Environmental Science & Technology

Article

pubs.acs.org/est

Comparison of Asian Aquaculture Products by Use of Statistically Supported Life Cycle Assessment

Patrik J. G. Henriksson,[*,†] Andreu Rico,[‡] Wenbo Zhang,[§] Sk. Ahmad-Al-Nahid,[∥] Richard Newton,[⊥] Lam T. Phan,[⊥,▲] Zongfeng Zhang,[§] Jintana Jaithiang,[■] Hai M. Dao,[▽] Tran M. Phu,[▽] David C. Little,[⊥] Francis J. Murray,[⊥] Kriengkrai Satapornvanit,[■] Liping Liu,[§] Qigen Liu,[§] M. Mahfujul Haque,[∥] Froukje Kruijssen,[□] Geert R. de Snoo,[†] Reinout Heijungs,[†] Peter M. van Bodegom,[†] and Jeroen B. Guinée[†]

[†]Institute of Environmental Sciences, Leiden University, 2300 RA Leiden, The Netherlands

[‡]Alterra, Wageningen University and Research Centre, 6708 PB Wageningen, The Netherlands

[§]College of Fisheries and Life Science, Shanghai Ocean University, Beijing 100044, China

[∥]Faculty of Fisheries, Bangladesh Agricultural University, Mymensingh 2202, Bangladesh

[⊥]Institute of Aquaculture, University of Stirling, Stirling FK9 4LA, Scotland, United Kingdom.

[▲]Department of Inland Resources & Fisheries Capture, Research Institute for Aquaculture No. 2, Ho Chi Minh City, Vietnam

[▽]College of Aquaculture and Fisheries, Can Tho University, Can Tho, Vietnam

[■]Faculty of Fisheries, Kasetsart University, Bangkok 10900, Thailand

[□]WorldFish, Amsterdam, the Netherlands

Supporting Information

ABSTRACT: We investigated aquaculture production of Asian tiger shrimp, whiteleg shrimp, giant river prawn, tilapia, and pangasius catfish in Bangladesh, China, Thailand, and Vietnam by 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 each system's global warming, eutrophication, and freshwater ecotoxicity impacts. The contribution to these impacts and the overall dispersions relative to results were propagated by Monte Carlo simulations and dependent sampling. Paired testing showed significant ($p < 0.05$) differences between the median impacts of most production systems in the intraspecies comparisons, even after a Bonferroni correction. For the full distributions instead of only the median, only for Asian tiger shrimp did more than 95% of the propagated Monte Carlo results favor 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 runoff 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 toward 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.

1. INTRODUCTION

Aquaculture is the only solution for meeting the growing demand for aquatic products in a world where capture fishery catches have stagnated.[1,2] Asia is the main producing region with 88% of global aquaculture production by volume, and the European Union (EU) is the largest single market with 36% of total world imports by value.[1] However, while consumption trends have rapidly increased in the European Union, concerns have been raised regarding the environmental sustainability of fish and crustacean products imported from Asia. These concerns are

Received: February 16, 2015
Accepted: October 29, 2015
Published: October 29, 2015

associated with detrimental environmental consequences such as global warming, eutrophication, ecotoxicity, land-use and land-use change (LULUC), excessive energy use, and freshwater use.[3−5]

Environmental impacts related to aquaculture commodities have been quantified in various life cycle assessment (LCA) studies.[3] However, only a handful of these have focused on Asian aquaculture. Four LCA studies have evaluated Vietnamese pangasius catfish,[6−9] three have studied shrimp farming,[4,10,11] two have focused on Indonesian finfish,[12,13] and one has studied Thai finfish.[14] Only three of these quantified the uncertainties related to results.[4,11,15] 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.[9] Seafood standards are, for example, starting to incorporate carbon footprints into their recommendations,[16] and a PAS2050 (publicly available specification) standard has been developed for seafood and other aquatic food products.[17] For such standards to be realistic and effective, differences in impact need to be statistically substantiated.

In the present study, we performed LCAs and statistically evaluated the environmental impacts for some of the most common Asian aquaculture commodities found on European markets[15] (see Table 1). From this selection, the most important producing regions and production systems were identified and evaluated.[15,18,19] Noteworthy is that some of these production systems currently are not eligible for export due to existing import regulations into the European Union (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.[15,19]

The present study builds upon the final LCA case study report[15] (available at: http://media.leidenuniv.nl/legacy/d35-annexreport.pdf) of the Sustaining Ethical Aquaculture Trade project (www.seatglobal.eu) but also includes calculated freshwater aquatic ecotoxicity potential characterization factors (FAETPs) for a number of aquaculture-related chemicals by use of the USEtox model, including uncertainty estimates for characterization factors.[20]

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 life-cycle impacts of commodities originating from different aquaculture system were equal (e.g., system A = system B).

Two approaches were used for testing the differences between paired results as obtained in dependent sampling:[9] one used significance tests (H_0: $m_A = m_B$ at $\alpha = 0.05$), and the other analyzed the percentage of Monte Carlo (MC) runs in which the difference was lower or higher than 0 [$p(x_A - x_B < 0)$ or $p(x_A - x_B > 0)$ at $p = 0.95$]. This dual approach was chosen as each answers different questions. Significance tests for the median analyze whether 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,[21] comparisons were made only across countries and systems, not across species.

2. MATERIALS AND METHODS

2.1. Goal and Scope. The study aimed to evaluate the comparative eco-efficiency per functional unit of 1 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 were evaluated: global warming, eutrophication, and freshwater toxicity. 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 comparison problems, diversity of inventory flows and impacts (e.g., acidification gave similar

Table 1. Farming Systems Evaluated in This Study[a]

code	species	region	key characteristics
		Bangladesh	
BD K	giant river prawn	**Khulna**	avg 2 kg of fish coproduced/kg of prawn
BD B	giant river prawn	**Bagerhat**	avg 3.3 kg of fish coproduced/kg of prawn
BD S&P	giant river prawn and Asian tiger shrimp	both	**integrated** with Asian tiger shrimp
BD W	Asian tiger shrimp	**West**	lower stocking density, not always fed, with fish
BD E	Asian tiger shrimp	**East**	higher stocking density, no fish
BD S&P	Asian tiger shrimp and giant river prawn	West	**integrated** with giant river prawn
		China	
CN HL	whiteleg shrimp	Guangdong	lined **high-level** ponds with pumped water exchange
CN LL	whiteleg shrimp	Guangdong	**low-level** earthen ponds with tidal water exchange
CN GD	tilapia	**Guangdong**	intensive to semi-intensive farms, <30 postlarvae·m^2
CN HI	tilapia	**Hainan**	intensive to semi-intensive farms, <30 postlarvae·m^2
CN R	tilapia	both	farmed in freshwater **reservoirs**
CN IG	tilapia	Guangdong	ponds fertilized by **integrated** pigs on dikes
		Thailand	
TH E	whiteleg shrimp	**East**	electricity as main energy source on farm
TH S	whiteleg shrimp	**South**	LPG as main energy source on farm
		Vietnam	
VN SI	Asian tiger shrimp	Soc Trang and Bac Lieu	**semi-intensive** with <30 shrimp postlarvae·m^2
VN I	Asian tiger shrimp	Soc Trang	**intensive** with >30 shrimp postlarvae·m^2
VN I	whiteleg shrimp	Ben Tre	**intensive** with >30 shrimp postlarvae·^{2}
VN S	pangasius catfish	An Giang and Can Tho	**small** farms with no full-time labor
VN M	pangasius catfish	An Giang and Can Tho	**medium** farms, privately owned with full-time labor
VN L	pangasius catfish	An Giang and Can Tho	**large** corporate farms

[a]Systems will hereafter be referred to by the code or characteristic shown in boldface type.

DOI: 10.1021/acs.est.5b04634
Environ. Sci. Technol. 2015, 49, 14176−14183

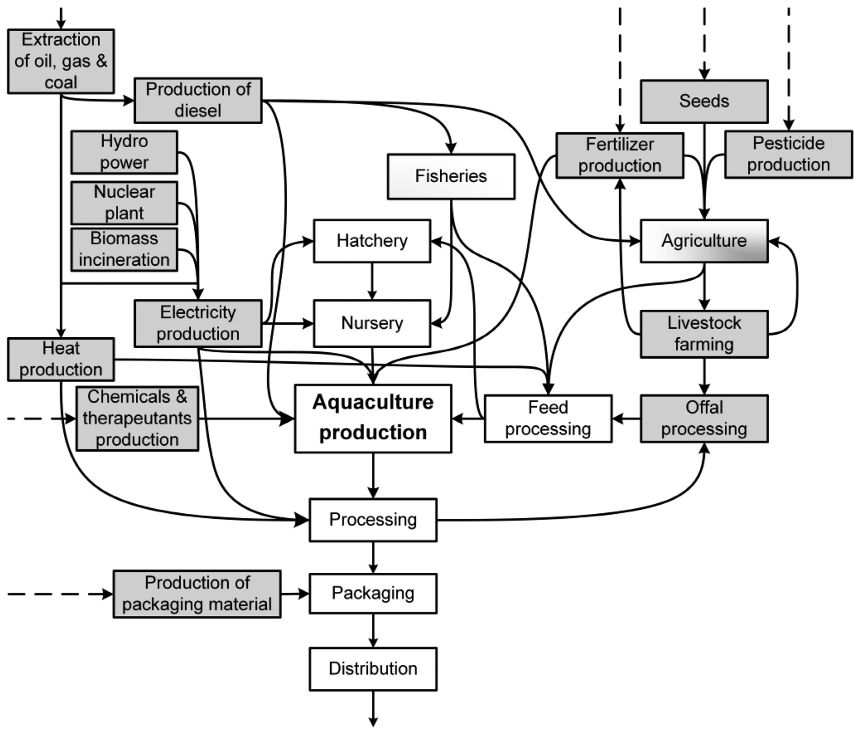

Figure 1. Simplified flowchart of the processes included in this LCA, where arrows symbolize transportation, dashed lines indicate upstream processes, unshaded boxes indicate processes modeled from primary data, and shaded boxes indicate processes modeled from secondary data.

outcomes to global warming[15]), and the different uncertainties they are subject to. Impacts were allocated among multiple coproducts 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[3] 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.)[3] was accounted for as part of the variable distribution and therefore considered in the statistical evaluation. Other modeling decisions that could influence outcomes (e.g., cutoff) 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.[15,18] and Supporting Information.

The data sourcing procedure was based upon the protocol presented in Henriksson et al.[22] 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 representativeness and use of data), defined by the numerical unit spread assessment pedigree and quantitative uncertainty factors in Frischknecht et al.[23] Overall dispersions were quantified as the sum of inherent uncertainty, spread (variability resulting from averaging), and unrepresentativeness, in accordance with the protocol.[24] Life cycle inventory (LCI) models were constructed, propagated, and characterized by use of CMLCA 5.2 software (www.cmlca.eu) and subsequently aggregated toward the functional unit over 1000 MC simulations with dependent sampling.[9] Covariance was not accounted for in the current models because of methodological limitations. Distributions were tested by the Anderson−Darling goodness-of-fit test in 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 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 nonparametric Wilcoxon signed-rank test rather than equality of means with a paired *t*-test. Significant differences were considered as $\alpha = 0.05$. 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 $\times$ 2 allocation factors $\times$ 3 impact categories)]. A Bonferroni correction was therefore implemented, adjusting the α level to $\alpha_b = 0.05/216 = 0.00023$.

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[25] and Huijbregts et al.[26]

2.2. Life Cycle Inventory Data Collection. Primary data for the current study involved several actors in the aquaculture value chains (Figure 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 (in total, about 1400 farmers were interviewed). Farm selection was performed by a random sampling design of farm clusters representing the most important production methods.[19] From this data set, 20 production systems were identified as systematically different based upon basic parameters such as feed used, energy sources, and integrated species[18] (Table 1). A follow-up in-depth survey was then

DOI: 10.1021/acs.est.5b04634
Environ. Sci. Technol. 2015, 49, 14176−14183

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 and as an annex to SEAT deliverable D3.5[18] (available at http://media.leidenuniv.nl/legacy/d35-annexreport.pdf).

2.3. Life Cycle Impact Assessment Data. Eutrophying emissions were characterized on the basis of the Redfield ratio, with the assumption of an average phytoplankton biomass composition of 106 carbon atoms, 16 nitrogen atoms, and 1 phosphorus atom, as suggested by Heijungs et al.[27] and neglect of any uncertainty. Emissions resulting in global warming were characterized by use of the characterization factors and uncertainty estimates presented in the fifth IPCC report (Table S1).[28,29] Characterization factors for freshwater ecosystem impacts were derived from Rosenbaum et al.[20] or, for noncharacterized chemicals used in aquaculture farming, calculated via the USEtox model (Tables S2–S4). Ecotoxicity data for potentially toxic chemicals applied in aquaculture farms used in the model were sourced primarily from Rico et al.[30] and Van den Brink,[31] and secondarily from the U.S. Environmental Protection Agency's (EPA) ECOTOX database (cfpub.epa.gov; accessed 25 May 2014) (Tables S3 and S4). 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 U.S. EPA). All chemicals applied to agricultural fields and ponds were assumed to be lost to the environment, in consistency with ecoinvent v2.2. For acute exposure, EC_{50} and LC_{50} values were considered, and for chronic exposure, no observed effects concentration (NOEC) and lowest observed effects concentration (LOEC) values were used (see Table S2). Dispersions around the FAETPs were calculated as the sum of dispersions around acute and chronic effect concentrations within and among genera, and the unrepresentativeness of these data. No dispersions were available, however, for the FAETPs readily available in Rosenbaum et al.[20]

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 Supporting Information (Tables S5–S34 and Figures S1–S3). Dispersions related to the contribution analysis could unfortunately not be quantified by 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.[32]

3.1. Asian Tiger Shrimp. 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. 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 but worse with regard 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 regard 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 2).

Table 2. Ranking of Relative Environmental Performance Related to Asian Tiger Shrimp Provided to European Consumers[a]

Rank	Global warming		Eutrophication		Ecotoxicology	
	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]

[a]VN = Vietnam; BD = Bangladesh; I = intensive; SI = semi-intensive; W = west; E = east; S&P = shrimp and prawn. Different letters indicate significantly different ranges identified by the Wilcoxon signed-rank test, and different colors indicate ranges where more than 95% of the runs favored the green alternative over the red.

3.2. Whiteleg Shrimp. For all three impacts, the median related to the production of frozen peeled whiteleg shrimp was 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 3), however, had similar global warming and eutrophication impacts, while low-level farms had lower freshwater ecotoxicity impacts. The environmental impacts of whiteleg shrimp farming in China were also similar to those of farming in Vietnam, while the allocation factor used greatly influenced results due to more extensive use of fishmeal from mixed fisheries and livestock byproducts in feeds. None of the impacts was significantly different when analyzing the entire distribution of differences between systems.

3.3. Giant River Prawn. Allocation also had a large influence on the outcomes of the Bangladeshi giant river prawn systems (Table 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 regard 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.

3.4. Tilapia. Among the Chinese tilapia systems, fillets from ponds in Guangdong were associated with significantly lower median impacts compared to fillets from Hainan (Table 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.

3.5. Pangasius Catfish. All evaluated environmental median impacts caused by the production of pangasius catfish 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). Small-scale farms also resulted in significantly lower median eutrophication impacts than medium-scale farms. Distributions of differences did not differ among systems.

DOI: 10.1021/acs.est.5b04634
Environ. Sci. Technol. 2015, 49, 14176–14183

Table 3. Relative Environmental Performance of Whiteleg Shrimp Provided to European Consumers[a]

	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

[a]VN = Vietnam; TH = Thailand; CN = China; I = intensive; E = east; S = south; LL = low-level; HL = high-level. Different letters indicate significantly different ranges identified by the Wilcoxon signed-rank test. For none of the comparisons did 95% of the runs favor one alternative over the other.

Table 4. Relative Environmental Performance of Giant River Prawn Provided to European Consumers[a]

	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

[a]BD = Bangladesh; B = Bagerhat; K = Khulna; S&P = shrimp and prawn. Different letters indicate significantly different ranges identified by the Wilcoxon signed-rank test. For none of the comparisons did 95% of the runs favor one alternative over the other.

Table 5. Relative Environmental Performance of Tilapia Fillets Provided to European Consumers[a]

	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

[a]CN = China; GD = Guangdong; HI = Hainan; I = integrated; R = reservoir. Different letters indicate significantly different ranges identified by the Wilcoxon signed-rank test. For none of the comparisons did 95% of the runs favor one alternative over the other.

Table 6. Relative Environmental Performance of Pangasius Catfish Fillets Provided to European Consumers[a]

	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

[a]VN = Vietnam; SL = small; MD = medium; LG = large. Different letters indicate significantly different ranges identified by the Wilcoxon signed-rank test. For none of the comparisons did 95% of the runs favor one alternative over the other.

4. DISCUSSION

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,[11] consideration of only relative uncertainties reduced type II statistical errors (incorrectly accepting the null hypothesis). Of the systems tested, most differed significantly, despite the conservative Bonferroni correction.[33] 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).[34] 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, not only is true for Monte Carlo but also is a danger of large real samples, and it is an inherent characteristic of classical hypothesis testing.[35] Using the alternative to significance tests showed that only the comparison of Asian tiger shrimp systems deviated in more than 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 that needs answering: 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,[9] statistical significance should not always be taken at face

DOI: 10.1021/acs.est.5b04634
Environ. Sci. Technol. 2015, 49, 14176−14183

value.[35−37] However, differences that are proclaimed to be "significant" should be supported by statistical tests.

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,[38] and like most salmon farming, the provision of feed (including fisheries, agriculture, and livestock) was related to most greenhouse gas emissions[39] (see Figures 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.[40,41] In the present research this also stood out as one of the major causes for global warming and eutrophication for many systems (see Figures S1 and 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 be achieved only 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 many of these are still discarded (e.g., shrimp byproducts in Bangladesh). This would not only reduce pressure on wild fish stocks[41,42] but also reduce eutrophying emissions at landfills and recycle nutrients.[6] Finally, 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 to optimize 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 life-cycle 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.

4.3. Limitations and Future Research Needs. When chemical and other emissions are considered, it is important to acknowledge that LCA has limited capacity to account for spatiotemporal aspects in both LCI and life cycle impact assessment (LCIA) phases.[43,44] 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 1, implying a potential risk to important structural end points of aquatic ecosystems not accounted for in the LCAs.[31] Similarly, for eutrophication, discharge of sediments and/or sludge from postharvested 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 spatiotemporal windows and mixture toxicity approaches currently used in risk assessment alongside LCA.[31] A risk assessment approach could also provide better insights into other impacts that have been deemed as relevant for aquaculture LCAs,[45] such as reduced dissolved oxygen levels, introduction of nonindigenous species, and spread of disease and parasites.

The large dispersions around the characterization factors for freshwater ecotoxicity originated partially from ecotoxicological 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 by use of QSARs. Given that these values are purely based upon the theoretical properties of molecules, QSAR estimates can differ greatly from reality.[37] Many other parameters related to inventory and impact assessment models also lack confidence estimates,[46,47] which in some cases were almost impossible to quantify.[48,49] For example, in the present research no uncertainty estimates were assigned to 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 nutrients, fate of emissions, abiotic factors, and nutrient compositions in receiving environments.[50] More recent impact assessment methods that address these challenges by presenting country- or even region-specific characterization factors[51,52] 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.[48] 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 capture only part of the underlying uncertainty. Significant differences thus consider only the dispersions quantified, confirming the strict relative meaning of comparative LCAs.[9] Other types of uncertainties, including several methodological choices, may also be more easily illustrated by performing sensitivity analyses[49] until more sophisticated approaches become available.[53,54]

DOI: 10.1021/acs.est.5b04634
Environ. Sci. Technol. 2015, 49, 14176−14183

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.[4] 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.

■ ASSOCIATED CONTENT

Supporting Information

The Supporting Information is available free of charge on the ACS Publications website at DOI: 10.1021/acs.est.5b04634.

Additional text, three figures, and 34 tables showing characteristic factors and chemical properties, comparative analyses, and contribution analysis (PDF)

Excel file with detailed contribution analyses for all systems and impact categories (ZIP)

■ AUTHOR INFORMATION

Corresponding Author

*Phone +31 (0)71 527 7461; e-mail Henriksson@cml.leidenuniv.nl. Present address: Stockholm Resilience Centre/WorldFish, The Beijer Institute, The Royal Swedish Academy of Science, Box 50005, 104 05, Stockholm, Sweden. Phone +46 8 673 95 38; e-mail patrik.henriksson@beijer.kva.se.

Notes

The authors declare no competing financial interest.

■ ACKNOWLEDGMENTS

This work is part of the Sustaining Ethical Aquaculture Trade (SEAT) project, which is cofunded by the European Commission within the Seventh Framework Programme—Sustainable Development Global Change and Ecosystem (Project 222889); www.seatglobal.eu.

■ REFERENCES

(1) *The State of World Fisheries and Aquaculture 2014*; Food and Agriculture Organization of the United Nations, 2014; http://www.fao.org/assets/infographics/FAO-infographic-SOFIA-2014-en.pdf.

(2) Duarte, C. M.; Holmer, M.; Olsen, Y.; Soto, D.; Marbà, N.; Guiu, J.; Black, K.; Karakassis, I. Will the Oceans Help Feed Humanity? *BioScience* **2009**, *59*, 967−976.

(3) Henriksson, P. J. G.; Guinée, J. B.; Kleijn, R.; de Snoo, G. R. Life cycle assessment of aquaculture systems—a review of methodologies. *Int. J. Life Cycle Assess.* **2012**, *17*, 304−313.

(4) Jonell, M.; Henriksson, P. J. G. Mangrove-shrimp farms in Vietnam − comparing organic and conventional systems using life cycle assessment. *Aquaculture* **2015**, *447*, 66−75.

(5) Pelletier, N. L.; Ayer, N. W.; Tyedmers, P. H.; Kruse, S. a.; Flysjo, A.; Robillard, G.; Ziegler, F.; Scholz, A. J.; Sonesson, U. Impact categories for life cycle assessment research of seafood production systems: Review and prospectus. *Int. J. Life Cycle Assess.* **2007**, *12*, 414−421.

(6) Phong, L. T.; de Boer, I. J. M.; Udo, H. M. J. Life cycle assessment of food production in integrated agriculture−aquaculture systems of the Mekong Delta. *Livest. Sci.* **2011**, *139*, 80−90.

(7) Bosma, R.; Anh, P. T.; Potting, J. Life cycle assessment of intensive striped catfish farming in the Mekong Delta for screening hotspots as input to environmental policy and research agenda. *Int. J. Life Cycle Assess.* **2011**, *16*, 903−915.

(8) Huysveld, S.; Schaubroeck, T.; De Meester, S.; Sorgeloos, P.; Van Langenhove, H.; Van linden, V.; Dewulf, J. Resource use analysis of Pangasius aquaculture in the Mekong Delta in Vietnam using Exergetic Life Cycle Assessment. *J. Cleaner Prod.* **2013**, *51*, 225.

(9) Henriksson, P. J. G.; Heijungs, R.; Dao, H. M.; Phan, L. T.; de Snoo, G. R.; Guinée, J. B. Product carbon footprints and their uncertainties in comparative decision contexts. *PLoS One* **2015**, *10*, e0121221.

(10) Mungkung, R. T. *Shrimp Aquaculture in Thailand: Application of Life Cycle Assessment to Support Sustainable Development*. Ph.D. Thesis, University of Surrey, England, 2005.

(11) Cao, L.; Diana, J. S.; Keoleian, G.; Lai, Q. Life Cycle Assessment of Chinese Shrimp Farming Systems Targeted for Export and Domestic Sales. *Environ. Sci. Technol.* **2011**, *45*, 6531−6538.

(12) Pelletier, N.; Tyedmers, P. Life Cycle Assessment of Frozen Tilapia Fillets From Indonesian Lake-Based and Pond-Based Intensive Aquaculture Systems. *J. Ind. Ecol.* **2010**, *14*, 467−481.

(13) Mungkung, R. T.; Aubin, J.; Prihadi, T. H.; Slembrouck, J.; van der Werf, H. M.; Legendre, M. Life Cycle Assessment for environmentally sustainable aquaculture management: a case study of combined aquaculture systems for carp and tilapia. *J. Cleaner Prod.* **2013**, *57*, 249−256.

(14) Pongpat, P.; Tongpool, R. Life Cycle Assessment of Fish Culture in Thailand: Case Study of Nile Tilapia and Striped Catfish. *Int. J. Environ. Sci. Dev.* **2013**, *4*, 608−612.

(15) Henriksson, P. J. G.; Zhang, W.; Nahid, S. A. A.; Newton, R.; Phan, L. T.; Dao, H. M.; Zhang, Z.; Jaithiang, J.; Andong, R.; Chaimanuskul, K.; et al. *Final LCA Case Study Report: Results of LCA Studies of Asian Aquaculture Systems for Tilapia, Catfish, Shrimp, and Freshwater Prawn*; SEAT Deliverable D3.5; Sustaining Ethical Aquaculture Trade, Leiden, The Netherlands, 2014; http://media.leidenuniv.nl/legacy/d35-final-case-study-report.pdf.

(16) Madin, E. M. P.; Macreadie, P. I. Incorporating carbon footprints into seafood sustainability certification and eco-labels. *Mar. Policy* **2015**, *57*, 178−181.

(17) British Standards Institution. *Assessment of Life Cycle Greenhouse Gas Emissions - Supplementary Requirements for the Application of PAS 2050:2011 to Seafood and Other Aquatic Food Products*; Publicly Available Specification (PAS) 2050-2:2012, 2012.

(18) Henriksson, P. J. G.; Zhang, W.; Nahid, S. A. A.; Newton, R.; Phan, L. T.; Dao, H. M.; Zhang, Z.; Jaithiang, J.; Andong, R.; Chaimanuskul, K.; et al. *Final LCA Case Study Report: Primary Data and Literature Sources Adopted in the SEAT LCA Studies*. SEAT Deliverable D3.5, Annex report; Sustaining Ethical Aquaculture Trade, Leiden, The Netherlands, 2014; http://media.leidenuniv.nl/legacy/d35-annexreport.pdf.

(19) Murray, F. J.; Zhang, W.; Nietes-Satapornvanit, A.; Phan, L. T.; Haque, M. M.; Henriksson, P. J. G.; Little, D. C. SEAT Deliverable 2.8, Report on Boundary Issues; Sustaining Ethical Aquaculture Trade, Stirling, U.K., 2014.

(20) Rosenbaum, R. K.; Bachmann, T. M.; Gold, L. S.; Huijbregts, M. a. J.; Jolliet, O.; Juraske, R.; Koehler, A.; Larsen, H. F.; MacLeod, M.; Margni, M.; et al. USEtox—the UNEP-SETAC toxicity model: recommended characterisation factors for human toxicity and freshwater ecotoxicity in life cycle impact assessment. *Int. J. Life Cycle Assess.* **2008**, *13*, 532−546.

DOI: 10.1021/acs.est.5b04634
Environ. Sci. Technol. 2015, 49, 14176−14183

(21) Schau, E. M.; Fet, A. M. LCA Studies of Food Products as Background for Environmental Product Declarations. *Int. J. Life Cycle Assess.* **2008**, *13*, 255−264.
(22) Henriksson, P. J. G.; Guinée, J. B.; Heijungs, R.; de Koning, A.; Green, D. M. A Protocol for Horizontal Averaging of Unit Process Data - Including Estimates for Uncertainty. *Int. J. Life Cycle Assess.* **2013**, *19*, 429−436.
(23) Althaus, H.; Doka, G.; Dones, R.; Heck, T.; Hellweg, S.; Hischier, R.; Nemecek, T.; Rebitzer, G.; et al. *Overview and Methodology: Data v2.0 (2007)*; Frischknecht, R., Jungbluth, N., Eds.; Ecoinvent Report No. 1, Dübendorf, Switzerland, 2007; http://www.ecoinvent.org/files/200712_frischknecht_jungbluth_overview_methodology_ecoinvent2.pdf.
(24) Henriksson, P. J. G.; Guinée, J. B.; Heijungs, R.; de Koning, A.; Green, D. M. A protocol for horizontal averaging of unit process data - including estimates for uncertainty. *Int. J. Life Cycle Assess.* **2014**, *19*, 429−436.
(25) Heijungs, R.; Kleijn, R. Numerical Approaches Towards Life Cycle Interpretation. *Int. J. Life Cycle Assess.* **2001**, *6*, 141−148.
(26) Huijbregts, M. a. J.; Gilijamse, W.; Ragas, A. M. J.; Reijnders, L. Evaluating Uncertainty in Environmental Life-Cycle Assessment. A Case Study Comparing Two Insulation Options for a Dutch One-Family Dwelling. *Environ. Sci. Technol.* **2003**, *37*, 2600−2608.
(27) Heijungs, R.; Guinée, J.; Huppes, G.; Lankreije, R.; Udo De Haes, H.; Wegener Sleeswijk, A. Ansems, A.; Eggels, P.; van Duin, R.; de Goede, H. *Environmental Life Cycle Assessment of Products: Guide and Backgrounds*; Leiden University, Leiden, The Netherlands, 1992; https://openaccess.leidenuniv.nl/handle/1887/8061.
(28) *Working Group I Contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change, Climate Change 2013: The Physical Science Basis*; Final Draft Underlying Scientific-Technical Assessment; IPCC, Geneva, Switzerland, 2013; http://www.climatechange2013.org/report/review-drafts/.
(29) Myhre, G.; Shindell, D.; Bréon, F.-M.; Collins, J.; Fuglestvedt, J.; Huang, J.; Koch, D.; Lamarque, J.-F.; Lee, D.; Mendoza, B.; et al. Anthropogenic and Natural Radiative Forcing, Supplementary Information. In *Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change, Climate Change 2013: The Physical Science Basis*; Stocker, T.; Qin, D.; Plattner, G.-K.; Tignor, M.; Allen, S.; Boschung, J.; Nauels, A.; Xia, Y.; Midgley, P., Eds.; IPCC, Geneva, Switzerland, 2013; p 44; https://www.ipcc.ch/pdf/assessment-report/ar5/wg1/supplementary/WG1AR5_Ch08SM_FINAL.pdf.
(30) Rico, A.; Phu, T. M.; Satapornvanit, K.; Min, J.; Shahabuddin, A. M.; Henriksson, P. J. G.; Murray, F. J.; Little, D. C.; Dalsgaard, A.; Van den Brink, P. J. Use of veterinary medicines, feed additives and probiotics in four major internationally traded aquaculture species farmed in Asia. *Aquaculture* **2013**, *412*−*413*, 231−243.
(31) Rico, A.; Van den Brink, P. J. Probabilistic risk assessment of veterinary medicines applied to four major aquaculture species produced in Asia. *Sci. Total Environ.* **2014**, *468*−*469*, 630−641.
(32) Heijungs, R.; Suh, S. *The Computational Structure of Life Cycle Assessment*; Eco-Efficiency in Industry and Science, Vol. *11*; Kluwer Academic Publishers: Dordrecht, The Netherlands, 2002.
(33) Narum, S. R. Beyond Bonferroni: Less conservative analyses for conservation genetics. *Conserv. Genet.* **2006**, *7*, 783−787.
(34) Fay, M.; Kim, H.; Hachey, M. On using truncated sequential probability ratio test boundaries for Monte Carlo implementation of hypothesis tests. *J. Comput. Graph. Stat.* **2007**, *16*, 946−967.
(35) Cohen, J. The earth is round (p<.05). *Am. Psychol.* **1994**, *49*, 997−1003.
(36) McCloskey, D.; Ziliak, S. The standard error of regressions. *J. Econ. Lit.* **1996**, *34*, 97−114.
(37) Doweyko, A. M. QSAR: dead or alive? *J. Comput.-Aided Mol. Des.* **2008**, *22*, 81−89.
(38) Ewoukem, T. E.; Aubin, J.; Mikolasek, O.; Corson, M. S.; Eyango, M. T.; Tchoumboue, J.; van der Werf, H. M. G.; Ombredane, D. Environmental impacts of farms integrating aquaculture and agriculture in Cameroon. *J. Cleaner Prod.* **2012**, *28*, 208−214.
(39) Pelletier, N.; Tyedmers, P.; Sonesson, U.; Scholz, A.; Ziegler, F.; Flysjo, A.; Kruse, S.; Cancino, B.; Silverman, H. Not All Salmon Are Created Equal: Life Cycle Assessment (LCA) of Global Salmon Farming Systems. *Environ. Sci. Technol.* **2009**, *43*, 8730−8736.
(40) Naylor, R. L.; Goldburg, R. J.; Primavera, J. H.; Kautsky, N.; Beveridge, M. C. M.; Clay, J.; Folke, C.; Lubchenco, J.; Mooney, H.; Troell, M. Effect of aquaculture on world fish supplies. *Nature* **2000**, *405*, 1017−1024.
(41) Cao, L.; Naylor, R. L.; Henriksson, P. J. G.; Leadbitter, D.; Metian, M.; Troell, M.; Zhang, W. China's aquaculture and the world's wild fisheries. *Science (Washington, DC, U. S.)* **2015**, *347*, 133−135.
(42) Newton, R.; Telfer, T.; Little, D. Perspectives on the utilization of aquaculture coproduct in Europe and Asia: prospects for value addition and improved resource efficiency. *Crit. Rev. Food Sci. Nutr.* **2014**, *54*, 495−510.
(43) Guinée, J. B.; Heijungs, R. A proposal for the classification of toxic substances within the framework of life cycle assessment of products. *Chemosphere* **1993**, *26*, 1925−1944.
(44) Pinsonnault, A.; Lesage, P.; Levasseur, A.; Samson, R. Temporal differentiation of background systems in LCA: relevance of adding temporal information in LCI databases. *Int. J. Life Cycle Assess.* **2014**, *19*, 1843−1853.
(45) Ford, J. S.; Pelletier, N. L.; Ziegler, F.; Scholz, A. J.; Tyedmers, P. H.; Sonesson, U.; Kruse, S. a.; Silverman, H. Proposed Local Ecological Impact Categories and Indicators for Life Cycle Assessment of Aquaculture. *J. Ind. Ecol.* **2012**, *16*, 254−265.
(46) Hauschild, M. Z.; Goedkoop, M.; Guinée, J.; Heijungs, R.; Huijbregts, M.; Jolliet, O.; Margni, M.; Schryver, A. D.; Humbert, S.; Laurent, A.; et al. Identifying best existing practice for characterization modeling in life cycle impact assessment. *Int. J. Life Cycle Assess.* **2013**, *18*, 683−697.
(47) Nemecek, T.; Schnetzer, J. *Methods of assessment of direct field emissions for LCIs of agricultural production systems*. Agroscope Reckenholz-Tänikon Research Station (ART), Zurich, Switzerland, 2011; file:///C:/Users/Scott/Downloads/ART%202012%20-%20Methods%20of%20assessment%20of%20direct%20field%20emissions%20for%20agricultural%20systems.pdf.
(48) Maurice, B.; Frischknecht, R.; Coelho-Schwirtz, V.; Hungerbühler, K. Uncertainty analysis in life cycle inventory. Application to the production of electricity with French coal power plants. *J. Cleaner Prod.* **2000**, *8*, 95−108.
(49) Björklund, A. E. LCA Methodology Survey of Approaches to Improve Reliability in LCA. *Int. J. Life Cycle Assess.* **2002**, *7*, 64−72.
(50) Ptacnik, R.; Andersen, T.; Tamminen, T. Performance of the Redfield Ratio and a Family of Nutrient Limitation Indicators as Thresholds for Phytoplankton N vs. P Limitation. *Ecosystems* **2010**, *13*, 1201−1214.
(51) Posch, M.; Seppälä, J.; Hettelingh, J.-P.; Johansson, M.; Margni, M.; Jolliet, O. The role of atmospheric dispersion models and ecosystem sensitivity in the determination of characterisation factors for acidifying and eutrophying emissions in LCIA. *Int. J. Life Cycle Assess.* **2008**, *13*, 477−486.
(52) Gallego, A.; Rodríguez, L.; Hospido, A.; Moreira, M. T.; Feijoo, G. Development of regional characterization factors for aquatic eutrophication. *Int. J. Life Cycle Assess.* **2010**, *15*, 32−43.
(53) Jung, J.; von der Assen, N.; Bardow, A. Sensitivity coefficient-based uncertainty analysis for multi-functionality in LCA. *Int. J. Life Cycle Assess.* **2014**, *19*, 661−676.
(54) Beltran, A.; Guinée, J.; Heijungs, R. A statistical approach to deal with uncertainty due to the choice of allocation methods in LCA. *Proceedings of the 9th International Conference on Life Cycle Assessment in the Agri-Food Sector*, San Francisco, CA, October 8−10, 2014; http://lcafood2014.org/papers/163.pdf.

DOI: 10.1021/acs.est.5b04634
Environ. Sci. Technol. 2015, 49, 14176−14183