as other environmental impacts. Therefore, the production of FMFO from diverse sources must be evaluated for its abiotic and biotic impacts to discern differences among these products, and the extent of the impacts from this sub-sector. To evaluate the biotic impacts of these products with respect to their diversity, it was necessary to improve the current method of evaluation to be more context specific to the products and its source species and ecosystems. This research thus informs biophysical sustainability implications of reduction fisheries themselves, and the current dominant user, the aquaculture industry.

Reduction fisheries represent a substantial portion of total world fisheries. These fisheries support the global expansion of fed aquaculture. However, the alternative of a greater utilization of by-products and by-catch from currently established direct human consumption fisheries is becoming more commonly accepted. Therefore, this thesis will aim to illustrate the environmental impacts associated with reduction fisheries and a partial comparison to some current direct human consumption fisheries by-products. For feed producers to pursue sustainable aims, proper information on environmental and ecological impacts must be available. This thesis will thus hopefully serve to illustrate the relevant efficiency and scale concerns in current reduction fisheries to demonstrate the merits and demerits of different species, ecosystems, fisheries, and the sector as a whole.

1.7 Organization Of Thesis

The balance of this thesis is organized around two manuscripts that have been prepared for submission to academic journals along with a final concluding chapter. Chapter 2 is a manuscript for publication that reviews the current use of the PPR indicator, and proposes practice to become more region and species-specific in its quantification of ecosystem impact. The current use of this primary production focused indicator of biotic impact has become more common in the LCA literature of animal food production systems, but there is disparity in its application. This chapter focuses on clarifying current use, and the application of a proposed revised method to the 2012 average Norwegian salmon feed.

Chapter 3 is an evaluation of the abiotic and biotic impacts of fishmeal and fish oil products from dedicated reduction fisheries covering approximately 50% of this sector for 2012. These products' abiotic and biotic impacts are evaluated by their cumulative greenhouse gas emissions and their primary production required, respectively. Results are expressed in terms of tonnes of CO_2-e and sea area required for primary production (km^2) per tonne of fishmeal and oil products produced. Estimates of the total scale of PPR and greenhouse gas emissions from total global FMFO production from reduction fisheries are made and compared to total source ecosystem primary production and estimated thresholds for global climate change (350 parts per million), respectively.

Chapter 4 presents a broader discussion of the findings of the thesis. The major themes articulated are: 1) concerns of improving efficiency in food production as compared to the total impact these systems are having at a planetary scale; 2) the potential limitations of biophysical accounting tools; and 3) the future of aquaculture and reduction fisheries given the growth of aquaculture to meet demand of the growing affluent human population. Finally, conclusions and outlooks for future research are also presented.

Appendix A contains data tables, and other results not detailed in Chapter 3 as it was intended as a manuscript for publication.

The manuscripts for publication were both co-authored, where Tim Cashion, the author of this thesis, was the primary author for all writing and analyses. Chapter 2 was co-authored with Sara Hornborg, Friederike Ziegler, Erik Hognes and Peter Tyedmers all assisting in conceptual development and editorial assistance. Chapter 3 was co-authored with Peter Tyedmers who had editorial and supervisory oversight, and assisted in concept development, and Robert Parker who assisted with the data and analysis of fuel use of fisheries.

Chapter 2. A Review And Advancement Of The Marine Biotic Resource Use Metric In LCAs: A Case Study Of Norwegian Salmon Feed

2.1 Introduction

As human consumption of Earth's biocapacity continues to grow at unprecedented rates (Foley, Monfreda, Ramankutty, & Zaks, 2007; Rockström et al., 2009) with food provisioning as a leading driver (Foley et al., 2011), understanding and limiting the scale of our food-related impacts is critical. A suite of techniques has been developed for this purpose that employ a variety of metrics underpinned and motivated by a range of theoretical backgrounds. One of the increasingly popular frameworks is life cycle assessment (LCA), intending to quantify the scale of material and energy resource requirements and resulting waste stream implications associated with the provision of a product or process from 'cradle-to-grave' (ISO, 2006). However, while these biophysical accounting tools may quantify many environmental impacts, quantifying the extent of human dependence on biotic or living resource concerns is still underdeveloped. Here we set out to review and propose refinements to emerging practice within LCA to account for marine living resource utilization.

Techniques for understanding and measuring the extent of human utilization of and/or impact on living resources are diverse and reflect differences in the analysts' motivation and theoretical background. Ecological impacts may be quantified through land use changes (Foley et al., 2005; Lindeijer et al., 2002), ecosystem and biodiversity assessments after human activities (Chapin et al., 2000), loss of ecosystem services (Worm et al., 2006), and they are also predicted through environmental impact assessments (Noble & Storey, 2005). In turn, the human use of biotic resources has been quantified through their energy content specifically to energy return on investment (Draganovic et al., 2013; Pelletier et al., 2010; Tyedmers, 2000), remaining useful energy through emergy analysis (Wilfart, Prudhomme, Blancheton, & Aubin, 2013), total biomass used (fish-in fish-out ratios used in aquaculture, input-output analysis; Shepherd and Jackson 2013), the primary production required (PPR) to produce said biomass (PPR, or human appropriation of net primary production; Pauly and Christensen 1995), and through area measures that quantify resources used and area required for waste assimilation (ecological footprint; Wackernagel and Rees 1996; Folke et al 1998; Haberl et al 2004). Previously, LCA did not account for ecological impacts or use of biotic resources in a nuanced way (Pelletier et al., 2007). However, researchers have developed indicators of relevant ecological impacts for fisheries and aquaculture including area of seafloor damaged by trawling gear (Ziegler et al., 2003), the incidental catch of vulnerable or endangered species (Hornborg, Svensson, et al., 2013), discarding of non-target species into the ocean (Vázquez-Rowe et al., 2012), and have proposed ways to assess biodiversity impacts and local eutrophication effects from net-pen salmon farms (Ford et al., 2012). These broader ecological impacts are traditionally not accounted for in LCA; therefore, these methods attempted to quantify the ecological impacts of fisheries and aquaculture more holistically within the LCA framework. However, the most commonly adopted ecologically based measure in LCA has been PPR.

Quantifying the extent of PPR as an indicator of biotic (i.e. living) resource use (BRU) dependency has had limited use in broader LCA practice, but has become common in recent LCAs of seafood production systems. The initial inspiration and continued methodological basis of the quantification of PPR research follows the logic and methods from Pauly & Christensen's (1995) paper on the PPR to sustain global fisheries using Equation 1. The PPR method for marine ingredients sets out to estimate the net primary production required to yield an amount of marine biomass at a trophic level above primary production. Through estimating the carbon content of the target species and the loss of energy through each trophic transfer, an amount of PPR can be estimated.

$PPR = C/M * (1/T)^{L-1}$ Equation 1

Adapted from Pauly and Christensen (1995)

Where C is the mass of catch, M is ratio of wet weight biomass to carbon, T is transfer efficiency, and L is trophic level.

This method yields a result of a mass of carbon, originally derived from photosynthesis, that is required to yield a specified mass of product of biological origin. It thus quantifies the human appropriation of primary production as a resource that has ecological impacts when removed. PPR has also been applied to terrestrial inputs that originate from primary production such as crops and livestock products.

The PPR method, however, has been applied in different ways and with different methodological choices. Furthermore, analysts applying the PPR measure commonly use estimates of average values for key input parameters that may not be representative of all species and ecosystems, nor do they reflect the inherent uncertainty and variability in their physical reality (Libralato, Coll, Tudela, Palomera, & Pranovi, 2008; Parker & Tyedmers, 2012b). This diversity of practice has its greatest potential impact when: a) modeling higher trophic level biotic inputs, because of the exponential effect of trophic level (Equation 1); and b) inputs are derived from wild ecosystems, where the amount of energy required to yield a higher trophic level species is unknown compared to controlled ecosystems like livestock production. These confounding challenges both arise when attempting to model the implications of wild-caught marine resource inputs to production systems such as aquaculture (Pelletier & Tyedmers, 2007). Thus, this chapter focuses solely on how marine inputs are considered in the PPR metric used in LCA, and the potential advancement of this metric with more specificity to the ecological and species attributes of these marine inputs.

2.1.2 Aim

This chapter has two main objectives: (1) reviewing use to date of the PPR method as it has been employed to quantify the extent of marine biotic resource dependency in LCA and related research; and (2) propose and apply a set of methodological improvements to the current marine PPR methodology for assessing primary production requirements. To illustrate the application of these proposed changes, we model marine resource dependencies of marine-derived inputs to the case study of Norwegian salmon aquaculture feed production in 2012.

2.2 Literature Review Of Primary Production Measures Of Seafood

The literature review was performed through attempting to collect all English-language published seafood LCA studies that included the PPR measure. Google Scholar was used as the primary database with the following search terms: life cycle assessment, LCA, biotic resource use, net primary production, primary production required. The studies were then analyzed for their subject, term used, functional unit, allocation, reason for quantification, and modeling undertaken to distinguish patterns, similarities, and divergences of practice.

2.2.1 General Patterns In The Quantification Of Primary Productivity Requirement To Date

In a review of LCAs of seafood literature (Table 1), there are a few major patterns that arise in the use of terms and rationale motivating the use of this metric. The term 'biotic resource use' and its associated acronym BRU have been applied most frequently (11 of 26)[1] and arise in both aquaculture and fisheries LCAs. The term 'net primary productivity used' and its associated acronym NPPU are as widely used (11 of 26) but only in relation to aquaculture studies. The more general, and arguably progenitor term 'primary production required' and its acronym PPR has been used less frequently (3 of 26) and exclusively for fisheries studies. The term PPR is used here because of the specificity with which it refers to its potential impact, rather than the more vague biotic resource use term.

Most studies justify the inclusion of this measure on the desire to quantify the amount of primary production necessary to form a certain product (Hornborg, Belgrano, et al., 2013; Pelletier et al., 2009), and the resulting demand/pressure this places on ecosystems (Pelletier & Tyedmers, 2007). This justification was also expressed as "a biotic resource…being unavailable for other purposes," in that if humans appropriate this

[1] While in total 26 studies were surveyed (Table 1), many of the items that are compared in the following sections are not applicable to all studies. Therefore, the number of studies indicated is not always in relation to all 26 studies, but relative to those studies where it was applicable.

primary production, which broadly represents the available energy of ecosystems, it is not available for other organisms to use (Papatryphon et al 2004, p. 318). Many studies did not explicitly state the reason for quantification of PPR; however, the motivation was assumed to be the same as the paper cited for the PPR method (i.e. method author). Thus, in all but two of the studies (D'Orbcastel, Blancheton, & Aubin, 2009; Jerbi, Aubin, Garnaoui, Achour, & Kacem, 2012) the motivation for the use of this measures is to quantify reliance, and potential impact on ecosystems, that result from the harvesting of the products of primary production. These other two studies use NPPU because "it measures the trophic level of the rearing system" under study (D'Orbcastel et al 2009, p. 115; Jerbi et al 2012, p. 4).

2.2.2 Similarities Of Use

Despite the variation in the descriptors employed, there is a broad similarity amongst studies in regards to biotic inputs included, method of calculation, and functional unit (i.e. the basis of analysis and comparison) choice. All studies included marine ingredients in their calculation of PPR. Every aquaculture study surveyed included agricultural products, marine products, and livestock products when used, in the calculation of total PPR. Only two studies explicitly included bait (Vázquez-Rowe et al., 2012, 2014) and three studies included a measure or accounted for the discarding of non-target species back into the ocean (Hornborg et al., 2012; Vázquez-Rowe et al., 2012, 2014).

All studies employed Pauly & Christensen's (1995) method (Equation 1) for quantification of marine inputs. This was initially adapted for LCA by Papatryphon et al (2003) to quantify the PPR of cultured trout production. This original adaptation, and all subsequent studies, adopted the estimated 10% average for transfer efficiency (Pauly & Christensen, 1995). Agricultural inputs into aquaculture feeds are accounted for through the crops' carbon content (Papatryphon, Petit, van der Werf, & Kaushik, 2003). Accounting for the PPR of livestock ingredients (mainly poultry by-products) was relevant to only three studies (Pelletier and Tyedmers 2007; Pelletier et al 2009; McGrath et al 2015). The use of PPR allows for various biotic inputs (e.g. agricultural, marine,

Table 1. Methodological Choices and Characteristics of PPR in Seafood LCAs arranged chronologically

Subject (Reference)	Term[a]	Method Author[b]	Allocation[c]	Explicit Reason for quantifying PPR	Modeling[d]
Aquaculture Trout (1)	BRU	(2)	E	None given	-
Salmonid Feeds (2)	NPPU	*	E	"Being unavailable for other purposes" (p. 318)	FU, SA, SM
Aquaculture Turbot (3)	NPPU	(2)	E	"Being unavailable for other purposes" (p.1262)	-
Aquaculture Salmon Feed (4)	BRU	(2)	N	Appropriation places increased pressure on ecosystems	AC, SM
Aquaculture Finfish (5)	NPPU	(2)	E	"Being unavailable for other purposes" (p. 356)	-
Aquaculture Trout (6)	NPPU	(2)	E	“It measures the trophic level of the rearing system” (p.115)	SA, SM
Aquaculture Salmon (7)	BRU	(4)	N	Net PPR to sustain feedstuffs	SM
Aquaculture Tilapia (8)	BRU	(4)	N	Net PPR to sustain feedstuffs	-
Salmonid Feeds (9)	NPPU	(2)	E	"Being unavailable for other purposes" (p. 64)	AC, SM
Aquaculture Shrimp (10)	BRU	(4)	N	None given	SA, SM
Aquaculture Seabass (11)	NPPU	(2)	E	“it measures the trophic level of the rearing system” (p. 4)	-
Swedish Nephrops Fishery (12)	PPR	(2)	E	Amount of PP to yield a product	SA, FU
Antarctic Krill Fishery (13)	BRU	*(2, 3, 7)	N	None given	SA, AC, SM
Multiple Fisheries (14)	BRU	*	-	“Removed carbon that was fixed through photosynthesis” (p. 539)	-
Poly-aquaculture (15)	NPPU	-	E	Amount of NPP to produce a product	-
Aquaculture Carp and Tilapia (16)	NPPU	(2)	E	NPPU refers to biotic resource use	-
Various Aquaculture species (17)	NPPU	(2)	E	"Being unavailable for other purposes" (p. 99)	SA, AC
Peruvian anchovy uses (18)	BRU	*	M	None given	SA
Global Fisheries (19)	NPP	(2)	-	Change in biomass production capacity	-
Peruvian anchovy products (20)	BRU	*	M	Primary production consumed by organism	-
European pilchard Fishery (21)	BRU	(2)	E	None given	SA, FU, SM
Sardine Fishery (22)	PPR	*	M	Estimate of magnitude of primary production needed	-
Aquaculture Salmon (23)	BRU	(5, 7, 11)	N	None given	SA, AC, SM, MC
Aquaculture Trout (24)	NPPU	(2)	E	Pressure on biotic resources	SA
Prawn Fishery (25)	PPR	*	-	Primary production consumed by organism	-
Poly-aquaculture (26)	NPPU	(2)	-/E and N	NPPU refers to biotic resource use	SA, AC

References: (1) Papatryphon et al 2003; (2) Papatryphon et al 2004; (3) Aubin et al 2006; (4) Pelletier and Tyedmers 2007; (5) Aubin et al 2009; (6) D’Orbcastel et al 2009; (7) Pelletier et al 2009; (8) Pelletier and Tyedmers 2010; (9) Boissy et al 2011; (10) Cao et al 2011; (11) Jerbi et al 2012; (12) Hornborg et al 2012;

(13) Parker and Tyedmers 2012b; (14) Vázquez-Rowe et al 2012; (15) Efole Ewoukem et al 2012; (16) Mungkung et al 2013; (17) Wilfart et al 2013; (18) Avadí et al 2014b; (19) Langlois et al 2014; (20) Avadí et al 2014a (21) Vázquez-Rowe et al 2014; (22) Almeida et al 2014; (23) McGrath et al 2015; (24) Chen et al 2015; (25) Farmery et al 2015; (26) Aubin et al 2015.

a. Biotic Resource Use (BRU); Net Primary Production Use (NPPU); and Primary Production Required (PPR). b. Method author is the author (or group of authors) cited in the paper when describing the PPR method used, with * indicating the method established by Pauly and Christensen (1995). c. Economic (E); Gross Nutritional Content (N); Mass (M); No method established or necessary (-). d. Modeling choices undertaken that directly affected the calculation of PPR were separated into different elements: Sensitivity Analysis (SA); Allocation Choice (AC); Scenario Modeling (SM); Functional Unit choice (FU); and Monte Carlo simulation (MC).

livestock) to be measured in a single unit of mass of carbon appropriated, which was universally presented among studies.

Most studies (25 of 26) employed a mass-based functional unit, for example per unit mass of fish or the amount of feed milled. This has many advantages, as it is broadly understandable and relatively easy to quantify and compare. However, a mass-based functional unit can obscure critical attributes of products (Pelletier, Ardente, Brandão, De Camillis, & Pennington, 2014). In the case of feeds, the nutritional quality of inputs as well as with the final composite feed can vary greatly as can the yield of edible portion from one tonne of fish. One study used instead the protein content of resulting products as the functional unit (Vázquez-Rowe et al., 2014). The common unit of mass makes comparability across studies delivering relatively substitutable products simple; however, this can be obscured by the reasons cited above.

Almost all studies in which meals and oils derived from fish and other aquatic organisms were used as an input to a feed or ultimately an aquaculture system (16 of 18) did not report specific meal and oil yield rates (typically expressed as a percentage fraction of meal or oil mass to round fish mass) employed in their methods. This makes it difficult to discern if average or species-specific yields were used. There can be a large variance in meal and particularly oil yield rates of different species, as well as within species depending on time of year, body condition, size of animals, etc. as well as attributes of the reduction plants themselves. While there are many factors of variability that influence yield rates, species-specific yield rates are an advancement upon average yield rates (Parker & Tyedmers, 2012b). More broadly, relevant assumptions of product yield for agricultural and livestock products were also not included in most studies.

2.2.3 Dissimilarities Of Use

Across all studies reviewed, the methodological decisions were most different in terms of allocation of inputs and impacts amongst outputs of multi-functional production systems, modeling of uncertainty, and terms used. In LCA, environmental burdens and inputs, including PPR, need to be allocated amongst co-products in multi-product systems to attribute environmental burdens to individual co-products. There are various bases upon which allocation can be based including mass, nutritional energetic content, and economic value. The rationale for and process of co-product allocation is highly debated in the literature (Ayer, Tyedmers, Pelletier, Sonesson, & Scholz, 2007; Pelletier et al., 2014; Weidema & Schmidt, 2010; Weinzettel, 2012), and can be very influential to the results of PPR as the relative attribute profile of co-products vary. In the reviewed studies, economic value was most common (13 of 22), while energetic content was second most common (6 of 22). While many studies do not include an explicit rational for their allocation choice, the primary use of the products (energy for feeds or food), or the driver of system (economic revenue or profit) are given as reasons for the allocation method (Boissy et al., 2011; Parker & Tyedmers, 2012a).

In addition to construction of base models of PPR, approximately half of the studies (14 of 26) explore implications of either uncertainty in model construction or parameterization or the implications of alternate scenarios. Sensitivity analyses were slightly more common than scenario modeling (11 of 26 and 9 of 26, respectively). Scenario modeling was conducted in relation to feed conversion ratio of aquaculture species (5 of 9), and for different feed formulations (4 of 9). Allocation was not necessary for all studies but sensitivity to allocation method choice was modeled in 6 of 22 studies. Other sensitivity analyses included modeling functional unit choice (3 of 26), and varying marine-input attributes such as trophic level and fishmeal and fish oil yield rates (2 of 26). One study employed Monte Carlo simulation to understand the impacts of describable uncertainty of input parameters on model outcomes.

2.2.4 Challenges To Current PPR Method

As noted above (Section 2.2.3), there is a diversity of practice that can greatly influence results and comparability of a study originating from allocation decisions and other assumptions not necessarily well detailed. Transparency of practice is a current challenge that must be addressed moving forward, to ensure adequate conclusions can be drawn from results. However, the similarities and dissimilarities outlined above (Section 2.2.2 and 2.2.3) demonstrate two alternative methods currently used: i) a ‘standard method’, in which a 10% transfer efficiency is assumed to apply across all trophic levels and typical or ‘average’ meal and oil yield rates (of between 21-24% for meal and 5-9% for oil) are applied to all marine derived products, and ii) a ‘yield-specific method’, in which a 10% transfer efficiency is again assumed to apply universally but species-specific yield rates of meal and oil are applied.

A significant limitation to both these approaches is the use of an average transfer efficiency value of 10%, irrespective of source ecosystem or species interactions, as this creates constraints to conclusions and comparisons of results. This 10% value was originally assumed (May, 1976; Slobodkin, 1962), but was not based concretely in evidence (Slobodkin, 2001). However, other studies broadly supported this as a global marine average in reviews of Ecopath with Ecosim models (Libralato et al., 2008; Pauly & Christensen, 1995). Importantly, however many studies, mainly based on Ecopath with Ecosim models, have found that transfer efficiency is highly variable among aquatic ecosystems and ecosystem types (Libralato et al., 2008), ranging between 3.51 and 38.1%. The range of values is large (Figure 4), but there is a tighter distribution of

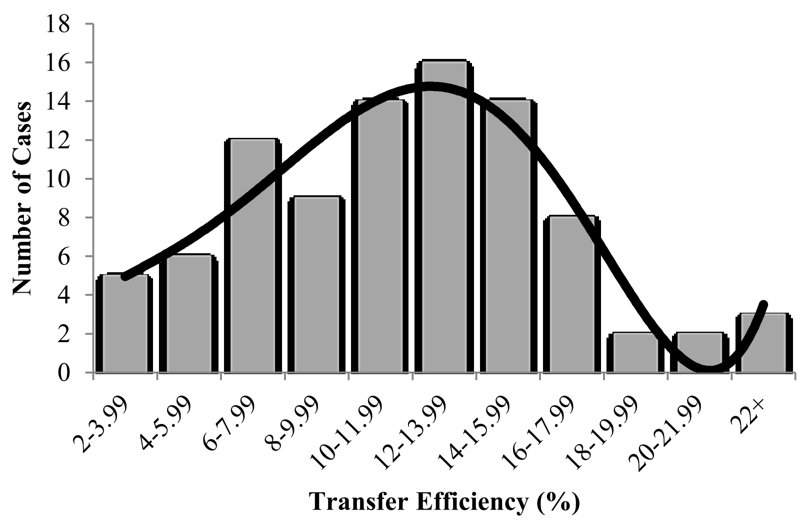


Figure 4. Distribution of transfer efficiency values of 91 ecosystems surveyed in Libralato et al 2008.

observations in the 6-16% range (Figure 4; Libralato et al 2008). Given the centrality of the transfer efficiency value in Equation 1, these differences in input values can have a substantial impact on outcomes (Parker & Tyedmers, 2012b).

Furthermore, it is unclear in many studies what values are used to convert quantities of fishmeal and oil inputs into quantities of round or live-weight fish, or vice versa. Using species-specific yield values will, of course, produce a more accurate estimation of PPR, rather than say average yield values of 22.5% for fishmeal and 5% for fish oil that are often employed for other calculations of resource dependency (A. Jackson, 2009; Tacon & Metian, 2008; Ytrestøyl et al., 2011).

When parameterizing models of PPR based on Equation 1, it is typical that analysts employ discrete values. However, all requisite inputs are subject to natural variability (i.e. the values are not static in nature), and uncertainty (i.e. our knowledge of the parameters is incomplete; Parker and Tyedmers 2012a), such as primary production and transfer efficiency which vary seasonally and annually in reality (Chavez, Messié, & Pennington, 2011; Libralato et al., 2008). The use of discrete values produces results that are discrete and static, which often results in values that appear to give a clear indication of merit or demerit of the production system under study, and thus can lead to potentially false claims in favour of one over the other (see Parker & Tyedmers 2012). Using discrete values in this case can lead to a misinterpretation of meaningful difference in results. In contrast, incorporating and presenting variability that better reflect fundamental and describable uncertainty will provide more realistic if nuanced results.

Lastly, current efforts to estimate PPR do not provide a measure of scale relative to total ecosystem productivity. Quantifying PPR from fisheries as a percentage of total source ecosystem primary production available provides an indication of the scarcity of this resource and degree of overfishing (Coll, Libralato, Tudela, Palomera, & Pranovi, 2008). Previous research has pointed to the increasing pressure on biodiversity associated with humans appropriating primary production (Foley et al., 2007; Krausmann et al., 2013), but little work has been done on the thresholds for sustainability of appropriation levels (Bishop, Amaratunga, & Rodriguez, 2009; Langlois et al., 2014), but see Coll et al.

(2008). Further challenges to the broader utility of the PPR indicator will be discussed below (Section 2.5.3).

2.3 Methods

2.3.1 Advancement Of Method

To improve the quantification and contextualization of PPR calculation in seafood LCAs, we made three modifications to the contemporary practice as most recently applied by Wilfart et al (2013) and Almeida et al (2014), hereafter referred to as the standard method. Specifically: 1) where and when possible, source large marine ecosystem (LME) specific transfer efficiency values are applied in lieu of the heretofore average value of 10%; 2) species-specific fishmeal and oil yield values were employed when available; and 3) to provide an indication of the scale of PPR relative to total source LME productivity, we will express the feed's PPR as a fraction of annual total ecosystem productivity. Taken together, this represents what hereafter is referred to as the refined method. To illustrate the functional utility and novel insight available through the application of this method, we apply it to the quantification of PPR associated with all marine-sourced inputs to Norwegian salmon feeds milled in 2012. A variant of the standard method, hereafter referred to as yield-specific method, that employed species-specific fishmeal and oil yield values was also defined because it was unclear if some previous studies used species-specific or average meal and oil yield values (Section 2.2.1). We then compare and contrast results of the refined method with those using the heretofore described standard and yield-specific method. In addition to the main analyses of the three methods, two sensitivity analyses and an uncertainty analysis were performed (Sections 2.3.4 and 2.3.5).

2.3.2 Characterization Of Inputs To Norwegian Feeds

Data on inputs into Norwegian salmon feeds for 2012 were solicited from the three largest producers of salmon feed in Norway. The marine focus here meant that the species and ecosystem of all sources of marine meal and oil were important, as well as the species-specific fishmeal and oil yield rates.

2.3.3 Standard And Refined Model Parameterization

Global average transfer efficiency of 10% was used for standard method and yield-specific method. Standard method used average fishmeal and oil yield rates of 22.5% and 9.3%, respectively (Ytrestøyl et al., 2011). Yield-specific method and the refined method used species-specific data on fishmeal and fish oil yields from feed producers that were incorporated. For all three methods, fish trophic levels were obtained from FishBase (Froese & Pauly, 2012), and the same species harvested from different ecosystems was assumed to have the same trophic level value. To compare the different meal and oil sources along a commonly used unit, a mass-based functional unit of one tonne of fishmeal or fish oil was used throughout all three methods.

The refined method's parameters differed from both standard and yield-specific methods for: 1) LME-specific transfer efficiency, and 2) indication of scale of PPR as a fraction of the source ecosystem's annual total primary production. Ecosystem specific transfer efficiency values rather than the global average of 10% were used (Pauly & Christensen, 1995), and these ecosystem-specific estimates were obtained from a summary of Ecopath with Ecosim models (Libralato et al., 2008). These models demonstrate substantial variance from the global average of 10% (3.51% to 16.5% for ecosystems modeled here), but correspond to the specificity of the ecosystems they are harvested from instead of the global average (Libralato et al., 2008). Data on LME primary production were accessed from the Sea Around Us Project (2011). The primary production estimates are derived from satellite data (from the ten year period of 1998 to 2007) that calculates primary production from chlorophyll pigment concentration (Platt & Sathyendranath, 1988).

Energetic allocation was employed for this analysis. Fish oil was applied an equal energetic content of 39.3 MJ/kg, whereas meal energy density varied by species (18.0-23.8 MJ/kg). Where species-specific information was not available, three categories were employed for the meal energy density: herring-type fish with a meal crude protein content of 68-72% (including herring, capelin, mackerel, sprat, and sandeel; 20.0 MJ/kg); sardine type fish with a meal crude protein content of 65% (including anchovy, sardine, and horse mackerel; 19.0 MJ/kg); and whitefish meal derived from by-products with a meal crude protein content of 65% (18.4 MJ/kg; FAO, 1986; Sauvant, Perez, & Tran, 2004).

2.3.4 Sensitivity Analysis

A sensitivity analysis examines the implications of assumptions and methodological decisions made throughout the analysis on final results. Here we evaluate, first, the choice of functional unit (PPR per: kilogram of fishmeal; fish oil; kilogram of round fish, or 100 gigajoules), as it has previously been shown to have a large effect on the distribution of impacts and how they are represented (Parker & Tyedmers, 2012b). Secondly, we evaluated the influence of three input parameters (transfer efficiency, trophic level, meal and oil yield rates) on the PPR by altering these parameter values by +/- 10% for Atlantic herring from the North Sea. These input parameters were evaluated individually and in various combinations. The results were then compared as a percentage in relation to the results of the refined method. While 10% does not represent the actual variance of these input parameters, this analysis demonstrates their relative contribution and influence towards the final results, and thus which parameters are most influential on the PPR results.

2.3.5 Uncertainty Analysis

The uncertainty of the PPR calculation was modeled through a Monte Carlo analysis to give a distribution of results. Monte Carlo analysis utilizes the range within input parameters to model distributions over 10,000 iterations. Each iteration randomly selects a value from within each parameter based on the mean, standard deviation, and truncations inputted in the data (see Coll et al 2008; Parker and Tyedmers 2012b). Monte Carlo analysis was run through Pallisade Corporation's @Risk addition to Microsoft Excel to yield histogram results of the 10,000 iterations of PPR.

To model uncertainty with Monte Carlo analysis, a justifiable range, standard deviation, and mean must be available. In this chapter, we aimed in the refined method to use specific values rather than average values, which thus had less data available for them. Therefore, species-specific fishmeal and oil yield rates and ecosystem specific transfer efficiencies were modeled as discrete parameters in the uncertainty analysis, while trophic level incorporated uncertainty into modeling because of the uncertainty and natural variability in estimates. The standard deviation was assumed to be the standard error of the sample as provided by Fishbase. The trophic level range was limited by a lower bound of 2, because trophic levels lower than 2 indicate autotrophy, and an upper bound of 5, representing a rarely reached trophic level of apex predators.

2.4 Results

2.4.1 2012 Norwegian Salmon Feeds

Norway is the largest producer of cultured salmon globally (Ytrestøyl et al., 2011), and data from the three largest Norwegian feed producers who together accounted for approximately 95% of all salmon feed milled in Norway in 2012 were assembled. Approximately 66% of the mass of all feeds were derived from crops, with the balance originating from the marine environment or were micro ingredients (Table 2). Interestingly, 23% of the fish oil and 32% of the fishmeal were derived from by-products of various direct human consumption fisheries that were treated as co-products of these systems in this analysis (Table 2 and 3). All marine inputs and their species and ecosystem properties are detailed in Table 3. The micro ingredients are mainly composed of phosphate substances such as mono-calcium-phosphate, vitamins, minerals, pigments and amino acids (Table 2). The total volume of marine inputs in processed form (i.e. fishmeal and fish oil) by these three companies to salmon feed in 2012 was 488,702 tonnes.

Table 2. Coarse sources of inputs to Norwegian salmon feeds milled for 2012

Ingredients	Mass [tonnes]	Notes
Marine oils	182 362 (12%)	23% derived from by-products
Marine meals	306 340 (19%)	32% derived from by-products
Crop meals	617 032 (39%)	63% of this soy protein concentrate
Crop starch/carbohydrates	122 158 (8%)	Mainly wheat starch
Crop oil	298 991 (19%)	100% rape seed oil
Micro ingredients	43 807 (3%)	Mainly phosphate, vitamins, minerals and amino acids.
TOTAL	1 577 233 (100%)	Total feed consumption in Norwegian salmon aquaculture industry in 2012 was 1 663 894 tonnes.

Table 3. Species and ecosystem properties of marine inputs to 2012 Norwegian salmon feed

Species	Ecosystem and T[a]	L[b]	Meal Yield[c]	Oil Yield[c]	Meal[e]	Oil[e]	Meal PPR[f]	Oil PPR[f]
Reduction Fisheries								
Antarctic krill (*Euphasia superba*)	SO (14)	2.2	0.16	0.0008	2,946	-	7.29E+03	-
Atlantic herring (*Clupea harengus*)	IS (14)	3.2	0.204	0.115	1,987	50	2.06E+04	3.66E+04
Atlantic herring (*C. harengus*)	NWS (3.51)	3.2	0.204	0.115	2,233	959	4.31E+05	7.67E+05
Atlantic herring (*C. harengus*)	NS (11.6)	3.2	0.204	0.115	2,748	4,104	3.11E+04	5.53E+04
Atlantic mackerel (*Scomber. scombrus*)	NS (11.6)	3.7	0.194	0.186	516	6,396	6.67E+04	1.31E+05
Blue whiting (*Micromesistius poutassou*)	NS (11.6)	4	0.197	0.019	5,786	501	2.98E+05	6.52E+05
Boarfish (*Capros aper*)	NS (11.6)	3.1	0.216	0.034	3,448	540	1.68E+04	3.68E+04
Capelin (*Mallotus villosus*)	BS (11.6)	3.2	0.165	0.077	24,844	15,395	5.57E+05	1.09E+06
Capelin (*M. villosus*)	IS (14)	3.2	0.165	0.077	29,082	13,519	2.66E+04	5.22E+04
Chilean jack mackerel (*Trachurus murphyi*)	HC (6.6)	3.5	0.194	0.186	507	-	1.77E+05	-
European sprat (*Sprattus sprattus*)	NS (11.6)	3	0.188	0.079	22,518	15,909	2.41E+04	4.73E+04
Gulf menhaden (*Brevoortia patronus*)	GM (9.7)	2.2	0.24	0.13	1,463	2,806	3.60E+03	7.41E+03
Norway pout (*Trisopterus esmarkii*)	NS (11.6)	3.2	0.204	0.115	94	599	1.28E+04	5.81E+04
Peruvian anchovy (*Engraulis ringens*)	HC (6.6)	2.7	0.23	0.05	101,358	78,209	3.35E+04	7.16E+04
Sandeels (*Ammodytes tobianus*)	NS (11.6)	2.7	0.197	0.0424	8,018	2,280	1.54E+04	3.03E+04
South American pilchard (*Sardinops sagax*)	HC (6.6)	3.1	0.23	0.18	615	-	5.56E+04	-
By-products								
Atlantic cod (*Gadhus morhua*)	NS (11.6)	4.4	0.17	0.017	2,906	1,191	4.39E+05	9.38E+05
Atlantic herring (*C. harengus*)	IS (14)	3.2	0.2	0.04	15,951	13,535	9.71E+03	1.73E+04
Atlantic herring (*C. harengus*)	NWS (3.51)	3.2	0.2	0.04	31,770	10,387	2.04E+05	3.62E+05
Atlantic herring (*C. harengus*)	NS (11.6)	3.2	0.2	0.04	17,307	9,014	1.47E+04	2.61E+04
Atlantic mackerel (*S. scombrus*)	NS (11.6)	3.7	0.194	0.186	2,418	442	3.22E+04	6.33E+04
Capelin (*M. villosus*)	BS (3.51)	3.2	0.165	0.077	1,511	193	2.58E+05	5.07E+05
Capelin (*M. villosus*)	IS (14)	3.2	0.165	0.077	9,404	689	1.23E+04	2.42E+04
Fish hydrolysate[g]	NWS (3.51)	3.2	0.21	0.1	28	-	2.04E+05	-
Fish protein concentrate[g]	NWS (3.51)	3.2	0.21	0.1	14,936	3,850	2.04E+05	3.62E+05

a. Southern Ocean: SO; North Sea: NS; Icelandic Shelf: IS; Norwegian Sea: NW; Barents Sea: BS; Humboldt Current: HC; Gulf of Mexico: GM. Transfer efficiency (T; %) of source ecosystem in brackets. b. Method as establish by Fishbase (1+ weighted average trophic level of prey as establish from diet and model studies). c. Tonnes of meal or oil produced from 1 tonne of round fish. d. Transfer Efficiency (T) from Libralato et al (2008) sourced from ecosystem models. e. Meal and Oil are both in tonnes. f. Primary Production Required (PPR) in kilograms of carbon per tonne of meal or oil. g. Both fish hydrolysate and fish protein concentrate are modeled as by-products of Atlantic herring from the Norwegian Sea.

2.4.2 Implications Of Standard And Refined Methods

The standard and yield-specific methods can have results similar to the refined method, but often differ from the refined method's results based on the values used for transfer efficiency and meal and oil yields (Figure 5 and 6). The magnitude of this effect is most pronounced for ecosystems with transfer efficiencies further from the traditionally assumed average of 10%, like the Icelandic Shelf (Figure 5) or the Barents Sea (Figure 6). For ecosystems with very low or very high transfer efficiencies, the 10% average value under or over-reports the PPR, respectively, because the transfer efficiency value has an exponential effect. For example, the different transfer efficiencies for Atlantic herring across ecosystems can cause differences in PPR by a factor of 20 (Figure 5). Atlantic herring sourced from Icelandic Shelf is the most efficient, because of its high transfer efficiency (Table 3) while the Norwegian Sea is the least efficient having the lowest transfer efficiency out of these three ecosystems (Table 3, and Figure 5). This difference is obscured by the standard and yield-specific methods, which do not incorporate this difference, and perform almost identically to each other (Figure 5).

A similar effect can be observed for the use of average yield rates (22.5% and 9.3% for meal and oil, respectively), although with less pronounced effects. The species that perform far outside the range of average values for fishmeal and oil yield rates, Antarctic krill and South American pilchard at either end of the spectrum, are most different from the use of average values. In summary, the major differences, and thus results, between the refined method and the standard and yield-specific methods is demonstrated as species and ecosystem values move farther away from the averages.

Using the refined method, there is a great variance of PPR across the sources of fishmeal and oil included in the feed, but there are general patterns that influence these results. Consistently, high trophic level species and species sourced from ecosystems with low transfer efficiency have the largest PPR per tonne of meal or oil produced (Table 3). Additionally, species with low fishmeal and oil yield rates had higher PPRs (e.g. blue whiting and capelin [Barents Sea]), but this pattern was difficult to separate from the relationship of trophic level and transfer efficiency. Overall, use of the refined method resulted in PPR estimates over three times as large as the standard method (Figure 7). This substantial variation can be attributed to the influence of the high PPR inputs (based on reasons noted above). The yield-specific method performed much closer to the standard method, giving another indication of the influence of ecosystem-specific transfer efficiency.

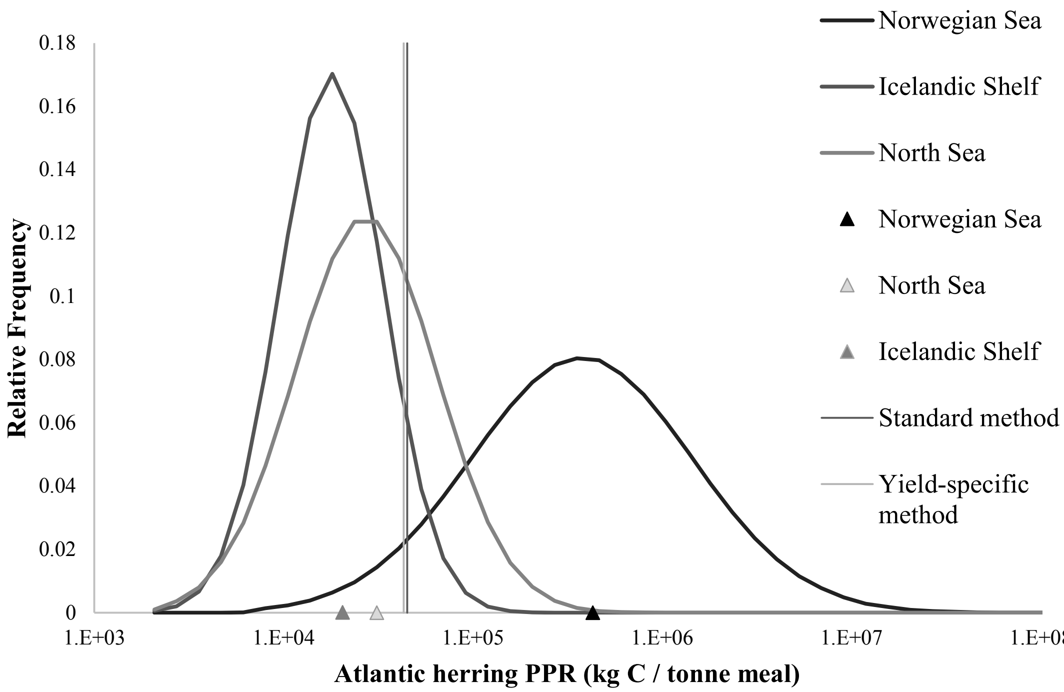


Figure 5. PPR (kg C/ tonne meal) on a logarithmic scale of Atlantic herring sourced from three different ecosystems. The curves and triangles represent the Monte Carlo distribution of results and refined method results, respectively, for Atlantic herring modeled from three different source ecosystems: Norwegian Sea (black), Icelandic Shelf (dark grey), and North Sea (light grey). The two vertical lines represent results for standard method (dark grey) and yield-specific method (light grey).

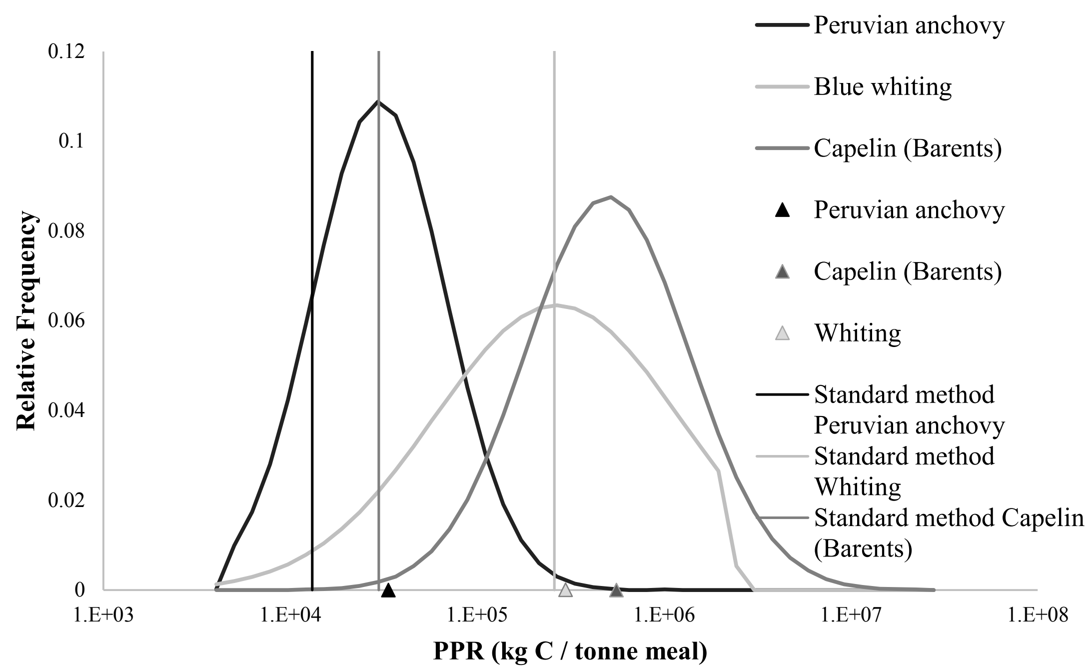


Figure 6. PPR (kg C/ tonne meal) on a logarithmic scale of blue whiting, capelin (Barents Sea), and Peruvian anchovy. The curves, triangles, and vertical lines represent the Monte Carlo distribution of results, refined method results, and standard method results, respectively, for Peruvian anchovy (black), blue whiting (light grey), and capelin (dark grey).

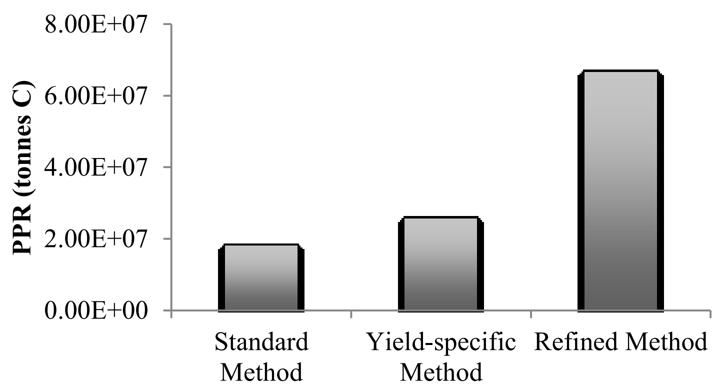


Figure 7. PPR of marine inputs to Norwegian salmon feed by method

2.4.3 PPR To Sustain Marine Inputs To 2012 Norwegian Salmon Feed

Applying the refined PPR method, the total PPR for the marine portion of the 2012 Norwegian salmon feed is 6.66E+07 tonnes of carbon that accounts for 4.89E+05 tonnes of meals and oils (Table 4). The Humboldt Current is the source of a large portion of marine inputs into the feeds modeled (over 37%), but these removals represent a relatively small percentage of total annual primary production of the ecosystem. In contrast, meals and oils from the Norwegian Sea, with a relatively low transfer efficiency and low annual primary production (Table 3 and 4, respectively), has a much greater percentage of primary production appropriated, while only providing 13% of marine inputs by mass (Table 4). The effect of sources of fishmeal and oil from low transfer efficiency ecosystems having higher PPR (e.g. Barents Sea and Norwegian Sea), in turn, increases the total ecosystem PPR with the Norwegian Sea reporting high levels of ecosystem PPR (Table 4).

Table 4. PPR by ecosystem compared to ecosystem primary production

Source Ecosystem	Contribution to marine portion of feed by mass (meal and oil combined; %)[a]	PPR to support meal and oil provision (tonnes C)	Total salmon feed marine PPR from each ecosystem (%)	Annual total primary production of source ecosystems (tonnes C / year)	% Appropriated
Humboldt Current	37	9.12E+06	13.69	8.38E+08	1.09
Antarctic Sea	0.61	2.15E+04	0.03	3.47E+08	0.00618
Icelandic Shelf	17	2.04E+06	3.07	1.05E+08	1.95
Norwegian Sea	13	1.64E+07	24.60	1.99E+08	8.24
North Sea	21	7.75E+06	11.64	2.81E+08	2.76
Celtic-Biscay	0.82	7.79E+04	0.12	2.67E+08	.0291
Barents Sea	8.7	3.12E+07	46.82	3.06E+08	10.2
Gulf of Mexico	0.88	2.60E+04	0.04	3.18E+08	0.00818
Total	100	6.66E+07	100	2.66E+09	X

a. Total may not add to 100% because of rounding errors.

2.4.4 Sensitivity Analyses And Effect Of Parameters

The sensitivity analysis of PPR values to choice of functional unit was reported to the average PPR value for each functional unit, and showed that species performance varies strongly based on the functional unit chosen (Figure 8). Varying performance is caused by the meal and oil yields, energetic content, as well as the PPR per kg round fish. As an example, Chilean jack mackerel has a very high oil yield of 18.6% while blue whiting has a very low oil yield of 1.9% that causes the large disparity in the reported PPR per kg of fish oil (Table 3). Species like Atlantic herring (North Sea) can display higher relative PPR values for round fish than fish oil in this analysis (Figure 8) because it is based on the comparison to the average PPR of all sources for each functional unit. This species has results high above the average when round fish is the functional unit, but not for fish oil due to its high oil yield. Species with high trophic levels can thus perform relatively efficiently depending on the functional unit choice because of high meal and oil yields having a compensatory effect. Thus, the choice of functional unit can influence the interpretation of results, and display different patterns in the relative performance of different meals and oils.

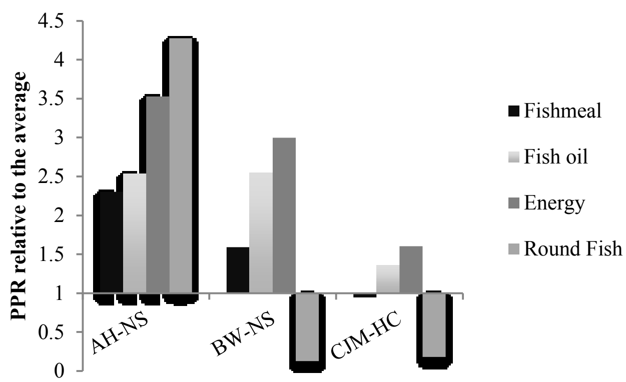


Figure 8. PPR in comparison to the average (set at 1) PPR of all sources for various functional units of Atlantic herring from Norwegian Sea (AH-NS), blue whiting from North Sea (BW-NS), Chilean jack mackerel from Humboldt Current (CJM-HC).

In the formal sensitivity analysis of different parameter's relative influence on results, trophic level had the greatest influence of any single parameter on the PPR (Table 5; 50% to 199% of refined method result), while transfer efficiency was slightly less influential (Table 5; 81% to 126%). The sensitivity analysis showed a large variance in results, especially when multiple parameters were altered leading to compounding their effect on results (Table 6; 39% to 294%). Thus, this analysis supports the other findings (Table 3) that higher trophic level and lower source ecosystem transfer efficiency are predictive factors that species will have higher impacts on the PPR indicator (Table 5 and 6).

Table 5. Sensitivity analysis of influence of PPR parameters for Atlantic herring meal (North Sea) individually and in various combinations presented relative to refined method result (100%).
T: transfer efficiency; Y: meal yield rate; L: trophic level; +: increase of 10% to parameter value; -: decrease of 10% to parameter value.

	No Factor	T+	T-	Y+	Y-
No Factor	100%	81%	126%	93%	113%
L+	199%	157%	260%	185%	226%
L-	50%	42%	61%	47%	57%
Y+	X	75%	117%	X	X
Y-	X	92%	143%	X	X

Table 6. Sensitivity Analysis of influence of PPR parameters for Atlantic herring meal (North Sea) individually and in various combinations presented relative to refined method result (100%).
T: transfer efficiency; Y: meal yield rate; L: trophic level; +: increase of 10% to parameter value; -: decrease of 10% to parameter value. T+/- and L+/- includes both these factors to compare the effect of all three parameters together.

	Y+	Y-
T+ L+	145%	178%
T+ L-	39%	48%
T- L+	241%	294%
T- L+	57%	69%

2.4.5 Uncertainty Analysis

The results of the Monte Carlo analysis (Figures 5 and 6) demonstrate a wide dispersion of potential results based on uncertainty and natural variability of underlying modeled input parameters. Furthermore, this dispersion causes substantial overlap in the ranges of possible PPR values, and includes the results from all three methods (Figures 5 and 6). The three methods analyzed produce results from discrete parameters often with clear differences; however, when uncertainty and variability are accounted for, the difference is less clear. Accounting for these factors thus demonstrates that there is not always a clearly more or less impactful option.

2.5. Discussion

2.5.1 Implications For LCA Of Feed Production

There is a great variance in the PPR of different fishmeals and oils. This largely confirms previous research in this domain (Parker & Tyedmers, 2012b; Ytrestøyl et al., 2011), and reinforces the need for greater consideration of the varied impacts of these products. As a whole, the marine portion of the Norwegian salmon feeds milled in 20123 appropriated a non-trivial amount of primary production from large marine ecosystems across the globe. Given the scarcity of this resource (Chassot et al., 2010; Watson, Zeller, & Pauly, 2014), and the higher scarcity in less productive and less efficient ecosystems (Coll et al., 2008), the refined method developed and tested including the indication of scale of appropriation is an important step taken in this analysis. The results demonstrate a fairly large proportion of appropriation from these ecosystems solely for the feed provision to sustain Norwegian farmed Atlantic salmon production in 2012. Given the context of other fisheries co-occurring in these source ecosystems, the high levels of appropriation in some of these ecosystems represents a challenge to the sustainability of these fisheries.

The refined method put forward in this chapter adds to the diversity in current LCA practice, but with the intention of refining the PPR measure to make it a more realistic and representative quantification of the biophysical reality. This method's use of ecosystem-specific transfer efficiency had a large effect on the results for this study, and was confirmed with a sensitivity analysis. The large impact this ecosystem-specific value can have on the final result is thus important when considering previous analyses that may be under or over estimating the PPR. For example, inputs like capelin and Atlantic herring sourced from the Norwegian Sea and Barents Sea would be a much lower impact when considered in other studies using average transfer efficiency than the refined method results in this study. The challenge of using more specific values for transfer efficiency, and meal and oil yields is that these data are based on fewer observations than the averages. The challenge remains that these data should represent a specific parameter more accurately; however, this cannot be confirmed without more knowledge of the input parameters.

Another challenge to interpreting current PPR results is the diversity of practice found in the LCA literature review. This diversity impacts the comparability of studies when allocation and functional unit decisions vary. Decisions on the allocation of impacts among co-products can contribute to alternative potential results of the study, and thus it is important to align these decisions with the goal of the study. While the variance in these decisions confounds comparability, it is not possible to be prescriptive for functional unit or allocation choice as they are study specific.

This chapter modeled the uncertainty of marine PPR using Monte Carlo analysis. Through modeling uncertainty in this quantification, we can explore the range of options and certainty with which we can make claims of the impacts of certain inputs. Reporting results solely in discrete values can give the impression of major differences between species utilized, while reporting within a range shows the likely ambiguity between some species and the clear differences between others (Parker & Tyedmers, 2012b). The use of more specific values enabled more specific estimates, rather than relying on global averages which increases uncertainty.

2.5.2 Limitations Of This Study

A key limitation of this study is that we did not focus on impacts other than biotic resource dependency of marine inputs into aquafeeds as quantified by PPR. While this was the focus of this study, it meant that other impacts associated with fisheries, agriculture, transport, and processing were not accounted for. Additionally, this study represents a snapshot in time of the marine inputs into the average 2012 Norwegian salmon feed that is not necessarily representative of other years or global aquafeed production.

This study is subject to the uncertainty and natural variability of the input parameters. We attempted to account for this uncertainty through presenting a range of values and using Monte Carlo analysis, but we are still subject to a lack of perfect data to be used for many parameters (yield rates, primary production, transfer efficiency, and trophic level). The specific values of meal and oil yields, trophic level, and transfer efficiency are subject to natural variability and uncertainty, but sufficient data (e.g. ranges, standard deviation, and mean) are not currently available to properly parameterize these values for all ecosystems and species within Monte Carlo analysis.

2.5.3 Limitations Of PPR

Further challenges that were not addressed in this paper relate to the broader utility of PPR as an indicator of ecological impact on ecosystems more generally. While PPR is relatively simple to quantify in comparison to other types of ecological impacts and is relatable to a functional unit, this simplicity means that the measure does not indicate: i) if the appropriation of primary production by society is ultimately sustainable; or ii) what impacts are on targeted and associated populations. For example, PPR as a measure of appropriation in terrestrial ecosystems does not encompass effects on biodiversity through land-use changes and practices, nor can it distinguish between source ecotypes of very different value (e.g. old-growth forests and agricultural land). Even though the PPR measure is frequently used to measure impacts in marine ecosystems, it does not consider impacts like discarding or the endangered status of by-catch species (Hornborg, Svensson, et al., 2013; Vázquez-Rowe et al., 2012). The redirection of energy in marine ecosystems caused by discarding, and the reduction in population and genetic diversity associated with by-catch of endangered species are clearly two important human impacts on marine ecosystems. Measuring either of these focuses more on the distribution of impacts within ecosystems, whereas PPR gives a broad indication of the whole ecosystem impact and pressure from human appropriation of this limited resource when contextualized within total ecosystem productivity. These challenges to the indicator more broadly were outside the scope of this paper.

This paper has solely treated the PPR metric as an indicator of ecosystem pressure of various marine inputs. PPR is only one measure of this, and a coarse measure of the impact society is having on these marine ecosystems through reduced primary production being available for other species in the ecosystem (Papatryphon et al., 2004). This measure, thus does not speak to the sustainability of fish stocks exploitation, or the impact on removing species from a certain trophic level from these ecosystems. Recent research has focused on the impacts of fisheries for small pelagic species on sea birds and marine mammals, and the potential thresholds of over-harvesting small pelagic fish on higher trophic level species (Cury et al., 2011; Smith et al., 2011). While our study incorporates the annual ecosystem primary production as a reference value, it does not make any claim on the sustainability of high levels of primary production appropriation. PPR thus must be recognized as a relative and descriptive measure of performance, rather than an absolute measure of sustainability.

2.6 Conclusion

In this chapter we have:

- Surveyed current practice of PPR as an indicator of impact on marine biotic resources
- Defined and applied a refined method for more accurate, region and species-specific LCA account of marine inputs
- Contextualized PPR to the source ecosystem to yield an indication of scarcity
- Modeled the uncertainty of this PPR estimation through presenting a distribution of results

Thus, future practice should use: i) species-specific information for fishmeal and oil yield rates; ii) ecosystem-specific values for transfer efficiency; and iii) contextualize the PPR of production systems to their source ecosystem to give an indication of the scale of appropriation. Thus, the PPR measure can serve as a relative measure of performance between these species and ecosystems, but is only operationalized to measure potential ecosystem pressure.

Chapter 3. Global Reduction Fisheries And Their Products In The Context Of Sustainable Limits

3.1 Introduction

Globally, one-sixth of capture fisheries are destined for the production of fishmeal and fish oil (i.e. reduction fisheries) which are currently overwhelming utilized by fed aquaculture (FAO, 2014a). Although this number has declined substantially from being almost one-third of total capture fisheries landings (Naylor et al., 2000), this still represents a large subsection of global catch. In general, the fish targeted for reduction are small pelagics and serve as forage fish in their ecosystems for higher trophic level species (Tacon & Metian, 2009). Many different sources are used globally for the production of fishmeal and fish oil (FMFO), but little concern has been given to the divergent environmental impacts of FMFO products based on the source: species, ecosystem, and fishing gear. The major impacts can be characterized broadly as biotic, impacting the populations and ecosystems from which they are harvested, and abiotic, negative changes to the immediate ecosystem’s environment or the broader atmospheric system.

The current major concerns regarding reduction fisheries relate broadly to biotic impacts caused by their removal from ecosystems. The biotic impacts of reduction fisheries include decreased population of the target species, by-catch, and disruption of energetic flows within their source ecosystem leading to reduced food energy available for trophic levels above the target species including marine mammals and seabirds (Alder et al., 2008; Cury et al., 2011; Naylor et al., 2009). All of these individual impacts can result in cumulative effects on ecosystems and their constituent parts. These fisheries are traditionally managed using individual stock assessment techniques and quotas, and occasionally with the goal to minimize target by-catch through the use of more species and size specific gear (Caddy & Cochrane, 2001; Pauly et al., 2002). The main motivation of managing individual stocks is to maximize human benefit that could be sustained indefinitely, and maximum sustainable yield and maximum economic yield reflect this motivation (Iversen, 1996; Kelleher et al., 2009; Royce, 1996). These techniques focus solely on the affected population(s) and often do not consider other external factors (Pauly et al., 2002).

The rise of ecosystem based management represents a shift in focus to consider other species affected, and the effect of harvesting one species on the rest of the ecosystem (Caddy & Cochrane, 2001). Some reduction fisheries are increasingly being considered under this management framework, notably amongst those undertakin in the North Sea (Dickey-Collas et al., 2013). Considering reduction fisheries in ecosystem based management is important as mid to low trophic level species targeted by reduction fisheries play an important role in the transfer of energy to higher trophic levels (Cury et al., 2011). Various indicators have been proposed to account for ecosystem impacts under this framework, including the trend of fishing down the food web (e.g. mean trophic level; Pauly et al. 1998), and the proportion of high trophic level species to low trophic level species harvested (e.g. fishing-in-balance index; Pauly and Watson 2005). These are important for measuring biotic impacts on marine ecosystems; however, many environmental or abiotic impacts of FMFO provisioning are still not considered in management decisions (Ziegler & Hornborg, 2014).

Environmental impacts of FMFO production are diverse and include greenhouse gas (GHGs) emissions from fuel use during the fishing stage and other activities along the supply chain including processing energy, and pollutants released from the fishing vessel including anti-foulant paints and refrigerants. The fuel use for fishing and the processing energy have been previously found to contribute substantially to abiotic impacts of FMFO production in some settings (Avadí & Fréon, 2013; Pelletier & Tyedmers, 2007; Pelletier et al., 2009; Tyedmers, 2000). The use of fossil-fuel energy, which is partially tied to gear use (Driscoll & Tyedmers, 2010; Parker, Vázquez-Rowe, & Tyedmers, 2014; Tyedmers et al., 2005) in fisheries has also been shown to be a proxy for other environmental impacts including acidification and eutrophication (Hospido & Tyedmers, 2005). These impacts have received increased attention recently, but are still not brought into management measures even though many well managed fisheries also result in relatively low abiotic impacts (Ziegler & Hornborg, 2014).

These abiotic and biotic impacts must be considered in the context of humanity's current impact on the planet. Previous research has focused on human carrying capacity and viewing human population, affluence, and technology in this light (Borgstrom, 1974; Catton, 1987; Cohen, 1995; Rees, 1992). Many analyses have focused on potential human population expansion and its challenges because of the limits of food production (Borgstrom, 1974; Cohen, 1995), while others have broadened this to include human impacts on natural systems (Rees, 1992; Wackernagel & Rees, 1996). Here, we measure impacts on the environment and ecosystems, while considering the sustainable limits of these impacts. Sustainable limits are boundaries that humanity should not exceed to ensure the proper functioning of ecosystems, and a stable abiotic environment to exist within (Rockström et al., 2009).

Both cumulative GHG emissions and biomass appropriation represent current challenges to environmental sustainability (Foley et al., 2007; Rockström et al., 2009; Running, 2012). Both of these areas of concern have theoretical safe boundaries that should not be exceeded if humanity is to continue to exist in a 'safe operating space' (Rockström et al., 2009, p. 2; Running, 2012), although it has not been articulated fully for marine biomass appropriation. Therefore, the aim of this chapter is to articulate the impacts of global FMFO production from dedicated reduction fisheries in relation to sustainable limits of source ecosystem productivity and climate change beyond a 2°C warming limit. The global reduction fisheries sector represents substantial biomass appropriated for human use and contribution to global GHG emissions, and therefore should be quantified with respect to these sustainability limits. A secondary aim is to compare the abiotic and biotic impacts of the major FMFO products that are commonly used to demonstrate variance in environmental performance because of target species, source ecosystem, and gear used.

The abiotic and biotic impacts of FMFO production can be accounted for through two measures. A marine footprint serves as a proxy of broad-scale ecological impacts through measuring the appropriation of primary production to sustain the catch in a spatial measure (km^2; Parker & Tyedmers, 2012b; Pauly & Christensen, 1995; Watson et al., 2014). This serves as a coarse measure of the scale of dependence on the ecosystem as a human appropriation of limited photosyntheticly captured energy available in these source ecosystems. Evaluating these products within the context of their source ecosystem demonstrates the scarcity of primary production, which varies between ecosystems. A carbon footprint quantifies the cumulative life cycle emissions of greenhouse gases. For this analysis, the major sources of GHG emissions from FMFO production originate from the fuel use of the fishery and the processing energy (Pelletier & Tyedmers, 2007). Anthropogenic greenhouse gas emissions are occurring within an environment that has a limited 'safe operating space' that has been hypothesized at 2°C warming above pre-industrial levels, and thus the relative contribution towards these limits should be articulated (Rockström et al., 2009). The FMFO products can thus be evaluated at a sector level and at an individual product level. The FMFO products may then be evaluated in terms of both their CO_2-e emissions and km^2 of sea area required per tonne of product produced. Both the carbon footprint and marine footprint attempt to account for impacts that are not conventionally accounted for in understanding the performance of reduction fisheries. However, these measures cannot, and are not intended to, replace analyses of individual stocks or ecosystem health.

3.2 Methods

3.2.1 Inclusion And Scope

To form an overview of the major contemporary sources of FMFO, literature sources and expert opinions were sought for agreement and inter-reliability. We included all sources of FMFO that met the following criteria: i) at least an average of 100,000 tonnes of biomass of that species must be destined for reduction annually over the period of 2008-2012; ii) general agreement among sources that this species, or a substantial portion of its landings are regularly destined for reduction; and iii) adequate information exists on (a) the source species, including source ecosystem, trophic level, and meal and oil yields, and (b) the fisheries, including fishing nations, gear employed, fuel use intensity (FUI), and annual landings.

Those species that are also commonly used for direct human consumption (DHC), such as Atlantic herring (*Clupea harengus*), chub mackerel (*Engraulis japonicas*) and Chilean jack mackerel (*Trachurus murphyi*), have variable landings destined for FMFO production. These three examples can be classified as "prime food fish," but have a history and current usage for FMFO production (Wijkström, 2012). Species like this were included initially because they have high historical and potential future use for FMFO production, and current information supports that at least 100,000 tonnes are destined for reduction into FMFO. For species that are used for both DHC and reduction to FMFO, estimates of the total amount destined for reduction were made based on publicly available information to assess their contribution to the global scale of FMFO production (Table 7).

In addition to the reduction species included here, species whose processing by-products are now commonly used for FMFO production were included as well. These species are caught for DHC, but have large portions of by-products that can be converted into FMFO for use in other sectors. They are included in this analysis for comparison because of their increasing importance as an estimated 35% of fishmeal production in 2012 was derived from processing by-products (FAO, 2014a). Sources of meal and oil derived from by-products are excluded from the cumulative section of this analysis (detailed below) because the amount of by-products destined for reduction to FMFO products is highly variable and not publicly available information.

We compared individual FMFO products on the basis of their mass, specifically per tonne of meal or oil produced, as these are the units that these commodities are typically traded in. Results could have been expressed on the basis of nutritional energy content (e.g. MJ) or in terms of a specific nutritional attribute (e.g. protein, fat, Omega-3 content). However, these attributes obscure comparability of these FMFO products for many users. Where the need arose to allocate resource inputs among product streams, for example by-products destined for meal and oil and fillets in fisheries for DHC, mass or energetic content of streams was used depending on the stage of the production system concerned. Mass allocation was adopted and used because of the physical relationship of mass to fuel use during the fishing stage for the co-products originating from DHC fisheries. The environmental burdens arising from reduction fisheries and the processing stage were allocated to FMFO co-products based on their gross nutritional energy content to reflect the primary use of these products, where meal energetic density is species or species-type specific (e.g. whitefish, herring type, anchovy type), and oil was assigned a constant energy density (FAO, 1986; Parker & Tyedmers, 2012b; Sauvant et al., 2004).

3.2.2 Marine Footprint Methods

The primary production required (PPR) to yield FMFO products was quantified following the method developed in Chapter 2 that refines the analytical approach originally established by Pauly and Christensen (1995; Equation 1)

$$P_x = M/C * (1/T_x)^{L-1}$$

Equation 2

Where P is PPR, M is the mass of landings in tonnes, C is the ratio of wet weight biomass to carbon content of typical marine tissue (9:1 or 11.1% ; Pauly and Christensen 1995), T is transfer efficiency of ecosystem x, L is trophic level, and x is the ecosystem under study. The PPR per tonne of meal or oil (P_m or P_o, respectively) is then calculated (Equation 3) by the proportion of P_x attributed to meal or oil based on the relative energetic content of meal or oil compared to the total energetic content of meal and oil produced per tonne of wet weight biomass processed.

$$P_m = \frac{Y_m * E_m}{(Y_m * E_m + Y_o * E_o)} * P_x$$

Equation 3

Where Y is yield, E is energetic content, m is meal, and o is oil. Equation 4 was used to express the area, F_x, required to sustain production of one tonne of fishmeal or fish oil.

$$F_x = P_m/R_x$$

Equation 4

Adapted from Parker and Tyedmers (2012)

Where F is marine footprint in $km^2 year^{-1}$, and R is the ecosystem primary production rate, expressed in tonnes carbon per km^2 per year, of ecosystem x.

Global landings data were obtained from FishStatJ using 2012 as the reference year (FAO, 2014b). Ecosystem specific transfer efficiency values rather than the global average of 10% was used (Pauly & Christensen, 1995), and these ecosystem-specific estimates were obtained from a summary of Ecopath with Ecosim models (Libralato et al., 2008). These models demonstrate substantial variance from the global average of 10% (3.51% to 14.8% for ecosystems modeled), but correspond to the specificity of the ecosystems from which animals are harvested from instead of the global average (Libralato et al., 2008). However, because they are not obtained from multiple data points they do not have the same degree of confidence as an average value has. Fish trophic levels were obtained from FishBase (2014), and the same species harvested from different ecosystems was assumed to have the same trophic level value. Data on LME primary production were accessed from the Sea Around Us Project (2011). The primary production estimates are derived from satellite data (from the ten year period of 1998 to 2007) that calculates primary production from chlorophyll pigment concentration (Platt & Sathyendranath, 1988). The Institute for Environment and Sustainability of the EU Joint Research Center produced these estimates under the supervision of Nicolas Hoepffner and Frédéric Mélin. Lastly, species-specific fishmeal and oil yield rates were obtained from public and private sources.

To complete the analysis of cumulative impacts on ecosystems, only those FMFO sources whose source ecosystems were known, or could be assigned with some confidence to a discrete large marine ecosystem were analysed. This analysis was conducted by drawing upon previous analyses of fisheries catches (Sea Around Us Project, 2014), and country specific reports of fishery landings. Thus, many species with an extensive distribution of fisheries were excluded from this section of the analysis.

3.2.3 Fishing And Processing Energy Use

The FUI, in liters of fuel per tonne of round weight landings, for each fishery was estimated from available published sources and unpublished analyses by the authors (Table A1; Parker & Tyedmers, 2014). Original data include both direct reports of fuel consumption by individual vessels or fleets, as well as estimates calculated from rates of fishing effort. Sources include national and international energy analyses, fishery life cycle assessments, government and industry reports, and fishing vessel energy audits. Each fishery was matched to records of FUI by both target species and fishing gear. Where multiple reported FUI values were available for a single fishery, estimates were weighted on the basis of sample size (number of vessels reporting).

Energy inputs to reduction plants were modeled based on previous studies to reflect species- and region-specific technologies and thus efficiencies where these data were available. Data regarding types and quantities of energy inputs to wet fish and invertebrate reduction processes were assembled from available public and private sources. These data were applied directly to the associated sources of meal and oil and reduction settings modeled where possible. For all sources of meal and oil for which direct reduction energy use data were not available, average values were applied per tonne of wet biomass processed based on known data from similar reduction settings. For example, where fish are reduced in settings in which natural gas is readily available as the primary thermal energy source, data from known reduction facilities that use steam from natural gas were applied. On-board processing is used for a minor amount of FMFO production, but these were modeled based on previous studies of FMFO products.

3.2.4 Carbon Footprint

The direct and indirect GHG emissions, or carbon footprint, of FMFO products were quantified through the summation of GHG emissions from fuel use of the fisheries stage and energy use during the processing stage of FMFO production. Greenhouse gas emissions from fossil fuel combustion also accounted for all associated upstream processes (e.g. extraction, refining, transport). The Ecoinvent 2.0 database from the Swiss Centre for Life Cycle Inventories was used for the GHG emissions of various fuel sources. The carbon intensity of electricity inputs into reduction facilities for mechanical operations and controls were modeled based on country-specific electricity mixes from the World Resources Institute's (2011) analysis for 2007.

The carbon footprint of each meal and oil product was used in conjunction with annual landings and portion destined for reduction to estimate the total carbon footprint of FMFO production from dedicated reduction fisheries analysed along with their contribution to the sustainable limits of greenhouse gas emissions (Pelletier & Tyedmers, 2010a). Assuming the dedicated reduction fisheries analysed were representative of global reduction fisheries, an estimate was then made of the total carbon footprint of FMFO production from reduction fisheries in 2012. The sustainable boundary was assumed to be 350 ppm CO_2 (Rockström et al., 2009), and thus an annual anthropogenic emission limit of 8.9 gigatonnes of CO_2 equivalents was assumed (Allison et al., 2009; Pelletier & Tyedmers, 2010a).

3.3 Results

3.3.1 Sources Of Meal And Oil Analyzed

Of all dedicated reduction fisheries that contribute to global FMFO availability, 18 discrete combinations of species, source ecosystem and gear met our inclusion criteria (Table 7). Together, they represent an estimated 52% of the 16.3 million tonnes of wet weight biomass landed by dedicated reduction fisheries in 2012 (Table 8). The majority of these species' landings originated from nine large marine ecosystems with a concentration in the North Atlantic region. All dedicated reduction fisheries studied used purse seine or pelagic trawl fishing gear, except for a bottom-trawl fishery targeting sandeel (*Ammodytes marinus*). The sources of FMFO were mainly small pelagic species that occupied a middle trophic level. However, some important sources of FMFO were derived from relatively low (e.g. Antarctic krill; L=2.2) and high trophic level organisms (e.g. blue whiting; L=4.0). Furthermore, sandeel and blue whiting both inhabit benthic environments regularly, although not exclusively, in contrast to most of the other species that occupy pelagic environments. Three DHC fisheries targeting Alaska pollock (*Theragra chalcogramma*), Atlantic cod (*Gadus morhua*), and haddock (*Melanogrammus*

Table 7. Fishery characteristics and impacts per meal and oil sorted by carbon footprint of meal

Species	FUI (L/ tonne) (Gear)[a]	Fishing stage contribution (%)[b]	Trophic Level	Destined for Reduction (%)	Yield (kg/tonne) Meal	Oil	Processing Energy (MJ) Thermal (Source)	Electricity (Country)	Ecosystem Properties T (%)	PP (mg C $m^{-2}day^{-1}$	Size (km^2)
Anchovetta (HC)	18 (P)	33.2	2.7	100	240	50	1518 (NG)	74 (Peru)	6.6	876	2,619,386
Gulf Menh. (GM)	37 (P)	48.9	2.2	100	210	160	1486 (NG)	92 (USA)	9.7	570	1,530,387
Capelin (IS)	23 (P)	40.2	3.2	95	165	77	1486 (NG)	92 (Nor)	14	551	521,237
A Herring (IS)	43 (P)	55.7	3.2	30	200	110	1486 (NG)	92 (Nor)	14	551	521,237
A Herring (NS)	43 (P)	55.7	3.2	30	200	110	1486 (NG)	92 (Nor)	11.6	1115	690,041
A Menh. (NE)	29 (P)	42.8	2.3	100	240	50	1486 (NG)	92 (USA)	14.8	1536	308,544
Cal. Pil. (CC)	100 (P)	72.2	2.4	25	230*	180*	1486 (NG)	92 (Mex)	4	613	2,224,665
Euro. Pil. (CnC)	109 (P)	76.1	3.1	50	230	180	1486 (NG)	92 (Nor)	5	1196	1,120,439
Capelin (IS)	102 (P)	74.9	3.2	95	165	77	1486 (NG)	92 (Nor)	14	551	521,237
A Herring (IS)	142 (T)	80.6	3.2	30	200	110	1486 (NG)	92 (Nor)	14	551	521,237
A Herring (NS)	142 (T)	80.6	3.2	30	200	110	1486 (NG)	92 (Nor)	11.6	1115	690,041
B. Whit. (NS)	85 (P)	71.3	4.0	95	197	19	1486 (NG)	92 (Nor)	11.6	1115	690,041
B. Whit. (NWS)	85 (P)	71.3	4.0	95	197	19	1486 (NG)	92 (Nor)	3.51	491	1,109,613
Pollock (GA)	64 (T)	50.2	3.5	/	170*	17*	2212(FO)	0	14.2	906	1,491,252
Pollock (WBS)	64 (T)	50.2	3.5	/	170*	17*	2212(FO)	0	12.1	586	2,182,768
B. Whit. (NS)	111 (T)	76.4	4.0	95	197	19	1486 (NG)	92 (Nor)	11.6	1115	690,041
B. Whit. (NWS)	111 (T)	76.4	4.0	95	197	19	1486 (NG)	92 (Nor)	3.51	491	1,109,613
Sandeel (NS)	149 (B)	81.3	2.7	100	197	42.4	1486 (NG)	92 (Nor)	11.6	1115	690,041
Euro. Sprat (NS)	371 (T)	91.6	3.0	50	188	79	1486 (NG)	92 (Nor)	11.6	1115	690,041
Krill (AS)	141 (T)	72.8	2.2	70	160	0.8	418 (MDO)/ 1507 (IFO)	0	14	273	3,486,169
Cod (BS)	533 (M)	94.0	4.4	/	170	17	1486 (NG)	92 (Nor)	3.51	1910	396,838
Cod (IS)	533 (M)	94.0	4.4	/	170	17	1486 (NG)	92 (Nor)	14	551	521,237
Haddock (BS)	679 (M)	95.2	4.1	/	170*	17*	1486 (NG)	92 (Nor)	3.51	1910	396,838
Haddock (IS)	679 (M)	95.2	4.1	/	170*	17*	1486 (NG)	92 (Nor)	14	551	521,237

a. Fuel Use Intensity (FUI) with (P) indicating purse seine fisheries, T indicating pelagic trawl fisheries, B indicating bottom trawl fisheries, and M indicating mixed-gear. b. The percentage of the carbon footprint from the fishing stage energy use. * Indicates non-species specific yields used, with Atlantic cod yield rates used for other whitefish species of haddock and Alaska pollock, and European pilchard used for California pilchard.

a. Anchovetta (*Engraulis ringens*); A Herring (*Clupea harengus*); A Menh. (*Brevoortia tyrannus*); B. Whit. (*Micromesistius poutassou*); Cal. Pil. (*Sardinops sagax*); Cap. (*Mallotus villosus*); Cod (*Gadus* morhua); Euro. Pil. (*Sardina pilchardus*); Euro. Sprat (*Sprattus sprattus*); Gulf Menh. (*Brevoortia patronus*); Haddock (*Melanogrammus aeglefinus*); Krill (*Euphausia superba*); Poll. (*Theragra chalcogramma*); and Sandeel (*Ammodytes marinus*).Antarctic

Shelf (AS); Barents Sea (BS) California Current (CC); Canary Current (CnC); Gulf of Alaska (GA); Gulf of Mexico (GM); Humboldt Current (HC); Icelandic Shelf (IS); Northeastern United States Continental Shelf (NE); North Sea (NS); Norwegian Sea (NWS).

aeglefinus), each with substantial by-product utilization rates were included in this analysis for comparison (Table 7).

3.3.2 Carbon And Marine Footprints Of FMFO Products

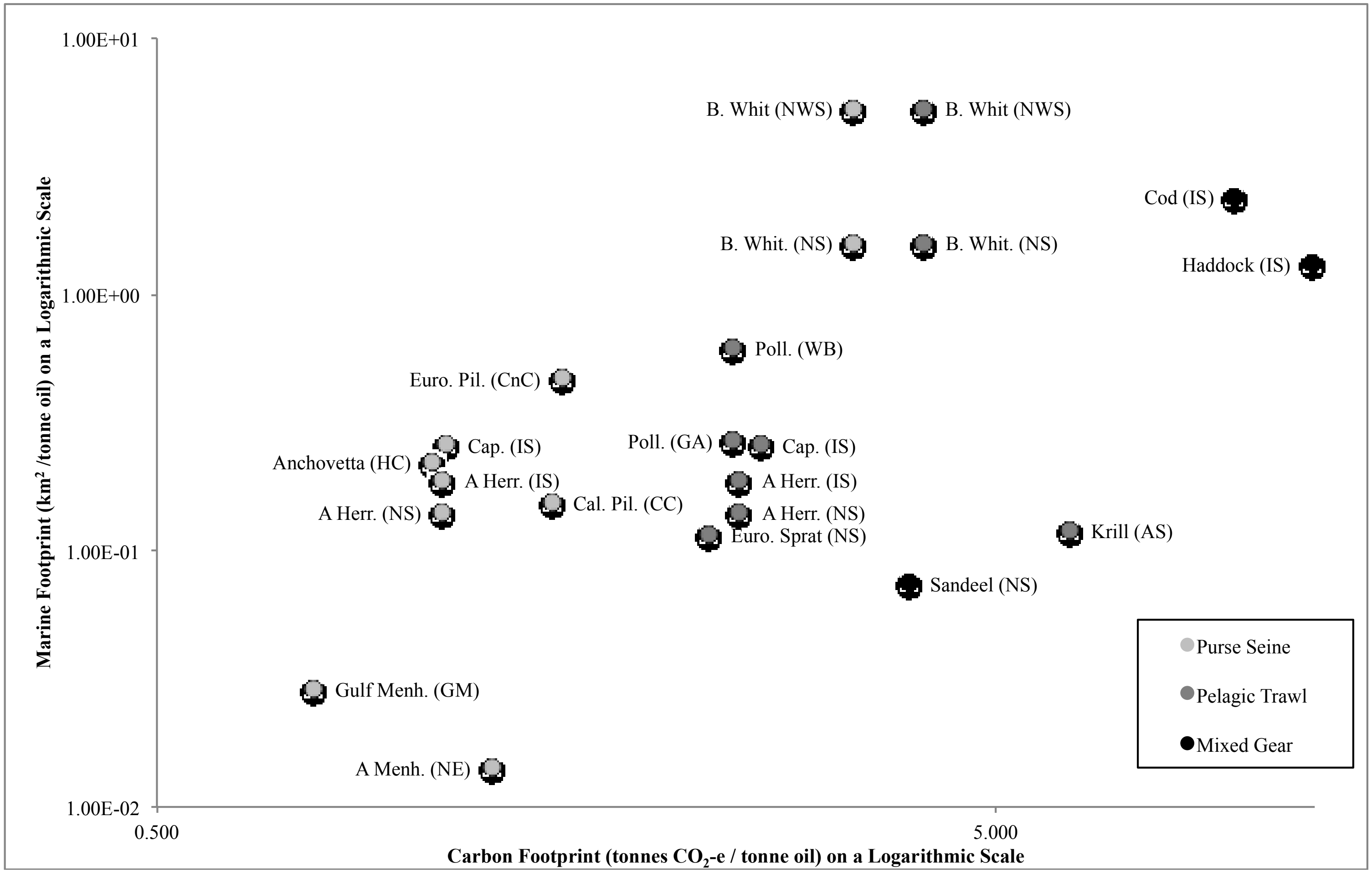
Three major patterns emerged when comparing the biotic and abiotic impacts of various FMFO products: i) the relative importance of fishing gear and source of processing energy on carbon footprint; ii) the effect of ecosystem transfer efficiency and trophic level on marine footprint; and iii) contrasting results of reduction fisheries and DHC fisheries.

On the basis of one tonne of fishmeal or fish oil produced, those sourced from fisheries employing purse seine gear had generally lower direct fishing FUI, through overlap exists with some fisheries using pelagic trawl gear of one form or another (Table 7, Figure 9, Table A2). Mixed gear fisheries for Atlantic cod and haddock performed the worst on this measure (Figure 9). The proportion of the carbon footprint attributed to the fishing stage varied substantially between sources of meal and oil (Table 7). Not surprisingly, those sources with higher fuel inputs during the fishing stage have a much greater proportion of GHG emissions associated with that stage. Based on the data available, the aggregate energy to process a tonne of wet weight biomass is fairly static (1600-2200MJ; Table 7). However, its associated carbon footprint varies more dramatically based on the source of thermal and electrical energy inputs and their relative proportions. The differences in amount and type of processing energy used are trivial in terms of the carbon footprint of fishmeal and oil products in all but the most fuel-efficient fisheries (<100L/tonne).

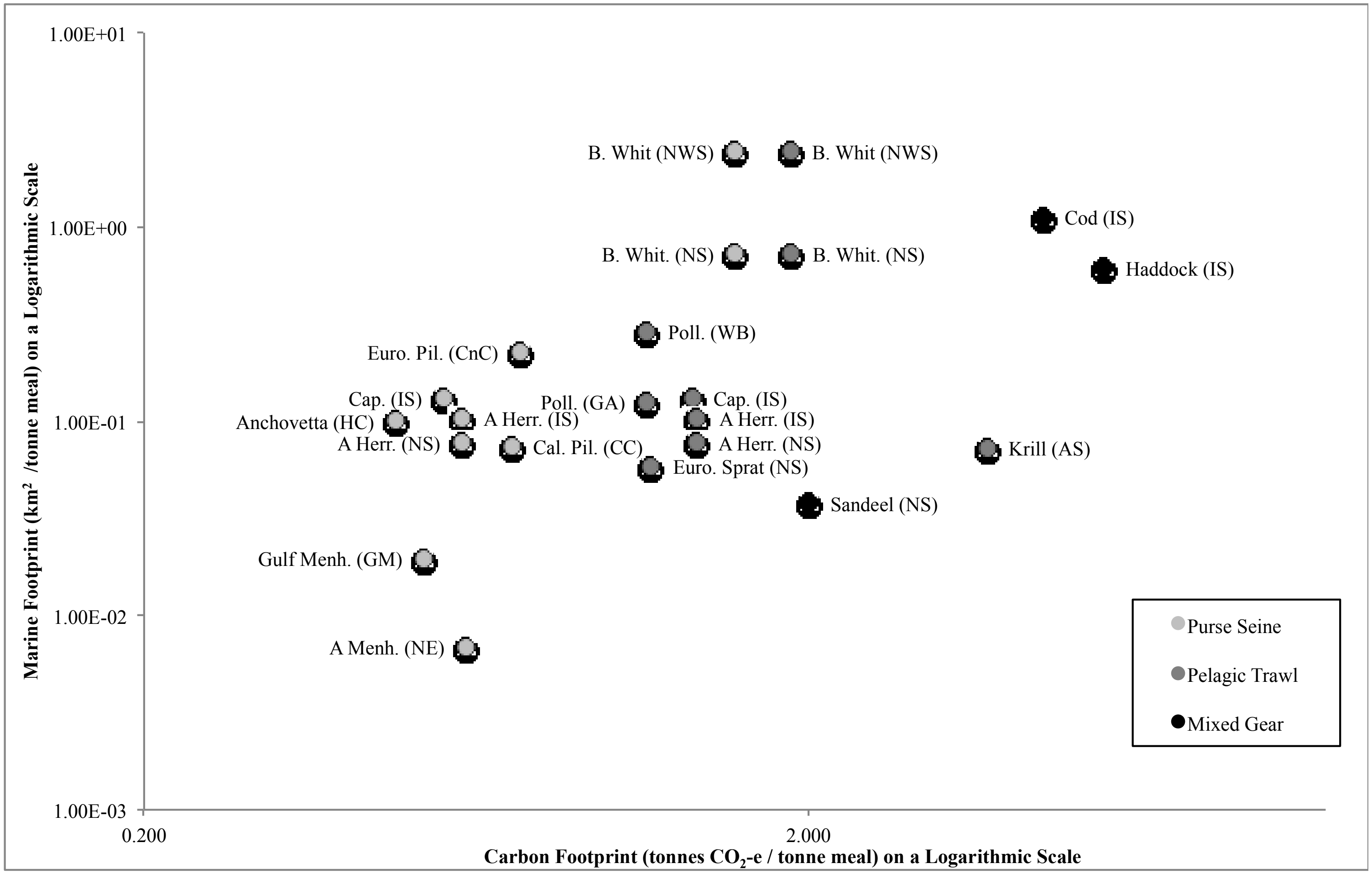
The sources of meal and oil that performed well on the marine footprint measure were derived primarily from low trophic level animals (Figure 9). However, the influence of source ecosystem specific transfer efficiency and productivity also played a substantial role. For example, the Barents Sea had the lowest transfer efficiency modeled (transfer efficiency= 3.51%), as compared to the North Sea (transfer efficiency= 11.6%) and the Icelandic Shelf (transfer efficiency= 14%). Consequently, the low transfer efficiency coupled with the very high trophic level species harvested from it (Atlantic cod= 4.2,

Figure 9. Carbon footprint and marine footprint of fish oil (A) and fishmeal (B) products on a logarithmic scale for both axes. Individual data points are listed by species with source ecosystem in brackets. Fishing method is denoted by shade of light grey (seine), dark grey (pelagic trawl), and black (mixed gear and bottom trawl).

Shortened form follows same convention in both figures and table: a. Anchovetta (*Engraulis ringens*); A Herring (*Clupea harengus*); A Menh. (*Brevoortia tyrannus*); B. Whit. (*Micromesistius poutassou*); Cal. Pil. (*Sardinops sagax*); Cap. (*Mallotus villosus*); Cod (*Gadus* morhua); Euro. Pil. (*Sardina pilchardus*); Euro. Sprat (*Sprattus sprattus*); Gulf Menh. (*Brevoortia patronus*); Haddock (*Melanogrammus aeglefinus*); Krill (*Euphausia superba*); Poll. (*Theragra chalcogramma*); and Sandeel (*Ammodytes marinus*).Antarctic Shelf (AS); California Current (CC); Canary Current (CnC); Gulf of Alaska (GA); Gulf of Mexico (GM); Humboldt Current (HC); Icelandic Shelf (IS); Northeastern United States Continental Shelf (NE); North Sea (NS); Norwegian Sea (NWS). Two data points both from the Barents Sea are excluded from both figures because they lie far outside the rest of the entries.


1.00E+01
1.00E+00
1.00E-01
1.00E-02
0.500
5.000
Marine Footprint (km² /tonne oil) on a Logarithmic Scale
Carbon Footprint (tonnes CO₂-e / tonne oil) on a Logarithmic Scale
B. Whit (NWS)
B. Whit (NWS)
Cod (IS)
B. Whit. (NS)
B. Whit. (NS)
Haddock (IS)
Poll. (WB)
Euro. Pil. (CnC)
Cap. (IS)
Poll. (GA)
Cap. (IS)
Anchovetta (HC)
A Herr. (IS)
A Herr. (IS)
Cal. Pil. (CC)
A Herr. (NS)
A Herr. (NS)
Euro. Sprat (NS)
Krill (AS)
Sandeel (NS)
Gulf Menh. (GM)
A Menh. (NE)
Purse Seine
Pelagic Trawl
Mixed Gear

A


1.00E+01
1.00E+00
1.00E-01
1.00E-02
1.00E-03
0.200
2.000
Marine Footprint (km² /tonne meal) on a Logarithmic Scale
Carbon Footprint (tonnes CO₂-e / tonne meal) on a Logarithmic Scale
B. Whit (NWS)
B. Whit (NWS)
Cod (IS)
B. Whit. (NS)
B. Whit. (NS)
Haddock (IS)
Poll. (WB)
Euro. Pil. (CnC)
Cap. (IS)
Poll. (GA)
Cap. (IS)
Anchovetta (HC)
A Herr. (IS)
A Herr. (IS)
A Herr. (NS)
Cal. Pil. (CC)
A Herr. (NS)
Krill (AS)
Euro. Sprat (NS)
Sandeel (NS)
Gulf Menh. (GM)
A Menh. (NE)
Purse Seine
Pelagic Trawl
Mixed Gear

B

haddock= 4.1) results in very large marine footprints for these two sources of meal and oil. Other ecosystems also had low transfer efficiencies (Humboldt current= 6.6%; California current= 4.0%), however, target species from these ecosystems were of lower trophic levels resulting in a smaller marine footprint than would otherwise be the case.

Sources of meal and oil that had high yield rates generally performed well. If the inputs of energy into the fishing and processing stage were comparable to other species, the higher level of products produced reduced the relative impact per unit of meal or oil. Yield rates were much more variable for oils than for meals, and this aided in dividing the burdens among more products and reduced the impact of both products when oil yields were high. Thus species with high oil yields like Gulf menhaden (16% oil yield rate) performed particularly well, and species with low oil yields, like blue whiting with a typical oil yield rate of 1.9%, performed particularly poorly.

In general, FMFO from by-products of DHC fisheries perform worse on both carbon and marine footprint measures than did products from dedicated reduction fisheries (Figure 9). This is mainly attributable to the relatively fuel intensive gear types deployed (bottom trawls, gillnets and long-lines), and the high trophic level sources of meal and oil. The allocation of inputs and emissions of co-products of DHC fisheries based on their relative mass, functionally sets the stage for this finding. Importantly, however, FMFO from DHC fisheries are not destined to perform poorly. This is illustrated by the performance of FMFO from Alaska Pollock, a species mainly caught with pelagic trawls in fisheries with low FUI that target a relatively low trophic level (3.5) animal. Fishmeal and oil from Alaska pollock not only performs better than the meals and oils from other DHC fisheries, it is just below the median value of all sources of FMFO assessed for both carbon footprint (median = 1.34 tonnes CO_2-e/tonne meal) and marine footprint (median = 0.129 km^2/tonne meal). In contrast, blue whiting has a relatively high trophic level for a species largely dedicated to reduction (4.0), and is on the higher end for its carbon footprint performing worse than Alaska pollock on both measures. Thus, there are exceptions to the general finding that sources of meal and oil from DHC fisheries perform worse than dedicated reduction fisheries on these measures (Figure 9).

3.3.3 Global Reduction Fisheries

The dedicated reduction sources of meal and oil modeled represent a live weight biomass that accounts for 52% of total reported landings destined for reduction in 2012 (FAO, 2014a). Together, the GHG emissions associated with producing the FMFOs analyzed here amounted to 1.96 megatonnes of CO_2-e in in 2012. This represents just 0.022% of humanity's estimated maximum annual CO_2-e emissions (8.9 gigatonnes) to stay within its sustainable boundary. Scaling up to global 2012 FMFO production from reduction fisheries, their combined carbon footprint can be estimated at 3.80 megatonnes of CO_2-e, or 0.0427% of the sustainable boundary. This can be considered a conservative estimate as some minor sources of GHG emmissions have been excluded.

To complete the analysis of cumulative impacts, only those FMFOs whose sources were known, or could be assumed, were analysed. In some cases, assigning the entire annual catch used for reduction to a single ecosystem would have yielded misleading results. Thus, many species with an extensive distribution of fishing were excluded from this section of the analysis. For example, the fishery for blue whiting spans five large marine ecosystems over most of the Northeast Atlantic region. While the major source ecosystems were modeled for this analysis for comparative impacts, assigning the entire marine footprint of blue whiting derived FMFOs to the North Sea or the Norwegian Sea would misrepresent the appropriation from multiple ecosystems. Therefore, the ecosystem impact can only be properly estimated for the Antarctic Shelf, the Gulf of Mexico, the Icelandic Shelf, the Northeast United States continental shelf, the North Sea, and the Humboldt Current (Table 8). The Humboldt Current experiences the largest estimated impact from reduction fisheries with 6.3% of primary production appropriated, but it also is responsible for 4.7 million tonnes of fish destined for reduction in 2012 and these landings historically have constituted the largest portion of this LME's annual landings (Sea Around Us Project, 2014).

Table 8. Global Reduction Fisheries Production and Impacts

Source	Landings1 (tonnes)	Meal Production (tonnes)	Oil Production (tonnes)	Carbon Footprint (Mt CO_2-e)	Ecosystem Area Appropriated (%)
Antarctic krill	131,703	21,072	105	0.0790	0.045
Atlantic herring	554,991	110,998	61,049	0.1757	X
Atlantic menhaden	224,404	53,857	11,220	0.0469	0.17
Blue whiting	359,854	70,891	6,837	0.1614	X
California pilchard	91,097	20,952	16,397	0.0392	0.18
Capelin	956,206	157,774	73,628	0.1705	4.6
European pilchard	509,696	117,230	91,745	0.2257	X
European sprat	204,255	38,400	16,136	0.0812	0.60
Gulf menhaden	578,693	121,526	92,591	0.1353	0.33
Peruvian anchovy	4,692,855	1,126,285	234,643	0.7877	6.3
Sandeels	107,577	21,193	4,561	0.0605	0.17
Total	8,411,330	1,860,178	608,913	1.9630	X

1. Landings destined for reduction to FMFO based on proportion of landings destined for reduction (Table 7) and 2012 landings (FAO, 2014b).

3.4 Discussion

3.4.1 Findings

Results of this research support many previous findings from LCAs of fisheries and aquaculture systems and studies of primary production required to sustain fisheries. The importance of the fishing stage to overall GHG emissions has previously been cited in many LCAs and related studies (Fréon, Avadí, Vinatea Chavez, & Iriarte Ahón, 2014; Ziegler et al., 2003). However, as found in a study of organic and conventional salmon feeds (Pelletier & Tyedmers, 2007), the processing stage of FMFO can be a relatively large contributor to the overall life cycle of FMFO products, particularly when fishery-related FUI is low. Results of the current study affirm these findings as demonstrated through the large variance in the proportion of the carbon footprint attributed to processing. While direct fuel consumption in a fishery may be a reasonable proxy of the carbon footprint of most DHC fisheries (Parker & Tyedmers, 2014), it is a less robust surrogate for reduction fisheries because of their relatively low fuel intensities and consistent, non-trivial processing-related emissions.

An important and relatively unique result of this work is the illumination of the substantial differences that pertain in terms of both biotic and abiotic impacts of reduction fisheries, and their products. While prior research has hinted at the existence of substantial differences in the impacts of specific FMFOs (Pelletier and Tyedmers 2007, Pelletier et al. 2009, McGrath et al. 2015) or addressed impacts of a limited set of meals and oils (Parker and Tyedmers 2012), the extent to which impacts of meals and oils can vary has not been fully appreciated. Consequently, these nutritionally and economically valuable products should not be treated as environmentally equivalent or interchangeable. Indeed, given the highly divergent environmental 'costs' associated with many of them, feed formulators and other consumers of meals or oils seeking to produce more sustainable products should attend closely to their unique characteristics. In this context, this thesis echoes previous work that highlights the importance of harvesting low trophic level species from ecosystems with high transfer efficiencies for minimized biotic impact (Parker & Tyedmers, 2012b), and from fisheries that are fuel efficient, either because of the gear used, stock status, or species characteristics (Parker & Tyedmers, 2014; Ziegler & Hornborg, 2014). These 'principles' function well in the absence of other available information on the abiotic and biotic impacts of these fisheries products.

The reduction fisheries surveyed would contribute about 0.022% of humanity's sustainable boundary of CO_2-e emissions. While this may seem insignificant, this is one sector within global fisheries that represents a small portion of global food production in comparison to cereals, fruits, vegetable, and livestock production (Troell et al., 2014). However, this may represent one of the lowest carbon footprint animal production systems in comparison to DHC fisheries or livestock production systems (Parker & Tyedmers, 2014; Pelletier & Tyedmers, 2010a). The products of FMFO, however, are an intermediate product and not intended for human consumption. If an expanded human consumption market were available for more of the species destined largely for reduction, as it has expanded already for some of the species traditionally used for reduction, many of the species surveyed would perform better than other animal protein products on these two measures. In their current primary use as inputs to aquafeeds, they often deliver an important environmental and ecological benefit to this production system by using lower impact feed inputs compared to other existing or proposed aquaculture feed inputs (Boissy et al., 2011; McGrath et al., 2015; Papatryphon et al., 2004; Pelletier & Tyedmers, 2007; Pelletier et al., 2009; Samuel-Fitwi, Meyer, Reckmann, Schroeder, & Schulz, 2013). The broader benefits that reduction fisheries' products deliver to society is out of the scope of this thesis, but the relative efficiencies and value they deliver to society is an important area for future comparison.

3.4.2 Methodological Choices

The relative performance of FMFO products from dedicated reduction fisheries and DHC fisheries does lay on a methodological foundation that treats all utilized co-products of a production system as biophysically 'valuable' or important. In other words, the approach used here and throughout much of the extant related literature (McGrath et al., 2015; Pelletier & Tyedmers, 2007; Pelletier et al., 2009) does not discount impacts of FMFO from DHC just because they are secondary. This analysis attempts to reflect the environmental burden of the products' source fishery, including those that are sourced from by-products of direct human consumption fisheries. This may seem contradictory to recent efforts to reduce demand on fisheries for aquafeed production and reduce waste from seafood supply chains; however, to accurately model the impacts associated with these products, allocation of burdens is necessary and was done in a way that is consistent with the biophysical realities of these systems. Furthermore, the primary purpose of all co-products from these fisheries is to deliver energy to humans, either directly or indirectly, and so energy was chosen to distribute environmental burdens between co-products of both types of fisheries. While efforts should continue to be made to minimize human biotic and abiotic impacts, analyses like this demonstrate that by-products are not 'free' of environmental burdens. If by-products are not used these burdens are attributed solely to the products that are used by society, and thus using by-products reduces burdens apportioned to the main products (e.g. fillets). This approach reflects the biophysical reality of these products; however, it is acknowledged that if allocation of inputs and impacts was based on relative economic values of co-products, results could vary substantially (Ayer et al., 2007).

3.4.3 Limitations

Given the lack of publicly available data on reduction fisheries, this study was inherently limited by data availability and accuracy. The inclusion criteria for reduction species (Section 3.2.1) were established to attempt to model the most important dedicated species and fisheries used for the production of FMFOs. Furthermore, the analysis was limited by the uncertainty of key parameters including transfer efficiency, primary production rates, and yields of FMFO. These same parameters have limited data availability, but are also subject to natural variability annually and seasonally. The inter-annual variation of marine primary production is debated in the literature (Swartz, Sala, Tracey, Watson, & Pauly, 2010; Watson et al., 2014), but recent research has shown stronger variation that may be linked to a changing climate (Dalpadado et al., 2014). Ecosystems like the Barents Sea, which is partially covered by sea ice for portions of the year, are likely to be greatly affected by warmer temperatures, resulting in increased primary production rates (Dalpadado et al., 2014). Other ecosystems could see declines in primary production leading to potentially increased %PPR if current fishing exploitation rates do not change (Chavez et al., 2011).

Lastly, the PPR metric is limited by the scope of what it addresses. This measure is meant to be a coarse indicator of potential biotic impact, but the sustainability of higher levels of appropriation has not yet been examined. The measure, thus, does not give an indication of the impact of higher ecosystem PPR, and what levels of appropriation may be sustainable. However, this measure has been used with other criteria to inform the probability of sustainability of current and past fisheries practices (Libralato et al., 2008). Other studies have demonstrated the existence of threshold levels of overharvesting middle trophic level fish (forage fish) that demonstrated negative impacts on seabirds (Cury et al., 2011). While giving a broad indication of the level of impact, the ecosystem-wide PPR does not show these kinds of impacts on individual species and their prey/predators in the ecosystem. This indicator, while giving a broad overview of potential ecological concerns and impacts, is not specific to where biotic impacts may be distributed within ecosystems.

3.5 Conclusion

Evaluating major sources of fishmeal and oil globally, using both carbon footprint and marine footprint demonstrates substantial differences in the impacts of these products. The information presented in this chapter can be used to inform feed formulation decisions based on biotic and abiotic criteria for the most widely utilized FMFO products.

Hopefully, this will promote the use of less impactful ingredients in the formulation of feeds for aquaculture and other livestock sectors that wish to meaningfully address environmental sustainability concerns. Furthermore, FMFO products and their respective fisheries should be examined in comparison to other production systems' contribution towards sustainable limits and their relative value to society.

Chapter 4. Discussion And Conclusion

This thesis began with two broad objectives: 1) to clarify and refine the current method of quantifying marine biotic resources in LCA; and 2) to evaluate global reduction fisheries products of fishmeal and fish oil on a biotic and abiotic measure for relative efficiencies and their cumulative impacts. In achieving the first objective I demonstrated the current variance of practice and need for a more accurate quantification of PPR, which was undertaken in Chapter 2, and then applied to marine inputs to Norwegian salmon feeds in 2012 and more generally to a large cross-section of sources of FMFO in Chapter 3.

Chapter 2 presented this clarification of current practice through a literature review of relevant LCAs and related studies. I found the consistent use of non-region-specific transfer efficiencies and non-species-specific fishmeal and oil yields in the calculation of PPR. This led to a refined method for quantifying PPR being proposed and applied to a case study of marine inputs to Norwegian salmon feed. I compared the previously employed methods to the proposed method and demonstrated a large variance that is caused by using species and region-specific values when compared to average values for these two parameters.

Chapter 3 explored the relative efficiencies and total impacts of the FMFO products used globally. It is estimated that this analysis captured 52% of fish destined for reduction in 2012. This also demonstrated a non-negligible appropriation of primary production at the ecosystem level, and contribution towards humanity's safe operating space with respect to GHG emissions. Results of this work demonstrated the relative benefits of using lower impact FMFO products, but also points to the bigger picture of environmental impacts when these activities are co-occurring with other human activities.

The most important findings that could not be fully explored in these two chapters are: 1) broader concerns of improving efficiency in food production as compared to the total impact these systems are having at a planetary scale; 2) the potential limitations of biophysical accounting tools; and 3) the future of aquaculture and reduction fisheries given projected growth of the aquaculture sector. These topics will be discussed separately followed by the limitations of this thesis, prospects for future research, and conclusions to be drawn from this research.

4.1 Efficiency And Scale

The concern of efficiency compared to the contributions made to the total scale of impacts of a given activity has been a consistent theme throughout this thesis. As previously stated, this concern is reflected in the choice of biophysical accounting tools. Methods like LCA, material flow analysis, and energy return on investment focus on the efficiency or eco-efficiency of products or processes. These methods have thus demonstrated relative inefficiencies in production systems from the material intensity of glass Coca-Cola bottles (Baumann & Tillman, 2004), to the higher impacts of animal by-products in feeds (Pelletier & Tyedmers, 2007; Pelletier et al., 2009). These techniques evaluate the material and energy use of different products and processes to target potential improvements from a material and energy standpoint, which often have environmental benefits. I cannot dismiss the value these techniques and analyses have to these industries in improving their performance economically and often environmentally.

Efficiency improvements are extremely important to economic activity and growth (Huesemann & Huesemann, 2008). The constant change and improvements of technology has resulted in less human-power to accomplish similar tasks (Catton, 1987; Edwards-Jones et al., 2000; Huesemann & Huesemann, 2008). This reduction of labour needs, and often material and energy needs as the process of technological advancement continues, has led to significant economic gain (Huesemann & Huesemann, 2008). This is now being pursued anew in the context of sustainable development to reduce the amount of material and energy throughput of economies under eco-efficiency (Jänicke, 2008). Whole economies have become more eco-efficient and we have witnessed a partial or absolute decoupling of particular environmental impacts and economic growth (OECD, 2014). Therefore, real environmental and economic gains are being made through the pursuit of efficiency, but eco-efficiency does not consider the impacts of other co-occurring activities of human society.

Reduction fisheries alone will not compromise humanity's safe operating space. They have had in the past, and may continue to result in large disruptive effects on marine ecosystems and fish stocks (Myers et al., 1997). However, these impacts are often cumulative when reduction fisheries are considered in the context of all other marine fisheries and their impacts (Baum & Worm, 2009; Pauly & Christensen, 2000), and other environmental stresses on marine ecosystems such as eutrophication (Millennium Ecosystem Assessment, 2005), ocean acidification (Mora et al., 2013), and climate change (Merino et al., 2012). Just as we must consider the impacts to marine ecosystems from a variety of stressors, GHGs from reduction fisheries are added to the cumulative emissions of humanity globally. While reduction fisheries may represent a small sub-sector of these global emissions, all sectors contributing must be evaluated for the value they deliver to humanity in relation to their environmental impacts. On a planetary scale, we have already exceeded the safe operating space for climate change (Rockström et al., 2009).

This exceeding of planetary boundaries can be explained by a number of factors. Past improvements in efficiency, such as steam and automobile engines, have actually led to increases in their use that rapidly outpaced the effect of efficiency improvements (Jänicke & Lindemann, 2010; Jänicke, 2008). Thus, increasing efficiency in the past has led to increases in environmental impacts because of a 'rebound effect' (Alcott, 2005; Jänicke & Lindemann, 2010). This is because of the decreased cost to engage in these activities, but also to the rising population and affluence of the global human population. Affluence has often led to greater environmental impacts as previous analyses of developed and developing countries has shown (Dietz, Rosa, & York, 2007; Ehrlich & Holdren, 1971; T. Jackson, 2005; York & Rosa, 2003). The rising nature of affluence, which has compounding impacts because of rising population, is likely outstripping any gains in efficiency.

Thus, this thesis has attempted to not only consider the relative improvements that can be made through choice of FMFO products, but also the total impacts of this sector. Analyzing the impact on marine ecosystems from the multiple stressors mentioned earlier was outside the scope of this thesis, but this kind of holistic analysis would be necessary to more fully understand human impacts on marine ecosystems, and more broadly on a planetary scale.

4.2 Biophysical Accounting Tools

Life cycle assessment and broadly related tools like ecological footprint analysis account for material and energy inputs in human society that are not accounted for through other means. Life cycle assessment accounts for and 'points to' material and energy use and waste throughout the life cycle of the product, and thus often the best places to target for efficiency gains to reduce environmental impacts (Baumann & Tillman, 2004). Alternatively, EFA is used to illustrate the magnitude of resource use and waste assimilation required for human activities in terms of the spatial area of ecosystem support required (Wackernagel & Rees, 1996). Neither was designed to directly address the ecological impact that results from these activities, but both imply that higher resource use and higher waste outputs (solids, liquids, and gases) likely lead to negative consequences for the natural environment, and often human-health as well.

This thesis has advanced methods used in both LCA and EFA to better understand the biotic impacts of fisheries and aquaculture. Primary production required is often used as a proxy for ecological impact that is difficult to measure directly, but it has been supplemented with other measures of biotic impacts. This is an area of challenge in LCA more broadly, how many indicators are appropriate, and which impacts must be included in an analysis? More and more methods have been developed to quantify abiotic and biotic impacts relative to a functional unit within an LCA framework, but the amount and kind of impacts that are reported in the average seafood LCA article often remain the same (Avadí & Fréon, 2013; Henriksson, Guinée, Kleijn, & Snoo, 2011).

The challenge remains to ensure LCA is an accurate reflection of production systems, products, and processes. Social LCA is in its early stages of development to account for hidden social impacts, both positive and negative, in supply chains of products (Kloepffer, 2008). Economic LCA and life cycle thinking and management have also been promoted to apply this method to organizations' views of their products (Heiskanen, 2002). If LCA is supposed to be a universal holistic method for accounting for sustainability and its three pillars, all of these aspects must be considered (Kloepffer, 2008). However, I would argue this is pushing LCA outside its domain to the point where it is no longer useful. It is helpful to address the social and economic impacts of the globalized supply chains that undergird world trade today, but forcing them into a framework that accounts for material and energy flows and measurement of potential environmental impacts seems to be a stretch. When designing or creating a new product, companies may look at their supply chains from cradle-to-grave to understand their social, economic, and environmental impacts, but forcing them into this BAT will likely not lead to progress on these issues. Additionally, LCA is not meant to be a decision-making tool like cost-benefit analysis, but to inform decisions based on new information. The two techniques of LCA and EFA are both limited in their scope and were not designed to be tools that measure everything about a product or a society, and nor should they be.

The suite of BATs should be focused on delivering clear information on the often unseen side of human activities. Ecological footprint analysis can clearly demonstrate the resource and assimilation needs of society at regional, national and global levels. Life cycle assessment can produce comparisons of different technologies, production systems, or substitutable products to inform future decision-making, and model potential improvements to systems. However, neither capture the actual ecological impacts of these effects, even with the advancements recently undertaken, as noted in Chapter 1. Life cycle assessment especially is used to quantify the *potential* for impact that a system is having, but not the realized impact it is *actually* causing. For example, including the seafloor swept as an impact of a trawl fishery quantifies the area impacted (Nilsson & Ziegler, 2007; Ziegler et al., 2003), but does not quantify the breeding grounds and coral reefs lost because of the trawling (Pauly et al., 2002). This continues to be an unresolved challenge in much of LCA practice in that its impacts are measured at the mid-point where they leave the economic system and not at the endpoint where the impacts are actually occurring (Haes & Jolliet, 1999; Hertwich, Pennington, & Bare, 2002; Jolliet et al., 2004).

There are two inter-related issues for assessment of impacts in LCA: endpoint measurement and aggregation of impacts to a single score. Endpoint measurement attempts to quantify the realized impact of production systems converting the resource use and waste streams of production systems into general areas of concern such as years of human life lost and species extinctions. Aggregation of impacts into a single score equalizes different types of emissions and resource uses into a unitless score to compare between production systems (Rack, Valdivia, & Sonnemann, 2013). Both of these methods involve assumptions that further enhance uncertainty and potential errors (Finnveden et al., 2009). Furthermore, the uncertainty and errors can be supplemented when aggregating impacts with value choices of how much a certain abiotic impact influences an area of protection like human health when there is limited data available. These methods attempt to simplify and express results in actual areas of concern enhance uncertainties of the final results. While these advancements could be useful in the future, the estimates they make cannot be done with the certainty for them to be useful at this time.

4.3 The Future Of Aquaculture And Reduction Fisheries

This thesis has focused on the subsector of reduction fisheries in the context of fisheries and aquaculture globally. I have addressed the current resource demands of this sector and impacts of both biotic and abiotic natures. However, this snapshot of relatively contemporary production does not indicate how these resource demands will evolve in the future. The prospect of a human population of over 9 billion in 2050, that is increasing demand for animal-protein on a relative and absolute basis, will require a growing aquaculture sector (FAO, 2014a). Aquaculture is expected to grow from 53 million tonnes in 2008 to 94 million tonnes by 2030, while capture fisheries landings will remain largely stagnant (FAO, 2014a). This will increasingly be supplied by a growing FMFO production sector that's marginal growth will almost entirely draw upon fish by-products (FAO, 2014a).

This growing sector's reliance on fish by-products for FMFO is potentially problematic based on results of this thesis. Fish by-product FMFO has been previously found (Ayer et al., 2007; Pelletier & Tyedmers, 2007; Pelletier et al., 2009), and largely confirmed in this thesis (Chapter 3) to be more impactful than reduction fisheries. While this cannot be taken as a firm rule, the fish that are caught by DHC fisheries, and whose by-products are used for FMFO production, are often those fish with characteristics of more impactful fisheries (high trophic level, low stock size, high fuel use, etc.; Parker & Tyedmers, 2012, 2014; Ziegler & Hornborg, 2014). The result being that the aquaculture industry is already likely utilizing the most eco-efficient sources of FMFO. As the industry must use less eco-efficient sources of FMFO to meet its growing demand, the industry will thus grow less eco-efficient in this area.

To meet growing demand for animal-protein, all sectors of animal production are projected to increase (Alexandratos & Bruinsma, 2012). As animal-protein food production is the most impactful sector of food production (Gerber, Wassenaar, & Rosales, 2007), and food production a highly impactful human activity (Foley et al., 2011), society should focus on minimizing these impacts through favouring eco-efficient forms of production. Capture fisheries are constrained by their natural ecosystems (Chassot et al., 2010; Merino et al., 2012; Pauly & Christensen, 1995), and aquaculture is partially constrained by the availability of FMFO and other feed ingredients (Naylor et al., 2009; Troell et al., 2014). However, as aquaculture systems perform environmentally favourably in many areas relative to other animal production systems (Pelletier & Tyedmers, 2007; Pelletier, 2010; Torrissen et al., 2011), their contribution to human diets should be maximized in comparison to other livestock products. Lastly, as future animal protein production alone presents potentially dangerous levels of environmental impacts (Pelletier & Tyedmers, 2010a), there should be a reduction of these products relative contribution to human diets (Foley et al., 2011).

A scenario that cannot be analyzed in depth here is the status of fisheries in a changing climate. Climate change will have significant effects on marine primary production and thus marine ecosystems globally by 2050 (Merino et al., 2012). Some ecosystems will thus not be able to sustain current catch levels (e.g. Humboldt Current), while others will likely increase their potential catch (e.g. Barents Sea; Merino et al., 2012). The high importance of the Humboldt Current to annual FMFO production because of Peruvian anchovy is particularly troubling given this scenario. While fisheries globally are expected to have a minor increase under this climate change scenario and under future management (FAO, 2014a; Merino et al., 2012), the distribution of those catches will be significantly altered. Thus, fisheries management will need to adapt to a changing climate within the near future.

4.4 Limitations

Uncertainty was addressed directly in Chapter 2 of this thesis. However, the issue of uncertainty pervades the quantification and analysis of production systems, and even more so for biological production systems (McGrath et al., 2015; Parker & Tyedmers, 2012b). The challenge I addressed was managing uncertainty with limited data. To address parameter uncertainty robustly through quantification methods, it is necessary to have even more data on the parameters of interest (van der Sluijs et al., 2005). This presents a situation where it is likely that as one can better quantify parameter uncertainty, it becomes more unnecessary because you have more information on the area of interest. This, however, does not pertain to the uncertainty created by natural variability, which can increasingly be accounted for and quantified through better knowledge of the area of interest, but does not become unnecessary in the same way as parameter uncertainty quantification. I addressed this lack of knowledge through eliminating the parameters of transfer efficiency and fishmeal and oil yield rates from my uncertainty analysis, which used Monte Carlo method. This was not the ideal choice for the situation, as modeling more parameters' uncertainty would be useful to estimate the range and confidence of the primary production required more effectively. However, it is unreasonable to quantitatively model uncertainty in Monte Carlo analysis when there is a large amount of both parameter uncertainty and lack of data (Huijbregts, 1998; van der Sluijs et al., 2005). Since I was using species and ecosystem specific parameter values for fishmeal and oil yield rates and transfer efficiency, respectively, I did not have the same number of observations as other parameters to be able to properly parameterize the Monte Carlo analysis. Therefore, this section of the analysis was limited to parameters where this modeling was reasonable.

This thesis was limited by data availability. The need to address uncertainty is a symptom of this, but this affected other parts of the analysis as well. The global analysis of reduction fisheries had to rely on several criteria to form an initial list of species, and this relied upon estimations on many of their parameters, including the amounts destined for reduction. Until data on the use of fisheries landings (e.g. reduction, DHC, etc.) and their respective supply chains (e.g. processing energy use, inputs and outputs of reduction processing, etc.) are more publicly available, a complete analysis cannot be done on this sector. Additional challenges are the constantly changing nature of these sectors that are responding to price fluctuations of the commodities, changing management guidelines for the total allowable catch allotted to different countries, and restrictions on use of the landed fish. This means this analysis of 2012 could vary significantly in other years, and is likely influenced by the most variable populations including Peruvian anchovy.

4.5 Future Research

This research has given me a brief opportunity to explore the nature of BATs, and their relation to the scale of the human enterprise. Many BATs, including LCA, do not focus their lens of analysis on the total scale of human activity but focus at a micro scale that often ends its concerns with efficiency. Efficiency, or eco-efficiency, can be used to reduce resource use and environmental impacts, although this is potentially problematic (see '4.1 Efficiency and Scale' above). This can also be used to inform decisions between alternative production systems or products to have the same effect. However, a concern solely with efficiency does not point to the larger issue of scale. Unless these gains in efficiency are considered in their proper context of humanity within a finite ecosphere, they will not progress towards environmental sustainability (Huesemann & Huesemann, 2008). This thesis has attempted to apply this same logic of evaluating the relative efficiencies between species and ecosystems for reduction fisheries, but also to measure their potential impact at the scale of the large marine ecosystem. Any single activity will likely not exceed planetary sustainability boundaries (Rockström et al., 2009); however, considering the important sectors and their contribution towards the unsustainability of society's total activities should force society to reconsider the current pattern of our activities (Pelletier & Tyedmers, 2010a).

In line with the previous discussion, and previous research in this direction, I believe more can be done to incorporate ecological impacts in BATs. There exist many recent advancements to incorporate ecological impacts like by-catch (Hornborg, Svensson, et al., 2013), discards (Vázquez-Rowe et al., 2012), damage to the benthos (Ziegler et al., 2003), and localized eutrophication (Ford et al., 2012) into LCA. These have not been widely applied, but could be considered more in the future. These advancements have been done with the intent of making LCA account more holistically for the ecological impacts of human activities, specifically fisheries and aquaculture. However, further advancements should also critically explore if LCA is able to accommodate more indicators of abiotic and biotic impacts, or if some of these impacts are best considered outside of this framework.

4.6 Conclusions

This thesis has achieved its two objectives, and presented a coherent vision of the quantification of marine resources in two methods: life cycle assessment and ecological footprint analysis. I argue that the greatest benefit of these methodologies is in their application towards sustainable boundaries, as operationalized in Chapter 3. Human society must refocus its attention on understanding our role and position within the planet's ecosystems. While this thesis demonstrates this role in the subsector of reduction fisheries, this is not the only subsector that should be examined nor the only relationship we can have to the natural world. Through using the framework of sustainable limits or a safe operating space, we can view our collective activities within the context of our planetary system. We must understand ourselves in this manner, if we are to continue to coexist in this safe operating space.

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Appendix A: Supplemental Information And Data For Global Reduction Fisheries And Their Products

Table A1. Fuel Use Intensity (FUI) Estimates and Information of Fisheries Analyzed

Species[1]	Fishing Gear[2]	FUI (L/tonne) Estimate	FUI Low	FUI High	# of Sources	# of data points	Quality of Sources
Alaska pollock	Pelagic Trawl [a]	65.1	36	85	3	10	Published; Thesis
Antarctic krill	Pelagic Trawl [b]	140.9	101	233	1	7	Published
Atlantic cod	Weighted average [3,a]	532.8	222	2724	11	90	Published and unpublished
Atlantic herring	Pelagic Trawl [c]	141.4	78	500	6	32	Published and unpublished
Atlantic herring	Seine [c]	43.4	8	153	4	19	Published
Atlantic menhaden	Seine [d]	28.7	17	42	2	16	Published
Blue whiting	Pelagic Trawl [e]	110.5	39	212	2	3	Published and unpublished
Blue whiting	Seine [e]	85	85	85	1	1	Published
California (S. American) pilchard	Seine [a]	100.4	29	217	3	14	Published and unpublished
Capelin	Pelagic Trawl [f]	102.2	45	112	2	6	Published
Capelin	Seine [f]	23.3	19	64	1	13	Published
Chilean jack mackerel	Seine [f]	10	10	10	1	1	Unpublished
Chub mackerel	Pelagic Trawl a	x	x	x	0	0	No Data
Chub mackerel	Seine [a]	x	x	x	0	0	No Data
European pilchard	Seine [a]	108.6	90	140	1	7	Published
European sprat	Pelagic Trawl e	94	94	94	1	1	Published
Gulf menhaden	Seine [d]	36.7	34	44	1	4	Published
Haddock	Weighted average [3,a]	678.7	471	1400	3	7	Published and unpublished
Japanese anchovy	Pelagic Trawl [a]	x	x	x	0	0	No Data
Japanese anchovy	Seine [a]	x	x	x	0	0	No Data
Peruvian anchovy	Seine [e]	18	10	19.5	2	7	Published and unpublished
Sandeel	Bottom-trawl [b,e]	147.1	55	204	2	8	Published and unpublished

1. Italicized entries were not included in final analysis.
2. Fishing gear was determined from multiple source agreement a. FAO Fisheries & Aquaculture 2014. b. Parker and Tyedmers 2012. c. Tyedmers 2004. d. Ruttan and Tyedmers 2007. e. Hasan and Halwart 2009. f. Tyedmers 2001.
3. Bottom Trawl, gillnets, and lines based on these fisheries contribution to total catch of their major fishing nations.

Table A2: Carbon footprint and marine footprint of fishmeal and oil products

Species	Ecosystem	Gear	Carbon Footprint				Marine Footprint			
			Carbon Footprint				Meal Footprint			
			Meal		Oil		Meal		Oil	
			(CO_2-e/ tonne)	Rank	(CO_2-e/ tonne)	Rank	(km^2/ tonne)	Rank	(km^2/ tonne)	Rank
Alaska Pollock	West Bering Sea	Pelagic Trawl	1.14	8	2.43	9	2.92E-1	13	6.23E-1	13
Alaska Pollock	Gulf of Alaska	Pelagic Trawl	1.14	8	2.43	9	1.26E-1	10	2.70E-1	11
Antarctic Krill	Antarctic Sea	Pelagic Trawl	3.72	15	6.13	15	7.32E-2	5	1.21E-1	5
Atlantic Cod	Icelandic Shelf	Weighted average	4.52	16	9.64	16	1.14E+0	16	2.43E+0	16
Atlantic Cod	Barents Sea	Weighted average	4.52	16	9.64	16	1.67E+2	19	3.56E+2	19
Atlantic Herring	North Sea	Pelagic Trawl	1.36	11	2.48	10	7.79E-2	7	1.42E-1	6
Atlantic Herring	North Sea	Seine	0.60	4	1.09	3	7.79E-2	7	1.42E-1	6
Atlantic Herring	Icelandic Shelf	Pelagic Trawl	1.36	11	2.48	10	1.04E-1	9	1.90E-1	8
Atlantic Herring	Icelandic Shelf	Seine	0.60	4	1.09	3	1.04E-1	9	1.90E-1	8
Atlantic Menhaden	Northeast US	Seine	0.61	5	1.25	5	6.93E-3	1	1.43E-2	1
Blue Whiting	North Sea	Pelagic Trawl	1.88	13	4.11	14	7.33E-1	15	1.60E+0	15
Blue Whiting	North Sea	Seine	1.55	12	3.38	12	7.33E-1	15	1.60E+0	15
Blue Whiting	Norwegian Sea	Pelagic Trawl	1.88	13	4.11	14	2.45E+0	17	5.36E+0	17
Blue Whiting	Norwegian Sea	Seine	1.55	12	3.38	12	2.45E+0	17	5.36E+0	17
California Pilchard	California Current	Seine	0.72	6	1.48	6	7.47E-2	6	1.55E-1	7
Capelin	Icelandic Shelf	Pelagic Trawl	1.34	10	2.63	11	1.32E-1	11	2.59E-1	10
Capelin	Icelandic Shelf	Seine	0.56	3	1.11	4	1.32E-1	11	2.59E-1	10
European Pilchard	Canary Current	Seine	0.74	7	1.52	7	2.28E-1	12	4.72E-1	12
European Sprat	North Sea	Pelagic Trawl	1.16	9	2.28	8	5.91E-2	4	1.16E-1	4
Gulf Menhaden	Gulf of Mexico	Seine	0.53	2	0.77	1	1.98E-2	2	2.89E-2	2
Haddock	Icelandic Shelf	Weighted average	5.57	17	11.90	17	6.18E-1	14	1.32E+0	14
Haddock	Barents Sea	Weighted average	5.57	17	11.90	17	5.99E+1	18	3.45E+1	18
Peruvian Anchovy	Humboldt Current	Seine	0.48	1	1.06	2	1.00E-1	8	2.24E-1	9
Sandeels	North Sea	Bottom-trawl	2.01	14	3.94	3	3.79E-2	3	7.45E-2	3

Table A3. Underlying information for estimates including sources and assumptions for fuel use sources, meal and oil yields, amount destined for reduction

Species	FUI	Meal and Oil Yields Source and Specificity	Percent destined for reduction	2008-2012 Average Landings (000s tonnes)	Processing Location
Alaska pollock	#	Alantic cod[b, e]	X	N/A†	USA- Alaska
Antarctic krill	#	Speciesa	70[3]	173	On-board
Atlantic cod	°	Speciese	X	N/A†	Norway
Atlantic herring	#	Speciese	<30[3]	2165	Norway
Atlantic menhaden	#	Speciesc	1002	209	USA
Blue whiting	#	Speciesa	>95[2,3]	592	Norway
California pilchard	*	European pilchard[b, e]	255	640	Mexico
Capelin	#	Speciese	95[2]; 50-100[3]	597	Norway
Chilean jack mackerel	#	Speciese	25[5]	905	Chile
Chub mackerel	*	Chilean jack mackerel[e]	25[5]	1700	Chile
European pilchard	#	Species[b, e]	50[3]	1122	Norway
European sprat	#	Species[e]	<50[3]	565	Norway
Gulf menhaden	#	Species[a]	100[4]	503	USA
Haddock	°	Alantic cod[b, e]	X	N/A†	Norway
Japanese anchovy	*	Peruvian anchovy[a]	67[2]	1233	China
Peruvian anchovy	#	Species[a,e]	>99[1,2]	6309	Peru
Sandeel	*	Species[e]	1003	341	Norway

FUI: # Species and gear specific estimate of FUI. °Based on weighted average of most used fishing techniques of bottom-trawl, gillnets and lines *Species specific estimate not available and is based on target group, except sandeel which are reported at the family level. Meal and Oil yields: a. Parker and Tyedmers 2012. b. Bimbo, 2015 (in press). c. Winther et al., 2009. d. Chiu et al., 2013. e. Ytrestøyl et al., 2011. Percent destined for reduction references: 1. (Tacon & Metian, 2009); 2. Wijkström, 2012; 3. Hasan & Halwart, 2009; 4. Seafish, 2011 5. No available data on amount destined for reduction but included in Wikjström (2012) and Tacon& Metian (2009) as common reduction species, so a conservative estimate of 25% was assumed. X: DHC Fishery. Landings: All landings are an average of the total landings of the species for the 2008-2012 period (FAO, 2014b). †: of DHC fisheries was not necessary because they were not included in advanced stages of analysis with respect to ecosystem appropriation and contribution of reduction fisheries to global carbon dioxide equivalent emissions. Processing Location: Specific country where it was processed that is applicable to processing technology and electricity type (see Table A4).

Table A4. Species Characteristics

Species	Trophic Level	Meal Yield[a]	Oil Yield[a]	Meal energy Density (MJ/kg)[b]	Source Ecosystem(s)[c]
Alaska pollock	3.5	170	17	18.4	WBS, GA
Antarctic krill	2.2	160	0.8	23.84	AS
Atlantic cod	4.4	170	17	18.4	IS, BS
Atlantic herring	3.2	200	110	22.1	NS, IS
Atlantic menhaden	2.3	240	50	19.1	NEUS
Blue whiting	4.0	197	19	18	NS, NWS
California pilchard	2.4	230	180	19	CC
Capelin	3.2	165	77	20	IS, BS
European pilchard	3.1	230	180	19	CNC
European sprat	3.0	188	79	20	NS
Gulf menhaden	2.2	210	160	19.1	GM
Haddock	4.1	170	17	18.4	IS, BS
Sandeel	2.7	215	47.5	20	NS

a. Meal and oil yields are in kilograms per tonne of round fish. b. Species types and meal energetic density was sourced from (FAO, 1986; Parker & Tyedmers, 2012b; Sauvant et al., 2004). c. Antarctic Shelf: AS; Barents Sea: BS; California Current: CC; Canary Current: CNC; East China Sea: ECS; Gulf of Alaska: GA; Gulf of Mexico: GM; Humboldt Current: HC; Icelandic Shelf: IS; North Sea: NS; Northeast US Continental Shelf: NEUS; Norwegian Sea: NWS; West Bering Sea: WBS.

Table A5. Processing Energy Information

Type	Corresponding Species [a]	Fuel Source	Amount (MJ/ tonne wet fish)	Carbon Intensity (CO2-e kg /MJ)[b]	Source/Date
Average Natural Gas Plant	Atlantic cod (N), Atlantic herring (N), Atlantic menhaden (U), blue whiting (N), California pilchard (M), capelin (N), European pilchard (N), European sprat (N), Gulf menhaden (U), haddock (N), sandeel (N)	Natural Gas	1486	0.0713[b]	Pelletier (2007), LCA of Food (2000), and Tyedmers (2000).
		Electricity	92	M: 0.149[c] N: 0.00108[c] U: 0.152[c]	
Peruvian anchovy Natural Gas	Peruvian anchovy (P)	Natural Gas	1518	0.0713[b]	Avadi et al. (2015) and Avadi (pers. comm.)
		Electricity	74	P: 0.0518[c]	
Diesel only	Alaska pollock (N/A)	Diesel	2212	0.0889[d]	Assumption based on diesel being used in place of other fuel oils, and for generation of electricity in remote processing facilities.
On-board Processing	Antarctic krill (N/A)	Marine diesel oil	418	0.0700[e]	Parker (2011)
		Intermediate fuel oil	1507	0.0887[e]	

a. Species (Country of processing): Mexico: M; Norway: N; Peru: P; USA: U. b. (EcoInvent process 'Heat, natural gas, at industrial furnace >100kW/RER U'. c. World Resources Institute 2011. d. US Life Cycle Inventory for Diesel, combusted in industrial boiler. e. Calculated from Parker (2011) thesis. g. Calculated based on an average of 3 natural gas reduction plants from previously published data. The data showed remarkable similarity across regions of USA, Canada, and Denmark.

Table A6. Reduction Fisheries Considered and Inclusion Criteria

Species[a]	Average Landings (tonnes; 2008-2012)	Percentage for Reduction or Comment on Quality	Sources of Agreement	Exclusion Reason
Peruvian anchovy	6,309,639	>99% [1]; 98% [2]	1, 2, 3, 4, 6, 7, 8, 9, 10, 11	N/A
Atlantic herring	2,165,976	European catches <30% [3]; 50% Iceland [2]	1, 2, 3, 4, 6, 7, 8, 10, 11	N/A
Chub mackerel	1,700,786	Well established food market [4]	1, 2, 3, 6, 8	Failure of Criteria C)
Japanese anchovy	1,233,833	67% China [2], 50% Japan [2]	1, 2, 4, 6, 8	Failure of Criteria C)
European pilchard	1,122,478	European catches 50% [3]	1, 2, 3, 6, 8, 11	N/A
Chilean jack mackerel	905,143	Well established food market [4]	1, 2, 3, 4, 6, 7, 8, 9, 11	Failure of Criteria C)
Atlantic Mackerel	813,155	0% [5]	3, 11	Failure of Criteria A)
California pilchard	640,061	No information found on HDC or reduction	1, 2, 3, 6, 8	N/A
Capelin	597,212	95% [3]; 100% Faroe Islands, 75% Iceland, 50% Norway, 0% Canada [2]	1, 2, 3, 4, 6, 7, 8, 9, 11	N/A
Blue whiting	592,569	>95% [3]; 95% Iceland [2]; 100% Norway, Denmark and Faroe Islands [2]	1, 2, 4, 6, 7, 8, 9, 10, 11	N/A
European sprat	565,528	European catches <50% [3]	1, 2, 3, 4, 6, 7, 8, 9, 11	N/A
European anchovy	554,666	0% [5]	2, 3, 8	Failure of Criteria A)
Gulf menhaden	503,326	100% [4]	1, 2, 3, 4, 6, 8, 9, 10, 11	N/A
Pacific herring	353,493	Well established food market [2]	2, 3, 8	Failure of Criteria A)
Sandeel	341,313	100% [3]	1, 2, 3, 4, 6, 7, 8, 9, 11	N/A
Japanese jack mackerel	228,214	Prime Food Fish [2]	2, 8	Failure of Criteria A)
Atlantic menhaden	209,921	100% [4]	1, 2, 3, 4, 6, 8, 9	N/A
Atlantic horse mackerel	209,331	European catches <20% [3]	3, 6	Failure of Criteria A)
Antarctic krill	173,335	70% [3]	9, 10, 11	N/A
Boarfish	71,510	100%	9, 11	Failure of Criteria A)
Norway pout	54,766	100% [3]	1, 2, 3, 6, 7, 8, 9, 11	Failure of Criteria A)

a. Italicized entries were not included in final analysis.

1. Tacon & Metian 2009; 2. Wijkström, 2012; 3. Hasan & Halwart, 2009; 4. Seafish, 2011; 5. FAO Species Profile, 2015; 6. Tacon 2005; 7. Seafish 2014; 8. Péron, 2010 ; 9. Bimbo 2015 (in press); 10. Parker & Tyedmers 2012; 11. Ytrestøyl et al. 2011

Table A7. Reduction Fisheries Ecosystems Considered

Species[a]	Potential Source Ecosystems [1,2]	Source Ecosystem(s) Analyzed	Reason
Antarctic krill	Antarctic Shelf	Antarctic Shelf	Only source LME 1,2
Atlantic herring	North Sea, Baltic Sea, Icelandic Shelf	North Sea, Icelandic Shelf	No robust TE for Baltic Sea
Atlantic menhaden	Northeast US Continental Shelf; Southeast US Continental Shelf	Northeast US Continental Shelf	Low landings from Southeast US Continental Shelf 2
Blue whiting	North Sea, Icelandic Shelf, Norwegian Sea,	North Sea, Norwegian Sea	Based on Norway's dominant share of the TAC for the 2012-2015 period 3
California pilchard (S. American pilchard)	Gulf of California, California Current, Pacific Central American Coastal	California Current	Lowest catches were in Pacific Central American Coastal 2
Capelin	Icelandic Shelf, Barents Sea	Icelandic Shelf	Barents Sea reported catch was 0 in 2006
Chilean jack mackerel	Humboldt Current	Humboldt Current	Large majority of catches occur here 1,2
Chub mackerel	Humboldt Current	Humboldt Current	Large majority of catches occur here 1,2
European pilchard	Canary Current	Canary Current	Large majority of catches occur here 1,2
European sprat	North Sea, Baltic Sea	North Sea	No robust TE for Baltic Sea
Gulf menhaden	Gulf of Mexico	Gulf of Mexico	Only source LME 1,2
Japanese anchovy	Kuroshio Current, Sea of Japan, East China Sea	East China Sea	No TE information on other ecosystems
Peruvian anchovy	Humboldt Current	Humboldt Current	Only source LME 1,2
Sandeel	North Sea, Norwegian Sea	North Sea	Large majority of catches occur here 2,3
Atlantic cod	Icelandic Shelf, Barents Sea, Norwegian Sea	Icelandic Shelf, Barents Sea	North Sea Cod stock is highly depleted 3
Haddock	Icelandic Shelf, Barents Sea, North Sea, Celtic-Biscay Shelf	Icelandic Shelf, Barents Sea	North Sea catches are comparatively low 2
Alaskan pollock	West Bering Sea, East Bering Sea, Gulf of Alaska	East Bering Sea, Gulf of Alaska	Lowest catches were in West Bering Sea 2

a. Italicized entries were not included in final analysis.
1. FAO Species Profile (2015); 2. Sea Around Us Project (2014); 3. European Commission (2012); http://ec.europa.eu/fisheries/documentation/publications/poster_tac2012_en.pdf