Evaluating European imports of Asian aquaculture products using statistically supported life cycle assessments
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Chapter 6

A comparison of Asian aquaculture products using statistically supported LCA

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Abstract

We investigated aquaculture production of Asian tiger shrimp, whiteleg shrimp, giant river prawn, tilapia and pangasius in Bangladesh, China, Thailand and Vietnam using life cycle assessments (LCAs), with the purpose of evaluating the comparative eco-efficiency of producing different aquatic food products. Our starting hypothesis was that different production systems are associated with significantly different environmental impacts, as the production of these aquatic species differs in intensity and management practices. In order to test this hypothesis, we estimated the systems global warming, eutrophication and freshwater ecotoxicity impacts. The contribution to these impacts and the overall dispersions relative to results were propagated using Monte Carlo simulations and dependent sampling. Paired testing showed significant ($p<0.05$) differences between the median impacts of most production systems in the intra-species comparisons, even after a Bonferroni correction. For the full distributions, instead of only the median, only for Asian tiger shrimp more than 95% of the propagated Monte Carlo results favored certain farming systems. The major environmental hot-spots driving the differences in environmental performance among systems were fishmeal from mixed fisheries for global warming, pond run-off and sediment discards for eutrophication, and agricultural pesticides, metals, benzalkonium chloride and other chlorine releasing compounds for freshwater ecotoxicity. The Asian aquaculture industry should therefore strive towards farming systems relying upon pelleted species-specific feeds, where the fishmeal inclusion is limited and sourced sustainably. Also, excessive nutrients should be recycled in integrated organic agriculture together with efficient aeration solutions powered by renewable energy sources.

6.1 Introduction

Aquaculture is the only solution for meeting the growing demand for aquatic products in a world where capture fishery catches have stagnated (Duarte *et al.* 2009; FAO 2014a). Asia is the main producing region with 88% of global aquaculture production by volume, and the European Union (EU) the largest single market with 36% of total world imports by value (FAO 2014a). However, while consumption trends have rapidly increased in the EU, concerns have been raised regarding the environmental sustainability of the fish and crustacean products imported from Asia. These concerns are associated with detrimental environmental consequences such as global warming, eutrophication, ecotoxicity, land-use and land-use change (LULUC), excessive energy use and freshwater use (Pelletier *et al.* 2006; Henriksson *et al.* 2012c; Jonell and Henriksson 2014).

The environmental impacts related to aquaculture commodities have been quantified in various life cycle assessment (LCA) studies (Henriksson *et al.* 2012c). However, only a handful of these have focused on Asian aquaculture. Four LCA studies have evaluated Vietnamese Pangasius catfish (Phong *et al.* 2011; Bosma *et al.* 2011; Huysveld *et al.* 2013; Henriksson *et al.* 2015a), three shrimp farming (Mungkung 2005; Cao *et al.* 2011; Jonell and Henriksson 2014), two Indonesian finfish (Pelletier and Tyedmers 2010b; Mungkung *et al.* 2013) and one Thai finfish (Pongpat and Tongpool 2013). Only three of these quantified the uncertainties related to results (Cao *et al.* 2011; Henriksson *et al.* 2014a; Jonell and Henriksson 2014). Little is therefore known about the level of confidence behind conclusions made in previous studies, despite the increasing importance of LCA results in policy contexts (Henriksson *et al.* 2015a). Seafood standards are, for example, starting to incorporate carbon footprints into their recommendations (Madin and Macreadie 2015) and a PAS2050 standard has been developed for seafood and other aquatic food products (BSI 2012). For such standards to be realistic and effective, differences in impacts need to be statistically substantiated.

In the present study, we performed life cycle assessments (LCAs) and statistically evaluated the environmental impacts for some of the most common Asian aquaculture commodities found on European markets (Henriksson *et al.* 2014a) (**Table 6.1**). From this selection, the most important producing regions and production systems were identified and evaluated (Henriksson *et al.* 2014a; Murray *et al.* 2014; Henriksson *et al.* 2014b). Noteworthy is that some of these production systems currently are not eligible for export due to existing import regulations into the EU (e.g. tilapia integrated with pigs in China). System characterization was based on farm scale, pond type, species combination and other features of the production systems (Henriksson *et al.* 2014a; Murray *et al.* 2014).

The present study builds upon the final LCA case study report (Henriksson *et al.* 2014a) of the Sustaining Ethical Aquaculture Trade project (www.seatglobal.eu), but also includes calculated freshwater ecotoxicity characterization factors (FAETPs) for a number of aquaculture related chemicals using the USEtox model, including uncertainty estimates for characterization factors (Rosenbaum *et al.* 2008).

Table 6.1: Farming systems evaluated in the study. The systems will hereon be referred to by the code or the characteristic in bold.

Country	Code	Species	Region	Key characteristics
Bangladesh	BD K	Giant river prawn	**Khulna**	Avg. of 2 kg fish coproduced per kg prawn
	BD B		**Bagerhat**	Avg. of 3.3 kg fish co-produced per kg prawn
	BD S&P		Both	**Integrated** with Giant tiger shrimp
	BD W	Asian tiger shrimp	**West**	Lower stocking density and not always fed, with fish
	BD E		**East**	Higher stocking density, no fish
	BD S&P		West	**Integrated** with Giant river prawn
China	CN HL	Whiteleg shrimp	Guangdong	Lined **high-level** ponds with pumped water exchange
	CN LL		Guangdong	**Low-level** earthen ponds with tidal water exchange
	CN GD	Tilapia	**Guangdong**	Intensive to semi-intensive farms, <30 post-larvae m^2
	CN HI		**Hainan**	Intensive to semi-intensive farms, <30 post-larvae m^2
	CN R		Both	Farmed in freshwater **reservoirs**
	CN IG		Guangdong	Ponds fertilized by **integrated** pigs on dikes
Thailand	TH E	Whiteleg shrimp	**East**	Electricity as main energy source on farm
	TH S		**South**	LPG as main energy source on farm
Vietnam	VN SI	Asian tiger shrimp	Soc Trang & Bac Lieu	**Semi-intensive** with <30 shrimp post-larvae stocked per m^2
	VN I		Soc Trang	**Intensive** with >30 shrimp post-larvae stocked per m^2
	VN I	Whiteleg shrimp	Ben Tre	Intensive with >30 shrimp post-larvae stocked per m^2
	VN S	Pangasius	An Giang & Can Tho	**Small** farms with no full-time labor
	VN M			**Medium** privately owned with full time labor
	VN L			**Large** corporate farms

In order to provide a level of confidence behind conclusions, the hypothesis "different production systems providing the same aquaculture commodity to European consumers are associated with different environmental impacts" was tested statistically. The null hypothesis tested assumed that the environmental lifecycle impacts of commodities originating from different aquaculture system were equal (e.g. system A = system B).

Two approaches were used when testing the differences between paired results as obtained in dependent sampling (Henriksson *et al.* 2015a), one using significance tests (H_0:m_A=m_B at α=0.05) and the other analyzing the percentage of Monte Carlo (MC) runs in which the difference was lower or higher than zero ($p(x_A-x_B<0)$ or $p(x_A-x_B>0)$) at p=0.95). This dual approach was chosen as

each of them answers different questions; significance tests for the median analyze if the distribution of differences has a median that deviates significantly from zero, while MC frequencies indicate how often a type of farming system is expected to perform better than another. Given the large differences in nutritional, culinary and monetary value of the different species (Schau and Fet 2008), comparisons were only made across countries and systems, not across species.

6.2 Materials and methods

6.2.1 Goal and scope

The study aimed to evaluate the comparative eco-efficiency per functional unit of one tonne of frozen product for some selected aquaculture commodities commonly imported to Europe from Bangladesh, China, Thailand and Vietnam. The products surveyed were frozen peeled tail-on (PTO) whiteleg shrimp (Litopenaeus vannamei), PTO Asian tiger shrimp (Penaeus monodon), headless shell-on (HLSO) giant river prawn (Macrobrachium rosenbergii), tilapia fillets (mainly Oreochromis niloticus) and pangasius catfish fillets (Pangasianodon hypophthalmus). The production chains were modeled up to European ports, assuming that any processes (e.g. retailing, cooking and composting) downstream of this system boundary would be equivalent.

Three impact categories, global warming, eutrophication and freshwater toxicity, were evaluated. The selection of these represents a trade-off among access to good quality data (e.g. important emissions driving some impact categories could not be specified for Asian processes, such as halon causing ozone layer depletion or palladium resulting in abiotic resource depletion), avoidance of extensive multiple comparisons problems, diversity of inventory flows and impacts (e.g. acidification gave similar outcomes to global warming (Henriksson *et al.* 2014a)), and the different uncertainties they are subject to. Impacts were allocated among multiple co-products originating from the same process (e.g. fillets and heads from fish processing) based upon mass and economic proceeds (monetary value times mass), in order to evaluate the sensitivity of this highly influential methodological choice (Henriksson *et al.* 2012c) and to strengthen conclusions. These two allocation methods were chosen as they generally constitute two extreme outcomes and since they can be consistently applied to all allocation situations. Sensitivity in many other pivotal parameters of aquaculture LCAs (amount of feed used, emissions from agricultural fields and aquatic systems, characterization factors, etc.) (Henriksson *et al.* 2012c) were accounted for as part of the variable distributions and therefore considered in the statistical evaluation. Other modeling decisions that could influence outcomes (e.g. cut-off) were not evaluated in the present research as they were deemed to be of only limited importance to our comparative setup. For a more complete set of impact categories and methodological choices, please see Henriksson *et al.* (2014a), Henriksson *et al.* (2014b) and the supporting information of this article. This information is available free of charge via the Internet at http://pubs.acs.org/.

The data sourcing procedure was based upon the protocol presented in Henriksson *et al.* (2012a). Following this protocol, secondary data were weighted (in this study based upon the squared coefficient of variation, wt = $1/CV^2$) according to their inherent uncertainty (inaccuracies in measurements and models) and unrepresentativeness (mismatch between the representativeness and use of data), defined by the Numerical Unit Spread Assessment Pedigree and quantitative

uncertainty factors in Frischknecht et al. (2007b). Overall dispersions were quantified as the sum of inherent uncertainty, spread (variability resulting from averaging) and unrepresentativeness, in accordance to the protocol (Henriksson et al. 2013). LCI models were constructed, propagated and characterized using the CMLCA 5.2 software (www.cmlca.eu) and subsequently aggregated towards the functional unit over 1000 MC simulations using dependent sampling (Henriksson et al. 2015a). Covariance was not accounted for in the current models because of methodological limitations. Distributions were tested using the Anderson-Darling goodness-of-fit test in the EasyFit v5.5 software (www.mathwave.com) and significance tests were conducted in SPSS v21 (for a more detailed description of the statistical approach, see supporting information).

The median impact of each system was pairwise tested against that of all other systems used to produce the same commodity, for all three impact categories. Since the distributions were quite skewed, we decided to test equality of medians with the non-parametric Wilcoxon signed-rank test rather than equality of means with means with a paired t-test. Significant differences were considered as α = 0.05. However, since 216 comparisons were made among the five species and 20 systems, for two allocation factors and three impact categories, there is over 99.99% probability that at least one of our hypothesis would be a false positive (1-(1-0.05)^(36 comparisons*2 allocation factors*3 impact categories)). A Bonferroni correction was therefore implemented, adjusting the alpha level to $α_b$ = 0.05/216 = 0.00023.

The alternative approach, looking at the cumulative frequency of one alternative to be favorable to another according to the MC runs, was assumed to hold if cumulative frequencies were higher than 95%, as described by Heijungs and Kleijn (2001), and Huijbregts et al. (2003).

6.2.2 LCI data collection

Primary data for the current study involved several actors in the aquaculture value chains (**Fig. 6.1**). Initial data collection on basic farming practices was conducted between October 2010 and February 2011 for approximately 200 farmers for each species in each of the four countries (a total of about 1400 farmers were interviewed). Farm selection was performed by a random sampling design of farm clusters representing the most important production methods (Murray et al. 2014). From this dataset, 20 production systems were identified as systematically different based upon basic parameters such as feed used, energy sources and integrated species (Henriksson et al. 2014b) (**Table 6.1**). A follow-up in-depth survey was then conducted between 2011 and 2013 with focus on more LCI specific data and other actors in the aquaculture value chain, including feed mills, capture fisheries and agricultural producers. A complete set of data is available as supporting information to this article and as an annex to SEAT deliverable D3.5 (Henriksson et al. 2014b).

6.2.3 LCIA data

Eutrophying emissions were characterized based upon the Redfield ratio, assuming an average composition of phytoplankton biomass of 106 carbon atoms, 16 nitrogen atoms and 1 phosphorus atom, as suggested by Heijungs et al. (1992) and neglecting any uncertainty. Emissions resulting in global warming were characterized using the characterization factors and uncertainty estimates presented in the fifth IPCC report (IPCC 2013; Myhre et al. 2013). Characterization factors for freshwater ecosystem impacts were derived from Rosenbaum et al. (2008), or, for non-characterized
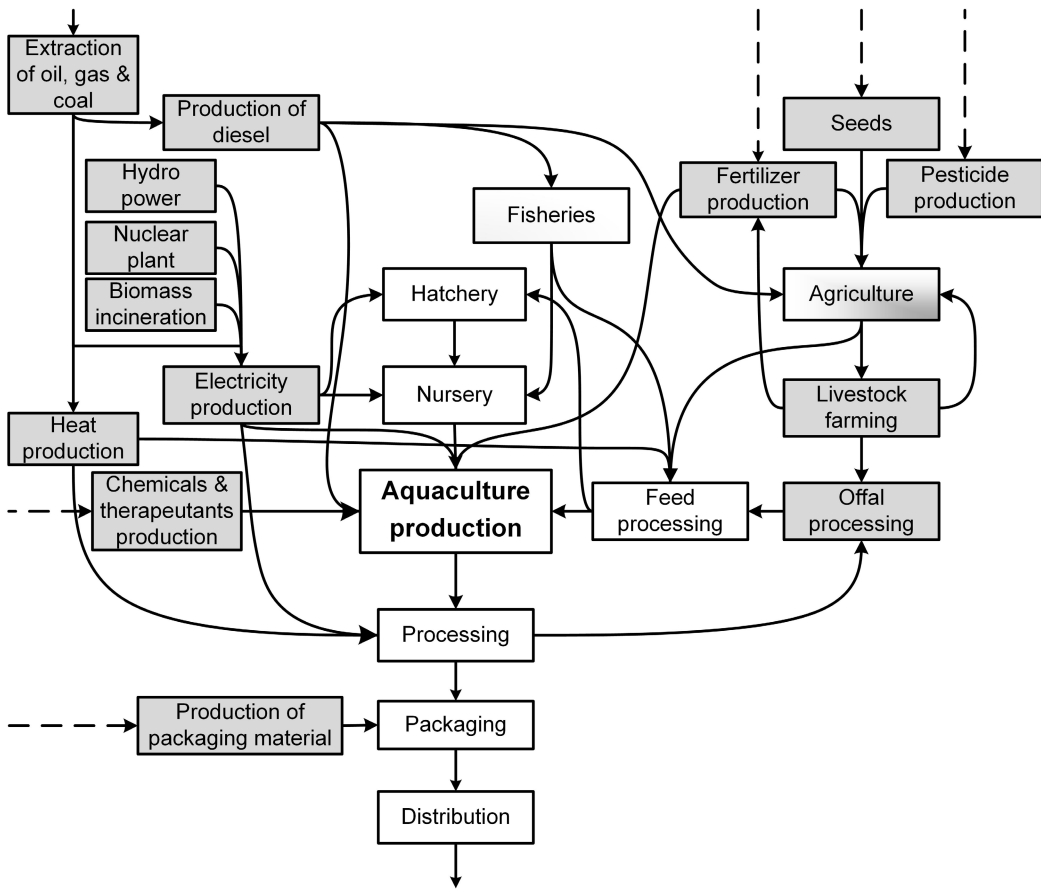
Fig 6.1: Simplified flow-chart of the processes included in this LCA, where arrows symbolize transportation, dashed lines indicate upstream processes, white filled boxes indicate processes modeled from primary data and grey fill boxes indicate processes modeled based upon secondary data.

chemicals used in aquaculture farming, calculated using the USEtox model. Ecotoxicity data for potentially toxic chemicals applied in aquaculture farms which were used in the model were primarily sourced from Rico *et al.*(Rico and Van den Brink 2014), and secondarily from the US Environmental Protection Agency's (EPA) ECOTOX database (cfpub.epa.gov; accessed 25-May-2014). For chemical characteristics, measured data were prioritized (primarily from sitem. herts.ac.uk/aeru/vsdb/atoz.htm; accessed 25-May-2014) before quantitative structure–activity relationships (QSARs) were used (toxnet.nlm.nih.gov, accessed 25-May-2014; Episuite v4.11 from US EPA). All chemicals applied to agricultural fields and ponds were assumed lost to the environment, in consistency with ecoinvent v2.2. For acute exposure EC50 and LC50 values were considered, and for chronic exposure NOECs and LOECs. Dispersions around the FEATPs were calculated as the sum of dispersions around acute and chronic effect concentrations within and among genera, and the unrepresentativeness of this data. No dispersions were, however, available for the FAETPs readily available in Rosenbaum *et al.* (2008).

6. 3 Results and interpretation

Significant conclusions among systems for each species are summarized below. Only conclusions that held for both allocation factors were considered. Relative differences as percentages and contribution analyses are available in the supporting information of this article. Dispersions related to the contribution analysis could unfortunately not be quantified using the present approach. These values are instead based upon the so-called baselines (point-value estimates), which in the current study were defined by arithmetic means, in line with the arithmetical structure of CMLCA (Heijungs and Suh 2002).

Asian tiger shrimp farming in Western Bangladesh was related to significantly lower median global warming and eutrophication impacts than all other systems, and also had the lowest median freshwater ecotoxic emissions alongside intensive farming in Vietnam (**Table 6.2**). This is explained by the fact that many Asian tiger shrimp farms in Western Bangladesh use limited feed and/or fertilizer inputs, resulting in a net sink for nutrients. The median eutrophying impacts of Bangladeshi farms in the east were, in the meantime, comparable with those from either of the Vietnamese shrimp farming systems, and worse with regards to freshwater ecotoxicity. Asian tiger shrimp integrated with prawn performed the worst for all impact categories except global warming. The poorer performance of the Bangladeshi systems with regards to toxicity was largely due to more extensive use of agricultural products as feed, for which pesticides are used. In Vietnam, intensive production of Asian tiger shrimp had significantly lower ecotoxicological and eutrophying impacts as compared to semi-intensive production, but similar global warming impacts.

Table 6.2: Ranking of the relative environmental performance related to Asian tiger shrimp at European consumers. VN = Vietnam; BD = Bangladesh; I = Intensive; SI = Semi-intensive; W = West; E = East; S&P = Shrimp and Prawn. Different superscripted letters indicate significantly different ranges identified using the Wilcoxon signed-rank test and different colors indicate ranges where more than 95% of the runs favored the green alternative over the red.

	Global warming		Eutrophication		Ecotoxicology	
Rank	Mass	Economic	Mass	Economic	Mass	Economic
Best	BD W[a]	BD W[a]	BD W[a]	BD W[a]	BD W[a]	BD W[a]
	BD E[b]	BD E[b]	BD E[b]	VN I[b]	VN I[b]	VN I[a]
	BD S&P[c]	VN SI[c]	VN I[c]	VN SI[c]	VN SI[c]	VN SI[b]
	VN I[d]	VN I[d]	VN SI[d]	BD E[c]	BD E[d]	BD E[c]
Worst	VN SI[d]	BD S&P[e]	BD S&P[e]	BD S&P[d]	BD S&P[e]	BD S&P[d]

For all three impacts, the median of related to the production of frozen peeled whiteleg shrimp were significantly larger for the Thai farms compared to the Vietnamese farms. Farming in low-level ponds in China was also related to lower median environmental impacts compared to farming in eastern Thailand. Chinese high and low-level farms (**Table 6.3**), however, had similar global warming and eutrophication impacts, while low-level farms were related to lower freshwater ecotoxicity impacts. The environmental impacts of whiteleg shrimp farming in China were also similar to farming in Vietnam, while the allocation factor used greatly influenced results due to a more extensive use of fishmeal from mixed fisheries and livestock byproducts in feeds. None of the impacts were significantly different when analyzing the entire distribution of differences between systems.

Table 6.3: Relative environmental performance of whiteleg shrimps at European consumers. VN = Vietnam; TH = Thailand; CN = China; I = Intensive; E = East; S = South; LL = Low-level; HL = High-level. Different superscripted letters indicate significantly different ranges identified using the Wilcoxon signed-rank test. For none of the comparisons, 95% of the runs favored one alternative over the other.

	Global warming		Eutrophication		Ecotoxicology	
Rank	Mass	Economic	Mass	Economic	Mass	Economic
Best	CN HL[a]	VN I[a]	VN I[a]	VN I[a]	CN LL[a]	VN I[a]
	CN LL[a]	CN LL[b]	CN LL[a]	CN LL[b]	CN HL[b]	CN LL[b]
	VN I[b]	CN HL[bc]	CN HL[a]	CN HL[b]	VN I[b]	CN HL[c]
	TH S[c]	TH S[bc]	TH S[b]	TH S[c]	TH S[c]	TH S[d]
Worst	TH E[d]	TH E[c]	TH E[b]	TH E[d]	TH E[d]	TH E[d]

Allocation also had a large influence on the outcomes of the Bangladeshi giant river prawn systems (**Table 6.4**). Farms where such prawn were polycultured with Asian tiger shrimp had more favorable median outcomes than prawn from Khulna province farmed without shrimp with regards to global warming and eutrophication, while the situation was the opposite in terms of freshwater ecotoxicity impacts. Distributions of differences did not differ among systems.

Table 6.4: Relative environmental performance of Giant River prawn at European consumers. BD = Bangladesh; B = Bagerhat; K=Khulna; S&P = Shrimp and Prawn. Different superscripted letters indicate significantly different ranges identified using the Wilcoxon signed-rank test. For none of the comparisons, 95% of the runs favored one alternative over the other.

	Global warming		Eutrophication		Ecotoxicology	
Rank	Mass	Economic	Mass	Economic	Mass	Economic
Best	BD B[a]	BD S&P[a]	BD S&P[a]	BD S&P[a]	BD B[a]	BD S&P[a]
	BD S&P[a]	BD B[b]	BD B[b]	BD K[b]	BD S&P[b]	BD B[b]
Worst	BD K[b]	BD K[b]	BD K[c]	BD B[c]	BD K[c]	BD K[b]

Among the Chinese tilapia systems, fillets from ponds in Guangdong were associated with significantly lower median impacts compared to fillets from Hainan (**Table 6.5**). The Hainan farms were also related to larger median eutrophication and ecotoxicity impacts than farms integrated with pigs and reservoir systems. Distributions of differences did not differ among systems.

Table 6.5: Relative environmental performance of tilapia fillets at European consumers. Tilapia. CN = China; GD = Guangdong; HI = Hainan; I = Integrated; R = Reservoir. Different superscripted letters indicate significantly different ranges identified using the Wilcoxon signed-rank test. For none of the comparisons, 95% of the runs favored one alternative over the other.

	Global warming		Eutrophication		Ecotoxicology	
Rank	Mass	Economic	Mass	Economic	Mass	Economic
Best	CN GD[a]	CN GD[a]	CN GD[a]	CN GD[a]	CN GD[a]	CN GD[a]
	CN R[b]	CN R[a]	CN INT[b]	CN INT[b]	CN INT[b]	CN R[a]
	CN INT[c]	CN INT[b]	CN R[c]	CN R[b]	CN R[b]	CN INT[b]
Worst	CN HI[d]	CN HI[b]	CN HI[d]	CN HI[c]	CN HI[c]	CN HI[c]

All evaluated environmental median impacts caused by the production of pangasius fillets were found to be significantly lower in the studied large-scale farms as compared to those calculated for small- and medium-scale farms. (**Table 6.6**). Small-scaled farms also resulted in significantly lower median eutrophication impacts than medium-scaled farms. Distributions of differences did not differ among systems.

Table 6.6: Relative environmental performance of Pangasius fillets at European consumers. VN = Vietnam; SL = Small; MD = Medium; LG = Large. Different superscripted letters indicate significantly different ranges identified using the Wilcoxon signed-rank test. For none of the comparisons, 95% of the runs favored one alternative over the other.

	Global warming		Eutrophication		Ecotoxicology	
Rank	Mass	Economic	Mass	Economic	Mass	Economic
Best	VN LG[a]	VN LG[a]	VN LG[a]	VN LG[a]	VN LG[a]	VN LG[a]
	VN SL[b]	VN SL[b]	VN SL[b]	VN SL[b]	VN SL[b]	VN SL[b]
Worst	VN MD[b]	VN MD[b]	VN MD[c]	VN MD[c]	VN MD[b]	VN MD[b]

6.4 Discussion

6.4.1 Analytical approach

Unlike previous comparisons of point-values, the current approach offered a level of confidence to support conclusions; and unlike previous comparisons of ranges (Cao *et al.* 2011), by only considering relative uncertainties, type II statistical errors (incorrectly accepting the null hypothesis) were reduced. Of the systems tested, most came out to differ significantly, despite the conservative Bonferroni correction (Narum 2006). This is largely due to the large sample size used (n=1000), a sample size deemed as sufficient, but not excessive. Historically, the number of MC iterations has been limited by computing power, and mathematical solutions for calculating the number of iterations needed to achieve a desired confidence level have even been proposed (so called sequential stopping boundaries) (Fay *et al.* 2007). One could therefore argue that by increasing the number of MC runs any hypothesis test on means or medians will always produce significant results. This, by the way, is not only true for Monte Carlo, but it is also a danger of large real samples, and it is an inherent characteristic of classical hypothesis testing (Cohen 1994). Using the alternative to significance tests showed that only the comparison of Asian tiger shrimp systems deviated in more 95% of the MC runs in their environmental impacts.

From a naive point of view, the two statistical approaches give contradictory answers, but in reality they answer different questions. The more suitable of the two approaches therefore depends upon the question needing answering, e.g. is the median of A significantly different from the median of B, or is a random pick of A demonstrably better than a random pick of B. Thus, while significance tests provide a conventional answer with respect to the median (or mean) impact, the proportional outcomes favoring a certain type of farming system might be more informative for a policy decision. In alternative words, statistical tests are about comparing distribution parameters, while the other approach is about a random pick from a distribution. While our belief is that operating within the paradigm of statistical hypotheses testing is too valuable to discard (Henriksson *et al.* 2015a), statistical significance should not always be taken at face value (Cohen 1994; McCloskey and Ziliak 1996; Doweyko 2008). However, differences that are proclaimed to be "significant" should be supported by statistical tests.

6.4.2 Aquaculture findings

Reflecting on previous aquaculture LCAs, many of the conclusions in the current research confirm the general outcomes of LCAs of fed aquaculture systems worldwide. Like tilapia and African catfish farming in Cameroon, eutrophication was mainly related to farm effluent (Ewoukem *et al.* 2012); and like most salmon farming, the provision of feed (including fisheries, agriculture and livestock) was related to most greenhouse gas emissions (Pelletier *et al.* 2009) (see Fig S1-S3). Lowering the feed conversion ratio would consequently offer environmental improvements, where formulated feeds tailored to the nutritional needs of each species served in portions ensuring high availability (e.g. floating pellets) should be promoted. Reductions in aquaculture impacts, moreover, require agriculture to switch to less toxic pesticides or adopt organic farming practices to the extent possible. Developing models for reusing pond effluents and sediments locally as fertilizers, as already practiced in traditional Chinese aquaculture, would also reduce the impacts of both agri- and aquaculture, as nutrients in modern aquaculture systems are largely lost to adjacent water bodies where they result in eutrophication. Production systems with limited environmental interactions that allow for nutrients to be captured, and the influence by external parasites and bacterial diseases to be reduced (thus reducing the reliance on and discharge of therapeutants) should therefore also be favored.

Use of wild fish in aqua-feeds is one of the major critiques of the aquaculture sector, based on both environmental and socioeconomic arguments (Naylor *et al.* 2000; Cao *et al.* 2015). In the present research this also stood out as one of the major causes for global warming and eutrophication for many systems (see Figure S1-S2). Limiting the inclusion and choosing more sustainable sources of fishmeal in feeds therefore need to be priorities for reducing the environmental impacts of farmed aquatic products, especially for shrimp. This goal can only be achieved if both feed producers and farmers, who often believe that larger fishmeal inclusions result in faster growth, recognize advancements in dietary substitution and supplements. A more sustainable source could be derived from processing byproducts, as much of these are still discarded (e.g. shrimp byproducts in Bangladesh). This would not only reduce pressure on wild fish stocks (Newton *et al.* 2014; Cao *et al.* 2015), but would also reduce eutrophying emissions at landfills and recycle nutrients (Phong *et al.* 2011). Lastly, it is important to always favor feed ingredients, terrestrial or aquatic, that do not compete with their direct use as human food, as malnutrition still is widespread in some regions of Asia and elsewhere.

Intensity of systems had no clear correlation with the impacts evaluated in the present study. Paddle-wheel aerators were, however, more intensively used in ponds with higher stocking densities, with consequent global warming impacts. Monitoring oxygen levels in ponds could therefore help optimizing the use of paddle-wheels, and more energy efficient forms of aeration should be developed and promoted. The use of coal to generate the electricity that powers aerators and other activities also needs to be curbed or improved, as does the electricity efficiency of freezers.

On-farm chemical use made only small contributions to the overall lifecycle freshwater ecotoxicity impacts, with the exception of benzalkonium chloride and other chlorine releasing compounds used as disinfectants. Chlorine is volatile and therefore used in large quantities, but the presence of organic matter leads to chlorinated compounds (e.g. halogenated hydrocharbons)

that are more stable and induce long-term toxicity. The use of alternative, less toxic, biocidal or disinfection methods is therefore promoted.

6.4.3 Limitations and future research needs

When considering chemical and other emissions, it is important to acknowledge that LCA has limited capacity to account for spatio-temporal aspects in both the LCI and the LCIA phases (Guinée and Heijungs 1993; Pinsonnault *et al.* 2014). Thus, even if many of the local impacts related to the grow-out sites appeared not to exceed the buffering capacity of local ecosystems, they cannot be discounted as inconsequential. For example, with regards to therapeutant use in the present study, the peak predicted environmental concentrations for 61% of the treatments applied by grow-out farmers resulted in a risk quotient higher than one, implying a potential risk to important structural endpoints of aquatic ecosystems not accounted for in the LCAs (Rico and Van den Brink 2014). Similar for eutrophication, where discharge of sediments and/or sludge from post-harvested ponds could have severe ecological consequences through peaks in turbidity, oxygen depletion or ammonia toxicity. Neither are additive and synergistic effects of different stressors accounted for in current LCA methodology, highlighting the added value of adopting the refined spatio-temporal windows and mixture toxicity approaches currently used in risk assessment alongside LCA (Rico and Van den Brink 2014). A risk assessment approach could also provide better insights into other impacts that have been deemed as relevant for aquaculture LCAs (Ford *et al.* 2012), such as reduced dissolved oxygen levels, introduction of non-indigenous species, and spread of disease and parasites.

The large dispersions around the characterization factors for freshwater ecotoxicity originated partially from the eco-toxicological effect factors, with large discrepancies in experimental acute and chronic effect concentrations, and within and among genus. Chronic effects on different types of algae often expressed the largest irregularities. Many additional assumptions exist around the chemical properties, some of which had to be resolved using QSARs. Given that these values are purely based upon the theoretical properties of molecules, QSAR estimates can differ greatly from reality (Doweyko 2008). Many other parameters related to inventory and impact assessment models also lack confidence estimates (Nemecek and Schnetzer 2011; Hauschild *et al.* 2012), which in some cases were confidence estimates are almost impossible to quantify (Maurice *et al.* 2000; Björklund 2002). For example, in the present research no uncertainty estimates were assigned to the eutrophication potentials, as the uncertainty around the actual environmental consequences are hard to quantify given their complex nature and geographically specific context, with discrepancies induced by factors such as planktonic species assemblage, bioavailability of the nutrients, fate of emissions, abiotic factors and nutrient compositions in receiving environments (Ptacnik *et al.* 2010). More recent impact assessment methods that address these challenges by presenting country-, or even region-, specific characterization factors (Posch *et al.* 2008; Gallego *et al.* 2010) can, in the meantime, induce new uncertainty in the form of unknown locations of emissions.

In addition to this, uncertainties also arise from the limited number of distributions available to represent data in LCA at present and the general negligence of covariance (Maurice *et al.* 2000). Still, these are only some of the many assumptions made over the different phases of an LCA,

where quantitative uncertainty estimates remain incomplete or undefined, resulting in a fragile pyramid where the ranges of results only capture part of the underlying uncertainty. Significant differences thus only consider the dispersions quantified, confirming the strict relative meaning of comparative LCAs (Henriksson *et al.* 2015a). Other types of uncertainties, including several methodological choices, may also be more easily illustrated by performing sensitivity analyses (Björklund 2002) until more sophisticated approaches become available (Jung *et al.* 2013; Beltran *et al.* 2014).

More extensive data on emissions related to LULUC are warranted, as the removal of mangrove for pond constructs is known to greatly influence both global warming and eutrophication results (Jonell and Henriksson 2014). More inventory and characterization data related to freshwater ecotoxicity are also invited, as many emissions with possible environmental effects had to be excluded from the present study due to resource constraints. The inclusions of infrastructure, its maintenance and waste disposal might, for example, alter the conclusions made related to freshwater ecotoxicity, as metals were a major driver for this impact category. Moreover, it is important to acknowledge that the data in the present research represents farming practices between 2010 and 2011, while aquaculture practices are notable for changing rapidly. For example, an outbreak of early mortality syndrome led to a rapid shift from Asian tiger shrimp to whiteleg shrimp for many Vietnamese farmers during the period of this research. Wild fish stocks, agricultural yields and monetary values are also variable over time. More extensive databases and better software that allow for more rapid data processing and invite practitioners to utilize methodological advancements are therefore desired, in order to promote more scientifically robust conclusions in future LCA studies.

Acknowledgements

Chapter 6