

# Proposed LCA methodology revisions for aquaculture SEAT Life Cycle Assessment studies

- Focusing on South-East Asian Aquaculture Systems for Tilapia (*Oreochromis spp.*), Catfish (*Pangasius spp.*), Shrimp (*Penaeid spp.*) and Freshwater prawn (*Macrobrachium spp.*) -

Final version, February 2011

Patrik Henriksson  
Jeroen Guinée  
Rene Kleijn







# Contents

Contents .....	4
Preface.....	6
Executive summary.....	7
1.1 Introduction.....	11
2 Review of existing aquaculture LCA studies.....	13
2.1 Introduction.....	13
2.2 Life Cycle Assessment and Aquaculture .....	13
2.3 Goal and scope definition .....	15
2.3.1 Goal .....	15
2.3.2 Scope .....	15
2.3.2.1 Attributional or consequential.....	15
2.3.2.2 Temporal coverage.....	16
2.3.2.3 Spatial coverage .....	16
2.3.2.4 Technology coverage .....	17
2.3.2.5 Functional unit .....	17
2.4 Inventory analysis .....	18
2.4.1 System boundaries.....	18
2.4.2 Data availability and quality .....	19
2.4.3 Allocation .....	20
2.5 Life Cycle Impact assessment (LCIA).....	21
2.5.1 Impact categories.....	21
2.5.2 Characterization methods for baseline impact categories .....	24
2.5.2.1 Global Warming.....	24
2.5.2.2 Acidification .....	24
2.5.2.3 Eutrophication.....	25
2.5.2.4 Abiotic Resource use .....	26
2.5.2.5 Toxicity.....	26
2.5.2.6 Photochemical ozone formation .....	27
2.5.3 Characterization methods for non-baseline (“novel”) impact categories.....	28
2.5.3.1 Cumulative energy demand .....	28
2.5.3.2 Biotic resource use.....	29
2.5.3.3 Water dependence .....	29
2.5.3.4 Land use .....	30
2.5.3.5 Other impact categories .....	31
2.5.4 Normalisation and weighting .....	31
2.6 Interpretation.....	32
2.6.1 Contribution analysis.....	32
2.6.2 Sensitivity analysis .....	32
3 Selection of topics for which LCA methodology revisions are needed.....	35
3.1 Topics from the review .....	35
3.2 Discussion and selection .....	37
4 Revision proposals and their practical feasibility for selected topics .....	39
4.1 Topic 1: water use .....	39
4.2 Topic 2: salinity .....	43
4.3 Topic 3: benthic impacts .....	44

4.4 Topic 4: developing (Monte-Carlo based) methods to include multiple variations in parameters and choices / assumptions .....44

5 How will SEAT LCAs thus look like? .....46

6 Acknowledgement .....49

References.....50

Annex 1: Terms & abbreviations .....58

## Preface

Aquaculture has, over the last decades, grown faster than any other animal production sector and today supplies half of the world's finfish. Simultaneously the growth of the import of Asian aquatic products into the EU has increased steadily. Current EU policy supporting international trade between Asia and Europe concentrates on issues of food safety as measures of quality, whilst market-forces drive development of standards and labels that identify social and environmental parameters. The SEAT (*Sustaining Ethical Aquatic Trade*) project proposes to establish an evidence-based framework to support current and future stakeholder dialogues organised by third party certifiers. This will contribute to harmonising standards, helping consumers to make fully informed choices with regards to the sustainability and safety of their seafood. The 'Ethical Aquatic Food Index' (EAFI), a qualitative holistic measure of overall sustainability intended to support consumers' purchasing decisions, will be based on detailed research centred on a Life Cycle Assessment (LCA) of current processes. Systems thinking will be applied to analyse livelihood impacts along the global value chains (GVCs) of four farmed aquatic products, Tilapia (*Oreochromis spp.*), Shrimp (*Penaeid spp.*), Catfish (*Pangasius spp.*) and Freshwater prawn (*Macrobrachium spp.*) in China, Thailand, Vietnam and Bangladesh, all major producing countries. Initial assessments of environmental impacts by and on aquatic production and processing systems, and impacts on product safety and social equity will lead towards prioritisation of critical issues and supportive action research. Micro, small and medium enterprises (MSMEs) based in the EU and Asia will participate in this process, enhancing their relative competitiveness. By strengthening the knowledge base surrounding EU-Asia seafood trade the project will provide the evidence required to support further expansion whilst ensuring a fair deal for producers who are meeting appropriate social and environmental goals and offering a safe and sustainable product for consumers.

This document is deliverable D3.2 of work package 3 (WP3) of the SEAT project. D3.2 concerns a report on LCA methodology. In this report we will look into specific methodological issues for LCAs on aquaculture systems. For that, first a review of existing LCA studies on aquaculture systems has been made. Based on that, methodological problems were evaluated and a selection of issues was made for which further developments and/or improvements are suggested based on existing proposals. Based on the results of the review and further analyses, a basic outline of the SEAT LCA case studies is provided.

## Executive summary

Aquaculture production has increased over tenfold during the last three decades. It is expected that production will continue to grow to about 80 million tons per year, almost equalling the magnitude of fisheries. Europe's share in global aquaculture production is about 4.5 %, whereas Asia's share is about 88.9 %. Important European aquaculture products include salmon (Norway, Iceland and Scotland), sea bass and sea bream in the Mediterranean sea (Turkey, Greece, Italy, Spain) and mussels and oysters (Spain, France, Netherlands, Ireland). China is the biggest runner-up in the fast expansion of Asian aquaculture production. Important Asian aquaculture products include green mussels, shrimps, prawns, tilapia and catfish. Sustaining Ethical Aquaculture Trade (SEAT) is a large collaborative project within the "Food, Agriculture and Fisheries, and Biotechnology" theme of the EU 7<sup>th</sup> Framework Programme (FP). The overall aim of the SEAT project is to enhance the sustainability (environmental impact, social justice, economic efficiency, nutritional quality and safety) of four major aquatic food commodities farmed in Asia and exported to Europe by developing an improved framework for sustainability assessment of the trade in farmed aquatic products between Asia and Europe. Until now a range of different sustainability tools has been used to assess aquaculture production systems. Increasingly LCA (Life Cycle Assessment) is used for industrial and agricultural production, and since 2004 LCA has also been increasingly applied to aquaculture systems. Previous LCA studies within the aquaculture sector have mainly focused on production in developed countries, while the sector is dominated by developing countries. The application of LCA to some major aquaculture systems in Asia, which is one of the goals of the SEAT project, will therefore be an important step in the understanding of sustainability concerns involved with EU seafood imports.

This document is deliverable D3.2 of work package 3 (WP3) of the SEAT project. D3.2 concerns a report on LCA methodology. In this report we will look into specific methodological issues for LCAs on aquaculture systems. For that, first a review of existing LCA studies on aquaculture systems has been made. Based on that, methodological problems were evaluated and a selection of issues was made for which further developments and/or improvements are suggested based on existing proposals. Based on the results of the review and further analyses, a basic outline of the SEAT LCA case studies is provided:

Phase/step	Choices made
<b>Goal and scope definition</b>	
Goals	<ul style="list-style-type: none"> <li>Getting insight in the environmental impact and its causes of aquaculture systems for tilapia, catfish, shrimp and prawn in China, Vietnam, Thailand and Bangladesh.</li> <li>Getting insight in starting points ("hot spot identification") for improving the environmental performance of aquaculture systems for tilapia, catfish, shrimp and prawn China, Vietnam, Thailand and Bangladesh, which includes insight into the effects of choices in methods and data on the outcomes.</li> <li>Learning.</li> </ul>
Attributional or consequential	Attributional. <i>This topic will be addressed in a scientific paper, separate from individual LCA studies.</i>

<b>Phase/step</b>	<b>Choices made</b>
Temporal coverage	Average data over longer periods will be collected, properly reflecting potential fluctuations. For collection of farm data the reference year is 2010; for other processes a reference year will be adopted as close as possible to 2010.
Spatial coverage	Thailand: Chanthaburi and Surat Thani. China: Guangdong province and Hainan Island. Vietnam: Mekong delta. Bangladesh: Khulna and possible other regions.
Technology coverage	Different farming systems and mixed technologies will be covered.
Functional unit	1000 kg of edible yield (either or not corrected for nutritional value) of (frozen, packed or etc. is to be determined later) species X produced on farm type Y in country Z for consumption in the EU (see D2.4).
<b>Inventory analysis</b>	
System boundaries	A system boundary from farm - including the production of all upstream inputs to the farm - to consumption (“from farm to fork”) is adopted. Human consumption of fish itself is not included as explained in deliverable D2.4. Where applicable and appropriate we will apply EIOA (extended Input-Output Analysis) to estimate the potential significance of lacking data.
Data availability and quality	Main efforts will be put into collection of Asia foreground data of the actual aquaculture processes including fisheries, local feeds, nurseries, hatcheries, farms etc. correcting for losses due to parasites and diseases and including risks of failed production caused by intensive farming in the production figures, and paying attention to the public availability (or non-disclosure) of the resulting database. For this national life cycle inventory databases (if existing; Thailand, e.g., has such a database), peer-reviewed journals and technical reports from recognised organisations will be consulted.
Allocation	Thorough justification will be provided for allocation choices made. In addition, sensitivity analyses on these allocation choices will be conducted in compliance with ISO standards.
<b>Life Cycle Impact assessment (LCIA)</b>	
Global Warming	Latest Global Warming Potentials (GWPs) from IPCC (Intergovernmental Panel on Climate Change).
Acidification	Updated Guinée <i>et al.</i> (2002) baseline method for acidification and other more recent (variants of baseline) methods and characterisation factors, replenished by the midpoint ILCD LCIA methods and factors for acidification.
Eutrophication	Updated Guinée <i>et al.</i> (2002) baseline method for eutrophication and other more recent (variants of baseline) methods and characterisation factors, replenished by the midpoint ILCD LCIA methods and factors for eutrophication.
Abiotic Resource use	Updated Guinée <i>et al.</i> (2002) ADP (Abiotic Depletion Potential) method for abiotic depletion and with other more recent (variants of baseline) methods and characterisation factors, replenished by the midpoint ILCD LCIA methods and factors for abiotic depletion.
Toxicity	USEtox™ model with <i>additional CFs calculated for specific aquaculture chemicals for which CFs may be lacking</i> in the current USEtox CF-list.

Phase/step	Choices made
Photochemical ozone formation	Updated Guinée <i>et al.</i> (2002) baseline method for photochemical ozone formation and other more recent (variants of baseline) methods and characterisation factors, replenished by the midpoint ILCD LCIA methods and factors for photochemical ozone formation and updated POCPs (Photochemical Ozone Creation Potentials) for Southern Asia by Cheng <i>et al.</i> (2010).
Cumulative energy demand (CED)	CED will be included as a non-baseline indicator. The results of this indicator will be presented separately stressing the overlap it has with particularly abiotic depletion.
Water use	The Bayart <i>et al.</i> (2010) terminology and suggestions for LCI (Life Cycle Inventory) are adopted. Their future suggestions for impact assessment methodologies will be considered in a later stage of the SEAT project. The water quality aspect from freshwater use inventories and impact assessment is excluded from the SEAT LCAs. Within the SEAT LCAs, the following LCI data will be collected for freshwater use: <ul style="list-style-type: none"> <li>• withdrawal/ extraction in m<sup>3</sup> specified, if possible, in resource type (groundwater, surface water from lake, surface water from river, ..), spatial (watershed indication or GIS coordinates) and in temporal terms (dry or wet season);</li> <li>• emission extraction in m<sup>3</sup> specified, if possible, in resource type (to groundwater, to surface water lake, to surface water river, ..), spatial (watershed indication or GIS coordinates) and in temporal terms (dry or wet season).</li> </ul>
Land use	Where possible and relevant, indirect and direct land use occupation (in m <sup>2</sup> ·yr) and transformation (in m <sup>2</sup> ) will be inventoried. On top of this, the recommended midpoint ILCD LCIA method for land use and/or the UNEP-SETAC's operational characterization factors for land use impacts on biodiversity and ecosystem services will be applied, if feasible. Since land use impacts cannot (yet) be properly handled in LCA, other WPs of the SEAT project will have to supply additional information on land use impacts from aquaculture systems.
Biotic resource use	Will possibly be included depending on the results of the <i>scientific paper, reviewing the inter-relations between biotic resource use, resource depletion, land use and biodiversity impacts. In this paper the available methods within the LCA community and outside that community will be used as a starting point. A framework for addressing these impacts within LCA (or outside LCA) will be proposed.</i>
Salinity	It is currently not considered as feasible to include this impact category into the SEAT project. The data and modeling efforts needed in relation to the expected relevance of salinity impacts with respect to aquaculture systems is considered too huge.
Benthic impacts (from capture fisheries)	It is currently not considered as feasible to include this impact category into the SEAT project. Only if the work by Ziegler <i>et al.</i> at SIK (The Swedish Institute for Food and Biotechnology) provides practically applicable results, this decision may be reconsidered.
Normalisation and weighting	Updated CML normalisation methods will be applied. In principle, weighting will not be performed, since weighting is not needed for the prime goals of the SEAT LCAs. If weighting is

Phase/step	Choices made
	needed as yet, one or more weighting methods from a recent study on weighting methods by Huppel & Oers will be adopted.
<b>Interpretation</b>	
Contribution analysis	Contribution analyses will be performed at all relevant levels (inventory analysis, characterization and, if useful, normalization) and for different elements (processes and interventions). Results of these analyses will be used for tracing possible errors and as basis for identifying improvement options.
Sensitivity analysis	Several kinds of sensitivity analysis will be performed on a broad range of data and methodological choices. For this, <i>the use and development of (Monte-Carlo based) methods to include multiple variations in parameters and choices/assumptions will be explored.</i>

# 1 Introduction

Sustaining Ethical Aquaculture Trade (SEAT) is a large collaborative project within the “Food, Agriculture and Fisheries, and Biotechnology” theme of the EU 7<sup>th</sup> Framework Programme (FP). The overall aim of the SEAT project is to enhance the sustainability (environmental impact, social justice, economic efficiency, nutritional quality and safety) of four major aquatic food commodities farmed in Asia and exported to Europe by developing an improved framework for sustainability assessment of the trade in farmed aquatic products between Asia and Europe.

Until now a range of different sustainability tools has been used to assess aquaculture production systems. Increasingly LCA (Life Cycle Assessment) is used for industrial and agricultural production, and since 2004 LCA has also been increasingly applied to aquaculture systems although the number of cases is still limited (see Guinée *et al.* 2010; see also section 2.2, Table 2 of this report).

Previous LCA studies within the aquaculture sector have mainly focused on production in developed countries (non-shaded rows in Table 2), while the sector is dominated by developing countries. The application of LCA to some major aquaculture systems in Asia, which is one of the goals of the SEAT project (Guinée *et al.* 2010), will therefore be an important step in the understanding of sustainability concerns involved with EU seafood imports.

Although LCA is a quite well-developed and ISO-standardized tool (ISO, 2006a; ISO, 2006b), LCA is not a “silver bullet”. It focuses on an environmental analyses of an as broad as possible range of impact over the whole life-cycle of the aquaculture systems considered. It however does not address all sustainability dimensions and it even cannot address all environmental impacts properly. It is therefore explicitly placed among a portfolio of other tools, like risk assessment (RA), life cycle costing (LCC), global value chain (GVC), social, and ethical analyses. Recently, Klöpffer & Renner (2008), Klöpffer (2008), Zamagni *et al.* (2009), Heijungs *et al.* (2010) and Guinée *et al.* (*accepted*) have advocated broadening and deepening current environmental LCA to Life Cycle Sustainability Analysis (LCSA) including social and economic aspects and, for example, economic mechanisms into a life-cycle based analysis<sup>1</sup>. LCSA is expected to be a trans-disciplinary integration framework of models rather than a model in itself. LCSA works with a plethora of disciplinary models and guides selecting the proper ones, given a specific sustainability question. Structuring, selecting and making the plethora of disciplinary models practically available in relation to different types of life cycle sustainability questions is a major challenge and still requires a lot of research. Therefore, this study will work on the basis of LCA as described in Guinée *et al.* (2010) and from there on try to add other elements to it (e.g., LCC) as far as possible and feasible in practice.

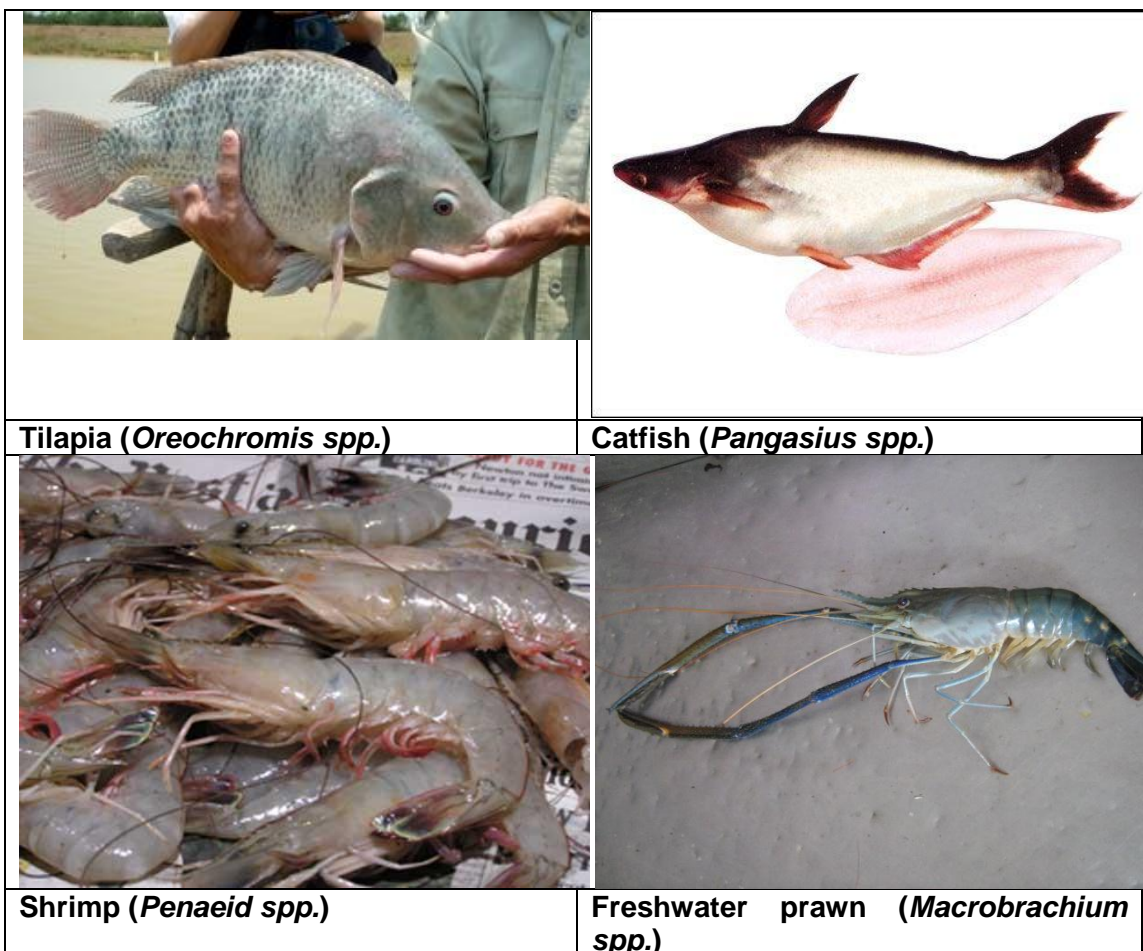
But even if broadened and deepened, LCSA is still not a “silver bullet”. LCSA is a framework for looking from one viewpoint, i.e. the life cycle viewpoint, to sustainability questions and only providing life cycle answers and no other; risk assessment (RA) is, for example, not part of this framework and is still needed as a complementary tool, as well as other tools and approaches.

The SEAT project focuses on four major cultured aquatic commodities and four producer countries in Asia (Table 1) and so will the LCA studies.

Table 1: Scope of the SEAT project in terms of cultured species.

<sup>1</sup> For a brief introduction into LCA, we here refer to annex 1 of Guinée *et al.* (2010).

Country/Species	Tilapia ( <i>Oreochromis</i> <i>spp.</i> )	Catfish ( <i>Pangasius</i> <i>spp.</i> )	Shrimp ( <i>Penaeid</i> <i>spp.</i> )	Freshwater prawn ( <i>Macrobrachium</i> <i>spp.</i> )
China	√√	√	√	?
Vietnam	√	√√	√√	(√)
Thailand	√	√	√√	√
Bangladesh	(√)	√	√√	√√
Europe	(√)	(o)	(o)	(o)



This document is deliverable D3.2 of work package 3 (WP3) of the SEAT project. D3.2 concerns a report on LCA methodology. In this report we will look into specific methodological issues for LCAs on aquaculture systems. For that, first a review of existing LCA studies on aquaculture systems has been made (Chapter 2). Based on that methodological problems were evaluated and a selection of issues was made (Chapter 3) for which further developments and/or improvements are suggested based on existing proposals (Chapter 4). Finally, the basic outline of the SEAT LCA case studies will be summarized (Chapter 5).

## 2 Review of existing aquaculture LCA studies

### 2.1 Introduction

Historical increases in food production have mainly been achieved by expanding fisheries, increasing farmland area, genetic selection, fertilizer and pesticide use, many of which are reaching their limits in enhancing production (Borlaug 2007; FAO 2009; Godfray *et al.* 2010). Global fish landings levelled off around 1990, followed by a six-fold increase in fuel consumption due to longer distances to productive fishing grounds and larger efforts needed to maintain catches (FAO 2009; Tlustý & Lagueux 2009). Expansions of farmland area, on the other hand, are challenged by competition with residential areas, bio-fuel, -fibre, feed and food production, biogenic carbon storage (in biotic stocks, e.g. natural forests) and nature conservation (Dias de Oliveira *et al.* 2005; FAO 2009; Godfray *et al.* 2010). Net additions to the world's food pool therefore needs to be achieved by efficiency improvements of existing production systems, better utilization of current food stocks and alternative production systems. Aquaculture has since the 1970s proven its ability, as an alternative production system, that can contribute in feeding a growing population; with global average per capita supply growing from 0.7 kg to 7.8 kg (FAO 2009).

Farming methods of aquatic organisms are highly diverse and the majority of production is conducted in developing countries on small scale for family consumption or local markets (Boyd *et al.* 2007). Aquatic plants and bivalves constitute 44 % of global production (by weight) while other systems are more dependent upon external feed inputs (Tacon & Metian 2008). Finfish make up 63.7 % of current aquaculture production (by weight, excluding seaweeds) and are in turn dominated by Cyprinid fishes that constitute almost 70 % of the world's farmed fin fishes (FAO 2009). The most common practice for farming finfish is either in ponds, in cages or more recently in land-based tanks. During the last decade the production of crustaceans has increased rapidly as well as marine finfish production, both generating high values (FAO 2009).

The rapid expansion of the sector has, however, been associated with many sustainability concerns such as emissions leading to climate change, eutrophication, (eco)toxic impacts, antibiotic use, land and water use needed for feed production, loss of biodiversity, introduction of non-indigenous species, spread/amplification of parasites and diseases, genetic pollution, dependence on capture fisheries and socio-economic concerns (Lewis 1997; Folke *et al.* 1998; Naylor *et al.* 2000; Naylor *et al.* 2001; Pauly *et al.* 2002; Brummett 2007; Pelletier & Tyedmers 2008; Naylor *et al.* 2009). In the process of trying to understand the true impacts of aquaculture, Life Cycle Assessment (LCA) has become a more frequently implemented complimentary tool for identifying best practices and estimate overall environmental performance (Pelletier & Tyedmers 2008).

In this paper, we review existing LCA studies in the field of aquaculture. The ambition is to clarify the methodological choices made in peer-reviewed published literature, identify methodological and data gaps and provide recommendations for future development of this field of research.

### 2.2 Life Cycle Assessment and Aquaculture

Originating from industrial production and processes, the initial LCA studies on food products were published in the early 1990s (Andersson *et al.* 1994). This has allowed for the identification of best production, processing, packing, distribution and

consumption patterns; as well as an overall understanding of the environmental burdens associated with the production of food (Roy *et al.* 2009). The first peer-reviewed published aquaculture related LCA was performed by Papatryphon *et al.* (2004) evaluating salmon feed. This was followed by an increasing number of full LCA studies<sup>2</sup> towards the second half of this decade (Table 2). The investigated production chains differ from feed production to post consumption. The research area has been dominated by the French INRA and IFRMER research institutes and by Dalhousie University, Canada. The overall effort has allowed for the identification of best production, processing, packing, distribution and consumption patterns, as well as an overall understanding of the environmental burdens associated with the production of food (Roy *et al.* 2009).

Author	Title	Journal
<b>Papatryphon <i>et al.</i> 2004</b>	Environmental impact assessment of salmonid feeds using Life Cycle Assessment (LCA)	Ambio
<b>Mungkung 2005</b>	Shrimp aquaculture in Thailand: application of life cycle assessment to support sustainable development	PhD thesis
<b>Aubin <i>et al.</i> 2006</b>	Characterisation of the environmental impact of a turbot ( <i>Scophthalmus maximus</i> ) re-circulating production systems using Life Cycle Assessment	Aquaculture
<b>Ellingsen &amp; Aanondsen 2006</b>	Environmental Impacts of Wild Caught Cod and Farmed Salmon – A Comparison with Chicken	Int. Journal of LCA
<b>Grönroos <i>et al.</i> 2006</b>	Life cycle assessment of Finnish cultivated rainbow trout	Boreal Environ. Research
<b>Pelletier &amp; Tyedmers 2007</b>	Feeding farmed salmon: Is organic better?	Aquaculture
<b>Aubin <i>et al.</i> 2009</b>	Assessment of the environmental impact of carnivorous finfish production systems using life cycle assessment	J. of cleaner production
<b>Ayer &amp; Tyedmers 2009</b>	Assessing alternative aquaculture technologies: life cycle assessment of salmonid culture systems in Canada	J. of cleaner production
<b>d'Orbcastel <i>et al.</i> 2009</b>	Towards environmentally sustainable aquaculture: Comparison between two trout farming systems using Life Cycle Assessment	Aquacultural Engineering
<b>Pelletier <i>et al.</i> 2009</b>	Not all salmon are created equal: Life cycle assessment (LCA) of global salmon farming systems	Environ. Sci. and Technol.
<b>Iribarren <i>et al.</i> 2010</b>	Revisiting the Life Cycle Assessment of mussels from a sectorial perspective	J. of cleaner production
<b>Pelletier &amp; Tyedmers 2010</b>	A life cycle assessment of frozen Indonesian tilapia fillets from lake and pond-based production systems	J. of Industrial Ecology

<sup>2</sup> As Papatryphon *et al.* (2004) limit their study to feed production, their study is not considered to be a full LCA and will therefore not be part of the in depth discussion afterwards.

Author	Title	Journal
Phong 2010	Dynamics of sustainability in Integrated Agriculture-Aquaculture systems in the Mekong Delta	PhD thesis

Table 2: LCA studies on aquaculture systems in order of publication year.

Below, we will discuss the peer-reviewed LCA studies listed in Table 2 with respect to the assumptions and choices made and data sources adopted for the different steps of Goal and Scope definition, Inventory Analysis, Impact Assessment and Interpretation. For each step, a brief description of its subject, methodological similarities and differences between the studies, and gaps and research or harmonization needs will be discussed briefly.

## 2.3 Goal and scope definition

### 2.3.1 Goal

The goal definition is one of the crucial steps of an LCA in order to comply with the ISO standard. It should include the intended application of the study, the reason for carrying out the study, the intended audience and whether it will produce comparative or disclosed results (ISO 14040). This step aims to provide guidance throughout the LCA study, to support the decision process and to make information easily available for interpreters.

Despite distinguishing the need for a goal definition, the extent and coverage of it differs substantially between studies. Most studies include the intent of their study, as well as ambition; while intended application and audience are generally missing. Ayer & Tyedmers (2009) do specify a complete goal definition; however, partially under different headings. Only Mungkung (2005; see also Mungkung *et al.* 2006) systematically distinguishes and describes a complete goal definition.

Overall, the need for a goal definition is commonly highlighted in the studies while much of the required content seems to be lost in the process of merging the definition into other topics, such as material and methods. It is therefore recommended to isolate the goal definition in order to agree with the ISO guidelines and to guide readers.

*As written in deliverable D2.4 (Guinée et al. 2010), the main goal of this LCA study is formulated as getting insight in:*

- *the environmental impact and its causes of aquaculture systems for tilapia, catfish, shrimp and prawn in China, Vietnam, Thailand and Bangladesh.*
- *starting points (“hot spot identification”) for improving the environmental performance of aquaculture systems for tilapia, catfish, shrimp and prawn in China, Vietnam, Thailand and Bangladesh, which includes insight into the effects of choices in methods and data on the outcomes.*

*On top of these main goals, learning (of the environmental ins and outs of aquaculture systems) is another important goal of this study.*

### 2.3.2 Scope

#### 2.3.2.1 Attributional or consequential

All studies in this review do not explicitly mention whether they adopted an attributional or a consequential LCA (CLCA) approach. Some of the studies apply system expansion for solving allocation problems. Schmidt refers to these approaches as semi-consequential (Schmidt 2010); see also D2.4 Goal and Scope

Report (Guinée *et al.* 2010). Little attention has though been given to macro-scale effects, despite that many of the feed inputs derive from expanding markets of internationally traded commodities associated with environmental concerns such as deforestation and collapsing fish stocks (Naylor *et al.* 2009; Galford *et al.* 2010). *As stated in deliverable D2.4 (Guinée et al. 2010) we will basically adopt an attributional approach in this study. In a later stage, when improvement options could be modelled, we may adopt an 'adapted' CLCA approach if available at that time. At least, we will draft a paper reviewing current practice of CLCA including an outlook into possible futures of CLCA.*

### **2.3.2.2 Temporal coverage**

The studies listed in Table 2 cover the period 2004-2010. Data used do not always reflect this period as most data stem from background databases covering data from 1990-2000.

Agricultural crop yields may experience large annual fluctuations (Naylor *et al.* 1997; Rööös *et al.* 2010). These fluctuations are driven by climate and weather events as well as fluctuations in oil prices due to the energy intensive nature of current food production (Naylor *et al.* 1997; Sandweiss *et al.* 2004). Fish stocks utilized as feed for aquaculture systems also experience annual fluctuations with the South American anchoveta fishery, the world's largest reduction fishery, being strongly influenced by El Niño-Southern Oscillation (ENSO) events (Sandweiss *et al.* 2004).

Temporal scales also influence consequences of farm emissions as flushing to remove residues from the pond sediments or draining at harvest may result in short term heavy polluting, eutrophication and spread of disease (Dierberg & Kiattisimkul 1996; Avnimelech & Ritvo 2003). The consequences of emissions will further depend upon seasonal changes, as monsoons and droughts will influence the carrying capacity and buffering potential of local ecosystems (Tang *et al.* 2004). Pollution spikes can therefore exceed the carrying capacity of surrounding ecosystems resulting in demise of wildlife, harmful algal blooms and anaerobic events.

*It is thus important to collect data as averages over longer periods properly reflecting these potential fluctuations. The reference year for data collection on farms is 2010; for other processes a reference year will be adopted as close as possible to 2010.*

### **2.3.2.3 Spatial coverage**

The studies reviewed here mainly focus on production systems in developed countries, while only two studies deal with aquaculture systems in developing countries (Mungkung 2005; Phong *et al.* 2010). Moreover, studies by the French INRA and IFRMER research institutes and by Dalhousie University, Canada mainly focused on aquaculture systems in their country of residence (France and Canada). Spatial scales may also be important in the impact assessment modelling of emissions (Potting & Hauschild 1997). Whereas for impact categories such as global warming spatial differences are hardly relevant, they may be very relevant for more regional impact categories such as eutrophication (Gallego *et al.* 2010). The carrying capacity of the water for eutrophying emissions depends on tidal flushing, current and assimilative capacity of the receiving water body (Wu 1995).

*It may thus be advisable to investigate the feasibility of adopting more regionalised impact assessment methods for the more regional- and local-scale impact categories.*

### 2.3.2.4 Technology coverage

All studies reviewed do not make any explicit statements on technology coverage in terms of weighted averages of the actual process mix, best available technologies or worst-operating ones.

The majority of the reviewed studies focuses on finfish production, which can be divided by farming intensity into (Naylor *et al.* 2000; Crespi & Coche 2008):

- *Extensive* – exclusion of predators and control of competitors yielding no more than 500 kg ha<sup>-1</sup> yr<sup>-1</sup>
- *Semi-intensive* – supplementary feed and fry, yielding 0.5 to 20 tonnes ha<sup>-1</sup> yr<sup>-1</sup>
- *Intensive* – provision of all nutritional requirements, yielding up to 200 tonnes ha<sup>-1</sup> yr<sup>-1</sup>
- *Hyper/super-intensive* – Usually pumped or gravity supplied water or cage-based, yielding more than 200 tonnes ha<sup>-1</sup> yr<sup>-1</sup>

While the bulk of production happens in extensive to semi-intensive ponds in Asia (Boyd *et al.* 2007), only Phong (2010) described extensive and semi-intensive systems. This is partially driven by a push from many stakeholders towards closed systems as it allows for better rearing conditions while minimizing proximal environmental effects. This in turn has raised the concern of greater energy dependence (Ayer & Tyedmers 2009). Another driver has been to get a better insight into seafood's performance compared to animal products (Ellingsen & Aanonsen 2006).

Apart from this, systems may also be divided into monoculture (one species), polyculture (several species) and integrated multi-trophic (several species of different trophic levels) aquaculture.

Non-fish species and integrated farming systems have been poorly covered by LCA methodology, despite showing potential to improve environmental performance (Troell 2009). Moreover, there is a lack of LCA studies covering offshore farming, which is becoming increasingly popular, as well as different processing techniques. *In line with deliverable D2.4 (Guinée et al. 2010), different farming systems and mixed technologies will be covered in the LCAs performed within the SEAT project mapping their environmental performance as well as identifying options for reducing their impacts.*

### 2.3.2.5 Functional unit

The functional unit describes the main function(s) fulfilled by a production system and indicates how much of this function is considered. In comparative LCAs the functional unit is the reference for the comparison. The choice of functional unit also defines the system boundaries that need to be included in the inventory. For example, if 'frozen fillets in the supermarkets' is chosen as a functional unit, the system boundaries need to be set to include processing, transportation and distribution.

The most commonly used functional unit between the articles discussed in this study is one tonne of live fish at farm gate (6 studies; pre-processing); with two other studies also limiting themselves to quantity at farm gate. Four studies defined their functional unit with regards to edible yield (post-processing; Table 5). A kg of salmon, for example, requires 2.3 kg of round fish compared to a kg of mussel flesh which requires 12.67 kg live mussels (Ellingsen & Aanonsen 2006; Iribarren *et al.* 2010). When edible yield was used this was defined as the main part of the organism that was marketed, in one case in terms of dry weight. Phong (2010) chose to present results using two different functional units, one weight based and one nutritional based, in order to represent nutritional value. This approach also avoided

allocation decisions between outputs from an agri-aquaculture system producing several feed outputs.

The choice of the functional unit is crucial for comparative LCAs between different species as the edible portion and nutritional value of products can differ widely (Mungkung & Gheewala 2007; Roy *et al.* 2009). This is further influenced by cultural differences where the definition of edible will differ depending upon consumer preferences (Phillips 2002; Edwards *et al.* 2004).

*In line with deliverable D2.4 (Guinée *et al.* 2010), it is therefore recommended to choose a post-processing functional unit (1000 kg of edible yield (either or not corrected for nutritional value) of - frozen, packed or etc. is to be determined later - species X produced on farm type Y in country Z for consumption in the EU), especially if the results are aimed for consumer guidance.*

## **2.4 Inventory analysis**

### **2.4.1 System boundaries**

Identifying which processes are to be included within the system boundaries can be difficult as it depends upon the intended audience, application of the study, assumptions made, impact categories, cut-off criteria, data and cost constraints (ISO 14040). In aquaculture, the often stated cradle-to-grave approach of LCA is rarely applied which may be the result of very large production systems (often incorporating both agriculture and livestock production), a wide range of processing methods and difficulties in tracing products to final markets. The level of processing and amount of seafood going to waste also differs between products (FAO 2009; Godfray *et al.* 2010).

Iribarren (2010) linked a great portion of the emissions from the lifecycle of mussel commodities to processing and consumption. The level of processing also influence the longevity of a products, and thereby amount of wasted (Sonesson *et al.* 2005). Infrastructure is often excluded (cut-off) due to the large time investments needed for calculating the total input in relation to the considered small impact (Ayer & Tyedmers 2009). For global warming, eutrophication and acidification, however, the contribution of infrastructure to overall emissions account for between 0 and 19.0 % between the reviewed articles (Table 4).

The key for making cut-offs in the quantification of impacts, however, is a lack of readily accessible data, implying disproportionate expenditure of funds and effort on data collection. Cut-off may substantially influence the outcome of an LCA study, however, and means that 'easy' LCAs come at a price. Today it is possible to better handle the cut-off problem by estimating the environmental interventions associated with flows for which no readily accessible data are available using environmentally extended Input-Output Analysis (EIOA; Suh *et al.* 2004).

Processing, retailing and consumption can significantly contribute to the overall environmental performance, depending upon the product in focus (Thrane, 2004; Thrane *et al.* 2009; Iribarren *et al.* 2010). A system boundary beyond farm-gate is therefore recommended if results are aimed for consumer guidance, as impacts otherwise will be underestimated. Further efforts should also be directed towards expanding current knowledge of infrastructures contribution, as it can significantly contribute to certain impact factors (Frischknecht *et al.* 2007).

*In this study we will adopt a system boundary from farm - including the production of all upstream inputs to the farm - to consumption (“from farm to fork”). Human consumption of fish itself is not included as explained in deliverable D2.4 (Guinée et al. 2010). Where applicable and appropriate we will apply EIOA to estimate the potential significance of lacking data.*

## **2.4.2 Data availability and quality**

Data availability and quality, and particularly reporting of data and their quality, and also inventory results are well-known problems in all LCA studies. It appears an even bigger problem in the aquaculture studies reviewed here. All reviewed studies model - to different extents - relevant agricultural, aquacultural and fishery processes.

Details of these modelling efforts are, however, not or incompletely published. Some studies do not publish any data, neither any inventory results. Others do extensively report data and sometimes also inventory results, but they focus on economic flow data and hardly provide any environmental data. For almost all studies reviewed it is unclear whether background database are used as estimations or real foreground data (site-samples) have been collected.

For example, Ellingsen & Aanonsen (2006) report: “Data are generally collected from various sources by both literature surveys, a study of available data sources, telephone conversations and meetings.” This provides unfortunately no clue as to which processes, data and data sources have been included in the study.

Consequently, reproducing results is often difficult or impossible.

With respect to the background data(bases) adopted by the various studies reviewed (see Table 5), it is striking that the 2006 study by Ellingsen & Aanonsen still uses Frischknecht *et al.* (1996) and BUWAL (1996) data. Furthermore, all studies more or less use the same merely European databases (different versions of ecoinvent), although these will not always properly represent local technologies very well.

Several studies do not report that they use SimaPro software with (all) included databases. As SimaPro includes many different databases (e.g., ecoinvent, US LCI database, US IO dbase, Danish IO dbase, Dutch IO dbase, LCA food dbase, Industry data, Japanese IO dbase, IVAM dbase; see

[http://www.pre.nl/simapro/inventory\\_databases.htm](http://www.pre.nl/simapro/inventory_databases.htm)), the actual data sources adopted by the study remain entirely unclear.

As a result of more user friendly LCA software and more extensive databases, many LCA studies take non-case specific data from the databases included in existing software packages. This will, however, result in highly flawed results as production practices differ widely between regions. Transparency is therefore recommended to allow for proper peer-reviewing of background data, at least to the extent possible in regards to sensitive industry inventory data. A good example how such background information can be supplied without comprising the focus of the article was given by Grönroos *et al.* (2006; see their background report on

<http://www.ymparisto.fi/download.asp?contentid=12889&lan=fi>) and by Pelletier *et al.* (2009) who published a supporting document describing inventories (although hardly any environmental data; see above), core processes, assumptions and calculations.

Those studies that also collected foreground data, focused data collection on the farms, specified data on economic data (feeds, energy inputs etc.) in detail, do not provide insight how they further model these foreground economic data (probably many use the background databases to model the feeds and energy inputs), and as far as foreground environmental data are collected, they focus on N/P balances.

These N/P balances are usually quite detailed balances as most practitioners come from aquaculture backgrounds and are very familiar with these types of balances, and these balances have historically been a major item of concern in aquaculture practices. However, there is very little or no emission data on, e.g., antibiotics,

copper agents for anti-fouling, metal emissions from fertilizers etc. These are obviously very hard to get.

For aquaculture systems the quality of the data available often depends upon the intensity of the system and region of data collection. Highly intensive systems, such as land-based systems of salmon, often keep more complete records of all inputs and outputs; while only general estimations will be available for most extensive pond systems in rural areas.

*Our conclusion from this review is that main efforts should be put into collection of Asian foreground data of the actual aquaculture processes including fisheries, local feeds, nurseries, hatcheries, farms etc. For this national life cycle inventory databases (if existing; Thailand, e.g., has such a database), peer-reviewed journals and technical reports from recognised organisations will be consulted.*

### **2.4.3 Allocation**

Most industrial processes yield more than one product, and they recycle intermediate or discarded products as raw materials. As a result, the materials and energy flows as well as associated environmental releases have to be allocated to the different products according to clearly stated procedures. The most commonly applied procedures are allocation based on mass, economic value, gross energetic content and system expansion. In aquaculture many of the feed inputs are co-produced in other production systems (e.g. rice bran, fisheries by-catch and co-products from livestock processing). Co-products also occur in the processing phase; these are two subsystems where allocation often needs to be implemented.

Some of the major methodological differences between the studies are driven by allocation. Even though all of the above mentioned allocation factors have historically been applied to seafood LCAs, economic value and gross nutritional energy content have been more commonly used in later publications (see Table 5). It is also the major methodological difference between the two major publishing institutions, with INRA and IFREMER favouring economic allocation while Dalhousie University commonly use gross energy content as their basis for allocation (Table 5). Grönroos *et al.* (2006) presented whole fish at farm gate to avoid allocation, while for feed inputs refer to a previous publication where mass allocation was applied. With a similar intent did Phong *et al.* (2010) partially avoid allocation by selecting kcal as one of his functional units. Iribarren *et al.* (2010), on the other hand, used system expansion for Spanish mussel production with the assumption that mussel shells replaced conventional production of calcium carbonate production. Other authors failed to distinguish their allocation decisions within their articles.

As the multi-functionality problem is an artefact of wishing to isolate one function out of many and as artefacts can only be cured in an artificial way, there is no 'correct' way of solving the multi-functionality problem, even not in theory. There are, however, demands one can make to solving this problem, like that the solution should be consistent in itself, and that it should be consistent with main methodological principles (Guinée *et al.* 2004).

*We therefore see no research needs here but recommend providing thorough justification for allocation choices made and conducting sensitivity analyses on these allocation choices in compliance with ISO standards.*

## 2.5 Life Cycle Impact assessment (LCIA)

### 2.5.1 Impact categories

Baseline impact categories that are most commonly included are depletion of abiotic resources, impacts of land use, global warming, ozone depletion, human toxicity, exotoxicology, photo-oxidant formation, acidification and eutrophication (Guinée *et al.* 2002). For the here reviewed studies, only global warming, acidification and eutrophication were consequently applied.

Impact category	Σ	Impact Assessment method
<b>Global warming</b>	12	Houghton <i>et al.</i> 2001 <sup>1,2,3,4,5,6,7,8,9,10,11,12</sup>
<b>Acidification</b>	12	Huijbregts 1999a <sup>1,2,5,6,7,8,9,10,11,12</sup> ; Goedkoop & Spriensma 1999 <sup>3</sup> ; Seppälä <i>et al.</i> 2006 <sup>4</sup> ; Heijungs <i>et al.</i> 1992 <sup>12</sup>
<b>Eutrophication</b>	12	Heijungs <i>et al.</i> 1992 <sup>1,2,5,6,7,8,9,10</sup> ; Goedkoop and Spriensma 2001 <sup>3</sup> ; Weidema <i>et al.</i> 1996 <sup>12</sup> ; Seppälä <i>et al.</i> 2002 <sup>4†</sup> ; Seppälä <i>et al.</i> 2004 <sup>4†</sup>
<b>Energy use</b>	8	VDI 1997 <sup>2,5,6,7,8,9,11</sup> ; Goedkoop & Spriensma 1999 <sup>3</sup> ; Article specific <sup>12</sup>
<b>Biotic resource use</b>	6	Papatryphon <i>et al.</i> 2004 <sup>2,6,8</sup> ; Pelletier & Tyedmers 2007 <sup>5,9,11</sup>
<b>Marine aquatic ecotoxicity</b>	4	Huijbregts 1999b <sup>5,7,10</sup> ; Meent & Klepper 1997 <sup>3*</sup>
<b>Abiotic depletion</b>	3	Guinée & Heijungs 1995 <sup>1,7,10</sup>
<b>Stratospheric ozone depletion</b>	3	WMO 1999 <sup>1,10</sup> ; Goedkoop & Spriensma 1999 <sup>3</sup>
<b>Human toxicity</b>	3	Huijbregts 1999b <sup>1,7,10</sup>
<b>Water dependence</b>	2	Article specific <sup>6,8</sup>
<b>Photochemical oxidant formation</b>	2	Derwent <i>et al.</i> 1998/Jenkin & Hayman 1999 <sup>1,10</sup>
<b>Freshwater aquatic ecotoxicity</b>	2	Meent & Klepper 1997 <sup>3*</sup> ; Huijbregts 1999b <sup>10</sup>
<b>Terrestrial ecotoxicity</b>	2	Meent & Klepper 1997 <sup>3*</sup> ; Huijbregts 1999b <sup>10</sup>
<b>Surface use</b>	2	Article specific <sup>8, 12</sup>
<b>Respiratory impacts from inorganics</b>	1	Goedkoop & Spriensma 1999 <sup>3</sup>
<b>Carcinogenic effects on humans</b>	1	Goedkoop & Spriensma 1999 <sup>3</sup>

<sup>†</sup>Aquatic and terrestrial eutrophication were reported separately

\*Ecotoxicity is summarized under one category

Table 3: Frequency of applying different impact categories in LCA studies on aquaculture, and impact assessment method used. Global warming, acidification and eutrophication were the only impact categories applied by all authors. References: <sup>1</sup>Mungkung 2005, <sup>2</sup>Aubin *et al.* 2006; <sup>3</sup>Ellingsen & Aanondsen 2006; <sup>4</sup>Grönroos *et al.* 2006; <sup>5</sup>Pelletier & Tyedmers 2007; <sup>6</sup>Aubin *et al.* 2009; <sup>7</sup>Ayer & Tyedmers 2009; <sup>8</sup>d'Orbcastel *et al.* 2009; <sup>9</sup>Pelletier *et al.* 2009; <sup>10</sup>Iribarren *et al.* 2010; <sup>11</sup>Pelletier & Tyedmers 2010; <sup>12</sup>Phong 2010.

The major impact assessment methodology used for characterization was CML 2 Baseline 2001 with only Ellingsen & Aanondsen (2006) applying end-point modelling (also name method here: EI'99). Grönroos *et al.* (2006) choose to use regio-specific characterization factors for eutrophication and acidification, where eutrophication was divided between aquatic and terrestrial emissions. There were also a few novel impact categories presented, describing biotic resource use, water dependence and surface use (Papatryphon *et al.* 2004; Pelletier *et al.* 2007; d'Orbcastel *et al.* 2009; Aubin *et al.* 2009). A summary of the results from some of the LCA studies in focus is presented in Table 4.

*In the SEAT LCAs, we will not limit ourselves to a selection of impact categories in advance, but we will try and adopt an as broad set of impact categories as possible and relevant and as far as practical methods are available.*

		Turbot re-circulating (avg. scenario) <sup>2</sup>	Rainbow trout raceways flow-through <sup>6</sup>	Sea-bass cages <sup>6</sup>	Turbot land-based re-circulating <sup>6</sup>	Turbot raceways flow-through <sup>8</sup>	Trou (FCR1.1) tanks. re-circulating <sup>8</sup>	Atlantic salmon net-pen <sup>2</sup>	Atlantic salmon floating bag <sup>2</sup>	Atlantic salmon land-based flow-through <sup>2</sup>	Atlantic salmon land-based recirculating <sup>2</sup>	Atlantic salmon global <sup>9</sup>	Tilapia lake <sup>11</sup>	Tilapia pond <sup>11</sup>	Shrimp conventional & C&C farm <sup>1</sup>	Rainbow trout net-cage <sup>4</sup>	Blue mussels ropes and rafts <sup>10</sup>
Global Warming	Feed	25,4%	73,0%	85,8%	31,9%	91,5%	90,7%	88,3%	86,3%	58,8%	9,4%	94,2%	91,9%	62,5%	24,5%	98,1%	0,0%
	On-site	69,4%	18,0%	5,6%	61,5%	7,0%	6,7%	2,7%	9,6%	41,1%	90,5%	3,5%	2,7%	33,6%	70,5%	1,2%	100,0%
	Smolt production	-	-	-	-	-	-	0,1%	0,6%	0,0%	0,0%	2,2%	5,4%	0,7%	5,0%	0,7%	-
	Infrastructure	5,2%	9,0%	8,6%	6,6%	1,5%	2,6%	8,9%	3,2%	0,1%	0,1%	-	-	-	-	-	-
	<b>kg CO<sub>2</sub> eq. per tonne</b>	<b>7560</b>	<b>2753</b>	<b>3601</b>	<b>6017</b>	<b>2015</b>	<b>2043</b>	<b>2073</b>	<b>1900</b>	<b>2770</b>	<b>28200</b>	<b>2160</b>	<b>1518</b>	<b>2103</b>	<b>4178</b>	<b>716</b>	<b>471,8</b>
Eutrophication	Feed	5,8%	7,1%	7,0%	5,0%	33,3%	45,5%	19,5%	19,5%	20,7%	41,8%	14,6%	9,1%	9,3%	3,7%	3%†	0,0%
	On-site	93,8%	92,7%	92,9%	94,6%	66,3%	54,2%	79,7%	80,2%	79,3%	57,0%	84,9%	90,7%	90,0%	96,3%	95,9%†	100,0%
	Smolt production	-	-	-	-	-	-	0,0%	0,1%	0,0%	0,5%	0,4%	0,1%	0,1%	0,1%	1,1%†	-
	Infrastructure	0,4%	0,2%	0,1%	0,3%	0,1%	0,2%	0,8%	0,1%	0,0%	0,5%	-	-	-	-	-	-
	<b>kg PO<sub>4</sub> eq. per tonne</b>	<b>66,9</b>	<b>65,9</b>	<b>108,9</b>	<b>77</b>	<b>28,5</b>	<b>21,1</b>	<b>35,3</b>	<b>31,8</b>	<b>29,9</b>	<b>20,1</b>	<b>49,3</b>	<b>47,8</b>	<b>45,7</b>	<b>16,9</b>	<b>48,3†</b>	<b>0,4</b>
Acidification	Feed	27,5%	68,0%	80,0%	29,0%	88,7%	88,7%	90,6%	92,4%	88,0%	8,9%	94,6%	94,5%	78,5%	36,9%	98,1%†	0,0%
	On-site	54,5%	19,0%	13,0%	52,0%	7,4%	7,4%	2,0%	4,4%	10,2%	89,7%	3,4%	2,2%	20,6%	59,6%	1,7%†	100,0%
	Smolt production	-	-	-	-	-	-	0,0%	0,6%	0,0%	0,0%	2,0%	3,3%	0,9%	3,4%	0,3%†	-
	Infrastructure	17,9%	13,0%	7,0%	19,0%	4,0%	4,0%	7,8%	2,5%	1,8%	1,3%	-	-	-	-	-	-
	<b>kg SO<sub>2</sub> eq. per tonne</b>	<b>50,5</b>	<b>19,2</b>	<b>25,3</b>	<b>48,3</b>	<b>13,4</b>	<b>13,3</b>	<b>18</b>	<b>15,8</b>	<b>16,6</b>	<b>255</b>	<b>20,4</b>	<b>20,2</b>	<b>23,8</b>	<b>27,5</b>	<b>4,4†</b>	<b>4,72</b>
<b>Institution</b>		<b>INRA/IFREMER</b>						<b>Dalhousie University</b>						<b>Independent</b>			

†Eutrophication and acidification were calculated to relevant units using characterization factors recommended by Guinée *et al.* 2002

Table 4: Origin of emissions from the different production systems. Direct comparisons between results should, however, be done with great caution as methodological differences between studies often constitute greater differences than the actual results. <sup>1</sup>Mungkung 2005; <sup>2</sup>Aubin *et al.* 2006; <sup>4</sup>Grönroos *et al.* 2006; <sup>6</sup>Aubin *et al.* 2009; <sup>7</sup>Ayer & Tyedmers 2009; <sup>8</sup>d'Orbcastel *et al.* 2009; <sup>9</sup>Pelletier *et al.* 2009; <sup>10</sup>Iribarren *et al.* 2010; <sup>11</sup>Pelletier & Tyedmers 2010.

## 2.5.2 Characterization methods for baseline impact categories

### 2.5.2.1 Global Warming

Common greenhouse gases (GHGs) from food production systems, apart from carbon dioxide (1 CO<sub>2</sub>-eq.), include methane (GWP equals 21 kg CO<sub>2</sub>-eq./kg CH<sub>4</sub> emitted) and nitrous oxide (GWP=298 kg CO<sub>2</sub>-eq./kg N<sub>2</sub>O emitted) (Houghton *et al.* 2001). In aquaculture systems these commonly originate from fertilizer production, agricultural practices, capture fisheries, livestock production, feed processing, fuel use on farm, electricity production, ponds, volatilization and land conversion.

The characterisation factors suggested by the international panel on climate change (IPCC)(Houghton *et al.* 2001) were the basis for all reviewed studies. In farms where ecosystem services to a large extent have been replaced by human activities may farm-site emissions make significant contributions to the overall GHG emissions from production (Table 4). Ayer & Tyedmers (2009) concluded that 36.8 % of the GHG emissions from land-based flow-through systems and 84.0 % from land-based re-circulating systems were contributed by electricity production for on-site activities. Similar trends were found for modelled trout farms where 24.4 % of the energy use in flow-through systems were energy sources consumed at farm-site and 60.8/66.7 % (Feed Conversion Ratio (FCR) of 1.1/0.8) in re-circulating systems (d'Orbcastel *et al.* 2009). The resulting GHG emissions from the trout farm was, however, lower as nuclear energy was considered as the major source of electricity (d'Orbcastel *et al.* 2009).

As a result of one highly resourceful centralised scientific body, the IPCC's guidelines have been adopted as the baseline for all midpoint characterisation in this review. This does simplify comparisons between studies, while high variability between feed production systems and limited information on the origin of feed input data remain an indistinct sector. Variation within farming and fishing practices is driven by variability in natural processes, local to regional conditions, cultivation practices and skipper experience (Naylor *et al.* 1997; Ruttan & Tyedmers 2007; Røos *et al.* 2010).

*For this study we propose to work with the latest GWPs from IPCC as included in the impact assessment spreadsheet published and downloadable from the CML website (<http://www.leidenuniv.nl/cml/ssp/databases/cmlia/cmlia.zip>).*

### 2.5.2.2 Acidification

Acidification of aquatic and terrestrial ecosystems is the result of sulphur (S) and nitrogen (N) emissions (Hicks *et al.* 2008). Anthropogenic emissions of SO<sub>2</sub> is about three times that of natural emissions and nitrogen over twice the size of the 8 Tg N (10<sup>12</sup> g) released annually from natural processes (Galloway 1995). Areas subject to high levels of emissions are subject to impacts on e.g. fish stocks, forests, acid-sensitive invertebrates and diatome communities (Skjelkvåle *et al.* 2003; Hicks *et al.* 2008). The seriousness of emissions do, however, depend upon precipitation, climate, soil depth, dissolved organic carbon (DOC), nitrogen saturation, deposition of marine ions and cat ion deposition (Skjelkvåle *et al.* 2003).

Only Grönroos *et al.* (2006) choose to include country specific characterization factors for acidification (Seppälä *et al.* 2006). The INRA/IFREMER group choose to use average European conditions in all of their published studies. Mungkung (2005) does state the inclusion of regional weighing factors while the background or implementation of these are not further described. The rest of the studies do not state

the region for characterisation factors, thereby the assumption of world averages based upon the chemical maximum hydrogen ion release is made. Most country-dependent characterisation factors for acidification currently only cover European countries (Amann *et al.* 1999; Seppälä *et al.* 2006; ILCD 2010) with a lack of knowledge in developing countries.

The effects of acidification on surrounding ecosystems is strongly influenced by the regional buffer capacity (Seppälä *et al.* 2006; Hicks *et al.* 2008). Therefore the implementation of maximum ion release, on the basis of a world average scenario, can be highly misleading. However, due to the globally widespread origin of many aquaculture processes and the currently limited coverage of country-dependent characterisation factors, accounting for local conditions can be difficult. Modelling systems developed for this purpose, include the RAINS model and EcoSense (Amann *et al.* 1999; ILCD 2010). Detailing can be done in regards to the ambition of the project, as long as the practice is consistent throughout the study.

*For this study we propose to work with the updated baseline method for acidification and other more recent (variants of baseline) methods and characterisation factors as included in the impact assessment spreadsheet available through the CML website (<http://www.leidenuniv.nl/cml/ssp/databases/cmlia/cmlia.zip>), replenished by the midpoint ILCD(2010) LCIA methods and factors for acidification (<http://lct.jrc.ec.europa.eu/publications>).*

### 2.5.2.3 Eutrophication

Complications such as oxygen depletion and red tide events as a result of eutrophication have been a major source of public concern related to aquaculture (Wu 1995; Tacon *et al.* 2006). Emissions originate from wasted feed and faecal material in solid form and dissolved wastes as metabolic by-products, originating from the gills and kidneys (Cho & Bureau 2001).

Nitrogen, phosphorus and carbon lost at farm-site accounted for 54.2 to 96.3 % of the eutrophying emissions from fed aquaculture systems (Table 4). Closed systems do have the ability to collect much of these sediments thereby reducing the nutrient release to surrounding ecosystems substantially. Both Ayer & Tyedmers (2009) as well as Aubin *et al.* (2009) concluded that cage systems have a higher eutrophying impact than flow-throw systems, which in turn are more polluting than re-circulating systems. Filter-feeding organisms, on the other hand, may even achieve negative eutrophying potentials by removing seston (=all particulate matter suspended in bodies of water such as lakes and seas, including plankton, organic detritus, and inorganic material: <http://en.wiktionary.org/wiki/seston>) through harvest or as particulate matter to the sediments (Newell *et al.* 2004; Iribarren *et al.* 2010).

Eutrophying emissions from fed aquaculture will be governed by either terrestrial emissions from agricultural activities or aquatic emissions from farm site activities (Cho & Bureau 2001; Grönroos *et al.* 2006). As terrestrial emissions originate almost solely (97.9%) from feed production and aquatic emissions from feed application (Grönroos *et al.* 2006), it is recommended to make a distinction between the two. Country, or even regional specific characterisation factors should also be implemented where available (Huijbregts & Seppälä 2001; Guinée *et al.* 2002; Gallego *et al.* 2010).

*For this study we propose to work with the updated baseline method for eutrophication and other more recent (variants of baseline) methods and*

characterisation factors as included in the impact assessment spreadsheet available through the CML website (<http://www.leidenuniv.nl/cml/ssp/databases/cmlia/cmlia.zip>), replenished by the midpoint ILCD (2010) LCIA methods and factors for eutrophication (<http://lct.jrc.ec.europa.eu/publications>).

#### 2.5.2.4 Abiotic Resource use

Abiotic resource depletion is the extraction of materials of limited reserves, such as iron, phosphorus, oil and uranium. Resource use can be assessed using different methodological approaches, depending upon the problem definition (Guinée *et al.* 2002; Stewart & Weidema 2005). The most commonly applied methodology is based upon reserve deposits in relation to current production (extraction; de-accumulation), while other approaches are based upon energy and material inputs, costs of recovery to initial state or aggregation of exergy/entropy impacts (Stewart and Weidema 2005).

All three studies that included abiotic resource use based their results upon the Abiotic Depletion Potential (ADP) approach of Guinée & Heijungs (1995), who in turn calculated their indicators upon ultimate reserve deposits with regards to production rate. The general conclusion was similar between the studies, that abiotic resource used for aquaculture systems is driven by energy use through the combustion of fossil fuels and extraction of uranium (Mungkung 2005; Ayer & Tyedmers 2009). Other processes that can influence the final outcome include mining of limestone, phosphorus and iron.

For aquaculture systems, abiotic resource use is largely governed by energy production. None of the authors reflected over the depletion of resources related to infrastructure or phosphorus extraction whereby these can be assumed to be limited.

Recently, the ADP approach has been slightly adapted, distinguishing between two impact categories:

- Abiotic Depletion of elements;
- Abiotic Depletion of fossil fuels.

The reason behind this distinction is that fossil fuels clearly constitute a depletion category of particular (energy) concern and that the depletion of fossil fuels fully dominated the results for abiotic depletion. Therefore abiotic depletion of fossil fuels was split off. In quantifying the ADPs for fossil fuels, fossil fuels are considered to be fully substitutable and using fossil fuels contributes to the depletion of the total energy reserve embodied in fossil fuels. The new ADPs for elements and fossil fuels can be found in the impact assessment spreadsheet available through the CML website (<http://www.leidenuniv.nl/cml/ssp/databases/cmlia/cmlia.zip>).

*For this study we propose to work with the updated ADP method for abiotic depletion and with other more recent (variants of baseline) methods and characterisation factors as included in the impact assessment spreadsheet available through the CML website (<http://www.leidenuniv.nl/cml/ssp/databases/cmlia/cmlia.zip>), replenished by the midpoint ILCD (2010) LCIA methods and factors for abiotic depletion (<http://lct.jrc.ec.europa.eu/publications>).*

#### 2.5.2.5 Toxicity

Toxicological effects may lead to either acute effects or subchronic/chronic concerns, with resulting consequences depending upon the type of organism and ecosystem (Owens 2002). This category is therefore often divided into human toxicity and ecotoxicity in freshwater, freshwater sediments, marine water, marine sediments and

terrestrial environments (Guinée *et al.* 2002). There is still an active debate within the scientific community on how account for toxicological effects (Guinée *et al.* 2002), which will hopefully come to an end with the USEtox characterization factors currently under review by UNEP-SETAC ([www.usetox.org](http://www.usetox.org); accessed 19-Oct-2010).

The lack of consensus for this impact category is reflected by the low adoption frequency of toxicity impact categories (only 3 studies adopted human toxicity and 2 studies adopted ecotoxicity as impact categories; see Table 3).

A mid-point approach based upon Huijbregts (1999b) was the most frequently used method for characterising toxicity effects. Ellingsen & Aanondsen (2006), however, used the endpoint approach suggested by Meent & Klepper (1997) with the same conclusions as Ayer & Tyedmers (2009), that feed production was the largest threat to human well-being. This went with the exception of land-based recirculating systems where emissions from electricity production were far greater (Ayer & Tyedmers 2009). Electricity production was also the largest contributor to marine toxicity for both land-based systems while release of copper and anti-fouling paints was the major concern of net-pen systems (Ellingsen & Aanondsen 2006; Ayer & Tyedmers 2009).

Toxicity modelling in general and aquatic and marine toxicity modelling of metals in particular, has received much criticism. Different methods were developed that used different modelling principles and were incomplete in substance coverage. On top of this, a group of specialists in the areas of LCA (Life Cycle Assessment), LCIA (Life Cycle Impact Assessment) and Risk Assessment “recognised that current ecotoxicity LCIA methods often produce a probably incorrect emphasis on the impact of metals” (<http://lcinitiative.unep.fr/includes/file.asp?site=lcinit&file=38D1F49D-6D64-45AE-9F64-578BA414E499>). This unsatisfactory situation was the background for launching a Task Force on Toxic Impacts under the auspices of the UNEP-SETAC Life Cycle Initiative. the Task Force made a comparison of existing characterization models and eventually proposed a scientific consensus model called USEtox™ (<http://www.usetox.org/>).

*In this study, we will adopt the USEtox™ model (which is the midpoint ILCD LCIA recommendation for toxicity) and try to add CFs for specific aquaculture chemicals (e.g., anti-fouling chemicals, antibiotics etc.) for which CFs may be lacking in the current USEtox CF-list*

*([http://www.springerlink.com/content/8217520256r12w36/MediaObjects/11367\\_2008\\_38\\_MOESM2\\_ESM.xls](http://www.springerlink.com/content/8217520256r12w36/MediaObjects/11367_2008_38_MOESM2_ESM.xls)).*

### 2.5.2.6 Photochemical ozone formation

Release of NO<sub>x</sub>, SO<sub>2</sub>, NH<sub>3</sub> and volatile organic compounds into the troposphere may, under the influence of UV-light, result in the formation of ozone (O<sub>3</sub>) (Guinée *et al.* 2002; Derwent *et al.* 2007). Elevated surface ozone levels are of concern to human health, visibility and climate change (Cheng *et al.* 2010). Common origins of these photo-chemically active compounds are motor vehicles and large combustion plants (Derwent *et al.* 2007). With rapid industrialization in many Asian countries, photochemical smog characterized by high concentrations of ozone and particulates have become a growing problem (Cheng *et al.* 2010). The reaction rate of these compounds can, however, vary by a factor of 10 000 (Guinée *et al.* 2002; Cheng *et al.* 2010).

The characterization factors for photochemical ozone formation suggested by Derwent *et al.* (1998) originate from a trajectory model, based on Master Chemical Mechanism. This model is, however, based upon the formation of ozone in a North-Western European scenario. Since the photochemical ozone creation potential (POCP) is influenced by regional conditions, alterations to existing factors may need to be made using regional data. An example of how local conditions can be taken into account is presented by Cheng *et al.* (2010) who report updated potentials for Southern China, a hot-spot for aquaculture production.

*For this study we propose to work with the updated baseline method for photochemical ozone formation and other more recent (variants of baseline) methods and characterisation factors as included in the impact assessment spreadsheet available through the CML website*

*(<http://www.leidenuniv.nl/cml/ssp/databases/cmlia/cmlia.zip>), replenished by the midpoint ILCD (2010) LCIA methods and factors for photochemical ozone formation (<http://lct.jrc.ec.europa.eu/publications>) and updated POCPs by Cheng *et al.* (2010).*

## **2.5.3 Characterization methods for non-baseline (“novel”) impact categories**

### **2.5.3.1 Cumulative energy demand**

Initial life-cycle studies of aquaculture systems focused on solar and auxiliary energy inputs into the production chain as energy intensity provides a good indicator of ecological footprint, abiotic resource depletion and GHG emissions (Folke 1988; Larsson *et al.* 1994; Ayer & Tydmers 2009; Huijbregts *et al.* 2010), as well for identifying food production systems vulnerable to fluctuations in energy prices. It has been frequently used in LCA studies on aquaculture using the characterization factors developed by VDI (1997).

Despite a more efficient conversion of feed to meat in salmon, salmon farming still has a higher cumulative energy demand (CED) than chicken farming (Ellingsen & Aanondsen 2006). The energy inputs usually come in the form of intensive use of machinery, feeds, transports, fertilizers, controlled growing conditions or the application of lime. The type of lime applied will be of importance as the production of hydraulic lime and quicklime consumes 13 respectively 15 times more energy than milling of limestone (ecoinvent v2.0; VDI 1997; Mungkung 2005).

Cumulative energy demand serves as a good indicator for highlighting areas of high anthropogenic intervention in aquaculture production systems, disregarding the noise of natural processes. It also allows for the inclusion of labour man-hours (Folke 1988; Nguyen *et al.* 2007). The inclusion of cumulative energy demand is recommended as it highlights the direct relationship between anthropogenic energy inputs and calorific food outputs (Troell *et al.* 2004). *In this study, we will therefore include CED as a non-baseline indicator and present the results of this indicator separately stressing the overlap it has with some baseline impact categories (particularly abiotic depletion).*

### 2.5.3.2 Biotic resource use

Biotic resource use has been adopted as impact categories by six of the thirteen studies reviewed. The approach adopted has been referred to as Biotic Resources Use (BRU; Pelletier *et al.* 2007). The BRU is calculated based on the carbon content of the fraction of crop used following Papatryphon *et al.* (2004). These were 460, 528, 607, and 465 g C/kg crop (dry matter) for wheat, soy bean, canola and corn respectively. Biotic resource use for fishery-derived ingredients was calculated following Pauly and Christensen (1995) using the formula:

$$P = \left(\frac{M}{9}\right) \times 10^{(T-1)}$$

where P is the mass (in kg) of carbon appropriated, M is the mass of fish required (wet weight), and T is the trophic level of the organism used (as reported in [www.fishbase.org](http://www.fishbase.org)). Here, carbon represents the transferable products of photosynthesis (net primary production: NPP). None of the studies have included indirect losses of wildlife resulting from aquaculture activities.

Biotic resource use may provide a useful indicator of the amount of natural resources needed for the conversion to marketable seafood. However, NPP has been used in LCA before in order to quantify land use impacts on biodiversity and life support systems before (Sas *et al.* 1997; Weidema & Lindeijer 2001). Furthermore, methods have been proposed for biotic resource depletion by Heijungs *et al.* (1992) and Guinée *et al.* (2002).

Biotic resource use, biotic resource depletion, land use impacts and biodiversity impacts are thus highly inter-related topics, for which similar and/or overlapping methods have been proposed. The proposals developed so far lack broad acceptance and there are no or only provisional recommended ILCD LCIA methods for these impacts. Within agro-LCAs various methods have been proposed that could be used as basis for further work. This work should build on the outcome of UNEP-SETAC's ongoing work on operational characterization factors for land use impacts on biodiversity and ecosystem services, due May 2011

([http://www.estis.net/sites/lcinit/default.asp?site=lcinit&page\\_id=337831BE-0C0A-4DC9-AEE5-9DECD1F082D8](http://www.estis.net/sites/lcinit/default.asp?site=lcinit&page_id=337831BE-0C0A-4DC9-AEE5-9DECD1F082D8)), and other ongoing projects (including certification systems).

*We therefore propose, as part of the SEAT project, to draft a scientific paper reviewing how biotic resource use, biotic resource depletion, land use impacts and biodiversity impacts are currently handled in agro- and aquaculture LCAs, reviewing which methods are available within the LCA community and outside that community, and proposing a framework for addressing these impacts within LCA (or outside LCA).*

### 2.5.3.3 Water dependence

Actual consequences of fresh water use are hard to estimate as it greatly depends upon the geographical location, temporal variations in water availability, type of water use (consumptive or not), level of water degradation and source of the water (Blackhurst *et al.* 2010). Water dependence can, in accordance to the U.S. Geological Survey, be divided into:

- *Use*, the total quantity of water utilized and available for reuse;
- *Consumption*, the quantity of water denied to others;
- *Depletion*, the quantity of water that is not sufficiently replaced by the hydrological cycle.

Water dependence was only detailed by Aubin *et al.* (2009) and d'Orbcastel *et al.* (2009) who both focused on water use at farm-site. Aubin *et al.* (2009) measured a

diversion of 52.6 m<sup>3</sup> river water per tonne produced trout in a flow-through system in Southern France, which was more than ten times the water needed for a re-circulating turbot system. The same article calculated an average water use for a cage system of 48 782 m<sup>3</sup> tonne<sup>-1</sup> live sea-bass, based upon the average current through the cages.

As aquaculture production is set in an aquatic environment, it is highly misleading to quantify water use, since the water quality of run-off water varies considerably and water degradation overlaps with other impact categories. Depending upon the type of use and number of reuses, aquaculture ponds may even be beneficial during drought periods as they act as water reservoirs (Verdegem *et al.* 2006). It is instead recommended to calculate freshwater consumption throughout the lifecycle; including agricultural production, animal production, evaporation, electricity production etc. Furthermore, new approaches to assess water use or dependence have emerged that should be considered being adopted (Bayart *et al.* 2010). *In this study, existing methods to quantify freshwater use will therefore be inventoried and applied taking into account the recommended midpoint ILCD (2010) LCIA method for water use.*

#### 2.5.3.4 Land use

The consequences of land use can be divided into land competition and environmental consequences resulting from the land use (Guinée *et al.* 2002). While land competition is easily expressed in m<sup>2</sup> ·yr, environmental consequences due to land use is complicated due to the long transition times and the fact that land rarely convert back to its original steady state (Guinée *et al.* 2002). Aquaculture farms are often competing for attractive coastal sites with other uses such as fisheries, tourism, residential areas and recreation (Folke 1998). It has also historically resulted in deforestation of mangrove which, in many places, has resulted in unproductive aquaculture due to failing ecosystem services (Rönnbäck 1999). Despite the importance of this impact category, only d'Orbcastel *et al.* (2009) and Phong *et al.* (2010) choose to include it as an impact category; Ellingsen & Aanondsen (2006) did instead cover the topic in the supporting text.

Despite the importance of this impact category, only d'Orbcastel *et al.* (2009) and Phong (2010) choose to include it as an impact category; while Ellingsen & Aanondsen (2006) did cover the topic in the supporting text.

Phong (2010) concluded that the on-farm area needed was similar to the off-farm area needed for the production of feed ingredients. Phong did, however, not include impacts on seafloor from capture fisheries that otherwise add another dimension to the validity of applying land use as an impact category for LCA for aquaculture (Ellingsen & Aanondsen 2006). The resulting disturbance from fishing activities is highly dependent upon the type of fishery and fishing location. Ellingsen & Aanondsen (2006) accounted for a land area of 2,76 m<sup>2</sup> for the production of salmon feed, whereas the bottom area covered by trawling was 494,5 m<sup>2</sup>. This differs greatly from the 2.10 – 2.75 m<sup>2</sup> of land area<sup>4</sup> needed to produce feed for 1 kg of trout in France (d'Orbcastel *et al.* 2009). Supporting areas to different aquaculture production systems differ widely, from 1 to 50 000 times the actual cultivation area (considering both local support and support by distant ecosystems for e.g. feeds) (Folke *et al.* 1998).

---

<sup>4</sup> Note that land area in Ellingsen & Aanondsen (2006) and d'Orbcastel *et al.* (2009) most likely refers to land occupation. In that case, the (occupation) dimension should be m<sup>2</sup> ·yr and not m<sup>2</sup> (see, e.g., Guinée *et al.* 2002, p.136 and 551, or Heijungs *et al.* 1992, Backgrounds - p. 35).

Area use related to aquaculture can be either direct (the area occupied by the farm), indirect (the area occupied for supporting services, largely driven by agricultural activities) or submerged (seafloor). As there still is no good methodology to account for the complex consequences from capture fisheries (Ziegler 2004), focus should be directed towards direct and indirect land use. *In this study, direct and –where possible and relevant - indirect land use occupation (in m<sup>2</sup>·yr) and transformation (when applicable; in m<sup>2</sup>) will be inventoried. When feasible the recommended midpoint ILCD (2010) LCIA method for land use will be applied. Since land use impact cannot (yet) be properly handled in LCA, other WPs of the SEAT project will have to supply additional information on land use impacts from aquaculture systems. Knowledge will also be drawn from the outcome of UNEP-SETAC's operational characterization factors for land use impacts on biodiversity and ecosystem services, due May 2011*

*([http://www.estis.net/sites/lcinit/default.asp?site=lcinit&page\\_id=337831BE-0C0A-4DC9-AEE5-9DECD1F082D8](http://www.estis.net/sites/lcinit/default.asp?site=lcinit&page_id=337831BE-0C0A-4DC9-AEE5-9DECD1F082D8)) and possibly also from certification systems.*

### 2.5.3.5 Other impact categories

Other impact categories that have been listed in the reviewed articles and in Pelletier *et al.* (2007) and Ziegler (2004) include salinity, benthic impacts (by seafloor trawling), by-catch and socio-economic impacts. For some of these impact categories, e.g. salinity, method proposals have been developed. “Ziegler *et al.* (2003) have estimated the area impacted by Swedish trawlers in the Baltic Sea and discussed the possible environmental impact. They indicate that bottom trawling can contribute to oxygen depletion, but conclude that more studies are needed in order to quantify the biological effects of trawling in a specific area. The seafloor swept by trawling is calculated based on data for average trawl dimensions, average trawling speed and average catch landed per day” (quote from Ellingsen & Aanondsen 2006). Guinée *et al.* (2006) inventoried and analysed methods for specific agriculture related impact categories such as erosion and soil fertility. The conclusion from this inventory was that methods for these impact categories have been drafted, but it would require significant additional efforts (in terms of both inventory data collection and further elaboration of impact assessment methods). This is considered unfeasible as yet within the scope of the SEAT project.

By-catch (which may also not be that relevant for aquaculture systems) and socio-economic impacts are not environmental impacts and therefore not addresses in the environmental LCAs. Socio-economic impacts are, however, part of the SEAT project, in WP5.

*Concluding we suggest that - because of their priority for aquaculture LCAs and their practical feasibility in terms of existing (or developing) practical methods - existing (or developing) methods to quantify salinity and benthic impacts will be inventoried and applied whenever feasible and appropriate.*

### 2.5.4 Normalisation and weighting

Only Ellingsen & Aanondsen (2006) and Grönroos *et al.* (2006) choose to weight their results. Ellingsen & Aanondsen (2006) used Eco-indicator 99 and also performed a sensitivity analysis, comparing the results with the EDIP indicator. The conclusion came to that the Eco-indicator resulted in higher eutrophication impacts, while the EDIP indicator disfavoured the eco-toxicity of salmon farming due to higher burdens related to copper emissions (Ellingsen & Aanondsen 2006).

*For this study we propose to work with the updated normalisation methods as included in the impact assessment spreadsheet available through the CML website (<http://www.leidenuniv.nl/cml/ssp/databases/cmlia/cmlia.zip>). In principle, we will not*

*perform any weighting, since weighting is not needed for the prime goals of our LCAs (mapping and hot spot identification). If weighting is needed as yet, we propose to adopt one or more weighting methods from a study on weighting methods by Huppes & Oers (2009; 2010) for the EC JRC-IES (Ispra, Italy). In this study several weighting methods are proposed as it is not possible to just define one overall method.*

## **2.6 Interpretation**

Interpretation is a phase of life cycle assessment in which the findings of either the inventory analysis or the impact assessment, or both, are combined consistent with the defined goal and scope in order to reach conclusions and recommendations (ISO 2006a). ISO further mentions that interpretation elements include:

- identification of the significant issues based on the results of the LCI and LCIA phases of LCA (contribution analysis);
- an evaluation that considers completeness, sensitivity and consistency checks;
- conclusions, limitations, and recommendations.

As conclusions and recommendations are case-specific and cannot be reviewed in general, we focus on contribution analysis and sensitivity analyses below as these are the only elements that have been included in some of the studies reviewed.

### **2.6.1 Contribution analysis**

The contribution (or dominance) analysis calculates the overall contribution to the results of the various factors. The contributions are usually expressed as percentages of the total. Contribution analysis answers questions about the contribution of specific environmental flows, processes or impacts to a given environmental score.

As can be derived from Table 4 and section 2.5.2, many of the reviewed studies performed contribution analyses. Contribution analyses can be very helpful in tracing potential flaws and errors and improvement options.

*In this study we will perform contribution analyses at all relevant levels (inventory analysis, characterization and, if useful, normalization) and for different elements (processes and interventions). We will use the results of these analyses for tracing possible errors and as basis for identifying improvement options. For the latter, we will also develop various types of sensitivity analyses (see section 2.6.2 and see Heijungs & Kleijn (2001)).*

### **2.6.2 Sensitivity analysis**

In order to use LCA as a tool for decision-making, information is needed on the robustness of the results. This element of the Interpretation phase assesses the influence on the results of variations in process data, model choices and other variables. In the sensitivity analysis, these changes are deliberately introduced in order to determine the robustness of the results with regard to these variations. Some of the reviewed articles performed sensitivity. Ayer and Tyedmers (2009) conducted a complete set of sensitivity analysis, whereof one addressed electricity sourcing for Canadian aquaculture production systems, with the conclusion that using a national electricity mix significantly changed the outcome compared to using specific electricity sources. Another study from Dalhousie University by Pelletier & Tyedmers (2007) concluded that allocation factor strongly influences the impact of different feeds. Both d'Orbcastel *et al.* (2009) and Pelletier *et al.* (2009) drew a parallel between FCR to GHG emissions in Chilean salmon production, where a theoretical reduction in FCR by about a quarter resulted in environmental impacts by

16 - 27 %. Mungkung (2005) conducted a sensitivity analysis on data assumptions for fishing practices as well as for different impact assessment methods, with the same general conclusions left standing. Overall, most of the sensitivity analyses were done on data values and not methodological differences. This is an area open to improvements, especially since allocation plays such a vital role. On top of that the results of sensitivity analyses are always presented and discussed for each analysis separately. However, the more sensitivity analyses are performed, the bigger the landscape of possible results. This landscape of results should be processed and presented in combination, e.g. running a Monte-Carlo analyses. No study does this yet, as far as known to the authors.

*For this study we strive to perform several kinds of sensitivity analysis on a broad range of data and methodological choices. For this (Monte-Carlo based) methods to include multiple variations in parameters and choices / assumptions need to be developed.*

Reference	Species	Farming system	Country	Functional unit	System boundary	Allocation factor	Software	Database	
Aubin <i>et al.</i> 2006	Turbot	Re-circulating	France	1 tonne live weight	Farm-gate	Economic value	SimaPro v.6.0	Buwal B250, Ecoinvent*	INRA/ IFREMER
Aubin <i>et al.</i> 2009	Rainbow trout, sea-bass and turbot	Various	France	1 tonne live weight	Farm-gate	Economic value	SimaPro v.6.0	Buwal B250, Ecoinvent*	
d'Orbcastel <i>et al.</i> 2009	Trout	Various	(Model)	1 tonne live weight	Farm-gate	Economic value	SimaPro v.6.0	EDF, 2004; etc.	
Pelletier & Tyedmers 2007	Atlantic salmon	Salmon feeds	Canada	1 tonne live weight	Farm-gate	Gross nutritional energy	SimaPro v.7.0	All included databases	Dalhousie
Ayer & Tyedmers 2009	Atlantic salmon & char	Various	Canada	1 tonne live weight	Farm-gate	Gross nutritional energy	SimaPro v.7.0	Ecoinvent 1.2; Franklin US LCI database.	
Pelletier <i>et al.</i> 2009	Atlantic salmon	Cage	Global	1 tonne live weight	Farm-gate	Gross nutritional energy	SimaPro v.7.1.8	ecoinvent v.2	
Pelletier & Tyedmers 2010	Tilapia	Net cage and ponds	Indonesia	1 tonne fillets	Market	Gross nutritional energy	SimaPro v.7.0	ecoinvent v.2	
Mungkung 2005	Shrimps	Ponds	Thailand	1.8 kg block of frozen shrimp	Waste	Economic value	SimaPro v.5.1	Buwal 250; etc.*	Independent
Grönroos <i>et al.</i> 2006	Rainbow trout	Net cage	Finland	1 tonne ungutted fish post-mortem	Farm-gate	Mass	KCL-ECO 2003	Silvenius & Grönroos, 2003	
Ellingsen & Aanonsen 2006	Atlantic salmon	Net cage	Norway	200 gram fillet	Market	Mass/Economic value	SimaPro v.6.0	Frischknecht <i>et al.</i> 1996; Buwal 250	
Iribarren <i>et al.</i> 2010	Blue mussels	Rafts	Spain	1 kg of dry edible mussel flesh	Market	System expansion	SimaPro v.6.0	ecoinvent v.2	
Phong <i>et al.</i> 2010	Various <sup>†</sup>	Ponds, integrated	Viet Nam	1 kg of live fish/ 1 kcal of produce	Farm-gate	Economic value	Not stated	Not stated	

Table 5: The ten articles and two PhD theses reviewed in this report, with general methodological choices highlighted.

\*Data has been supplemented by personal communication

<sup>†</sup>Tilapia, kissing gourami, giant gourami, silver barb, common carp, silver carp and striped catfish

### 3 Selection of topics for which LCA methodology revisions are needed

#### 3.1 Topics from the review

Based on the review of existing aquaculture LCA studies in Chapter 2, LCA issues need to be selected for which further developments and/or improvements will be suggested and implemented in the aquaculture LCAs that will be performed as part of the SEAT project. Issues are selected on the basis of:

- priority for aquaculture LCAs;
- feasibility within the scope and the time and financial budget of the project.

For the second criterion, we will prioritize issues for which proposals have been (partly) developed already (existing proposals).

For the goal and scope definition phase the following conclusions were drawn in section 2.3:

- Goal: the main goal of this LCA study is formulated as getting insight in:
  - the environmental impact and its causes of aquaculture systems for tilapia, catfish, shrimp and prawn in China, Vietnam, Thailand and Bangladesh.
  - starting points (“hot spot identification”) for improving the environmental performance of aquaculture systems for tilapia, catfish, shrimp and prawn China, Vietnam, Thailand and Bangladesh, which includes insight into the effects of choices in methods and data on the outcomes.

On top of these main goals, learning (of the environmental ins and outs of aquaculture systems) is another important goal of this study. No methodology revisions are needed for this.

- Attributional or consequential: an attributional approach is adopted in this study. *As ‘revision’, a scientific paper will be drafted reviewing current practice of consequential LCA including an outlook into possible futures of CLCA.*
- Temporal coverage: it is thus important to collect data as averages over longer periods properly reflecting these potential fluctuations. The reference year for data collection on farms is 2010; for other processes a reference year will be adopted as close as possible to 2010. No methodology revisions are needed for this.
- Spatial coverage: it may be advisable to investigate the feasibility of adopting more regionalised impact assessment methods for the more regional- and local-scale impact categories. Significant methodology revisions would be needed for this.
- Technology coverage: different farming systems and mixed technologies will be covered in the LCAs performed within the SEAT project mapping their environmental performance as well as identifying options for reducing their impacts. No methodology revisions are needed for this.
- Functional unit: a post-processing functional unit is adopted (see deliverable D2.4). No methodology revisions are needed for this.

For the inventory analysis phase the following conclusions were drawn in section 2.4:

- System boundaries: we will adopt a system boundary from farm - including the production of all upstream inputs to the farm - to consumption (“from farm to fork”). Human consumption of fish itself is not included as explained in deliverable D2.4. Where applicable and appropriate we will apply EIOA to

estimate the potential significance of lacking data. No methodology revisions are needed for this.

- Data availability and quality: main efforts should be put into collection of Asia foreground data of the actual aquaculture processes including fisheries, local feeds, nurseries, hatcheries, farms etc. correcting for losses due to parasites and diseases and including risks of failed production caused by intensive farming in the production figures, and paying attention to the public availability (or non-disclosure) of the database produced. For this national life cycle inventory databases (if existing; Thailand, e.g., has such a database), peer-reviewed journals and technical reports from recognised organisations will be consulted. No methodology revisions are needed for this.
- Allocation: we will provide thorough justification for allocation choices made and conduct sensitivity analyses on these allocation choices in compliance with ISO standards. No methodology revisions are needed for this.

For the impact assessment phase the following conclusions were drawn in section 2.5:

- Impact categories: In the SEAT LCAs, we will not limit ourselves to a selection of impact categories in advance, but we will try and adopt an as broad set of impact categories as possible and relevant and as far as practical methods are available. No methodology revisions are needed for this.
- Global warming: we will work with the latest GWPs from IPCC. No methodology revisions are needed for this.
- Acidification: we will work with the updated baseline method for acidification and other more recent (variants of baseline) methods and characterisation factors, replenished by the midpoint ILCD LCIA methods and factors for acidification. No methodology revisions are needed for this.
- Eutrophication: we will work with the updated baseline method for eutrophication and other more recent (variants of baseline) methods and characterisation factors, replenished by the midpoint ILCD LCIA methods and factors for eutrophication. No methodology revisions are needed for this.
- Abiotic resource depletion: we will work with the updated ADP method for abiotic depletion and with other more recent (variants of baseline) methods and characterisation factors, replenished by the midpoint ILCD LCIA methods and factors for abiotic depletion. No methodology revisions are needed for this.
- Toxicity: we will work with the USEtox™ model and *add CFs for specific aquaculture chemicals for which CFs may be lacking in the current USEtox CF-list, whenever possible.*
- Photochemical ozone formation: we will work with the updated baseline method for photochemical ozone formation and other more recent (variants of baseline) methods and characterisation factors, replenished by the midpoint ILCD LCIA methods and factors for photochemical ozone formation. No methodology revisions are needed for this.
- Cumulative energy demand (CED): we will include CED as a non-baseline indicator and present the results of this indicator separately stressing the overlap it has with some baseline impact categories (particularly abiotic depletion). No methodology revisions are needed for this.
- For biotic resource use, biotic resource depletion, land use and biodiversity impacts combined we will draft *a scientific paper reviewing the inter-relations between these potential impact categories, available methods within the LCA community and outside that community, and proposing a framework for addressing these impacts within LCA (or outside LCA).*

- Water use: ***existing methods to quantify freshwater use will be reviewed and assessed for practical application in the SEAT LCA studies in Chapter 4, taking into account the recommended midpoint ILCD LCIA method for water use.***
- Land use: where possible and relevant, we will inventory (indirect, when applicable) land use occupation (in m<sup>2</sup>·yr) and transformation (in m<sup>2</sup>). On top of this, the recommended midpoint ILCD LCIA method for land use will be applied, if feasible. Since land use impact cannot (yet) be properly handled in LCA, other WPs of the SEAT project will have to supply additional information on land use impacts from aquaculture systems. No methodology revisions are needed for this. Knowledge will also be drawn from the UNEP-SETAC guidelines, due May 2011 ([http://www.estis.net/sites/lcinit/default.asp?site=lcinit&page\\_id=337831BE-0C0A-4DC9-AEE5-9DECD1F082D8](http://www.estis.net/sites/lcinit/default.asp?site=lcinit&page_id=337831BE-0C0A-4DC9-AEE5-9DECD1F082D8)) and possibly also from certification systems. On top of this we will review inter-relations of land use impact with other impact categories adopted within particularly agro- and aquaculture LCAs (see discussion above).
- Other impact categories: because of their potential priority for aquaculture, ***existing methods to quantify salinity and benthic impacts will be reviewed and assessed for practical application in the SEAT LCA studies in Chapter 4.***
- Normalisation and weighting: we will work with the updated CML normalisation methods. In principle, we will not perform any weighting, since weighting is not needed for the prime goals of our LCAs (mapping and hot spot identification). If weighting is needed as yet, we propose to adopt one or more weighting methods from a study on weighting methods by Huppés & Oers (2009; 2010) for the EC JRC-IES (Ispra, Italy). No methodology revisions are needed for this.

For the interpretation phase the following conclusions were drawn in section 2.6:

- Contribution analysis: we will perform contribution analyses at all relevant levels (inventory analysis, characterization and, if useful, normalization) and for different elements (processes and interventions). We will use the results of these analyses for tracing possible errors and as basis for identifying improvement options. No methodology revisions are needed for this.
- Sensitivity analysis: we strive to perform several kinds of sensitivity analysis on a broad range of data and methodological choices. For this ***(Monte-Carlo based) methods to include multiple variations in parameters and choices / assumptions will be explored, starting in Chapter 4.***

### 3.2 Discussion and selection

All topics listed in bold and italics in section 3.1 will be explored in Chapter 4 with regard to the existence of elaborated methodological proposals (preferably with associated practical data and/or CFs) or potential for (practically feasible) developments and/or revisions, except for:

- Attributional or consequential: for writing a scientific paper will be drafted reviewing current practice of consequential LCA including an outlook into possible futures of CLCA, no further methodological developments and/or revisions are needed.
- Spatial coverage: if we would like to develop more regionalised impact assessment methods, we would not only have to collect regional modelling data but also similar regional inventory data. This is a huge job, which is not

realistically feasible within the SEAT project and will therefore be skipped. Moreover, the SEAT project also applies Risk Assessment (RA) and RA can take a much more regional to local look into impacts than LCA would ever be able to do.

- Toxicity: adding characterisation factors (CFs) for specific aquaculture chemicals for which CFs may be lacking in the current USEtox CF-list, needs no further methodological developments and/or revisions. It just needs data collection. For some chemicals this data collection will be possible, for other chemicals the input data needed to run the USEtox model will not be available and for these chemicals it will not be possible to calculate CFs.

## 4 Revision proposals and their practical feasibility for selected topics

Below, the following topics will be explored with regard to the existence of elaborated methodological proposals (preferably with associated practical data and/or characterisation factors) or potential for (practically feasible) developments and/or revisions:

- water use;
- salinity;
- benthic impacts from capture fisheries;
- developing (Monte-Carlo based) methods to include multiple variations in parameters and choices / assumptions;

Three topics concern impact categories. The problem of these categories often is that accepted methods and associated characterisation factors (CFs) are missing. In the ILCD (2010) Handbook ([http://ict.jrc.ec.europa.eu/index\\_jrc](http://ict.jrc.ec.europa.eu/index_jrc)) it was concluded that additional research was necessary before a best practice could be identified and adopted for most of these categories and some of the “other impact” categories were not mentioned at all. As part of the SEAT LCAs, we will therefore either have to find the best existing method with CFs or develop new models and CFs for addressing these categories as long as an accepted best practice is lacking.

Below, each of these topics will be discussed in more detail working towards a conclusion on how we can address them as part of the SEAT LCAs.

### 4.1 Topic 1: water use

Bayart *et al.* (2010) give a good overview of all relevant methods that have been proposed to address the topic of water use. They write “Current life cycle inventory (LCI) data only provide information on the volume of freshwater used for product systems, with sometimes limited information about its origin (type of water resource) and none about its fate (volume, quality, and place of release). Life cycle impact assessment (LCIA) methods hardly provide assessment schemes and characterization factors for the potential environmental impacts of freshwater use and its depletion. These methods therefore generally overlook the significant environmental consequences of the decrease in freshwater quality and availability.” The objective of the LCI stage is defined by Bayart *et al.* (2010) as to quantify changes in freshwater availability. These changes are generated by both freshwater consumptive use (a reduction of the net volume of water within the watershed) and freshwater degradative use (a reduction in the availability of freshwater of initial quality in the watershed). In their article, Bayart *et al.* (2010) present the first outcomes of the “Assessment of use and depletion of water resources within the LCA Framework” (WULCA) project, undertaken in the second phase (2007–2010) of the UNEP–SETAC Life Cycle Initiative (<http://lcinitiative.unep.fr/>). They propose a conceptual framework for all identified impacts of off-stream freshwater compatible with the LCA methodology, with consistent terminology, a basic LCI methodology and some suggestions for impact indicators at midpoint and endpoint levels. The focus is, however, on LCI modelling of off-stream freshwater use in order to support quantitative modelling in current and future research. Operational impact assessment methods are not in the scope of the Bayart *et al.* (2010) paper, but remain the final objective of the WULCA project.

The terminology proposed by Bayart *et al.* (2010) is summarized in Table 6 and Table 7 and is partly based terminology proposals by Owens (2001) and Pfister *et al.* (2009).

<b>Terminology</b>	<b>Definition</b>
<b>Freshwater use</b>	Generic term that groups all types of human uses of freshwater resources; in-stream use leaves the resource available for ecosystems
<b>In-stream freshwater use</b>	Use of water in situ (e.g., navigational transport on a river)
<b>Off-stream freshwater use</b>	Use of water that requires human removal from a natural body of water or groundwater aquifer (e.g., pumping or diversion of water for municipal, agricultural, or industrial purposes)
<b>Freshwater degradative use</b>	Withdrawal of water and discharge into the same watershed after the quality of the water has been altered (includes both quality deterioration and improvement)
<b>Freshwater consumptive use</b>	Use of freshwater when release into the original watershed does not occur because of evaporation, product integration, or discharge into different watersheds or the sea
<b>Competition for freshwater Resources</b>	Temporary reduced freshwater availability for current users
<b>Freshwater depletion</b>	Net reduction in the amount/availability of freshwater in a watershed or/and fossil groundwater stock. Depletion occurs when freshwater consumptive use exceeds the renewability rate of the resource over a significant time period

Table 6: Key terminology and definitions related to water use (source: Bayart *et al.* 2010).

	<b>Water use</b>	
	<b>In-stream use</b>	<b>Off-stream use</b>
<b>Consumptive use (depletion: use &gt; renewability rate)</b>	Evaporative loss of canals and water reservoirs used for, e.g., transportation and hydropower generation, respectively	Evaporation of irrigated water in agricultural production Product integration, e.g., in food products Alpine hydropower (dissipative loss of freshwater due to diversion of water from the original flow)
<b>Degradative use</b>	Cooling water of, e.g., a power plant: quality degradation occurs through uptake of heat	Water withdrawals or domestic and industrial purposes and release of effluents from wastewater

and thermal releases into the aquatic environment	plants: generally the water quality is degraded to some extent, e.g., increase in biological oxygen demand (BOD)
---	--

Table 7: Examples illustrating the terminology as proposed in Table 6.

Based on the review of existing proposals to address the topic of water use, Bayart *et al.* (2010) distinguish between four classes of freshwater use, namely in-stream consumptive and degradative uses and off-stream consumptive and degradative uses (see Table 7). In order to support a proper future impact assessment, it is suggested that LCI data on elementary flows of water use:

- should be distinguished according to water *resource types* (e.g., groundwater, surface water) and water *functionalities* ('high-quality aquifer water', 'low-quality river water', etc.);
- should include *freshwater withdrawal as well as freshwater release* after its use to facilitate the calculation of freshwater consumption;
- if available, should (preferably) include *spatial* location (e.g., watershed indication or GIS coordinates) and *temporal* information (e.g., dry and wet season).

*Resource type* (e.g., groundwater, surface water) is the first parameter that should be considered for distinguishing among water types. This distinction is already made in some LCI databases, such as in the ecoinvent database (<http://www.ecoinvent.ch/>). Water quality is suggested by Bayart *et al.* (2010) as the second parameter for water type classification. They state that the definition of quantitative values defining water quality is a complex challenge, which is outside the scope of their paper. Then, they write that quality can be considered using two distinct approaches: distance-to-target or *functionality*. In the distance-to-target approach, the quality of the different water types is assessed by determining the equivalent effort necessary to process each water source to the same final quality. This can be done either by assessing the volume of water required to dilute a given water type to acceptable quality standards (e.g., drinking water quality), or by assessing the energy required to purify a resource at the same quality. In the latter, the quality is considered with a functionality approach, which assesses to which users the water withdrawn and released is functional. Water is considered functional for a particular user if its quality parameters respect accepted standards (for aquaculture, e.g., Svobodová *et al.* 1993) concerning this user. Subsequently, Bayart *et al.* – contrary to their statement that water quality is outside the scope of their paper - adopt the functionality approach. In water footprinting<sup>5</sup>, the water quality dimension is part of the grey water footprint.

<sup>5</sup> The water footprint of a product is the volume of freshwater used to produce the product over the full supply chain. It has three components:

- *Green water footprint* (in-stream freshwater use, excluded by Bayart *et al.* 2010): The volume of rainwater that evaporates during the production of goods; for agricultural products, this is the rainwater stored in soil that evaporates from crop fields.
- *Blue water footprint* (off-stream freshwater use, included by Bayart *et al.* 2010): The volume of freshwater withdrawn from surface or groundwater sources that is used by people and not returned; in agricultural products this is mainly accounted for by evaporation of irrigation water from fields.
- *Grey water footprint* (degradative use, included by Bayart *et al.* 2010, but proposed to be excluded in the SEAT LCAs): the volume of water required to dilute pollutants released in production processes to such an extent that the quality of the ambient water remains above agreed water quality standards.

Sources: WWF 2010; Hoekstra & Chapagain 2008; Hoekstra *et al.* 2009a; Hoekstra *et al.* 2009b.

However, in LCA the water quality aspect is already quantified as emissions to water with potential contributions to other impact categories such as acidification, eutrophication, human toxicity and aquatic ecotoxicity. Also including the quality aspect in the assessment of freshwater use leads to double-counting, at least in LCA. *Therefore we propose to exclude the water quality aspect from freshwater use inventories and impact assessment in the SEAT LCAs.*

In addition to quantifying the volume of freshwater entering the product system (*withdrawal*), LCA datasets should include the volume of water leaving it (*emission*). The balance of each elementary flow makes it possible to quantify the net changes (*consumption*) of availability for each of them.

Bayart *et al.* illustrate these three aspects of water use by an example for cotton crops. Pakistan consumes about 10 m<sup>3</sup> irrigation water per kilogram cotton, while the total irrigation water use amounts 25 m<sup>3</sup>/kg. This results in 15 m<sup>3</sup>/kg degradative water use, which is subject to quality alteration. Assuming irrigation water withdrawn from an aquifer of good quality and drained back to a river, the elementary flows are defined as follows for the water quality inventory method:

- withdrawal ('consumption' in Bayart *et al.* but that is not consistent with the consumption term used above) of 25 m<sup>3</sup> of water type 'high-quality aquifer water';
- emission of 15 m<sup>3</sup> water type 'low-quality river water' (gain of this water type).

With a functionality approach, this could result in the consumption of 25 m<sup>3</sup> of water 'functional for agriculture, domestic, and industrial', and emission of 15 m<sup>3</sup> water 'functional for agriculture, industrial, transport, and hydropower' for example.

As most current water use impact assessment methods preferably require high *spatial* resolution, such spatial information should preferably already be included in the inventory analysis of water use. Watersheds and GIS-information are just two forms of how this spatial information could look like. Water availability may also have a significant temporal variability, which on its turn may be an important parameter for impact assessment methodologies. Thus such temporal information (e.g., dry or wet season) should preferably already be included in the inventory analysis of water use.

*Summarizing, we propose:*

- *to adopt the Bayart et al. (2010) terminology and LCI suggestions in the SEAT project, and to consider the practical possibilities of impact assessment methodologies for off-stream freshwater use in a later stage of the SEAT project, since at that time the WULCA suggestions for impact assessment of off-stream freshwater use might be available and published as yet.*
- *to exclude the water quality aspect from freshwater use inventories and impact assessment in the SEAT LCAs.*
- *to collect the following LCI data for the SEAT LCAs for freshwater use:*
  - *withdrawal/ extraction in m<sup>3</sup> specified, if possible, in resource type (groundwater, surface water from lake, surface water from river, ..), spatial (watershed indication or GIS coordinates) and in temporal terms (dry or wet season);*
  - *emission extraction in m<sup>3</sup> specified, if possible, in resource type (to groundwater, to surface water lake, to surface water river, ..), spatial (watershed indication or GIS coordinates) and in temporal terms (dry or wet season).*

## 4.2 Topic 2: salinity

Aquatic and/or terrestrial salinity is a major problem in particularly South-Africa and in Australia. Efforts to develop impact categories for salinity impacts thus logically origin from these two regions. Two proposals for modelling salinity impacts have been developed, one by Feitz & Lundie (2002) and one by Leske & Buckley (2003; 2004a; 2004b)

The basis for the Salinisation Potential impact model by Feitz & Lundie (2002) is “the adjusted sodium adsorption ratio expression for irrigation waters. The Sodium Adsorption Ratio (SAR) of an irrigation water is a relationship between  $\text{Na}^+$ ,  $\text{Ca}^{2+}$  and  $\text{Mg}^{2+}$  concentrations that predicts the  $\text{Na}^+$  status of the soil exchange complex when the exchange of cations within the soil comes into equilibrium with the soil solution or infiltrating irrigation water. It is a common index for the suitability of water for irrigation”. Feitz & Lundie provide all necessary equations and parameters needed to calculate salinisation potentials, but the data for the parameters need to be collected for each LCA study separately.

Salinity could be included in the SEAT LCAs in a simple way and focusing on soil salinity by adopting Feitz & Lundie (2002) and collecting irrigation water specific data on:

- Electrical Conductivity (EC; siemens per metre ( $\text{S}\cdot\text{m}^{-1}$ ));
- Na (g/l);
- Ca (g/l);
- Mg (g/l);
- Alkalinity (as  $\text{CaCO}_3$ ; g/l);
- pH;
- $\text{SO}_4$  (g/l); and
- Total volume of irrigation water per functional unit (l/fu).

Feitz & Lundie (2002) developed a soil salinisation impact model indicating potential land degradation from poor irrigation practices. They write that “land salinity is closely related to other land degradation processes such as soil erosion, acidification and structural decline. Grossly salt-affected land, where water infiltration is poor, leads to a reduction or complete loss in vegetation and a greater susceptibility to erosion. Erosion of topsoil can also hasten the expression of land salinity by exposing saline undersoil, hence widening the problem to include dryland salinity and more than one form of land degradation. Hence, soil salinisation is a simple yet highly correlated indicator that can be incorporated into existing LCA as a measure of the potential for land degradation caused by a product.”

Salinity could also be included in the SEAT LCAs in a more sophisticated and comprehensive way adopting Leske & Buckley (2003; 2004a; 2004b). Leske & Buckley developed a nested global fate-exposure-effect model for ionic compounds comparable to USES-LCA developed by Huijbregts (1999b). Their ultimate aim was to develop a nested global fate and effect model, “with emissions entering a mathematically defined ‘standard South African environment’. This model should be able to calculate characterisation factors for all compounds (organic and inorganic) and should include a separate salinity impact category.” As a start, they developed a regional (using a ‘standard South African environment’) environmental fate and effect model that specifically accounts for salinity effects. The latter model is currently available for South African conditions and for the lumped sum of total dissolved salts (TDS) only (instead of individual ionic substances). They recommend that in future these individual ionic substance be modelled as yet, but for that some level of speciation should be built into the model and at the time of their work some data needed for that were lacking, most notably adsorption constants for individual ions.

Leske & Buckley (2003; 2004a; 2004b) developed their salinity impact model to indicate impact on the following subcategories:

- material damage;
- aesthetic effects;
- aquatic ecosystem effects; and
- terrestrial ecosystem effects, comprising separate effects on terrestrial plants and animals.

A total salinity potential for the release of a ionic compound into an initial release compartment.

Feitz & Lundie (2002) and the Leske & Buckley (2003; 2004a; 2004b) thus model different impacts, developed different indicators for similar impacts () and apply different models.

As both the Feitz & Lundie (2002) and the Leske & Buckley (2003; 2004a; 2004b) approaches demand either a substantial amount of field measurements and/or a substantial amount of modelling parameter adaptations (in order to adapt it to other countries than South-Africa).

*It is currently not considered as feasible to include this impact category into the SEAT project. The data and modelling efforts needed in relation to the expected relevance of salinity impacts with respect to aquaculture systems is considered too huge. Only if salinity would appear to be a serious concern, this impact category will be considered as yet.*

### **4.3 Topic 3: benthic impacts form capture fisheries**

Aquaculture is still heavily reliant on wild capture fisheries in order to provide fishmeal and oil to feed aquaculture (Naylor *et al.* 2009). The environmental consequences related to this dependence have long attracted negative publicity towards fish farming (Naylor *et al.* 2000; Naylor *et al.* 2009). One of these is the destruction of benthic communities from fishing gear (Hiddink *et al.* 2006). Benthic impacts from capture fisheries have, to date, been poorly covered in LCA methodology. There is, however, a need to implement these impacts in order to justifiably account for the concerns of aquatic resources (Ziegler 2003; Ellingsen & Aanondsen 2006).

As the impacts of trawling are highly complex, the implementation of this impact category will depend upon the work of Ziegler *et al.* at SIK (The Swedish Institute for Food and Biotechnology), Sweden, who are currently developing characterisation factors for just this (pers. com.).

### **4.4 Topic 4: developing (Monte-Carlo based) methods to include multiple variations in parameters and choices / assumptions**

Goal and scope (e.g., the life time of a product), inventory (e.g., process data) and impact assessment (e.g., GWPs) parameters may have different ranges of values, sensitivity analyses on methods (allocation, impact assessment) and assumptions (e.g., potential recyclability of a material) etc. etc. may all influence the eventual result (inventory result, characterisation result, normalisation result, weighting result) of an LCA. In current LCAs, sensitivity analyses are done here and there, partial uncertainty analyses are performed, and sometimes both of these analyses are performed as part of one LCA. However, seldom or never, these analyses are combined with a probability estimations and presented as probabilities of LCA

results. The results of uncertainty analyses and all possible sensitivity analyses should actually be combined as all possible and relevant combinations of these uncertainty and sensitivity analyses, plotting the associated LCA results. One way of doing this is applying a Monte Carlo method. Monte Carlo methods are a class of computational algorithms that rely on repeated random sampling to compute their results. Monte Carlo simulation methods are especially useful in studying systems with a large number of coupled degrees of freedom, such as fluids, disordered materials, strongly coupled solids, and cellular structures ([http://en.wikipedia.org/wiki/Monte\\_Carlo\\_method](http://en.wikipedia.org/wiki/Monte_Carlo_method)) .... and thus also LCA. LCA outcomes are highly uncertain, depend on broadly debated methods that can often only be partially validated and involve many assumptions. One way of getting better insight into how combinations of different assumptions, data etc. can influence a final LCA results, is to randomly draw the different ranges of parameters and combine these with randomly selected methods and assumptions. This will result in a pallet of different LCA results with a certain frequency, which can be plotted. The resulting frequency distribution of LCA results will provide a much more realistic basis for improvement options, decision-making etc. than LCAs usually do.

Aquaculture production is strongly coupled with the ecosystem and production outputs are therefore dependent upon natural fluctuations. Moreover, practices differ between species, countries, regions and individual farmers. Many of the sub-processes that support aquaculture production also experience annual changes in both in- and outputs. Agricultural crop yields, e.g., experience great changes over both temporal and spatial scales (Naylor *et al.* 1997; Röös *et al.* 2010). Despite this, no LCA study on aquaculture has to date applied any form of variation analysis into their computations. In order to do any sort of up-scaling of results, these variations need to be quantified. We will explore the possibilities applying Monte-Carlo analysis to quantify all possible variations and uncertainties in an integral way.

## 5 How will SEAT LCAs thus look like?

Below, the choices made for all methodological steps specified in chapters 2 and 4 and together this will constitute the basic outline of the SEAT LCA case studies are summarized in one table (Table 8).

Phase/step	Choices made
<b>Goal and scope definition</b>	
Goals	<ul style="list-style-type: none"> <li>Getting insight in the environmental impact and its causes of aquaculture systems for tilapia, catfish, shrimp and prawn in China, Vietnam, Thailand and Bangladesh.</li> <li>Getting insight in starting points (“hot spot identification”) for improving the environmental performance of aquaculture systems for tilapia, catfish, shrimp and prawn in China, Vietnam, Thailand and Bangladesh, which includes insight into the effects of choices in methods and data on the outcomes.</li> <li>Learning.</li> </ul>
Attributional or consequential	Attributional. <i>This topic will be addressed in a scientific paper, separate from individual LCA studies.</i>
Temporal coverage	Average data over longer periods will be collected, properly reflecting potential fluctuations. For collection of farm data the reference year is 2010; for other processes a reference year will be adopted as close as possible to 2010.
Spatial coverage	Thailand: Chanthaburi and Surat Thani. China: Guangdong province and Hainan Island. Vietnam: Mekong delta. Bangladesh: Khulna and possible other regions.
Technology coverage	Different farming systems and mixed technologies will be covered
Functional unit	1000 kg of edible yield (either or not corrected for nutritional value) of (frozen, packed or etc. is to be determined later) species X produced on farm type Y in country Z for consumption in the EU (see D2.4)
<b>Inventory analysis</b>	
System boundaries	A system boundary from farm - including the production of all upstream inputs to the farm - to consumption (“from farm to fork”) is adopted. Human consumption of fish itself is not included as explained in deliverable D2.4. Where applicable and appropriate we will apply EIOA (extended Input-Output Analysis) to estimate the potential significance of lacking data.
Data availability and quality	Main efforts will be put into collection of Asia foreground data of the actual aquaculture processes including fisheries, local feeds, nurseries, hatcheries, farms etc. correcting for losses due to parasites and diseases and including risks of failed production caused by intensive farming in the production figures, and paying attention to the public availability (or non-disclosure) of the resulting database. For this national life cycle inventory databases (if existing; Thailand, e.g., has such a database), peer-reviewed journals and technical reports from recognised organisations will be consulted.
Allocation	Thorough justification will be provided for allocation choices made. In addition, sensitivity analyses on these allocation

Phase/step	Choices made
	choices will be conducted in compliance with ISO standards.
<b>Life Cycle Impact assessment (LCIA)</b>	
Global Warming	Latest GWPs from IPCC.
Acidification	Updated Guinée <i>et al.</i> (2002) baseline method for acidification and other more recent (variants of baseline) methods and characterisation factors, replenished by the midpoint ILCD LCIA methods and factors for acidification.
Eutrophication	Updated Guinée <i>et al.</i> (2002) baseline method for eutrophication and other more recent (variants of baseline) methods and characterisation factors, replenished by the midpoint ILCD LCIA methods and factors for eutrophication.
Abiotic Resource use	Updated Guinée <i>et al.</i> (2002) ADP method for abiotic depletion and with other more recent (variants of baseline) methods and characterisation factors, replenished by the midpoint ILCD LCIA methods and factors for abiotic depletion.
Toxicity	USEtox™ model with <i>additional CFs calculated for specific aquaculture chemicals for which CFs may be lacking</i> in the current USEtox CF-list.
Photochemical ozone formation	Updated Guinée <i>et al.</i> (2002) baseline method for photochemical ozone formation and other more recent (variants of baseline) methods and characterisation factors, replenished by the midpoint ILCD LCIA methods and factors for photochemical ozone formation and updated POCPs for Southern Asia by Cheng <i>et al.</i> (2010).
Cumulative energy demand	CED will be included as a non-baseline indicator. The results of this indicator will be presented separately stressing the overlap it has with particularly abiotic depletion.
Water use	The Bayart <i>et al.</i> (2010) terminology and suggestions for LCI are adopted. Their future suggestions for impact assessment methodologies will be considered in a later stage of the SEAT project. The water quality aspect from freshwater use inventories and impact assessment is excluded from the SEAT LCAs. Within the SEAT LCAs, the following LCI data will be collected for freshwater use: <ul style="list-style-type: none"> <li>• withdrawal/ extraction in m<sup>3</sup> specified, if possible, in resource type (groundwater, surface water from lake, surface water from river, ..), spatial (watershed indication or GIS coordinates) and in temporal terms (dry or wet season);</li> <li>• emission extraction in m<sup>3</sup> specified, if possible, in resource type (to groundwater, to surface water lake, to surface water river, ..), spatial (watershed indication or GIS coordinates) and in temporal terms (dry or wet season).</li> </ul>
Land use	Where possible and relevant, indirect and direct land use occupation (in m <sup>2</sup> ·yr) and transformation (in m <sup>2</sup> ) will be inventoried. On top of this, the recommended midpoint ILCD LCIA method for land use and/or the UNEP-SETAC's operational characterization factors for land use impacts on biodiversity and ecosystem services will be applied, if feasible. Since land use impact cannot (yet) be properly handled in LCA, other WPs of the SEAT project will have to supply additional information on land use impacts from aquaculture systems.
Biotic resource use	Will possibly be included depending on the results of the <i>scientific paper reviewing the inter-relations between biotic</i>

Phase/step	Choices made
	<i>resource use, resource depletion, and land use and biodiversity impacts. In this paper the available methods within the LCA community and outside that community will be used as a starting point. A framework for addressing these impacts within LCA (or outside LCA) will be proposed.</i>
Salinity	It is currently not considered as feasible to include this impact category into the SEAT project. The data and modeling efforts needed in relation to the expected relevance of salinity impacts with respect to aquaculture systems is considered too huge.
Benthic impacts (from capture fisheries)	It is currently not considered as feasible to include this impact category into the SEAT project. Only if the work by Ziegler <i>et al.</i> at SIK (The Swedish Institute for Food and Biotechnology) provides practically applicable results, this decision may be reconsidered.
Normalisation and weighting	Updated CML normalisation methods will be applied. In principle, weighting will not be performed, since weighting is not needed for the prime goals of the SEAT LCAs. If weighting is needed as yet, one or more weighting methods from a recent study on weighting methods by Huppel & Oers will be adopted.
<b>Interpretation</b>	
Contribution analysis	Contribution analyses will be performed at all relevant levels (inventory analysis, characterization and, if useful, normalization) and for different elements (processes and interventions). Results of these analyses will be used for tracing possible errors and as basis for identifying improvement options.
Sensitivity analysis	Several kinds of sensitivity analysis will be performed on a broad range of data and methodological choices. For this, <i>the use and development of (Monte-Carlo based) methods to include multiple variations in parameters and choices/assumptions will be explored.</i>

Table 8: Basic outline of the SEAT LCA case studies.

## 6 Acknowledgement

This work is part of the SEAT project, which is co-funded by the European Commission within the Seventh Framework Programme - Sustainable Development Global Change and Ecosystem (project no. 222889).

## References

1. Amann M, Cofala J, Heyes C, Klimont Z, Schöpp W (1999): The RAINS model: A tool for assessing regional emission control strategies in Europe. *Pollution Atmosphérique* 20: 41–46.
2. Andersson K, Ohlsson T, Olsson P, (1994) Life cycle assessment (LCA) of food products and production systems. *Trends in Food Science and Technology* 5 (5): 134-138.
3. Aubin, J., Papatryphon, E., van der Werf, H., Petit, J., Morvan, Y. (2006). Characterisation of the environmental impact of a turbot (*Scophthalmus maximus*) re-circulating production systems using Life Cycle Assessment. *Aquaculture* 261: 1259-1268.
4. Aubin, J., Papatryphon, E., van der Werf, H.M.G. Chatzifotis, S. (2009). Assessment of the environmental impact of carnivorous finfish production systems using life cycle assessment. *J. Cleaner Prod.* 17 (3): 354-361.
5. Avnimelech Y., Ritvo G., (2003) Shrimp and fish pond soils: processes and management. *Aquaculture* 220: 549-567.
6. Ayer, N., Tyedmers, P. (2009). Assessing alternative aquaculture technologies: life cycle assessment of salmonid culture systems in Canada. *J. Cleaner Prod.* 17 (3): 362-373.
7. Bayart, J-B, Bulle, C, Deschênes, L, Margni, M, Pfister, S, Vince, F, Koehler, A (2010). A framework for assessing off-stream freshwater use in LCA. *Int J Life Cycle Assess* 15 (3): 439–453.
8. Blackhurst M., Hendrickson C., Vidal J. S., (2010) Direct and Indirect Water Withdrawals for U.S. Industrial Sectors. *Environ. Sci. Technol.* 44: 2126-2130.
9. Borlaug NE (2007) Sixty-two years of fighting hunger: personal recollections. *Euphytica* 157:287-297.
10. Boyd C.E., Tucker C., McNevin A., Bostick K., Clay J. (2007). Indicators of Resource Use Efficiency and Environmental Performance in Fish and Crustacean Aquaculture. *Reviews in Fisheries Science* (15): 327-360.
11. Brummett, R.E. (2007) Comparative analysis of the environmental costs of fish farming and crop production in arid areas, In D.M. Bartley, C. Brugère, D. Soto, P. Gerber and B. Harvey (eds). Comparative assessment of the environmental costs of aquaculture and other food production sectors: methods for meaningful comparisons, FAO/WFT Expert Workshop. 24-28 April 2006, Vancouver, Canada. FAO Fisheries Proceedings. No. 10. Rome, FAO. 2007: 221–228.
12. BUWAL (1996). *Ökoinventare für Verpackungen*, Band I u. II, Schriftenreihe Umwelt 250, BUWAL Bern.
13. Cheng H.R., Guo H., Saunders S.M., Lam S.H.M., Jiang F., Wang X.M., Simpson I.J., Blake D.R., Louie P.K.K., Wang T.J. (2010) Assessing photochemical ozone formation in the Pearl River Delta with a photochemical trajectory model. *Atmos. Environ.* 44: 4199-4208.
14. Cho C. Y., Bureau D. P. (2001). A review of diet formulation strategies and feeding systems to reduce excretory and feed wastes in aquaculture. *Aquaculture Research* 32 (1): 349-360.
15. Crespi, V.; Coche, A. (2008). Glossary of aquaculture/Glossaire d'aquaculture/Glosario de acuicultura, FAO, Rome.
16. d'Orbcastel, R.E., Blancheton, J.-E., Aubin, J. (2009): Towards environmentally sustainable aquaculture: Comparison between two trout farming systems using Life Cycle Assessment. *Aquacultural Engineering* 40: 113-119.

17. Dias de Oliveira M.E., Vaughan E., Rykiel E.J. JR, (2005) Ethanol as Fuel: Energy, Carbon Dioxide Balances, and Ecological Footprint. *BioScience* 55 (7): 593-602.
18. Derwent, R.G., Jenkin, M.E., Saunders, S.M., Pilling, M.J. (1998). Photochemical ozone creation potentials for organic compounds in Northwest Europe calculated with a master chemical mechanism. *Atmos. Environ.* 32 (14–15): 2429–2441.
19. Derwent, R.G., Jenkin M.E., Passant N.R., Pilling M.J. (2007) Reactivity-based strategies for photochemical ozone control in Europe. *Environ. Sci. Pol.* 10: 445-453.
20. Deutsch L., Gräslund S., Folke C., Troell M., Huitric M., Kautsky N., Lebel L., (2007). Feeding aquaculture growth through globalization: Exploitation of marine ecosystems for fishmeal. *Global Environmental Change* 17: 238-249.
21. Dierberg F. E., Kiattisimkul W. (1996). Issues, Impacts, and Implications if Shrimp Aquaculture in Thailand. *Environmental management* 20 (5): 649-666.
22. ECETOC (1994). Ammonia Emissions to Air in Western Europe. Technical report no. 62. European Chemical Industry Ecology & Toxicology Centre, Brussels, Belgium.
23. ecoinvent: <http://www.ecoinvent.ch/>.
24. EDF 2004 Rapport annuel EDF 2004.
25. Edwards, P., Le Anh Tuan, Allan, G.L. (2004). A survey of marine trash fish and fish meal aquaculture feed ingredients in Vietnam. ACIAR Working Paper No. 57.
26. Ellingsen, H., Aanondsen, S. (2006). Environmental Impacts of Wild Caught Cod and Farmed Salmon – A Comparison with Chicken. *Int J Life Cycle Assess* 11 (1): 60-65.
27. Frischknecht, R., Bollens, U., Bosshart, S., Ciot, M., Ciseri, L., Doka, G., Dones, R., Gantner, U., Hirschier, R., Martin, A. 1996: Ökoinventare von Energiesystemen. Grundlagen für den ökologischen Vergleich von Energiesystemen und den Einbezug von Energiesystemen in Ökobilanzen für die Schweiz. Auflage No. 3, Gruppe Energie - Stoffe - Umwelt (ESU), Eidgenössische Technische Hochschule Zürich und Sektion Ganzheitliche Systemanalysen, Paul Scherrer Institut, Villigen, Bundesamt für Energie (Hrsg.), Bern.
28. FAO 2009, FAO, State of world fisheries and aquaculture 2008, Rome. <ftp://ftp.fao.org/docrep/fao/011/i0250e/i0250e.pdf>.
29. Feitz, A.; Lundie, S. (2002). Soil Salinisation: A Local Life Cycle Assessment Impact Category. *Int J Life Cycle Assess* 7 (4): 244-249.
30. Folke C., (1988). Energy Economy of Salmon Aquaculture in the Baltic Sea. *Environmental Management* 12 (4): 525-537.
31. Folke C., Kautsky N., Berg H., Jansson Å., Troell M. (1998). The Ecological Footprint Concept for Sustainable Seafood Production: a Review. *Ecological Society of America* 8 (1): S63-S71.
32. Frischknecht, R., Althaus, H.-J., Bauer, C., Doka, G., Heck, T., Jungbluth, N., Kellenberger, D., Nemecek, T. (2007). The Environmental Relevance of Capital Goods in Life Cycle Assessments of Products and Services. *Int J Life Cycle Assess* 12 (Special Issue 1; 1A, 2A): 7-17.
33. Gaillard, G., Crettaz, P. and Hausheer, J. (1997). Inventaire Environnemental des Intrants Agricoles en Production Végétale. Base de Données pour l'Établissement des Bilans Énergétiques et Écologiques en Agriculture. Station Fédérale de Recherche en Économie et Technologies Agricoles. Tänikon, Switzerland.
34. Galford G.L., Melillo J.M., Kicklighter D.W., Cronin T.W., Cerri C.E.P., Mustard J.F., Cerri C.C. (2010). Greenhouse gas emissions from alternative futures of deforestation and agricultural management in the southern

- Amazon, Proceedings of the National Academy of Sciences, Early edition, doi: 10.1073/pnas.1000780107.
35. Gallego A, Rodríguez L, Hospido A, Moreira1 M, Feijoo G (2010). Development of regional characterization factors for aquatic eutrophication, *Int J Life Cycle Assess* (15)1: 32-43.
  36. Galloway J.N. (1995). Acid deposition: Perspectives in Time and Space. *Water, Air and Soil Pollution* 85: 15-24.
  37. Godfray H. C., Beddington J. R., Crute I. R., Haddad L, Lawrence D, Muir J. F., Pretty J., Robinson S., Thomas S. M., Toulmin C., (2010): Food Security: The Challenge of Feeding 9 Billion People. *Science* 327 (5967): 812-818.
  38. Goedkoop, M., Spriensma, R. (1999). The Eco-indicator 99 - A damage oriented method for life cycle Impact assessment – Methodology Report; PRé Consultants: Amersfoort, the Netherlands. Available at: <http://www.pre.nl/eco-indicator99/ei99-reports.htm>
  39. Grönroos J., Seppälä J., Silvenius F., Mäkinen T. (2006). Life cycle assessment of Finnish cultivated rainbow trout. *Boreal Environment Research* 11: 401-414.
  40. Guinée, J.B. and R. Heijungs, 1995. A proposal for the definition of resource equivalency factors for use in product Life-Cycle Assessment. *Environmental Toxicology and Chemistry* 14 (5): 917-925.
  41. Guinée, J.B., 1995. Development of a methodology for the Environmental Life-cycle assessment of products, with a case study on margarines. Leiden: Institute of Environmental Sciences, thesis, 225pp.
  42. Guinée J.B., Heijungs R., Huppes G. (2004). Economic allocation: Examples and derived decision tree. *Int J Life Cycle Assess* 9 (1): 23–33.
  43. Guinée, Jeroen, Lauran van Oers, Arjan de Koning and Wil Tamis, 2006. Life cycle approaches for Conservation Agriculture - Part I: A definition study for data analysis; Part II: Report of the Special Symposium on Life Cycle Approaches for Conservation Agriculture on 8 May 2006 at the SETAC-Europe 16th Annual Meeting at The Hague. Research commissioned by Syngenta Crop Protection AG, Basel. CML report 171, CML, Leiden. Available from: [http://www.leidenuniv.nl/cml/ssp/publications/life\\_cycle\\_conserv\\_agri.pdf](http://www.leidenuniv.nl/cml/ssp/publications/life_cycle_conserv_agri.pdf)
  44. Guinée, J. B., R. Heijungs, G. Huppes, A. Zamagni, P. Masoni, R. Buonamici, T. Ekvall and T. Rydberg (2011). Life cycle assessment: past, present and future. *Environ. Sci. Technol.* 45 (1): 90–96.
  45. Guinée, J.B. (Ed.), Gorrée, M., Heijungs, R., Huppes, G., Kleijn, R., Koning, A. de, Oers, L. van, Wegener Sleeswijk, A., Suh, S., Udo de Haes, H.A., Bruijn, J.A. de, Duin, R. van, Huijbregts, M.A.J. (2002). Handbook on Life Cycle Assessment: Operational Guide to the ISO Standards, Kluwer Academic Publishers. Dordrecht, the Netherlands.
  46. Guinée, J.B., Kleijn, R., Henriksson, P. (2010). Environmental Life Cycle Assessment of South-East Asian Aquaculture Systems for Tilapia, Pangasius Catfish, Penaeid Shrimp and Macrobrachium Prawns: Goal & Scope Definition Report. Deliverable 2.4 of the SEAT project (<http://seatglobal.eu/>).
  47. Heijungs R., Huppes, G., Guinée, J.B. (2010). Life cycle assessment and sustainability analysis of products, materials and technologies. Towards a scientific framework for sustainability life cycle analysis. *Polymer Degradation and Stability* 95 (3): 422-428.
  48. Heijungs, R., Guinée, J., Huppes, G., Lankreijer, R.M., Udo de Haes, H.A., Wegener Sleeswijk, A., Ansems, A.M.M., Eggels, P.G., Duin, R. van, Goede, H.P. de (1992). Environmental Life Cycle Assessment of products. Guide and Backgrounds. CML, Leiden University, Leiden.

49. Heijungs, R., J. Guinée, G. Huppes (1997). Impact Categories for natural resources and land use Leiden: Centre of Environmental Science, CML-report 138, ISBN: 90-5191-111-4.
50. Heijungs, R., Kleijn, R. (2001). Numerical Approaches Towards Life Cycle Interpretation - Five Examples. *Int J Life Cycle Assess* 6 (3): 141–148.
51. Hicks K.W, Kuylenstierna J.C.I., Owen A., Dentener F., Seipand H-M., Rodhe H., (2008) Soil Sensitivity to Acidification in Asia: Status and Prospects. *Ambio* 37 (4): 295-303.
52. Hiddink J.G., Jennings S., Kaiser M.J., Queirós A.M., Duplisea D.E., Piet G.J., (2006) Cumulative impacts of seabed trawl disturbance on benthic biomass, production, and species richness in different habitats. *Canadian Journal of Fisheries and Aquatic Sciences* 63 (4): 721-736.
53. Hoekstra, A.Y. and Chapagain, A.K. (2008). Globalization of water: Sharing the planet's freshwater resources. Blackwell Publishing, Oxford, UK.
54. Hoekstra, A.Y., Chapagain, A.K., Aldaya, M.M. and Mekonnen, M.M. (2009a). Water footprint manual: State of the art 2009. Water Footprint Network, Enschede, the Netherlands.
55. Hoekstra, A.Y., Gerbens-Leenes, W., Meer, Th. van der (2009b). Reply to Pfister and Hellweg: Water footprint accounting, impact assessment, and life-cycle assessment. *PNAS* 106 (40): E114.
56. Houghton JT, Ding Y, Griggs DJ, Noguer M, van der Linden PJ, Dai X, Maskell K, Johnson CA, (2001) Climate Change 2001: The Scientific Basis, Intergovernmental Panel on Climate Change (IPCC). Cambridge Univ Press, Cambridge, UK.
57. Huijbregts M., Hellweg S., Frischknecht R., Hendriks H W. M., Hungerbühler K., Hendriks J. A., (2010) Cumulative Energy Demand As Predictor for the Environmental Burden of Commodity Production *Environ. Sci. Technol.* 44: 2189-2196.
58. Huijbregts M.A.J., (1999a), Life Cycle Impact Assessment of Acidifying and Eutrophying Air Pollutants. Calculations of equivalency factors with RAINS-LCA, Interfaculty Department of Environmental Science, Faculty of Environmental Science, University of Amsterdam.
59. Huijbregts M.A.J., (1999b), Priority assessment of toxic substances in LCA, Development and application of the multi-media fate, exposure and effect model USES-LCA, IVAM environmental research, University of Amsterdam.
60. Huijbregts M.A.J., Seppälä J., (2001) Life Cycle Impact Assessment of Pollutants Causing Aquatic Eutrophication. *Int J Life Cycle Assess* 6 (6): 339-343.
61. Huppes, G., Oers, L. van (2009). Recommendations on an appropriate weighting approach for measuring the EU-27 overall environmental impact. D1: Technical review of existing weighting approaches in LCIA. EC Joint Research Centre, Institute for Environment and Sustainability, Ispra, Italy.
62. Huppes, G., Oers, L. van (2010). Recommendations on an appropriate weighting approach for measuring the EU-27 overall environmental impact. D2 & D3: Operational methods and combined weighting tool. EC Joint Research Centre, Institute for Environment and Sustainability, Ispra, Italy.
63. ILCD, International Reference Life Cycle Data System Handbook, (2010) Analysis of existing Environmental Impact Assessment methodologies for use in Life Cycle Assessment, JRC, European Commission .
64. International Organization for Standardization (2006a). Environmental management—Life cycle assessment—Principles and framework (ISO 14040:2006). International Organization for Standardization, Geneva, Switzerland.
65. International Organization for Standardization (2006b). Environmental management—Life cycle assessment—Requirements and guidelines (ISO

- 14044:2006). Geneva, Switzerland  
[http://www.iso.org/iso/iso\\_catalogue/catalogue\\_tc/catalogue\\_detail.htm?csnumber=3849](http://www.iso.org/iso/iso_catalogue/catalogue_tc/catalogue_detail.htm?csnumber=3849).
66. Iribarren, D., Moreira, M., Gumersindo, F. (2010). Revisiting the Life Cycle Assessment of mussels from a sectorial perspective. *J. Cleaner Prod.* 18 (2): 101-111.
  67. IUCN/WWF/UNEP (1991). *Caring for the earth. A strategy for sustainable living*. Gland, Switzerland.
  68. Jacquet J., Hocevar J., Lai S., Majluf P., Pelletier N., Pitcher T., Sala E., Sumaila R., Pauly D., (2009) Conserving wild fish in a sea of market-based efforts, *Oryx*, doi:10.1017/S0030605309990470.
  69. Jenkin, M.E., Hayman, G.D. (1999). *Photochemical ozone creation potentials for oxygenated volatile organic compounds: sensitivity to variations in kinetic and mechanistic parameters*. *Atmos. Environ.* 33 (8): 1275–1293.
  70. Klöpffer, W. (2008). Life Cycle Sustainability Assessment of Products. *Int J Life Cycle Assess* 13 (2): 89-94.
  71. Klöpffer, W., Renner, I. (2008). Life-Cycle Based Sustainability Assessment of Products. Chapter 5 in: Schaltegger, S., Bennett, M., Burritt, R.L., Jasch, C. (Eds.): *Environmental Management Accounting for Cleaner Production*. ISBN 978-1-4020-8912-1. Springer, pp. 91-102.
  72. Krkosek M., Lewis M.A., Volpe J.P. (2005). Transmission dynamics of parasitic sea lice from farm to wild salmon. *Proceedings of the Royal Society of Biology* 272: 689-696.
  73. Kruse S, Flygsjö A, Kasperczyk N, Scholz A (2009) Socioeconomic indicators as a complement to life cycle assessment—an application to salmon production systems. *Int J Life Cycle Assess* 14: 8-18.
  74. Larsson J., Folke C., Kautsky N., (1994) Ecological Limitations and Appropriation of Ecosystem Support by Shrimp Farming in Colombia. *Environmental Management* 18 (5): 663-676.
  75. Leske, T.; Buckley, C. (2003). Towards the development of a salinity impact category for South African environmental life-cycle assessments: Part 1 - A new impact category. *Water SA* 29 (3): 289-296.
  76. Leske, T.; Buckley, C. (2004a). Towards the development of a salinity impact category for South African life cycle assessments: Part 2 - A conceptual multimedia environmental fate and effect model. *Water SA* 30 (2): 241-251.
  77. Leske, T.; Buckley, C. (2004b). Towards the development of a salinity impact category for South African life cycle assessments: Part 3 – Salinity potentials. *Water SA* 30 (2): 253-265.
  78. Lewis D., (1997) Rethinking aquaculture for resource-poor farmers: perspectives from Bangladesh. *Food Policy* 22 (6): 533-546.
  79. Lindfors, L.G., 1996. *A Desk study on characterisation methods applicable in EU Ecolabeling programmes*. In: Udo de Haes, H.A., R. Clift, R. Griesshammer, L. Grisel & A.A. Jensen, 1996. *Practical guidelines for Life Cycle assessment for the EU ecolabeling programme*. Leiden.
  80. Meent D. van de, Klepper O., (1997), Mapping the Potential Affected Fraction (PAF) of Species as an indicator of Generic Toxic Stress, RIVM report 607504001, Bilthoven.
  81. Mungkung, R. (2005). Shrimp aquaculture in Thailand: application of life cycle assessment to support sustainable development . Ph.D. thesis. Center for Environmental Strategy, School of Engineering, University of Surrey, Surrey, UK.
  82. Mungkung, Rattanawan, Helias A. Udo de Haes, Roland Clift (2006). Potentials and Limitations of Life Cycle Assessment in Setting Ecolabelling Criteria: A Case Study of Thai Shrimp Aquaculture Product. *Int J Life Cycle Assess* 11(1): 55 – 59.

83. Mungkung, R. & Gheewala, S. 2007. Use of life cycle assessment (LCA) to compare the environmental impacts of aquaculture and agri-food products. In D.M. Bartley, C. Brugère, D. Soto, P. Gerber and B. Harvey (eds). *Comparative assessment of the environmental costs of aquaculture and other food production sectors: methods for meaningful comparisons*. FAO/WFT Expert Workshop. 24-28 April 2006, Vancouver, Canada. FAO Fisheries Proceedings. No. 10. Rome, FAO. 2007, pp. 87–96.
84. Naylor R, Falcon W, Zavaleta E (1997) Variability and Growth in Grain Yields, 1950-94: Does the Record Point to Greater instability? *Population and Development Review* 23 (1): 41-58.
85. Naylor R., Goldberg R., Primavera J., Kautsky N., Beveridge M., Clay J., Folke C., Lubchenco J., Mooney H., Troell M., (2000): Effect of aquaculture on world fish supplies. *Nature* 405: 1017-1024.
86. Naylor R., Williams S., Strong D. (2001): Aquaculture—A Gateway for Exotic Species. *Science* 294: 1655-1656.
87. Naylor R, Hardy R., Bureau D., Chiu A., Elliott M., Farrell A., Forster I., Gatlin D., Goldberg R., Hua K., Nichols P., (2009) Feeding aquaculture in an era of finite resources. *Proceedings of the National Academy of Sciences* 106 (36): 15103-15110.
88. Newell R.I.E. (2004). Ecosystem Influences of Natural and Cultivated Populations of Suspension-feeding Bivalve Molluscs: a Review. *Journal of Shellfish Research* 23 (1): 51-61.
89. Nguyen T.L.T., Gheewala S.H., Garivait S. (2007). Full chain energy analysis of fuel ethanol from cassava in Thailand. *Environ. Sci. Technol.* 41 (11): 4135–4142.
90. Owens JW (2001) Water resources in life-cycle impact assessment: considerations in choosing category indicators. *J. Ind. Ecol.* 5 (2):37–54.
91. Papatryphon E., Petit J., Kaushik S., van der Werf H. (2004). Environmental impact assessment of salmonid feeds using Life Cycle Assessment (LCA), *Ambio* 33 (6): 316–323.
92. Pauly D, Christensen V, Guénette S, Pitcher T, Sumaila U, Walters C, Watson R, Zeller D (2002) Towards sustainability in world fisheries. *Nature* 418: 689-695.
93. Pauly D., Christensen V., (1995): Primary production required to sustain global fisheries. *Nature* 374: 255–257.
94. Pelletier, N.L., Ayer, N.W., Kruse, S.A., Flysjo, A., Robillard, G., Scholz, A.J., Ziegler, F., Tyedmers, P.H. and Sonesson, U. (2007). Impact categories for life cycle assessment research of seafood production systems: review and prospectus. *Int J Life Cycle Assess* 12 (6), 414-421.
95. Pelletier N., Tyedmers P., (2008): Life cycle considerations for improving sustainability assessments in seafood awareness campaigns. *Environmental Management* 42: 918-931
96. Pelletier N., Tyedmers P., Sonesson U., Scholz A., Zeigler F., Flysjo A., Kruse S., Cancino B., Silverman H. (2009). Not all salmon are create equal: Life cycle assessment (LCA) of global salmon farming systems. *Environ. Sci. Technol.* 43 (23): 8730-8736.
97. Pelletier, N., Tyedmers, P. (2007). Feeding farmed salmon: Is organic better? *Aquaculture* 272: 399-416.
98. Pelletier, N., Tyedmers, P. (2010). A life cycle assessment of frozen Indonesian tilapia fillets from lake and pond-based production systems. *Journal of Industrial Ecology* 14 (3): 467-481.
99. Pfister S, Koehler A, Hellweg S (2009). Assessing the environmental impact of freshwater consumption in LCA. *Environ. Sci. Technol.* 43 (11): 4098–4104.

100. Phillips, M.J., (2002). Fresh water aquaculture in the Lower Mekong Basin, MRC Technical Paper No. 7, Mekong River Commission, Phnom Penh. 62 pp. ISSN: 1683-1489.
101. Phong L.T. (2010). Dynamics of sustainability in Integrated Agriculture-Aquaculture systems in the Mekong Delta. Ph.D. thesis. Wageningen Institute of Animal Sciences, Wageningen University, Wageningen, the Netherlands.
102. Potting J., Hauschild M., (1997) Predicted Environmental Impact and Expected Occurrence of Actual Environmental Impact. *Int J Life Cycle Assess* 2 (3): 171-177.
103. Rönnbäck P., (1999) The ecological basis for economic value of seafood production supported by mangrove ecosystems. *Ecological Economics* 29: 235-252.
104. Rööös E., Sundberg C., Hansson P-A. (2010). Uncertainties in the carbon footprint of food products: a case study on table potatoes. *Int J Life Cycle Assess* 15: 478-488.
105. Roy P., Nei R., Orikasa T., Xu Q., Okadome H., Nakamura N., Shiina T., (2009) A review of life cycle assessment (LCA) on some food products. *Journal of Food Engineering* 90: 1-10.
106. Ruttan L., Tyedmers P (2007) Skippers, spotters and seiners: Analysis of the “skipper effect” in US menhaden (*Brevoortia* spp.) purse-seine fisheries. *Fisheries Research* 83 (1): 73-80.
107. Sandweiss D.H., Maasch K.A., Chai F., Andrus C.F.T., Reitz E.J., (2004) Geoarchaeological evidence for multidecadal natural climatic variability and ancient Peruvian fisheries. *Quaternary Research* 61: 330–334.
108. Sas, H., Voet, E. van der, Corten, F.G.P., Huele, R., Kleijn, R. (1997). *Extraction of biotic resources: development of a methodology for incorporation in LCAs, with case studies on timber and fish*. VROM (Ministry of Housing, Spatial Planning and the Environment), publicatiereeks produktenbeleid nr. 1997/3. Zoetermeer.
109. Schmidt, J.H. (2010). Comparative life cycle assessment of rapeseed oil and palm oil. *Int J Life Cycle Assess* 15 (2):183–197.
110. Seppälä, J., Knuutila, S., Silvo, K. (2004). Eutrophication of aquatic ecosystems. A new method for calculating the potential contributions of nitrogen and phosphorus. *Int J Life Cycle Assess* 9 (2): 90–100.
111. Seppälä, J., Posch, M., Johansson, M., Hettelingh, J-P. (2006). Country-Dependent Characterisation Factors for Acidification and Terrestrial Eutrophication Based on Accumulated Exceedance as an Impact Category Indicator. *Int J Life Cycle Assess* 11 (6): 403-416.
112. Silvenius. F., Grönroos, J. (2003). Fish farming and the environment. Results of inventory analysis. Mimeograph series of the Finnish Environment Institute 276, Helsinki. Available at: <http://www.ymparisto.fi/download.asp?contentid=12889&lan=fi>
113. Skjelkvåle B.L., Evans C., Larssen T., Hindar A., Raddum G.G. (2003). Recovery from Acidification in European Surface Waters: A View to the Future. *Ambio* 32 (3): 170-175.
114. Sonesson U., Mattsson B., Nybrandt T., Ohlsson T. (2005). Industrial Processing versus Home Cooking: An Environmental Comparison between Three Ways to Prepare a Meal. *Ambio* 34 (4-5): 414-420.
115. Stewart M., Weidema B. (2005). A Consistent Framework for Assessing the Impacts from Resource Use. *Int J Life Cycle Assess* 10 (4): 240-247.
116. Suh, S., Lenzen, M., Treloar, G.J., Hondo, H., Horvath, A., Huppes, G., Jolliet, O., Klann, U., Krewitt, W., Moriguchi, Y., Munksgaard, J., Norris, G. (2004). System boundary selection in life-cycle inventories using hybrid approaches. *Environ. Sci. Technol.* 38 (3): 657-664.

117. Svobodová Z., Lloyd R., Máchová J., Vykusová B. (1993). Water quality and fish health. EIFAC Technical Paper. No. 54. Food and Agriculture Organization of the United Nations, Rome, Italy.
118. Tacon A.G.J., Metian M. (2008). Global overview on the use of fishmeal and fish oil in industrially compounded aquafeeds: Trends and future prospects, *Aquaculture* 285: 146-158.
119. Tacon, A.G.J., Hasan, M.R., Subasinghe, R.P. (2006). Use of fishery resources as feed inputs for aquaculture development: trends and policy implications. FAO Fisheries Circular. No.1018. Rome, FAO, 99p.
120. Tester, M., Langridge, P. (2010). Breeding Technologies to Increase Crop Production. *Science* 327 (5967): 747-906.
121. Thrane, M. (2004). Environmental impacts from Danish fish products – hot spots and environmental policies. PhD thesis, Ålborg University, Ålborg, Denmark.
122. Thrane, M., Nielsen, E., Christensen, P. (2009). Cleaner production in Danish fish processing – experiences status and possible future strategies. *J. Cleaner Prod.* 17 (3): 380–390.
123. Tlustý M. F., Lagueux K., (2009) Isolines as a new tool to assess the energy costs of the production and distribution of multiple sources of seafood. *J. Cleaner Prod.* 17: 408-415.
124. Troell M, Tyedmers P, Kautsky N, Rönnbäck (2004) Aquaculture and Energy Use. *Encyclopedia of energy* 1: 97-108.
125. Troell, M. 2009. Integrated marine and brackishwater aquaculture in tropical regions: research, implementation and prospects. In D. Soto (ed.). A global review of integrated marine aquaculture. FAO Fisheries and Aquaculture Technical Paper. No. 529. Rome. FAO, pp. 47–131.
126. VDI (1997) Cumulative energy demand—terms, definitions, methods of calculation. In: Ingenieure VD. VDI-Richtlinien 4600, Düsseldorf, Germany.
127. Verdegem M.C.J., Bosma R.H., Verreth J.A.J. (2006). Reducing Water Use for Animal Production through Aquaculture. *Water Resources Development* 22 (1): 101–113.
128. Weidema B.P., Lindeijer E., (2001) Physical impacts of land use in product life cycle assessment. Technical University of Denmark.
129. Weidema, B.P., Morteson, B., Nielsen, P., Hauschild, M. (1996). Elements of an Impact Assessment of Wheat Production. Institute for Product Development, Technical University of Denmark, Lyngby.
130. WMO (World Meteorological Organisation) (1999). *Scientific assessment of ozone depletion: 1998*. Global Ozone Research and Monitoring Project - Report no. 44. Geneva.
131. Wu, R.S.S., (1995): The Environmental Impact of Marine Fish Culture: Towards a Sustainable Future. *Marine Pollution Bulletin* 31 (4-12): 159-166.
132. WWF (World Wildlife Fund) (2010). Living Planet Report 2010. <http://assets.panda.org/downloads/lpr2010.pdf>
133. Zamagni, A., P. Buttol, R. Buonamici, P. Masoni, J.B. Guinée, G. Huppes, R. Heijungs, E. van der Voet, T. Ekvall, T. Rydberg, (2009). Blue Paper on Life Cycle Sustainability Analysis. Deliverable 20 of the CALCAS project (<http://www.estis.net/sites/calcas/>).
134. Ziegler F., (2004) Seafloor effects of seafood production, in: Mattsson B., Ziegler F., Environmental Assessment of Seafood Products through LCA, TemaNord 2004:546.

## Annex 1: Terms & abbreviations

CALCAS	Co-ordination Action for innovation in Life-Cycle Analysis for Sustainability
CF	Characterisation Factor
EC	European Commission
EIOA	Environmentally extended Input-Output Analysis
EU	European Union
EZ	(Dutch) Ministry of Economic Affairs
FCR	Feed Conversion Ratio
FP	Framework Programme
GWP	Global Warming Potential
IA	Impact Assessment
ILCD	International Reference Life Cycle Data System
IO	Input-Output
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organization for Standardization
JRC	Joint Research Centre
JRC-IES	Joint Research Centre - Institute for Environment and Sustainability
LCA	Life Cycle Assessment
LCC	Life Cycle Costing
LCIA	Life Cycle Impact Assessment
LCSA	Life Cycle Sustainability Analysis
LNV	(Dutch) Ministry of Agriculture, Nature Management and Fisheries
MSME	Micro, Small and Medium Enterprise
RA	Risk Assessment
SEAT	Sustaining Ethical Aquatic Trade
SETAC	Society for Environmental Toxicology and Chemistry
UK	United Kingdom
UNEP	United Nations Environment Programme
UNEP-SETAC	United Nations Environment Programme - Society for Environmental Toxicology and Chemistry
V&W	(Dutch) Ministry of Transport, Public Works and Water Management
VROM	(Dutch) Ministry of Housing, Spatial Planning and Environment
WP	Work Package
WRI	World Resources Institute