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LCA FOR ENERGY SYSTEMS AND FOOD PRODUCTS

Review and advancement of the marine biotic resource use metric in seafood LCAs: a case study of Norwegian salmon feed

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Abstract
Purpose Seafood life cycle assessment (LCA) studies have adopted the primary production required (PPR) indicator to account for the impact of these production systems (e.g., capture fisheries or aquaculture) on the ecosystems they harvest wild inputs from. However, there exists a large diversity in the application of methods to calculate PPR, and current practice often does not consider species- and ecosystem-specific factors. Here, we critically examine current practice and propose a refined method for applying the PPR metric in seafood LCAs.
Methods We surveyed seafood LCAs that quantify PPR, or its derivatives, to examine the diversity of practice. We then defined and applied a refined method to a case study of the average Norwegian salmon feed in 2012. This refined method incorporates species-specific fishmeal and oil yields, source ecosystem-specific transfer efficiencies and expresses results as a percentage of total ecosystem production that PPR represents. Results were compared to those using previously applied methods based on the literature review, and the impact of uncertainty and natural variability of key input parameters was also assessed using Monte Carlo simulation.
Results and discussion From the literature review, most studies do not incorporate species-specific fishmeal and oil yields or ecosystem-specific transfer efficiencies when calculating PPR. Our proposed method, which incorporated source species- and ecosystem-specific values for these parameters, provides far greater resolution of PPR than when employing global average values. When alternative methods to calculate PPR were applied to marine inputs to Norwegian salmon feeds, resulting PPR values were similar for some sources of fishmeal and oil. For other species, such as Atlantic herring from ecosystems with low transfer efficiencies, there was a large divergence in resulting PPR values. For combined inputs to Norwegian salmon feeds in 2012, the refined method resulted in a total PPR value that is three times higher than would result using the currently standard method signaling that previous LCA research may have substantially underestimated the marine biotic impacts of fishery products.
Conclusions While there exists a great diversity of practice in the application of the PPR indicator in seafood LCA, the refined method should be adopted for future LCA studies to be more specific to the context of the study.

Keywords Aquaculture · Biotic resource use · Fisheries · LCIA · Primary production required · Salmon feed

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1 Introduction

As human consumption of Earth's resources continues to grow at unprecedented rates (Foley et al. 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 and metrics has been developed for this purpose, which are underpinned and motivated by a range of theoretical backgrounds. An increasingly popular framework is life cycle assessment (LCA), intending to quantify the scale of material and energy resource requirements and resulting waste stream implications and potential environmental impacts associated with the provision of a product from "cradle-to-grave" (ISO 2006). However, while biophysical accounting tools may quantify many environmental impacts, quantifying the dependence and impact of products on biotic or living resource is still underdeveloped.

Techniques for understanding and measuring the extent of human utilization of, and impact on, living resources are diverse and reflect differences in analysts' motivation and disciplinary backgrounds. Ecological impacts have been quantified through measures of land use change (e.g., Lindeijer et al. 2002; Foley et al. 2005), assessment of ecosystem and biodiversity impacts after human activities (e.g., Chapin et al. 2000), or as a loss of ecosystem services (e.g., Worm et al. 2006). Direct human use of biotic resource inputs to aquaculture have been quantified through their energy content: examples include energy return on investment (Tyedmers 2001; Troell et al. 2004; Pelletier et al. 2011; Draganovic et al. 2013), emergy analysis (Wilfart et al. 2013), total biomass used such as fish-in/fish-out ratios used in aquaculture input-output analysis (Shepherd and Jackson 2013), the primary production required (PPR) to produce said biomass at a trophic level (TL) above primary producers as discussed below (or human appropriation of net primary production; Vitousek et al. 1986; Pauly and Christensen 1995), and measures of the area of functioning ecosystem required to sustain production of biotic resources and for waste assimilation, referred to as ecological footprints (Wackernagel and Rees 1996; Folke et al. 1998).

Early seafood LCAs did not account for ecological impacts or use of biotic resources in a nuanced way despite these production systems' profound dependence on these resources. Efforts to develop indicators of relevant ecological impacts for fisheries and aquaculture have included accounting for the area of seafloor damaged by bottom-trawling gear (Nilsson and Ziegler 2007), by-catch of vulnerable or endangered species (Hornborg et al. 2013b), discarding of non-target species into the ocean (Vázquez-Rowe et al. 2012), harvest and change in population size of target species (Langlois et al. 2014; Emanuelsson et al. 2014), or have proposed ways to assess biodiversity impacts and local eutrophication effects from salmon farming (Ford et al. 2012). Though each of these approaches to addressing ecological impacts of seafood systems have application in certain contexts, the most commonly adopted ecologically based measure in LCA has been PPR.

The use of PPR to indicate 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 methodological basis of the quantification of PPR research follows the logic and methods from Pauly and Christensen's (1995) paper on the PPR to sustain global fisheries (Eq. 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 TL 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.

$$PPR = \frac{C}{M} * (1/TE)^{TL-1} \tag{1}$$

(Adapted from Pauly and Christensen (1995))

Where C is the mass of catch, M is the ratio of wet weight biomass to carbon content of the species of interest, TE is trophic transfer efficiency of the source ecosystem expressed as a whole number , and TL is the trophic level of the species of interest. This method yields the mass of carbon originally derived from photosynthesis that is required to support the production of 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. In addition to marine products, the PPR indicator in LCA has also been used to quantify the primary production required to sustain terrestrial product inputs, such as crops and livestock products, to aquaculture (Pelletier et al. 2009). Here, we set out to review and propose refinements to current practice within LCA to improve the quantification of PPR to sustain marine living resource utilization.

1.1 Aim

This article has two main objectives: (1) to review use to date of the PPR method in seafood LCAs and (2) to propose methodological improvements of the current marine PPR metric used in seafood LCA. To illustrate the application of these proposed changes and the difference between common practice and the proposed refined method, we model marine resource dependencies of marine-derived inputs to Norwegian salmon aquaculture feed production in 2012 as a case study.

2 Literature review of the PPR indicator in seafood LCA

A literature review was conducted of seafood LCAs published in English that included the PPR indicator or some variation of it. Google Scholar was used as the primary database with the following search terms employed: life cycle assessment, LCA, biotic resource use, net primary production, and primary production required. The studies were then analyzed for their subject, term usage, functional unit, allocation practice, reason for quantification, and modeling undertaken to distinguish patterns, similarities, and divergences of practice.

2.1 General patterns in the quantification of primary productivity requirement to date

There are a few major patterns in the use of terms and rationale motivating the use of PPR in seafood LCAs (Table 1). Terminology differs when addressing PPR in LCA practice. The term "biotic resource use," or BRU, has been applied most frequently (11 of 26)[1] in both aquaculture and fishery LCAs. The term "net primary productivity used," or NPPU, is as widely used (11 of 26) but only in relation to aquaculture studies. The more specific and arguably progenitor term "primary production required," or PPR, has been used less frequently (3 of 26) and exclusively for fishery studies. Hereafter, the term PPR is used because of the specificity with which it refers to what it quantifies, rather than the ambiguity of the biotic resource use term.

Most studies justify the inclusion of this indicator by the desire to quantify the amount of primary production necessary to form a certain product (Pelletier et al. 2009; Hornborg et al. 2012) and the resulting pressure this places on ecosystems (Pelletier and Tyedmers 2007). This justification was also expressed as "a biotic resource…being unavailable for other purposes," in that if humans appropriate this 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 et al. 2009; Jerbi et al. 2012), the motivation for the use of this indicator is to quantify reliance and potential impact on ecosystems, which result from the harvesting of the products of primary production. These other two studies rationalized use of 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).

2.1.1 Similarities of use

Despite the variation in the descriptors employed, there is a broad similarity among studies. All studies included PPR of fishery landings and, in aquaculture, also agricultural and livestock products when used. Only two studies explicitly included bait (Vázquez-Rowe et al. 2012; Vázquez-Rowe et al. 2014), and three studies included a measure or accounted for the discarding of non-target species back into the ocean (Vázquez-Rowe et al. 2012; Hornborg et al. 2012; Vázquez-Rowe et al. 2014). The PPR methodology by Pauly and Christensen (1995) was initially adapted for LCA by Papatryphon et al. (2003) to quantify the PPR of cultured trout (*Oncorhynchus mykiss*) production. This original adaptation, and all subsequent studies, adopted the estimated 10 % average for TE (Pauly and Christensen 1995). Agricultural inputs into aquaculture feeds are accounted for through the crops' carbon content (Papatryphon et al. 2003). Accounting for the PPR of livestock ingredients (mainly poultry byproducts) occurred in only three studies (Pelletier and Tyedmers 2007; Pelletier et al. 2009; McGrath et al. 2015)

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 et al. 2014). In the case of feeds, the nutritional quality of inputs and the final composite feed can vary greatly. One study used instead the protein content of resulting products as the functional unit (Vázquez-Rowe et al. 2014).

Virtually, none of the studies (2 of 18) reported whether species-specific meal and oil yields were employed in their methods, but typically expressed them as a percentage or fraction of meal or oil mass to round fish mass. This makes it difficult to discern if average or species-specific yields were used in the calculation of PPR. There can be a large variance in meal and particularly in oil yields 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 fishmeal reduction plants themselves (Ytrestøyl et al. 2011; Parker and Tyedmers 2012a; Hognes et al. 2014). The use of species-specific fishmeal and oil yields are an advancement over average yields, although there are many other factors that influence yields (Parker and Tyedmers 2012a). More broadly, relevant assumptions of product yield for agricultural and livestock products were also not included in most studies.

2.1.2 Dissimilarities of use

Besides using varying terminologies for PPR, the analyses reviewed also differed in several methodological choices within the LCA method, including allocation of inputs and impacts among outputs of multifunctional production systems and modeling of uncertainty. In LCAs of multiple output systems, when allocation needs to be done, a basis for the division of environmental burdens and inputs among coproducts has to be chosen. The basis for allocation is a topic highly debated in the literature (Ayer et al. 2007; Weidema and Schmidt 2010; Weinzettel 2012; Pelletier et al. 2014) and can affect results considerably, including PPR. In the reviewed studies, economic value of coproducts was the most common basis of

[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.

Table 1 Methodological choices and characteristics of PPR in seafood LCAs arranged chronologically

Subject (reference)	Term	Method author[a]	Allocation	Explicit reason for quantifying PPR	Modeling[c]
Aquaculture trout (1)	BRU	(2)	E	None given	–
Salmonid feeds (2)	NPPU	[b]	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	(4)[b]	M	"Limits global fisheries yield…. [a] proxy for trophic interactions such as depletion of top predators" (p. 1194)	SA, FU
Antarctic Krill fishery (13)	BRU	[b](2, 3, 7)	N	None given	SA, AC, SM
Multiple fisheries (14)	BRU	[b]	–	None given	–
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	[b]	M	None given	SA
Global fisheries (19)	NPP	(2)	–	"Change in biomass production capacity" (p. 66)	–
Peruvian anchovy products (20)	BRU	[b]	M	Primary production consumed by organism	–
European pilchard fishery (21)	BRU	(2)	E	None given	SA, FU, SM
Sardine Fishery (22)	PPR	[b]	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	[b]	–	None given	–
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. 2014a; (19) Langlois et al. 2014; (20) Avadí et al. 2014b (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

BRU biotic resource use, *NPPU* net primary production use, *PPR* primary production required, *SA* Sensitivity Analysis, *AC* Allocation Choice, *SM* Scenario Modeling, *FU* Functional Unit choice, *MC* Monte Carlo simulation, *E* economic, *N* gross nutritional content, *M* mass, – no method established or necessary

[a] Method author is the author (or group of authors) cited in the paper when describing the PPR method used

[b] indicating the method established by Pauly and Christensen (1995)

[c] Modeling choices undertaken that directly affected the calculation of PPR were separated into different elements

allocation (12 of 22), and energetic content of coproducts was second most common (6 of 22). Though many studies do not include an explicit rationale for their allocation choice, the primary use of the products (energy for feeds or food) or the driver of the system (economic revenue or profit) are given as reasons for the allocation method (Boissy et al. 2011; Parker and Tyedmers 2012b). While mass allocation was least common in the reviewed studies (4 of 22), it is the recommended method in the PAS-2050 standard on assessing lifecycle greenhouse gas emissions of seafood products (BSI 2012).

Approximately half of the studies (14 of 26) explore PPR-related 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). Sensitivity to allocation method choice was modeled in 6 of 22 studies. Other sensitivity analyses included modeling functional unit choice (3 of 26) and marine-input attributes such as TL and fishmeal and fish oil yields (2 of 26). One study employed Monte Carlo simulation to understand the impacts of describable uncertainty of input parameters on model outcomes.

2.2 Limitations of current PPR method

As noted in Sect. 2.2, there is a diversity of practice of allocation decisions and assumptions that are not necessarily well described but that can greatly influence results. Lack of transparency of practice is a current challenge that must be addressed to ensure adequate conclusions can be drawn from results. The similarities and dissimilarities outlined above demonstrate two alternative methods are currently in common use: (i) a "standard method," in which a 10 % TE is assumed to apply across all TLs and "average" meal and oil yields (of between 21–24 % for meal and 5–10 % for oil) are applied to all marine products, and (ii) a "yield-specific method," in which a 10 % TE is again assumed to apply universally but species-specific yields of meal and oil are applied.

A significant limitation of both these approaches is the use of an average TE value of 10 %, irrespective of source ecosystem or species interactions, as this creates constraints to conclusions and limits the comparability of results. The value of 10 % was originally assumed (Slobodkin 1962; May 1976) but was not based concretely in evidence (Gmel 2010). Later studies supported this 10 % value as a global average of marine ecosystem TE through reviews of published and unpublished time dynamic ecosystem models completed through the EcoSIM software (Pauly and Christensen 1995), although this value has been contested by Baumann (1995) and Ryther (1969). Other studies based on EcoSIM models, however, have found that TE is highly variable among aquatic ecosystems and ecosystem types (Heymans et al. 2011). Transfer efficiency has been found to range between 3.51 and 38.1 % (average of 11.9 % and standard deviation of 5.45, Fig. 1) and can vary over time based on changing ecosystem interactions (Libralato et al. 2008). For modelling purposes, we adopt the common assertion that TE does not vary between TLs within an ecosystem, which is supported by EcoSIM models (Pauly and Christensen 1995), even though this assertion is contested by others. Given the centrality of the TE value to estimating PPR, these differences in input values can have a substantial impact on outcomes (Parker and Tyedmers 2012a).

Furthermore, it is unclear in many LCA 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 (Tacon and Metian 2008; Jackson 2009). While fish oil yields can vary substantially within species, accounting for geographic,

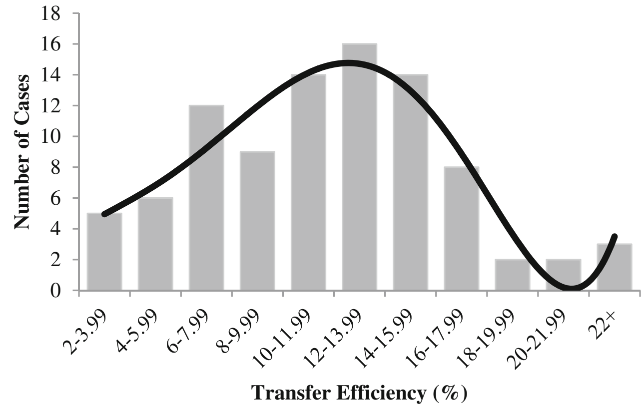

Fig. 1 Distribution of transfer efficiency values of 91 ecosystems surveyed in Libralato et al. (2008)

seasonal, or interannual variability is currently not possible with publicly available data.

When parameterizing models of PPR, it is typical that analysts employ discrete values. However, all requisite inputs, such as primary production and TE, 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; Libralato et al. 2008; Chavez et al. 2011). The use of static input values fails to recognize significant variability between systems in some cases (Parker and Tyedmers 2012a, 2012b) and can potentially result in misleading conclusions. Incorporating system-specific values and their associated uncertainties would avoid this source of error and better reflect natural variability.

Lastly, current efforts to estimate PPR do not provide a measure of scale of impact relative to total ecosystem productivity. Alternatively, 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 potential overfishing (Coll et al. 2008). Previous research has pointed to the increasing pressure on biodiversity that results from humans appropriating primary production (Vitousek et al. 1986; Foley et al. 2007; Krausmann et al. 2013). Currently, little work has been done to determine thresholds for sustainable appropriation as noted by Bishop et al. (2009) and Langlois et al. (2014), but there are some examples for marine ecosystems (Coll et al. 2008; Chassot et al. 2010). Further challenges to the broader utility of the PPR indicator will be discussed below (Sec. 5.2.2).

3 Methods

3.1 Advancement of method

To refine the PPR calculation in seafood LCAs, we made three modifications to contemporary practice, hereafter referred to as the standard method. Specifically, (1) where and when possible, large marine ecosystem (LME) source-specific TE values were applied instead of an 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 the total productivity of the source LME, the PPR to sustain provision of a biotic input was expressed as a fraction of annual total productivity of the ecosystem from which it was sourced. Taken together, this represents what hereafter is referred to as the refined method. To illustrate the outcome of the refinement, we applied it to marine-sourced inputs to Norwegian salmon feeds milled in 2012. Another variant of the standard method, hereafter referred to as the yield-specific method, which employed species-specific fishmeal and oil yield values, but average TE, was also defined because from the literature review, it was unclear if previous studies were using species-specific or average meal and oil yield values. We compared results of the analysis of PPR to sustain marine inputs to Norwegian salmon feeds using the refined method with those using the standard and yield-specific methods. In addition to the main analyses using the three methods, two sensitivity analyses and an uncertainty analysis were performed (Sects. 3.2.3 and 3.2.4).

3.1.1 Characterization of inputs to Norwegian feeds

Norway is the largest producer of cultured Atlantic salmon (*Salmo salar*) globally (Ytrestøyl et al. 2011). Data on inputs into Norwegian salmon feeds for 2012 were solicited from the three largest producers of salmon feed in Norway. Data were solicited in accordance with previous LCA and resource studies of salmon production (Ytrestøyl et al. 2011; Hognes et al. 2014). The exact origin of the marine inputs was only given by one of the producers and this information was used as a proxy for all of the marine inputs. Terrestrial-based inputs were not included in our analysis because of the marine focus of our study. A mass-based functional unit of 1 t of fishmeal or fish oil feed input was used throughout all three methods.

3.1.2 Standard and refined model parameterization

The standard method used average fishmeal and oil yields associated with typical inputs to Norwegian salmon feeds of 22.5 and 9.3 %, respectively (Ytrestøyl et al. 2011). The yield-specific method and the refined method used species-specific data on fishmeal and fish oil yields from solicited information, previously published studies, and lipid content of the species (Hognes et al. 2014). For all three methods, fish TLs were obtained from FishBase (Froese and Pauly 2012) and applied generally on the assumption that the same species harvested from different ecosystems, and regardless of size, have the same TL value.

Energetic allocation was employed for this analysis as it reflects the underlying energetic relationships between co-products as products of ecosystems, which are made up of energetic relationships of food webs. Fish oil was applied an energetic content of 39.3 MJ/kg, regardless of source, whereas meal energy density varied by species (18.0–23.8 MJ/kg; Sauvant et al. 2004). 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 % (20.0 MJ/kg); sardine-type fish with a meal crude protein content of 65 % (19.0 MJ/kg); and white-fish meal derived from byproducts with a meal crude protein content of 65 % (18.4 MJ/kg; FAO 1986; Sauvant et al. 2004).

Ecosystem-specific TE values were obtained from a summary of EcoSIM models; these models demonstrate substantial variance from the global average of 10 % (3.51 to 16.5 % for ecosystems modeled here) (Libralato et al. 2008). Data on LME primary production were obtained from the Sea Around Us Project (Sea Around Us Project 2014). The primary production estimates are derived from satellite data (from the 10-year period of 1998 to 2007) that calculates primary production from chlorophyll pigment concentration (Platt and Sathyendranath 1988).

3.1.3 Sensitivity analysis

A sensitivity analysis examines the implications of assumptions and methodological decisions made throughout the analysis on final results. We evaluated (a) the choice of functional unit, as this has previously been shown to have a large effect (Parker and Tyedmers 2012a), using 1-kg fishmeal; 1-kg fish oil; 1-kg round fish; and 100 GJ of meal and oil combined and (b) the influence of three input parameters (TE, TL, and fishmeal yields) on the PPR by altering these values by ±10 % for Atlantic herring (*Clupea harengus*) from the North Sea, individually and in various combinations. While 10 % does not represent the actual variance of these input parameters, this analysis demonstrates their relative contribution to, and influence on, the final results, and thus which parameters affect the PPR results most.

3.1.4 Uncertainty analysis

The uncertainty of the PPR calculation was modeled through a Monte Carlo analysis to generate a distribution of results. Monte Carlo analysis utilizes the range within input parameters to model distributions over a given number of iterations. Each iteration randomly selects a value from within each parameter based on the mean, standard deviation, and truncations entered 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 for 10,000 iterations of PPR.

To model uncertainty for Monte Carlo analysis, a justifiable range, standard deviation, and mean value for each parameter must be available. In this study, we aimed in the refined method to use specific values rather than average values, which consequently had less data available for them. Therefore, species-specific fishmeal and oil yields and ecosystem-specific transfer efficiencies were modeled as discrete parameters in the uncertainty analysis, while TL 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. The TL range was limited by a lower bound of 2, because TL lower than 2 indicates autotrophy, and an upper bound of 5, representing a rarely reached TL of apex predators.

4 Results

4.1 2012 Norwegian salmon feeds

The feed producers surveyed accounted for approximately 95 % of all salmon feed produced in Norway in 2012. Approximately 66 % of the mass of all feeds was derived from crops, while the balance was marine-derived ingredients or microingredients (Table 2). Of marine inputs, 23 % of the fish oil and 32 % of the fishmeal were derived from byproducts of various fisheries that were treated as coproducts of these systems in this analysis, rather than environmentally “free” products because they are not the primary product or the most valuable product of the fishery (Tables 2 and 3). All marine inputs and their species and ecosystem properties are detailed in Table 3. All feed producers reported all marine inputs at the species level; however, only one reported the associated source ecosystem. For some inputs, source ecosystem was safely assumed (e.g., Peruvian anchoveta (*Engraulis ringens*) from the Humboldt current). In other cases, it was derived from the information provided by other feed producers. The microingredients are mainly composed of phosphate substances such as monocalcium phosphate, vitamins, minerals, pigments, and amino acids (Table 2). The total mass of marine inputs for salmon feed, used in processed form (i.e., fishmeal and fish oil) by the three companies was 488,702 t in 2012. Applying the industry average economic feed conversion ratio for Norway in 2012 of 1.2 (Hognes et al. 2014), the total feed including terrestrial inputs would have produced approximately 1,314,000 t of live-weight salmon (Table 2).

4.2 Comparison of standard and refined methods

Unsurprisingly, the effect of the refined method on PPR is most pronounced for ecosystems with transfer efficiencies deviating most from the average of 10 %, as is the case for the Icelandic Shelf (Fig. 2) or the Barents Sea (Fig. 3). The TE effect is large because TE has an exponential effect on PPR. The importance of source ecosystem transfer efficiency on results using the refined method is illustrated by looking at the PPR required to sustain provision of Atlantic herring from three North Atlantic LMEs (Fig. 2). Atlantic herring sourced from the Icelandic Shelf is the most efficient because of its high TE (Table 3), while herring from the Norwegian Sea is the least efficient having the lowest TE of the three source ecosystems and a resulting PPR that is ~20 times that of herring derived from the Icelandic shelf (Table 3, and Fig. 2). This difference is obscured by the standard and yield-specific methods, which do not incorporate this difference, and perform almost identically to each other (Fig. 2). A similar effect can be observed for the use of average yields (22.5 and 9.3 % for meal and oil, respectively), although with less pronounced effects. PPR estimates of species that have yields that deviate most from the average, for example Antarctic krill (*Euphausia superba*) and South American pilchard (*Sardinops sagax*), are affected most by the refined method.

Using the refined method, a great variance of PPR is revealed across the sources of fishmeal and oils included in Norwegian salmon feeds (Table 3), but there are easily understood drivers of these differences. Meals and oils from high TL species and/or species sourced from ecosystems with low TE have the largest PPR per tonne of meal or oil produced (Table 3). For species with low fishmeal and oil yields, the pattern of PPR results is less clear; with the refined method, values of PPR were higher (e.g. blue whiting (*Micromesistius poutassou*) and Barents Sea capelin (*Mallotus villosus*)), but this pattern was difficult to separate from the relationship of TL and TE. However, the yield-specific method performed much closer to the standard method, which indicates the overwhelming influence of ecosystem-specific TE on results. Overall, the refined method reported a PPR that is over three times as large as that generated by the standard method for the total mass of marine-derived ingredients to Norwegian feeds milled in 2012 (Fig. 4).

4.3 PPR to sustain marine inputs to 2012 Norwegian salmon feed

The total marine PPR is 42.2 and 11.4 t of carbon per tonne of marine inputs to feed produced for the refined method and standard method, respectively. Applying the refined PPR method, the total PPR for the marine portion of the 2012 Norwegian salmon feed (4.89E+05 t of meals and oils) is 6.66E+07 t of carbon (Table 4). The Humboldt Current is the source of a large portion of marine inputs into the feeds modeled (over 37 % by mass), but these removals represent a relatively small percentage of total annual primary production of that source ecosystem. In contrast, meals and oils from the Norwegian Sea, with a relatively low TE and low annual primary production (Tables 3 and 4, respectively), represent

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

Ingredients	Mass (tonnes)	Percentage of feed (%)	Notes
Marine oils	182,362	12	23 % derived from byproducts
Marine meals	306,340	19	32 % derived from byproducts
Crop meals	617,032	39	63 % of this is 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 t

a much greater percentage of primary production appropriated, while only providing 13 % of marine inputs by mass to the salmon feeds modeled (Table 4). The fact that individual fishmeals and oils from low TE ecosystems (e.g., Barents Sea and Norwegian Sea) have high PPR, in turn, increases the total ecosystem PPR when multiple meals and oils are sourced from a single ecosystem such as the Norwegian Sea (Table 4).

4.4 Sensitivity analyses and effect of parameters

PPR values were found to be highly sensitive to the choice of functional unit, particularly in relation to the meal and oil yields of the species being assessed (Fig. 5). Species with high oil yields, such as Chilean jack mackerel (*Trachurus murphyi*) and Atlantic herring, are associated with particularly low PPR values when assessed on the basis of energy or mass of oil. Conversely, species with low oil yields display a lower PPR when assessed on the basis of round weight of fish rather than by energetic content, as the former does not take poor oil yields into consideration.

Sensitivity analysis of different parameters' relative influence on results indicate that, consistent with observations made in Sect. 4.2, small variations in TL had the greatest influence of any single parameter on PPR (Table 5; 50 to 199 % of refined method result), while similarly scaled differences in TE were slightly less influential (Table 5; 81 to 126 %). Not surprisingly, when multiple parameter values were altered at the same time, resulting PPR estimates varied widely (Table 5; 39 to 294 %).

4.5 Uncertainty analysis

Results of the Monte Carlo analysis (Figs. 2 and 3) demonstrate a wide distribution of potential results based on uncertainty and natural variability in TL values used to model PPR of various species. This distribution illustrates that use of discrete values can dramatically misrepresent the PPR to sustain the provision of fishery products. Furthermore, there is often substantial overlap in the distributions of PPR values for meal and oil products derived from different species (Figs. 2 and 3). This is in sharp contrast to results of the three methods analyzed when discrete parameters are used throughout the analyses. Instead of seemingly clear differences in the PPR values of different meals and oils, when uncertainty and variability are accounted for, the relative differences in PPR can be far less clear.

5 Discussion

5.1 Implications for LCA estimates of PPR from feed production

Given the increased importance of LCA practice as a means to inform understanding of impacts of production systems as well as guiding efforts to improve systems, it is essential that all major dimensions or forms of impacts be addressed. Currently, there is a diversity of practice in how the PPR method is employed in LCA research, but consistently, it has not utilized available species- and ecosystem-specific data. We have developed and applied a refined method that can improve specificity to species and source ecosystem characteristics and improves its differentiation of inputs to production systems sourced from marine ecosystems. Applying our refined method to fishmeals and oils used in the construction of salmon feeds demonstrated that there is a substantial variation in the PPR of different fishmeal and oils. This echoes previous research in this domain (Ytrestøyl et al. 2011; Parker and Tyedmers 2012a) and reinforces the need for greater consideration of the impacts of these products. More importantly, however, our refined method provided estimates of PPR that were, in many cases, markedly at odds with and often far higher than those generated using more conventional methods of quantifying PPR. Furthermore, by expressing aggregate PPR derived from an individual LME as a fraction of that ecosystem's total annual productivity advances our understanding of the potential impact of a production system, such as Norwegian farmed salmon, on ecosystems as a whole.

Table 3 Species and ecosystem properties of marine inputs to 2012 Norwegian salmon feeds along with PPR values for resulting fishmeals and oils calculated using the refined method

Species	Ecosystem	TE (%)[a]	TL[b]	Meal Yield (t meal/ t fish)	Oil yield (t oil / t fish)	Meal energy density (MJ/kg)	Meal (t)	Oil (t)	Meal PPR (kg C/t meal)	Oil PPR (kg C/t oil)
Reduction fisheries										
Antarctic krill (*Euphausia superba*)	SO	14	2.2	0.160	0.0008	23.84	2946	–	7.29E+03	–
Atlantic herring (*Clupea harengus*)	IS	14	3.2	0.204	0.115	22.1	1987	50	2.06E+04	3.66E+04
Atlantic herring (*C. harengus*)	NWS	3.51	3.2	0.204	0.115	22.1	2233	959	4.31E+05	7.67E+05
Atlantic herring (*C. harengus*)	NS	11.6	3.2	0.204	0.115	22.1	2748	4104	3.11E+04	5.53E+04
Atlantic mackerel (*Scomber scombrus*)	NS	11.6	3.7	0.194	0.186	20.0	516	6396	6.67E+04	1.31E+05
Blue whiting (*Micromesistius poutassou*)	NS	11.6	4	0.197	0.019	18.0	5786	501	2.98E+05	6.52E+05
Boarfish (*Capros aper*)	NS	11.6	3.1	0.216	0.034	18.0	3448	540	1.68E+04	3.68E+04
Capelin (*Mallotus villosus*)	BS	11.6	3.2	0.165	0.077	20.0	24,844	15,395	5.57E+05	1.09E+06
Capelin (*M. villosus*)	IS	14	3.2	0.165	0.077	20.0	29,082	13,519	2.66E+04	5.22E+04
Chilean jack mackerel (*Trachurus murphyi*)	HC	6.6	3.5	0.194	0.186	20.0	507	–	1.77E+05	–
European sprat (*Sprattus sprattus*)	NS	11.6	3	0.188	0.079	20.0	22,518	15,909	2.41E+04	4.73E+04
Gulf menhaden (*Brevoorti patronus*)	GM	9.7	2.2	0.240	0.130	19.1	1463	2806	3.60E+03	7.41E+03
Norway pout (*Trisopterus esmarkii*)	NS	11.6	3.2	0.204	0.115	20.0	94	599	1.28E+04	5.81E+04
Peruvian anchovy (*Engraulis ringens*)	HC	6.6	2.7	0.230	0.050	18.4	101,358	78,209	3.35E+04	7.16E+04
Sandeels (*Ammodytes tobianus*)	NS	11.6	2.7	0.197	0.042	20.0	8018	2280	1.54E+04	3.03E+04
South American pilchard (*Sardinops sagax*)	HC	6.6	3.1	0.230	0.180	19.0	615	–	5.56E+04	–
By-products										
Atlantic cod (*Gadhus morhua*)	NS	11.6	4.4	0.170	0.017	18.4	2906	1191	4.39E+05	9.38E+05
Atlantic herring (*C. harengus*)	IS	14	3.2	0.200	0.040	22.1	15,951	13,535	9.71E+03	1.73E+04
Atlantic herring (*C. harengus*)	NWS	3.51	3.2	0.200	0.040	22.1	31,770	10,387	2.04E+05	3.62E+05
Atlantic herring (*C. harengus*)	NS	11.6	3.2	0.200	0.040	22.1	17,307	9014	1.47E+04	2.61E+04
Atlantic mackerel (*S. scombrus*)	NS	11.6	3.7	0.194	0.186	20.0	2418	442	3.22E+04	6.33E+04
Capelin (*M. villosus*)	BS	3.51	3.2	0.165	0.077	20.0	1511	193	2.58E+05	5.07E+05
Capelin (*M. villosus*)	IS	14	3.2	0.165	0.077	20.0	9404	689	1.23E+04	2.42E+04
Fish hydrolysate[c]	NWS	3.51	3.2	0.210	0.100	22.1	28	–	2.04E+05	–
Fish protein concentrate[c]	NWS	3.51	3.2	0.210	0.100	22.1	14,936	3850	2.04E+05	3.62E+05

SO Southern Ocean, *NS* North Sea, *IS* Icelandic Shelf, *NW* Norwegian Sea, *BS* Barents Sea, *HC* Humboldt Current, *GM* Gulf of Mexico

[a] Transfer efficiency (TE) in % from Libralato et al. (2008) sourced from ecosystem models

[b] Trophic level (TL) according to method as establish by Fishbase (1+ weighted average TL of prey as establish from diet and model studies)

[c] Both fish hydrolysate and fish protein concentrate are modeled as by-products of Atlantic herring from the Norwegian Sea

The PPR appropriated to sustain marine inputs to all Norwegian salmon feeds milled in 2012 was substantial. Moreover, those inputs derived from North Atlantic ecosystems, in particular, represented a large portion of the net primary productivity of these LMEs. However, this must be considered in light of the benefits derived from salmon production that results from this feed (Troell et al. 2014) or against potential other "uses" of this resource (Alder and Pauly 2006). The fractions of total productivity claimed from various LMEs to sustain inputs to Norwegian salmon feeds

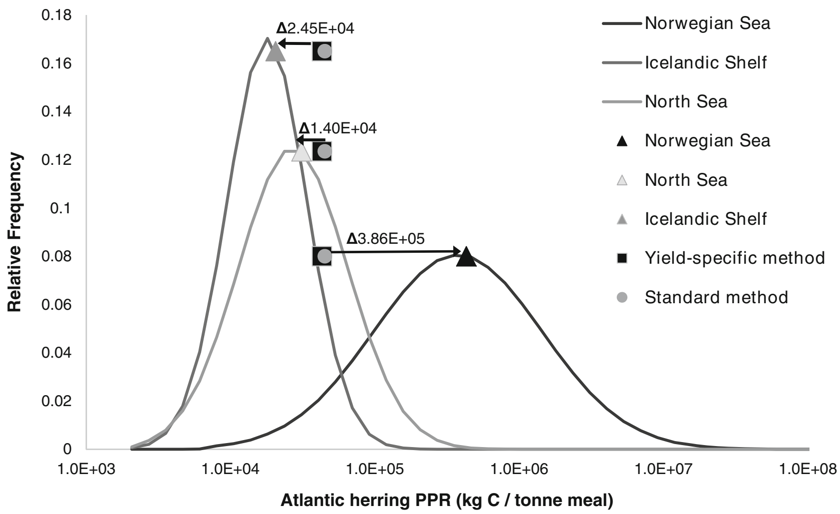

Fig. 2 PPR (kg C/ton meal) on a logarithmic scale (*X*-axis) of Atlantic herring sourced from three different ecosystems. The *curves* represent the Monte Carlo distribution of results along with their relative frequency (*Y*-axis) of occurrence in the Monte Carlo analysis for Atlantic herring modeled in three different source ecosystems. The *triangles* represent the results of the refined method for each ecosystem, which are irrespective of relative frequency. The *circles* (*black*) and *squares* (*light gray*) represent results for standard method and yield-specific method, respectively, which are irrespective of relative frequency. The *arrows* demonstrate the difference between the standard method and the refined method, with the difference between them indicated above

2012 are within the range of rates of PPR appropriated by global fishing fleets at the individual LME level (Swartz et al. 2010; Watson et al. 2014). Importantly, however, these studies did not include ecosystem-specific transfer efficiencies in their analyses but instead used the average rate of 10 % throughout. Based on our results, incorporating ecosystem-specific transfer efficiency values in future analyses of PPR for all fisheries in an LME is likely to markedly increase the total fraction claimed in certain ecosystems. This is concerning as prior studies have signaled an existing overconsumption of primary production in marine ecosystems for fisheries and aquaculture in general (Folke et al. 1998), threatening the long-term sustainability of these ecosystems (Coll et al. 2008). These factors are important to consider given the increasing demands placed on ecosystem productivity by fisheries (Chassot et al. 2010; Watson et al. 2014) and the greater scarcity of fisheries resources in less productive and less efficient ecosystems (Coll et al. 2008). Thus, including the scale of PPR appropriation from individual LMEs is an important step taken in this analysis and should be considered

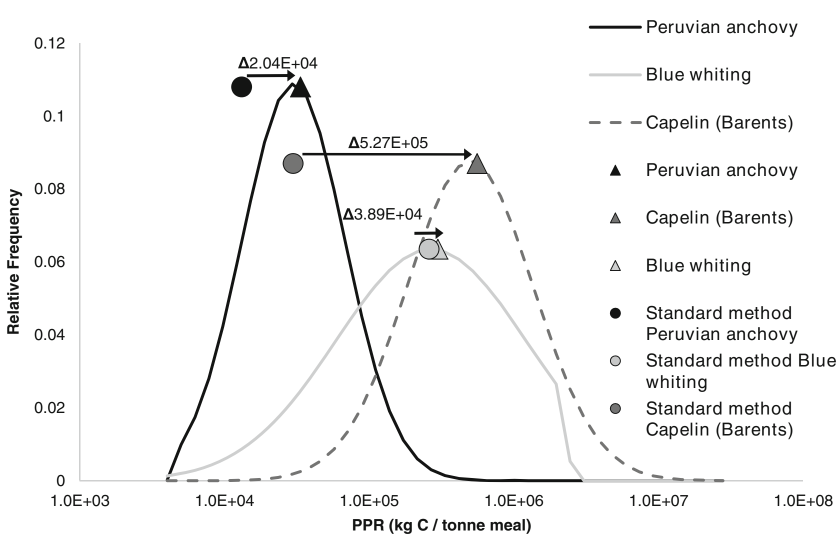

Fig. 3 PPR (kg C/ton meal) on a logarithmic scale (*X*-axis) of blue whiting, capelin (Barents Sea), and Peruvian anchovy. The *curves* represent the Monte Carlo distribution of results along with their relative frequency (*Y*-axis) of occurrence in the Monte Carlo analysis. The *triangles* represent the results of the refined method, which are irrespective of relative frequency. The *circles* represent the standard method results, which are irrespective of relative frequency. The *arrows* demonstrate the difference between the standard method and the refined method, with the difference between them indicated above

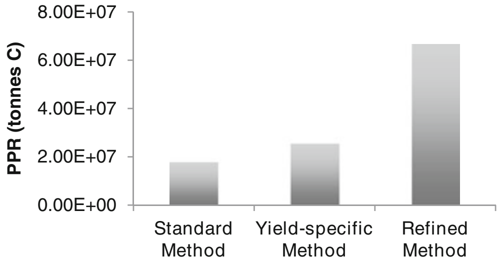

Fig. 4 PPR of marine inputs to Norwegian salmon feed by method

whenever analyses of entire aquaculture or fishery sectors are conducted.

Given concerns regarding the impact of global capture fisheries on marine ecosystems (Pauly et al. 2003; Worm et al. 2009), the emergence of fed forms of aquaculture in which compound diets are formulated from fishmeal and oils from low-trophic level organisms and crop inputs could be considered a technological innovation that improves the eco-efficiency of human food production by lowering the total PPR relative to what is needed to sustain harvest of fish, such as a salmon, sourced from the wild (Welch et al. 2010). However, this innovation shifts environmental burdens from solely marine ecosystems to terrestrial ones and places pressure on different species and environments with other ecosystem dynamics and resource scarcities. For example, substituting fishmeal with soybean meal in an aqua feed, while keeping all other conditions, such as crop system inputs, yields, and demands, constant, will inevitably require more agricultural land and thus potentially the conversion of existing natural habitats into agricultural land.

Future use of the refined method developed here would provide a more source- and species-specific quantification of PPR in LCA studies. The use of ecosystem-specific TE in the calculation of PPR for individual fishmeal and oils had a large effect relative to results generated using the standard method. The role of TE in calculating PPR is thus important when interpreting results of prior LCAs, particularly when biotic inputs are sourced from ecosystems with particularly low or high transfer efficiencies. For example, LCAs of systems that sourced inputs from the Norwegian Sea or Barents Sea using what we describe as the standard method have likely dramatically underestimated the PPR to sustain their provision given these ecosystems' low transfer efficiencies. Similarly, prior LCAs in which PPR was estimated using average fish oil yield values for species with unusually low (or high) oil yields, such as blue whiting or Antarctic krill, have also likely underestimated the PPR associated with the provision of these oils. A challenge of using more specific values for TE, as well as meal and oil yields for individual species, is that these data are based on fewer observations than are used when calculating average values. Consequently, the uncertainty associated with using source-specific values is likely to be higher but, in our opinion, this is an acceptable compromise in the interests of employing data that more accurately represent reality. What is clear though is that more knowledge regarding the dynamics and processes of marine ecosystems is needed to strengthen our understanding of their differences and similarities. Similarly, greater access to actual fishmeal and oil yield values associated with various species and under different conditions is essential if analyses of their impacts are to improve.

This article modeled uncertainty associated with estimates of marine PPR using Monte Carlo analysis when distributions for certain input parameters were used in place of discrete values. Similar to findings of Parker and Tyedmers (2012a), results of this modeling challenge the confidence with which we can make claims regarding the relative impacts of certain marine products. Reporting results solely as discrete values can give the impression of potentially major differences in the PPR between products derived from different species. In contrast, reporting distributions of results can illustrate the substantial overlap that can exist in estimates of PPR to sustain provision of products derived from some species and the clear differences between others.

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 (tons C)	Appropriation of total salmon feed marine PPR from each ecosystem (%)	Annual total primary production of source ecosystems (tons C/year)	Appropriated of total ecosystem production (%)
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 Shelf	0.82	7.79E+04	0.12	2.67E+08	0.0291
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	×

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

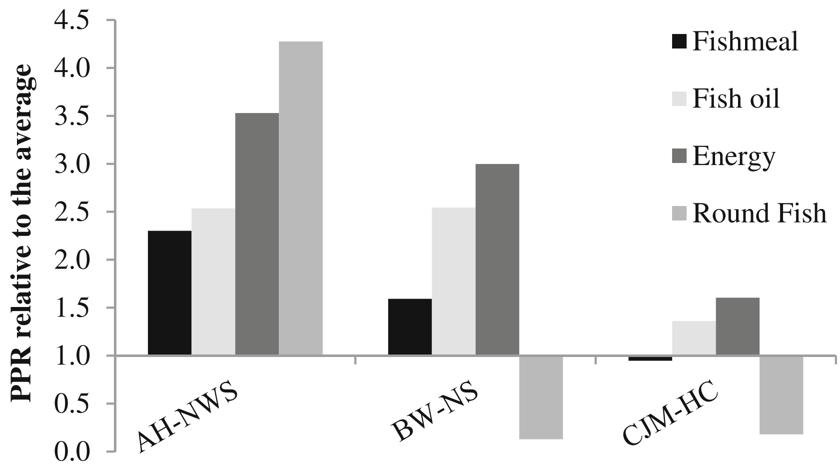

Fig. 5 PPR in comparison to the average (set at 1) PPR of all sources for various functional units of Atlantic herring from Norwegian Sea (AH-NS), blue whiting from North Sea (BW-NS), and Chilean jack mackerel from Humboldt Current (CJM-HC)

5.2 Limitations

5.2.1 Limitations of this study

A key limitation of this study is that we did not include impacts other than PPR of marine inputs into aqua feeds. Other impacts associated with seafood production, such as differential risks of stock collapse (Pinsky et al. 2011), were not accounted for. Additionally, we did not attempt to address issues associated with the quantification of PPR of crop inputs, nor did we include estimates of crop-related PPR in our case study analysis of inputs to Norwegian salmon feeds in 2012.

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 %)

A. Non-adjusted transfer efficiency			
	TL	TL +10 %	TL −10 %
Yield	100 %	199 %	50 %
Yield +10 %	93 %	185 %	47 %
Yield −10 %	113 %	226 %	57 %
B. Transfer efficiency +10 %			
	TL	TL +10 %	TL −10 %
Yield	81 %	157 %	42 %
Yield +10 %	75 %	145 %	39 %
Yield −10 %	92 %	178 %	48 %
C. Transfer efficiency −10 %			
	TL	TL +10 %	TL −10 %
Yield	126 %	260 %	61 %
Yield +10 %	117 %	241 %	57 %
Yield −10 %	143 %	294 %	69 %

Tables are presented in non-adjusted transfer efficiency (*A*); transfer efficiency increased by 10 % (*B*); and transfer efficiency decreased by 10 % (*C*)

Yield meal yield rate, *TL* trophic level, *+10 %* increase of 10 % to parameter value, *−10 %* decrease of 10 % to parameter value

This study is subject to the known uncertainty and natural variability of the input parameters modeled. 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 "true" data to be used for many parameters (yields, primary production, TE, and TL). These values are subject to natural variability and uncertainty, yet sufficient data regarding many key parameters, e.g., ranges, standard deviations, and means, are not currently available to properly parameterize these values in a Monte Carlo analysis.

Further challenges that are not addressed in depth in this paper relate to the broader utility of PPR as an indicator of ecological pressure on ecosystems more generally; a brief discussion on the topic follows.

5.2.2 Limitations of the PPR concept

A major benefit of the use of PPR in LCA is that it allows for various biotic inputs (e.g., agricultural, marine, livestock) to be measured in a single unit of mass of carbon appropriated. This indicator also has the advantage of demonstrating the increased absolute and relative dependence on ecosystem resources that occurs through harvesting higher TL species or when sourcing from ecosystems with either low TE or lower productivity. Thus, this indicator can be used to illustrate the limited capacity of marine ecosystems to produce fish biomass and that the appropriation of this biomass by humans is a direct exporting of energy from these food webs. However, the PPR measure as it is currently used in seafood LCAs does not address (i) if the overall appropriation of primary production by society is ultimately sustainable at either an ecosystem or global level (Coll et al. 2008); (ii) the status of targeted and associated populations and their potential risk of collapse or even extinction (e.g., Hutchings and Reynolds 2004); (iii) the risk of cascading effects on ecosystem structure and function from targeting key functional groups (e.g., Smith et al. 2011); or (iv) the fate and consequences of PPR associated with discards (that are not removed from the ecosystem). Even though

PPR is frequently used in seafood LCAs to gauge impact associated with marine resource use, it is insensitive to impacts on sensitive species, which when assessed separately, may even send contrasting signals to PPR (Hornborg et al. 2012; Hornborg et al. 2013a). 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 can only provide a broad indication of impact at an ecosystem level and pressure from human appropriation of this limited resource when contextualized within total ecosystem productivity.

This article has solely treated PPR as an indicator of human pressure on ecosystems; PPR is only one measure of this and is a coarse one. PPR thus speaks neither to the sustainability of harvesting fish stocks as in traditional fishery single-stock management methods (Emanuelsson et al. 2014), nor to the impact of removing species in certain TLs from these ecosystems as may be considered in an ecosystem-based management context. Recent research has illustrated the impacts of fisheries for small pelagic species on higher TL species such as sea birds and marine mammals and the importance of connectivity within the food web to potential cascading effects (Smith et al. 2011; Cury et al. 2011). Operationalizing insight from PPR results in LCA would generally favor a shift to products derived from low TL species. However, it is important to recognize that there is a limit to how much can be taken out of an ecosystem even at lower trophic levels. This aspect can better be understood from estimating PPR as a fraction of total production, as is proposed in the refined method, but reference values for sustainable thresholds are yet to be defined. The fractional PPR measure, however, can be used to demonstrate relative merit or demerit between different products' impacts on their respective source ecosystems and can be operationalized to reduce impacts attributed to products that incorporate relatively efficient inputs. PPR thus must be recognized as a relative and descriptive measure of performance in LCAs, rather than an absolute measure of sustainability.

6 Conclusions

It has become common to account for biotic impacts in seafood LCAs, especially through the use of the PPR metric. However, the analyses performed often involve the use of estimates and average values in place of ecosystem- and species-specific values. The uncertainty associated with PPR results relating to methodological choices, coupled with the demonstrated magnitude of PPR values and uncertainty around them in the example of Norwegian salmon feeds, demonstrates the importance of refining this method for use in future studies. The potential importance of this cannot be understated as marine PPR of Norwegian salmon feed is three times larger when using the refined method compared to the standard method. In addition, the use of average values obfuscates reality where the same species can have very different PPR values depending on from which ecosystem it is sourced.

Our refined method for the PPR indicator takes advantage of species- and ecosystem-specific data when available, which improves upon currently used methods. Furthermore, our results are contextualized within the productivity of the source ecosystems to reflect variation in productivity and efficiency of different marine ecosystems and demonstrate the reliance of seafood systems on marine productivity. This demonstrated that inputs to total Norwegian salmon feeds milled in 2012 sourced from just the Barents and Norwegian Seas accounted for between 8–10 % of annual primary production in these ecosystems.

Finally, the use of the PPR indicator in seafood LCAs allows for a more holistic assessment of the impacts of seafood production and their reliance on wild inputs. This allows for comparisons to be made across disparate foods and feeds as well as predictions of ecological cost of potential decisions such as increasing fish inputs to aqua feeds. However, the utility of this measure is ultimately determined by the accuracy and reliability of results, and in the case of ecological indicators, this requires careful consideration of the methods used and the data informing the calculation.

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