Exploring the Sustainability of Open-Water Marine, Integrated Multi-Trophic Aquaculture, Using Life-Cycle Assessment

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Dedications

I dedicate this thesis to my Mum and Dad, who have, throughout my life, provided me with food, shelter and discipline. It is further dedicated to all other members of my family.

I also dedicate this work to Nasr Al Jardani, who was an extremely good friend.

Abstract

Among efforts to develop sustainable approaches towards the intensive rearing of finfish within open marine waters, is the development of integrated aquaculture techniques. Integrated Multi-Trophic Aquaculture (IMTA), has been promoted as a way to reduce unwanted environmental impacts associated with the intensive production of marine finfish within net-pens. The principle aim of this concept, is the bioremediation of nutrient discharges from fish aquaculture. This is to be achieved by integrating fish cultivation with the growing of species from lower trophic levels, which use the nutrient discharges as a food source. Many studies have been performed that investigate the ability of various species of macroalgae to remove dissolved nutrient discharges, and the ability bivalves to remove solid-bound nutrients, presented as either fish faeces, or an enhanced production of phytoplankton that may be promoted by nutrients emitted by fish-farms. IMTA has also been suggested as a means to improve overall productivity per unit of feed applied to fish, through the conversion of nutrient emissions into additional biomass, such as the tissues of macroalgae or bivalves.

Within the research community which focuses upon the environmental impacts of aquaculture, there is a growing awareness that sustainable solutions to aquaculture production cannot be realised through a focus restricted to the growing-phase, and to a limited set of environmental impacts which may this activity may produce. This is because changes to a specific production phase often promote changes at phases located elsewhere along a products value chain. Life-Cycle Assessment (LCA), is a method employed for modelling the environmental impacts that may potentially be generated across the value chain of a product. It is particularly useful for identifying instances of environmental impact shifting; a term used to describe situations where efforts to reduce the contribution of a specific production phase towards one or more environmental impacts, has the effect of either displacing this contribution to another phase, or increases the contribution of production towards different environmental impacts.

Despite its apparent suitability, LCA has not previously been fully applied to the environmental impact modelling of open-water IMTA systems. The work presented in the following thesis advances this research front, by using LCA to explore the potential for environmental problem shifting occurring as a consequence of replacing intensive monoculture production, with IMTA. Comprehensive datasets have been acquired from the Chilean aquaculture industry, describing the production of aquafeed and

Salmo salar, as well as for the production of the Phaeophytic macroalga, *Macrocystis pyrifera*, and the molluscan bivalve, *Mytilus chilensis*. Using LCA methodology, the production of salmon feed, and the production of *S.salar*, *M.pyrifera* and *M.chilensis*, have been assessed for their capacity to contribute towards a variety of global-scale, environmental impacts. IMTA consisting of either *S.salar* and *M.pyrifera*, *S.salar* and *M.chilensis*, or all three of these species, and combined at ratios required for a bioremediation efficiency of 100 %, 50%, or 20 % of either nitrogen or phosphorous emission from fish, is compared to the monoculture production of *S.salar*. The comparison is based upon a standardised functional unit, with each species produced through IMTA, being modelled as part of the reference flow required to fulfil the functional unit. Monoculture is compared to IMTA upon the basis of nutritional function, by using a functional unit of mass-adjusted protein content, and mass-adjusted economic value. The use of economic value is based upon the 'best-case' assumption, that it serves as a proxy for the total nutritional function that each product offers to human society.

The LCAs presented in this study have produced a number of results. Salmon ingredients derived from agricultural crops and animals account for the majority (between 71 % to 98 %) of contributions towards the impacts of compound salmon feed. In general, agricultural crops ingredients contribute more to these impacts than do agricultural animal ingredients, and account for between 31 % and 87 % of the contributions from all ingredients and inputs. In contrast, the combined supply of fish meal and fish oil from capture fisheries is responsible for between 0.13 % and 11 % of all impacts. Vegetable oil accounts for the vast majority of contributions from ingredients derived from agricultural crops. Vegetable oil is modelled as a 50 : 50 blend of sunflower oil and rapeseed, oil, but sunflower oil accounts not only for most of the contributions from vegetable oil, it is responsible for over 50 % of the contributions that all agricultural crop based ingredients contributes towards some impact categories. Replacing sunflower oil with rapeseed oil reduces the contributions of salmon feed by between 6 % and 24 % across 10 out of the 11 impact categories. When compared upon the basis of equal weight, the contributions of fish oil are between 18 % and 99 % lower than those from rapeseed oil.

The production of feed is responsible for the majority of contributions (between 32 % and 86 %) to all impacts of salmon grow-out production. The production of salmon-smolts accounts for between 3 % and 18 %. The majority (64 %) of contributions towards the eutrophication potential of salmon production are from nutrient emissions, which are the result of fish metabolism, whilst nutrients released through the production of feed, the majority of these being from the agricultural production of crop and animals, account for 32 %. Feed production is also a major contributor to the impacts of land-based smolt production, but these contributions (between 12 % and 37 % across all impact categories) are of a lower magnitude than those from the supply of feed to the grow-out phase. Inputs of salt, and inputs of both electricity produced in a diesel power generator and obtained from the national electricity network, are also notable contributors (between 5 % and 67 %, 4 % and 29 %, and 2 % 47 %, respectively) towards the impacts of smolt-production.

The main contributors towards the potential impacts of kelp grow-out production (excluding eutrophication potential) are the supply of infrastructure (between 14 % and 89 %), operation of a diesel-powered motorboat for maintenance purposes (between 1 % and 89 %), and the supply 'of seeded cartridges' (between 9 and 49 %). The major contributors from the production of 'seeded cartriges' in a land-based facility are the supply of electricity from the national electricity network, the supply of fresh water, and the treatment of waste water. The impact potentials of producing seed in this facility might be reduced if the scale of operation is increased. Removal of nitrogen and phosphorous upon the harvesting of kelp is calculated based upon kelp tissue contents of these nutrients. The harvesting of 200 tonnes ha / yr^{-1}, results in a eutrophic potential with a negative value (-376.51 kg of phosphate equivalents). The removal of such a quantity of nutrients might be beneficial if the local marine environment is at risk of hypernutrification, but when no such problem is present, the potential for undesirable consequences of nutrient sequestration should be considered.

The major contributor towards the impacts of mussels is the provision of infrastructure (between 25 % and 99.5 %, excluding eutrophication potential). Infrastructure is also responsible for the majority of contributions from mussel seed production. The provision of cotton mesh bags, which are used to aid attatchment of seed to drop-ropes in the grow-out phase, account for between 37% and 99 % of the contributions from the infrastructure from the grow-out phase. This result suggest that either the impacts of mussel production can be reduced by using an alternative material with lower environmental impact potentials, or the inventory data describing the producing of cottonmesh bags requires some improvement.

The outcomes of the LCAs of the different IMTA scenarios, are interesting. The results show that choice of species, and the ratios of their combination as required for the different efficiencies of bioremediation, can have a significant effect upon the comparison between IMTA and monoculture. The study demonstrates a potential for environmental problem shifting as being a consequence of IMTA, especially when the functional unit is mass-adjusted economic value. As bioremediation efficiency increases, contributions towards eutrophication decrease. However, this reduction is achieved at the cost of increasing the contributions of IMTA towards those impact categories, such as ‘ozone layer depletion,’ for which it has a greater contribution than does monoculture. In general, it cannot be concluded from these results that open-water IMTA represents a more sustainable alternative to the monoculture production of Atlantic salmon. The sustainability of IMTA is shown to be dependent upon a variety of trade-offs, between individual environmental impacts, and between these impacts and the nutritional function that the system is capable of providing.

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Chapter 1. Introduction

Throughout the past 20 years there has been a body of published research focusing upon integrated multi-trophic aquaculture (IMTA). Much of this work has investigated the application of this concept to intensive aquaculture systems located within marine, open-water environments. A common assertion to be found throughout this literature, is that IMTA offers a more sustainable alternative to intensive monoculture production. This is based mainly upon the premise that in an IMTA system, nutrient emissions from intensively reared species, such as salmon, can be reduced through the cocultivation of species from lower trophic levels, such as bivalves and seaweeds.

As a concept, sustainability has been given various definitions. Perhaps the most common, is that provided by the Brundtland Commission (WCED 1987), which defined sustainability as “development that meets the needs of the present without compromising the ability of future generations to meet their own needs.” Although this definition is seemingly easy to comprehend, the task of determining the sustainability of economic production and services, has proven exceedingly difficult. Much of this difficulty is encountered because products can have multiple possible environmental impacts, and these impacts occur as a result of activities taking place at the various stages of the value chain. Changing a specific activity can influence other activities within the same stage of production, as well as activities in stages positioned elsewhere along the chain. In effect, this means that efforts to characterise the sustainability of a product must consider the entire chain of production.

In some cases, IMTA may result in a measurable reduction of localised environmental impacts associated with intensive, marine aquaculture production. However, the sustainability of IMTA from a value chain perspective has received little attention. Therefore, it appears that much more work needs to be done before we can understand the sustainability of IMTA within an appropriate context. Might reductions in nutrient emissions and their related impacts, achieved through growing multiple crops of different trophic levels, lead to an increase in emissions that contribute towards other types of environmental impact? At what stages of the value chain do these other emissions occur? How can we select a standard unit of production for which these emissions can measured, and against which other forms of aquaculture production can be compared?

This apparent dearth of understanding was the impetus behind the PhD research that I’ve conducted, and now present in this thesis. The opportunity to do so was made possible via a grant obtained through the University of Stirling Horizon scholarship scheme. and the willingness of Prof. Trevor

Telfer and Prof. David Little to provide supervision. The objective of the project was to answer the above questions, or at least, to succeed in making a valuable contribution towards doing so. In essence, this required the development of an environmental impact profile of IMTA and of the value chains upon which it is dependant. There was only one serious contender as a methodology for achieving this task.

Life-cycle assessment (LCA) is a methodological framework for describing the possible contributions of economic activity towards a range of environmental impacts that occur at a global scale. Typically, it assesses the entire value chain of a product (the products life-cycle), rather than one particular phase of production. It is increasingly being applied to the environmental analysis of food production systems. As a methodology, LCA is very complex. Ideally, it should be performed by a team of individuals, who as a collective, possess the range of expertise required for such a comprehensive task. For an LCA to be of good quality, there needs to be an expert understanding of LCA itself, as well as of the product to which it is being applied. When I began this project, my previous educational experience consisted mainly of the study of zoology and ecology at an undergraduate level, followed by a postgraduate (MSc) study of aquaculture with an emphasis upon sustainability. This certainly provided me with a knowledge that is both appropriate and necessary, but it did not include a complete set of skills required for the successful execution of an LCA. Had I fully realised the complex and resource intensive nature of LCA, I might have thought twice about proceeding. However, the amount of learning required to develop expertise in this subject, and the challenges associated with collecting a large and varied dataset, forced me to adapt, teaching me to seek solutions to difficult problems under difficult circumstances, and, ultimately, to have confidence in my own abilities as a researcher. It has also enabled me to view product and system sustainability from a life cycle perspective, a skill which is not ubiquitous even among those who are involved with the foregrounding of sustainability issues.

Dr Alejandro Buschmann, of i~mar Chile (centro de investigation y desarrollo de recursos y ambientes costeros), performs research in the field of aquaculture sustainability and IMTA. He offered me the opportunity to conduct LCA research at the institute, which is located in Puerto Montt, region X, the principle hub of Chilean aquaculture associated activities. Dr Buschmann was also involved in the development of an IMTA system, at the time, consisting of kelp and salmonids, as well as being involved with developing post-harvest utilisation options for cultivated kelp. Over a two-year period, this provided me with a perfect base for collecting the necessary data.

The thesis begins (in chapter 2) by examining the potential of open-water, marine IMTA, to reduce localised impacts associated with intensive fish farming. It proposes that when dealing with food products, sustainability should be understood within the context of food security. It argues, that from a life cycle perspective, the concept of IMTA as being able to provide a more sustainable alternative to intensive monoculture, is not proven at all. In chapter 3, the development of the Chilean salmon farming industry is explored from a socio-economic, and environmental perspective. The next chapter, chapter 4, is split into 2 parts. Part 1 describes the basic methodology of LCA, and it is written to be of use to those new to the subject. Part 2 describes the use of methodology as it relates to the LCAs performed in this study, as well as the methodologies employed for collecting data, and for calculating quantitative and qualitative uncertainty of values derived from the averaging of data from multiple sources. This is followed by five chapters presenting a collection of LCA studies, each with a description of the LCA model structure, its results, and a discussion of their significance. These LCAS describe the production of salmon compound feeds (chapter 5), farmed salmon (chapter 6), mussels (chapter 7), and kelp (chapter 8). The LCA of salmon feed is important as feed production is an essential input to intensive salmon farming. This LCA also covers the production of salmon feed ingredients. The salmon, mussel, and kelp LCAs are necessary and they as they are components of the IMTA system, the LCAs of which are presented in chapter 9. The LCAs in chapter 9 describe a various IMTA system scenarios, each consisting of various combinations of the aforementioned species. This latter chapter also discusses the applicability of different standardised units for which life-cycle impacts can be measured. Chapter 10 presents the final discussion and conclusions of the project. I have made efforts to ensure the thesis is as reader friendly as possible, because I intend the work to be of value to LCA experts and non-experts alike. This means that I have not provided lengthy, detailed, technical descriptions of every process modelled throughout the study. I hope my organisation of the document, combined with my chosen writing style, has enabled me to succeed in delivering a final product that is accessible to those with an interest in aquaculture and sustainable food production.

Chapter 2: Marine, Open-Water, Integrated Multi-Trophic Aquaculture: A Review of Evidence for Proof of Concept, and the Importance of a Value-Chain Perspective for Understanding its Sustainability.

2.1. Introduction

The forecasted growth in world population, combined with changes in consumption behaviour, is expected to increase food demand for most of this century (Godfray et al. 2010). Aquaculture is expected to play a prominent role in meeting the nutritional demands of a global population that is projected to reach approximately 9 billion by 2050 (FAO 2014). In 2012, marine-cultivated finfish represented 12.6 % of global farmed finfish production and 26% of their total value (FAO 2014). Important to this sector is the intensive production of high-value, carnivorous finfish species, reared in floating cages placed within coastal environments. Notably, cage farmed Atlantic salmon (*Salmo salar*) has become a major, internationally traded food commodity, with its production and associated activities having developed into economically important industry in Norway, Chile and Scotland. The similar cultivation of gilthead sea bream (*Sparus aurata*) and sea bass (*Dicentrarchus labrax*) is established in Mediterranean areas, and in Chinese coastal waters the cage rearing of variety of species is often wide-spread.

Notwithstanding the economic success of marine-water cage aquaculture, the prospect of its continued expansion is challenged by potential relationships with issues of environmental concern. Intensive production of carnivorous finfish is dependent upon exogenously sourced feed supplies. The acquisition of fish meal and oil for use as an ingredient in industrially produced compound feed, or as is sometimes the case in Asian nations, the use of whole or chopped 'trash'-fish, is largely dependent on capture fisheries (Tacon and Metian 2008), possibly pressuring wild populations (FAO 2009), and competing with direct human consumption if food grade fish are used (Tacon & Metian 2009). Not insignificantly, feed ingredients are also derived from terrestrial crops and livestock that are associated with their respective environmental impacts. More directly, cage- farming activities are a point source of nutrient emissions with the potential to modify the receiving environments. In water with poor dispersive capacity, localised organic loading of sediments though the settling of feed and faecal particles can lead to changes in benthic fauna species composition. Nutrient enrichment of the water column is possible through releases of dissolved forms of nitrogen (N), phosphorus (P) and carbon (C). Efforts to mitigate and prevent undesirable impacts relating to the provision of feed inputs and the subsequent discharges from fish metabolism have focused on the reduction and replacement of fish meal and oil in compound feeds, and upon changes to diet formulations and feeding methods that improve the conversion of feed to fish biomass and reduce nutrient outputs. Another area of research has explored variations upon an ecological engineering approach, by which the integrated production of distinct aquatic species is based upon shared resources. What is now commonly termed integrated multi-trophic aquaculture (IMTA), can be defined in basic terms as the cultivation of aquatic organisms within proximity to each other, with nutrient emissions from species of a higher trophic level being used as a nutrient input for the growth of species of a lower trophic level. Thus, from an optimistic viewpoint, IMTA methods can potentially increase the amount of cultivated biomass per unit of feed input through the recycling of nutrients, whilst simultaneously achieving bioremediation. The body of research focusing upon marine IMTA has been mainly enthusiastic, and the results from studies of biomass production and bioremediation, as well as economic viability and social acceptability, has been cited as sufficient evidence that IMTA principles represent a more sustainable alternative to conventional monoculture of marine finfish. This paper reviews evidence from the research applied to open-water, marine IMTA within the context of sustainability and food security. It argues that expanding the focus of research beyond its current limitations, to examine the functioning of IMTA within complex product value chains, is key to understanding its sustainability.

2.2. Evidence for achieving bioremediation of nutrients through the integrated cultivation of species from different trophic levels.

The production of high value-finfish in net pens is usually an activity that can be described as an open-water system. Open-water systems are cultivation systems which take place with open-waters, these mainly being waters which are not contained within natural or human made barriers, such as ocean bays, coastal waters, or waters offshore. Some freshwater activities, such as the rearing of fish in cages placed within large lakes, are also sometimes described as being open. Open-water systems differ from closed systems, which are aquaculture activities that take place in waters mostly contained within barriers, examples being the production of fish in ponds, or tanks, such as occurs within recirculation culture systems. In open-water, nutrient discharges exit the cage to be received by the surrounding environment, whereas closed-system usually offer some opportunity the mechanical or manual removal of some nutrients.

As a defining objective of the IMTA concept, bioremediation of nutrient discharges through the integrated cultivation of species must be demonstrable. Within the context of open-water, marine IMTA, the subject of this study, bioremediation of nutrient discharges from intensively[1] reared species has been most commonly investigated using macroalgae and bivalves as nutrient extracting species. To a lesser extent, bioremediation using sea urchins and polychaetes has also been studied.

2.2.1. Cultivating macroalgae for the bioremediation of dissolved organic nutrients

Within the context of bioremediation, a distinction between nutrient uptake rate and efficiency has been defined. Whereas uptake rate is the amount of nutrients removed per unit of time, uptake efficiency quantifies the reduction of nutrient concentration within the water, or the removal of nutrients entering the water. Thus, uptake-rate helps to define the physiological ability of algae to extract nutrients. Measurements of both of these can be standardised to an amount of biomass, such as a specific weight of algae, or standardised to an area of cultivation. Uptake rate is somewhat dependent on the nutrient concentration within the water, the relationship is often considered to be describable by Michaelis-Mentis kinetics (Harrison and Hurd 2001). Uptake rate is important because it is a determining factor in the ability of macroalgae to extract nutrients from its surrounding water, and therefore is also a factor influencing uptake efficiency, but ultimately it is uptake efficiency that serves as the measure of bioremediation performance. Laboratory experiments have been used to investigate uptake efficiency and uptake rate within varying parameters, to indicate the suitability of species of seaweed for bioremediation within IMTA. Some of these studies reported high uptake efficiencies even at high levels of nutrients, and demonstrate that some seaweeds can, to a variable extent, exhibit increased uptake rates in response to increasing levels of nitrogen (e.g. Kang et al. 2008; Carmona et al., 2006). High uptake efficiencies have also been achieved in land-based systems, such as those based in ponds (Neori et al. 2003) and raceways (Robertson-Andersson 2008). Data from laboratory and land-based systems are useful because they provide an understanding of the physiological capacity for the uptake of nutrients by macroalgae in response to specific variables under controlled conditions. However, measures of uptake rate and uptake efficiencies taken under these

[1] Intensive production refers to the cultivation of organisms that depend upon an exogenous supply of feed, often industrially produced, that is not available within the immediate environment in which the species are cultivated.

conditions cannot be accurately extrapolated to IMTA systems operating within uncontrolled, open-water, changeable environments. In addition to factors such as temperature and irradiance, nutrient uptake kinetics are influenced by cultivation depth, age of tissue, history of previous exposure to nutrients, the ratio of nutrients between their various forms, water-flux, and other chemical and physical variables (Lobban and Harrison 1997). Within the environments of open-marine cultivation, the complex interactions between these variables are temporally and spatially dynamic, with potential consequences for how bioremediation might function. In particular, high uptake efficiencies obtained in laboratory experiments and land-based systems are difficult to relate to open-water systems in which dissolved nutrients are not locally contained, but dispersed by water currents. Consequently, the concept of open-water IMTA is a challenged by the need to demonstrate and quantify the bioremediation of nutrients taking place within functional, open-water systems.

There is evidence for the successful bioremediation of nutrients associated with open–water aquaculture production in China. Huo et al. (2012), measured nutrient concentration in water samples taken from the integrated cultivation of *Gracilaria verrucosa* and *Pseudosciaena crocea* (yellow croaker). The experiment was performed in a Chinese bay with poor water exchange, severe eutrophication, and widespread fish production. Nutrient concentrations in the water samples were significantly lower than those taken from a reference area in which *P. crocea* was cultivated alone. By comparing these concentrations, uptake efficiency was calculated to be 57.8 % for phosphorous (as PO4-P), 47.7 % for nitrite (as NO2-N), 60.9 % for ammonium (as NH4-N), and 47.4 % for nitrate (as NO3-N). In this case, uptake efficiency could be measured using in-situ water samples because of persistently elevated nutrient levels in water with low water exchanged. It is unlikely that uptake efficiency can be measured in this way when aquaculture operations are located in environments with good dispersive capacities that avoid extreme nutrient pollution. If analysis of water samples cannot be used to measure bioremediation, other indicators are needed to help determine if seaweed take up nutrients emitted by aquaculture. Tissue nutrient content and growth parameters of seaweed have been used as indicators of bioremediation. Troell et al. (1997) reported that *Gracilaria chilensis* cultivated at a distance of 10 m from salmonids cage-rearing facilities, had a higher tissue content of N and P than cultivations placed at distances of 150 m and 1000 m. Similarly, Sanderson et al. (2012) reported that *Palmaria palmata* and *Saccharina latissima* cultivated adjacent to salmon rearing had higher tissue N content than those cultivated at reference sites. These results suggest that nutrient availability was higher at close proximity to the salmonid rearing sites than further away, resulting in a greater uptake of these nutrients. Examples of higher growth rates exhibited by algae grown close to finfish cultivation are also supportive. Growth rates of *G. chilensis* (Troell et al. 1997; Abreu et al. 2009) *P. palmata* (Sanderson et al. 2012) and *S. latissima* (Sanderson et al. 2012; Wang et al. 2014) were reported to be higher for cultivations placed close to salmon rearing, than those at reference sites. In the above studies, increased growth and higher nutrient content appear to be a response to increased nutrient availability, although their robustness as indicators is somewhat hampered by environmental variables that can influence the outcome of results. In addition, they do not reliably confirm direct uptake of nutrient emissions from fed aquaculture. To provide evidence of direct uptake, there is potential for the use of stable isotopes as markers that can identify the source of nutrients. Analysis of the nitrogen stable isotope $\delta^{15}N$ has been used to identify the uptake of nutrients from fish-farm by seaweed. Elevated ratios found in the tissue of seaweed cultivated close to salmonids have provided some suggestion of direct uptake (Sanderson 2006; Wang et al. 2014) but as yet, analysis of stable nitrogen isotopes has not delivered conclusive evidence of significant direct uptake.

It is not crucial to provide evidence for direct nutrient removal if bioremediation is viewed from a mass balance perspective. Regardless of the source, nutrients are removed when seaweed is harvested. Following this basic approach, bioremediation efficiency could be calculated as the total quantity of nutrient removed upon harvesting (accounting for the nutrient content of the juvenile algae introduced to the system as 'seed'), expressed as a percentage of the total nutrient input to the system from fed-species cultivation. This is expressed by the following equation:

$$\textit{Bioremediation efficiency} = ((NC_{\mathrm{algaeharvested}} - NC_{\mathrm{algaeseed}}) \div DNE_{fedspecies}) \times 100 \qquad \text{Eq.2.1}$$

Where $NC_{\mathrm{algaeharvested}}$ is the nutrient content of the algal tissue multiplied by the quantity of biomass harvested, $NC_{\mathrm{algaeseed}}$ is the nutrient content of algal seed multiplied by the biomass of algae introduced as seed, and $DNE_{fedspecies}$ is the quantity of dissolved nutrients entering the system as emissions from fed species cultivation. The nutrients considered can be dissolved forms of nitrogen, phosphorous or carbon, and the nutrient emissions from fed species may be calculated by including the leaching of dissolved nutrients from particulate material. More complex approaches have been used to model the influence of environmental and seasonal variables upon functions such as the uptake, storage and fixation of nutrients, and algal growth (Broch & Slagstad 2012; Broch et al. 2013; Hadley et al. 2015) In an absence of fully operational systems, modelling can be useful to provide information about species choice and site location in relation to bioremediation, but they are limited by uncertainty derived from model assumptions and the accuracy of input values.

2.2.2. Cultivating filter feeding species for the bioremediation of solid-bound nutrients.

For the bioremediation of solid-bound organic nutrients from fed species cultivation, most studies have focused on bivalves. Evidence that bivalves might be able to utilise particulate organic matter (POM) associated with fish farms is provided by assessments of growth performance based upon measurements such as growth rate and condition index. In several studies, bivalves cultivated in close proximity to fish cultivation displayed higher growth performance than bivalves cultivated further away (Wallace 1980; Jones and Iwama 1990; Sara et al. 2009; Lander et al. 2012). This suggests that fish farming activities may have led to quantitative or qualitative increases in the supply of organic material that bivalves consumed. In some cases, higher levels of POM have been measured in the vicinity of fish farms where improved growth performance was observed (Sara et al. 2009; Jones and Iwama 1990). Mussels (*Mytilus edulis*) placed directly next to salmon cultivation cages were exposed to higher levels of POM than mussels at reference locations and displayed significantly increased feeding activity (MacDonald et al. 2011). Other studies report no improvements in growth performance associated with increasing proximity to fish farming activities (Cheshuk et al. 2003; Navarrete-Mier et al. 2010). Experiments using stable isotopes and fatty acids as markers to help identify the sources of food, provide evidence of fish-farm wastes contributing to the diet of bivalves grown in integrated systems (Gao et al. 2006; Handå et al. 2012; Irisarri et al. 2015; Weldrick and Jelinski 2016).

Bivalves have a varying ability of to effectively capture and utilise organic particles as a food source, which is important to understand because it contributes to the effectiveness of IMTA. In general, bivalve feeding activity may be influenced by qualitative and quantitative aspects of a diets presentation (Ward & Shumway 2004; Cranford et al. 2011). These factors can determine both the proportion of waste POM within the diet, and the proportion of POM released by fed species that can be extracted by bivalve cultivations. For efficient capture, bivalves must be must be exposed to organic matter and be able to intercept and retain the particles. In cultivation environments where there is limited horizontal dispersion of solid waste, cultivated bivalves may receive little exposure to waste particles unless they are grown within the immediate vicinity of fish farm infrastructure. Particles with high settling velocities that are deposited below fish cultivation cages will not be exposed to bivalve feeding activity, limiting the maximum proportion of waste POM that can be captured. There are also limitations to the proportion of suspended solids that can be captured. A basic model developed by Cranford et al. (2013), suggests that capture efficiency can be significantly restricted by the speed at which waste particles pass through bivalve cultivations. Further limitation is imposed by the proportion of particles that, once intercepted, can be retained and digested. Intercepted particles may be subject to selection processes, according to various particle characteristics such as size, with rejected particles being ejected as pseudofaeces (Ward and Shumway 2004). This can create problems for optimising bioremediation as fish of different size classes produce wastes solids with differing characteristics, and bivalves of different sizes accept different sized particles, a situation compounded by environmental influences. There are few studies demonstrating that fish-farm organic wastes are a suitable food source for cultivated bivalves. Ideally, in addition to having properties that are within the appropriate range for efficient capture, the intercepted organic wastes should have a composition that can be easily digested. Mussels (*Mytilus spp*.) placed in experimental chambers with flowing water captured and digested salmon feed fines[2] and faeces with organic content absorption efficiency values comparable to those for mussels fed algal diets (Reid et al. 2010; MacDonald et al. 2011). Importantly, these studies demonstrate the possibility of bivalves utilising fish organic wastes as food source. However, particle selection and absorption are complex and variable processes that are difficult to examine (Ward and Shumway 2004; Cranford et al. 2011), and there is a paucity of information describing the processing and digestion of fish derived POM as part of mixed diet including natural seston presented in conditions of open cultivation. The characteristics of this ambient supply can change seasonally and fluctuate periodically due to events such as intermittent upwelling. If the supply of non-farmed derived, ambient POM is at a threshold level where its capture and digestion by bivalves is optimal, the addition of particulate waste from fish cultivation may have no beneficial influence upon feeding or growth performance. It may be that wastes from fish-farms can only make significant contributions the diet of the cultivated bivalves during conditions of low ambient POM. A lack of winter growth stoppage rings on shells of *M. edulis* grown close to salmonid cultivation suggests that the higher growth exhibited by these mussels than those from populations further away, was related to a supply of organic waste from fish cultivation during winter months when ambient POM was low (Wallace 1980). Solid wastes from fish farms might permit the cultivation of bivalves in areas otherwise unable to support growth, such as waters with waters low nutrients and little phytoplankton production. A model analysis of IMTA in an offshore environment with low ambient POM, found that waste solids from fish might increase the production of Pacific oysters (*Crassostrea gigas*) by up to 20% in comparison to oysters in an offshore monoculture (Ferreira et al. 2012).

When bioremediation is considered within the context of localised benthic impacts the consumption of fish farm solid wastes by bivalves does not by itself demonstrate a reduction in nutrient loading.

[2] Small, dust like particles of feed.

Bivalves filter suspended particulates, a proportion of which will be egested as settleable faecal and pseudofaecal material. Consequently, non-settleable fish-farm particles and ambient seston can be transferred from the water column to the seafloor via bivalve feeding activity, potentially contributing to benthic organic enrichment. Viewed basically, for a reduction in benthic organic loading, the quantity of fish-farm organic matter removed through bivalve cultivation must greater than the total organic content of bivalve faecal deposition. This can be expressed using a 'bioremediation index', presented here as an adaptation of the original equation by Cranford et al. (2013)[3]:

$$BI = A_{OFF} / (F_{OFF} + F_{OS}) + (PS_{OFF} + PS_{OS}) \qquad \text{Eq.2.2}$$

where A_{OFF} is the absorption rate of organic matter in fish faeces and fish-feed, F_{OFF} is the egestion rate of organic matter from undigested fish faeces and fish-feed, F_{OS} is the egestion rate of organic matter from undigested seston, PS_{OFF} is the pseudofaeces production rate of organic matter from rejected fish faeces and fish-feed, and PS_{OS} is the pseudofaeces production rate of organic matter from rejected seston. When the bioremediation index value is greater than 1, it is estimated that there will be a net reduction in benthic organic loading, whereas for index values lower than 1, it is estimated that the feeding activity of bivalves increases organic deposition[4]. This, and similar approaches, can be used to estimate the bioremediation potential of bivalve cultivations feeding on mixed diets with varying proportions of food from different sources. A model based upon absorption efficiencies and faecal production explored the threshold proportion of fish-waste-organic matter in the diet of mussels that must be exceeded for a net reduction in benthic organic loading (Reid et al. 2013). If the dietary proportion threshold cannot be maintained, increasing the number of cultivated bivalves will result in an increased deposition of organic matter. The proportion of fish-farms wastes in the diets is not easy to determine. Mixing models based upon isotopic concentrations of organic matter and bivalve tissue have been used to estimate the relative proportions of different sources of POM in the diet of bivalves grown in IMTA systems (Gao et al. 2006; Weldrick and Jelinski 2016). Weldrick and Jelinski (2016), estimated that the dietary proportion of fish waste in the diet of *Mytilus edulis* was approximately 54 % when grown next to Sablefish (*Anoplopoma fimbria*), although the experiment was not conducted during summer months when phytoplankton may have may have provided a significant source of food. As the supply of ambient POM in marine environments is commonly subject to variation, and the

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[3] Pseudofaeces production was considered insignificant in the conditions modelled by Cranford et al. (2013), and was omitted. Waste fish-feed was also omitted from the calculation. For the purpose of illustration, equation 2.2 is presented here as the original expression presented by Cranford et al. (2013), modified to include the production of pseudofaeces and the absorption and egestion of waste feed.

[4] If all solid wastes from fish farms settle within the cultivation area, the removal of fish organic wastes needs only to be higher than the production of faecal and pseudofaecal organic material from undigested seston (Cranford et al., 2013; Reid et al. 2013).

characteristic of fish-farm solid waste emissions can vary throughout the culture cycle, it may in some cases be challenging to permanently maintain the amount of fish waste in the bivalve diet at levels that exceed the dietary proportion threshold.

2.2.3. Potential role of other Invertebrates for the removal of solid-bound nutrients

2.2.3.1. Polychaetes

Some research has focused on the use of deposit-feeding polychaete worms for the bioremediation of solid material that settles on the seafloor below net-pens. In Japan, following the artificial introduction of a species of *Capitella* to organically enriched sediments beneath fish cultivation pens, significant reductions in organic content of the benthic sub-surface layers have been recorded, with a decomposition rate of organic matter that was estimated to equate, and sometimes exceed, the flux of organic matter to the seafloor (Tsutumi et al. 2005; Kinoshita et al. 2008). This decomposition may be partly due to heterotrophic bacteria which proliferated in the benthic environment created by the feeding activity of the *Capitella* population (Kunhiro et al. 2008; Kunhiro et al. 2011). In these studies, the populations could not be maintained beyond autumn and winter months, with benthic enrichment resuming in the absence of living *Capitella*. These experiments demonstrate temporally restricted reduction of benthic enrichment, and the potential for longer term bioremediation appears limited without harvesting of the *Capitella* biomass, which will decompose following population collapse. Some interest has been expressed in the use of polychaetes that might have economic value, such as the families *Sabellidae* and *Nereidae* (e.g. Barrington et al. 2009) although there has been little documented progress of its use in open-water IMTA.

2.2.3.2. Sea cucumbers

Sea cucumbers are deposit feeders with potential market value, and some species are collected or farmed for food markets. Laboratory trials have shown that some species are capable of feeding directly upon aquaculture derived wastes (Zhou et al. 2006; MacDonald et al. 2013), and analysis of stable carbon and nitrogen isotopes suggest that when placed below net-pens, the sea cucumber *Apostichopus japonicus* appears to digest particulate waste from fish cultivation (Yokoyama 2013; Park et al. 2015). Calculated organic matter absorption efficiencies, in addition to positive growth rates exhibited by some species when placed beneath fish net-pens (Hannah et al. 2013; Yokoyama et al. 2013) and bivalve cultivations (Zhou et al. 2006; Slater and Carton, 2007; Paltzat et al. 2008), support the possibility of using sea cucumbers for the bioremediation of particle wastes that settle to the seafloor. These studies focused on deposit feeding sea cucumbers cultivated in cages that are either placed directly on the seafloor (Slater & Carton 2007; Yu et al. 2014), or suspended below the fish or bivalve cultivation facility (Zhou et al. 2006; Paltzat et al. 2008; Yokoyama 2013). A suspension-feeding species of sea cucumber, *Cucumaria frondosa*, placed for short periods in cages suspended below salmon net-pens, fed on the surrounding particles with an absorption efficiency of organic matter which is comparable to that of *Mytlilus edulis* fed natural diets (Nelson et al. 2012). Sea cucumbers have also been placed directly within fish production nets. The red sea cucumber, *Parastichopus californicus*, reduced the amount of net fouling material when placed inside salmon fry cultivation cages, displaying higher organic matter absorption efficiencies and body wall muscle development than individuals feeding on natural sediments (Ahlgren et al. 1998). Although these results are encouraging, cultivation of sea cucumbers is presented with difficulties. Some species can enter a period of aestivation during which individuals cease feeding and the visceral organs degenerate. In cultivation, aestivation has been associated with weight loss and reduced growth (Paltzat et al. 2008; Yu et al. 2014). In some trials, growth has been reduced at higher stocking densities, possibly as a result of increased competition for food resources between individuals (Slater and Carton 2007; Hannah et al. 2013; Yokoyama 2013). Temporary cessation of feeding, as well as possible limitations to maximum stocking densities may have implications for commercial production and for potential bioremediation. Model analysis of IMTA systems suggest that positive growth and bioremediation is possible over complete production cycles (Ren et al. 2012; Cubillo et al. 2016), but it is yet to be fully demonstrated in practice.

2.2.3.3. Sea urchins

Deposit feeders placed inside bivalve cultivation cages might be capable of reducing fouling organisms. Biofouling can necessitate manual cleaning of culture equipment, and can reduce growth and survival in some cultivations (Adams et al. 2011). Two sea urchin species, *Echinus esculentus* and *Psammechinus miliaris,* placed within pearl nets containing scallops, reduced the amount of biofouling on both nets and scallop shells (Ross et al. 2004). Another sea urchin, *Lytechinus variegatus*, reduced the fouling of nets and shells in an oyster cultivation (Lodeiros and García 2004). Longer-term trials could provide information about the effects of sea urchin grazing on growth and survivorship of bivalves, and help determine if sea urchins produced using this method are of a marketable quality. The potential for sea urchins to feed upon particulate waste from finfish cultivations has also been investigated. In the laboratory, the sea urchin *Strongylocentrotus droebrachiensis* was able to feed upon a diet of sablefish waste, with an organic absorption efficiency comparable to that of individuals fed a more natural diet of kelp (Orr et al. 2014). Short-term experiments suggest that the sea urchin *Paracentrotus lividus* could survive and grow when placed in pearl nets below salmon cages, with fatty acid analysis suggesting that fish derived waste contributed to their diet (Cook and Kelly 2007). However, the production of sea-urchin roe of marketable quality may require a supplementary diet (Cook & Kelly 2007) or a period of conditioning in a controlled environment (Carboni 2013). As with sea-cucumbers, modelling analysis suggests that sea-urchins may contribute to bioremediation in IMTA systems in some circumstances (Lamprianidou et al. 2015), although there is a lack of demonstration of quantifiable bioremediation from full cultivation cycles.

2.3. Practical implementation of IMTA

2.3.1. Spatial considerations

A major challenge to the practical implementation of IMTA is that the addition of extractive species cultivations to monocultures can result in significant areal expansion. The removal of dissolved nutrients using seaweed cultivation requires a potentially large area relative to that occupied by fish cultivation. As the depth and density at which seaweed can be cultivated is limited by sunlight dependant growth, increases in its production must generally be met through expansion of a horizontal, rather than vertical nature. Based upon calculations from published studies, the estimated area of seaweed needed for complete removal of nitrogen released by producing of 1 tonne of salmon per year, ranges from 0.067 ha for *Gracilaria chilensis* (based upon Abreu et al. 2009), 0.1 ha for *Alaria esculenta* (Reid et al. 2013), and 0.129 ha (Reid et al. 2013) to 0.243 ha (extrapolated from Broch and Slagstad 2012) for *Saccharina latissima*. Basic calculations using data from Chilean Atlantic salmon production and a pilot cultivation of *Macrocystis pyrifera* grown close to salmonid culture in Chile, suggest that the area of *M.pyrifera* required to remove all nitrogen from the growth of 1 tonne of salmon per year is approximately 0.04 ha (authors calculations). Clearly, the annual production of salmonids at individual cultivation sites is much greater than 1 tonne. Even with a relatively small production of 1500 tonnes per year, the complete removal of nitrogen could require between 63 to 365 ha of seaweed. Of course, complete bioremediation is not necessarily an objective, with perhaps a more practical objective being the removal of enough nutrients to prevent nutrient related environmental problems (Chopin et al. 2001). However, when based upon the above values, even the removal of an arbitrary, although much reduced, 25% of nitrogen from a salmon farm producing 1500 tonne per year, would require an area of seaweed between approximately 16 to 91 ha. Seaweed cultivations are also large in comparison to fed species culture when measured in terms of biomass weight. Using a mass-balance modelling approach to quantify the amount of seaweed needed to remove the dissolved nitrogen from salmon production, the approximate weight ratio of seaweed to salmon was calculated to be 6.7:1 for *A. esculenta*, 12.9:1 for *S. latissima* (Reid et al. 2013). Based upon these ratios, complete removal of nitrogen from 1500 tonnes of fish would require approximately 10050-19350 wet tonnes of seaweed, with the removal of 25% requiring 2,513-4838 wet-tonnes.

This scale of expansion seems an unlikely occurrence within the regulatory frameworks of regions where the size and location of aquaculture facilities are limited. In countries where open-water IMTA exist, such as Chile (personal observation) and China especially (Ferreira et al. 2008), IMTA has developed incidentally rather than deliberately, a situation facilitated by policy that has stimulated the expansion of aquaculture in the absence of stringent regulation. However, if a nutrient mass balance approach to bioremediation is accepted, then localised expansion of the cultivation area may not be necessary. Bioremediation may be managed at larger scales, with seaweed cultures being fragmented and placed at distances that do not impede the operational activities of fed-species cultivation, whilst avoiding the occupation of large areas of water within one location (Reid et al. 2013). Of course, this approach cannot be adopted for the direct removal of solid waste when bivalves or other invertebrate species must be cultivated in close proximity to the source of emission. Unfortunately, cultivating extractive species in close proximity to fish cultivation will be an impractical arrangement for many cage farming operations. Fish farm sites require space for maintenance, for feeding and operational infrastructure, and for the manoeuvring of watercraft, such well-boats for harvesting. Being in close proximity to fed species may be disruptive to the cultivation of extractive organisms themselves, which must also be easily accessible for maintenance and harvesting (e.g. Slater and Carton, 2007). Overcoming these problems, whilst maintaining a transfer of nutrients, might entail radical changes to site design and construction, increasing the complexity of the production system which needs to be managed. Coordinating the production of several species will also increase system complexity.

Undoubtedly, such an expansion of production must be assessed for adverse environmental effects. There are reports of seaweed cultivations having influenced ecosystem functioning through its affects upon macroflora and macrofauna (Eklöf et al. 2005; Eklöf et al. 2006), and decomposing seaweed biomass can contribute to suspended and sediment organic matter (Ren et al. 2014). Assessments from sediments below a 1 ha cultivation of *M.pyrifera* in Chile but found no consistent increase in sediment organic matter over a period of 3 years, and there was no obvious benthic modification (Buschmann et al. 2014). Suspension feeding invertebrates, such as bivalves, may increase deposition of organic matter to the seafloor when the amount of organic matter egested by the extractive species exceeds that which it removes from the fish-farm wastes.

2.3.2 Economic Feasibility

Adoption of IMTA practices will only occur if they can form economically viable enterprises. Research has explored the circumstances in which IMTA can be a profitable activity. In a basic financial analysis of Scottish salmon and mussel production over a 20-year period, the net present value (NPV) was higher for a hypothetical integrated salmon and mussel system, than the combined NPV of salmon and mussel monocultures, attributable to the higher growth rates assumed for mussels in the integrated system (Whitmarsh et al. 2006). However, the profitability of the IMTA system was sensitive to price changes, with a fall of 2% per annum in salmon prices resulting in negative NPV even when the increase in mussel growth attained through integration was as high as 30 %. In a similar study using biological data from IMTA in Canada, NPV was higher for salmon, mussel and kelp IMTA than it was for salmon monoculture, with the profits from mussels and kelp maintaining profitability when salmon mortality was increased, or salmon prices dropped by 12% (Ridler et al. 2007). A cost benefit analysis in which an economic value was given to nutrient removal in a Chinese bay, an integrated scallop and kelp system was calculated to be more profitable per unit area than scallop or kelp monocultures, owing partly to the higher yield and nutrient removal of the IMTA system (Shi et al. 2013). The implementation of nutrient credit trading programs could act as an incentive for finfish monoculture operations to adopt IMTA practices, by providing a financial benefit to the reduction of nutrient discharges that would otherwise incur a monetary cost (Chopin et al. 2001; Ferreira and Bricker 2016). If aquaculture production takes place within a framework where nutrient emissions are capped, incur a tax, or are subject to a credit trading scheme, this might strengthen the economic performance of IMTA. Under such scenarios, the use of extractive species may allow the producers of fed aquaculture to avoid taxes or might enable producers to expand the production of fish species without exceeding the allowable limits of nutrient emissions.

Continued development of food production systems as sustainable and economically successful enterprises requires that their activities remain socially acceptable. Ethical issues concerning social acceptability, such as environmental sustainability are important credence cues in consumer product choice (Fernqvist and Ekelund 2014). A study of social perceptions of IMTA in Canada used focus groups consisting of members representing different groups of the population (Barrington et al. 2010). After being shown a video describing the Bay of Funday IMTA project, most participants felt that IMTA has the potential to improve the sustainability of aquaculture, and 50% were willing to pay more for environmentally friendly seafood. Seafood certification and ecolabel schemes enable consumers to identify products with perceived enhanced environmental attributes (Pelletier & Tyedmers 2008), and in some cases, are associated with increased willingness to pay (Fonner and

Sylvia 2015). If IMTA can reduce the environmental externalities of aquaculture, it may be possible for this to be recognised in the market place through the use of ecolabel schemes, potentially commanding premium prices, enabled through consumer willingness to pay for preferred attributes. Non-market valuation techniques have been used to estimate that availability of IMTA produced salmon in Canada could increase the aggregate benefit of salmon consumers, through a predicted increase in consumer surplus[5] (Martínez-Espiñeira et al. 2015). It has been proposed that non-use benefits could be experienced by members of public who do not consume salmon, estimated through their willingness to pay, through taxes, for a policy subsidising IMTA that achieves bioremediation (Martínez-Espiñeira et al. 2016).

The production of species such as bivalves and algae requires specialist knowledge and dedicated infrastructure. Producers of fish have business structures focused on the production and marketing of target species, with the production of distinct secondary species requiring access to these additional resources. For the fish production industry to move in this direction there must be significant incentive~~.~~ Market demand is a crucial factor in assessing the likelihood of successful IMTA development and its economic significance. Seaweeds are produced for the production of the hydrocolloids carrageenan, alginate and agar, serving a variety of industrial uses (McHugh et al. 2003; Bixler and Porse 2011). In general, there is some room for growth in hydrocolloid markets. There has been some difficulty in supplying the demand for seaweed in carrageenan markets, and there are possible future limitations to supply for some other hydrocolloid industries (Bixler and Porse 2011; Hurtado et al. 2015). Seaweed produced through IMTA methods may provide opportunities to secure a supply of various seaweed species and improved strains necessary for differentiated hydrocolloid products, especially if supported by research and development. However, seaweed produced for these uses may be geographically limited to environments where suitable species can be affordably cultured to yield required traits, and by proximity to hydrocolloid production. The majority of the world's production of cultivated seaweed takes place in Asia, a significant proportion of which is destined for direct human consumption (FAO 2016). Unlike East Asia, which has a general tradition of seaweed consumption, seaweed has no common presence in western gastronomy. A dramatic shift in consumption behaviour would be required for seaweed to become a prominent feature in western diets. The relative absence of existing markets for seaweed food products in western nations has encouraged innovative development of novel goods. Dried seaweed preparations are now available as snacks and condiments as niche market products (Walsh et al. 2011). There are also new markets for seaweed extracts as plant growth promoters (Bixler and Porse 2011). In Chile, supply of *M.pyrifera*

[5] Consumer surplus being measured as the difference between the price a consumer pays and the price the consumer is willing to pay.

as a feed ingredient to a growing abalone industry is subject to seasonal fluctuations, which may be alleviated through the cultivation of this alga in IMTA systems (Correa et al. 2016). Interest in the use of seaweeds as a feedstock for biofuels has resulted in considerable research activity (Roberts et al. 2015). In trials taking place in Chile, *Macrocystis pyrifera* has been grown close to salmonid farms to assess its suitability as a feedstock for the production of ethanol (Buschmann et al. 2014). However, worldwide there is currently no commercial production of seaweed biofuels at a meaningful scale and there are considerable challenges facing the development of a successful industry (Roberts et al. 2015).

2.4. IMTA, sustainability, and food security within the context of value chains

Population growth and environmental risk present a variety of pressures which challenge global food security (Godfray et al. 2010). Food systems must be capable of adequately supplying the global population with safe and nutritious food necessary for good health, produced as efficiently and sustainably as possible. Within this context, it has been suggested that IMTA can contribute towards satisfying these demands through the provision of food produced in a system with a more efficient use of resources and with lower environmental impacts than intensive fish monoculture (Neori et. 2008; Barrington et al. 2009; Chopin et al. 2012). Understanding the role that IMTA may play in future food provision requires determination of the nutritional functions of food that are needed to supply security. Nutritional insecurity is experienced through inappropriate access to calories, macronutrients such as protein, or micronutrients needed for good health (FAO, IFAD and WFP 2015). Provision of protein or calories alone does not result in food security, with other nutrients being required to fulfil this function. Historically, challenges to food security have been met by focusing upon increasing the supply of food through technologies such as those that increased crop yields (Godfray et al. 2010; Lang and Barling 2012). The present and future challenges to food security are characterised by an emphasis on the need to develop international, system-wide solutions that improve the supply of important food nutritional functions in economies challenged by issues of resource criticality and environmental impacts (Baulcombe et al. 2009). In general, the value of the contribution of IMTA towards achieving food security will be determined by its capacity to offer an efficient return of nutritional functions from the investment of limited resources and the generation of environmental impacts. Furthermore, this value is relative to the capacity of alternative food production systems to efficiently return nutritional function.

To understand the relative value of alternative food products, a standardised unit of comparison must be defined. This is an inherently complex task[6], but for the purpose of this discussion, a very simplified unit of comparison can be taken to be a unit quantity of a complex of the most important nutrients required for food security. An inventory can be developed quantifying emissions to the environment and the resources used for all the activities required to produce the basic unit of comparison. This approach is the basis of life-cycle assessment (LCA) methodologies, such as carbon footprint analysis, which are used to produce quantified environmental profiles of products. In a basic and hypothetical scenario, IMTA produced seaweed might be compared to an alternative food product with a greater nutritional content per unit of mass. It may be that the production of a certain mass of seaweed releases the same quantity of greenhouses gases than does the production of the equivalent mass of the alternative product (Figure 1A). However, when compared using the standardised unit of nutritional content, the quantity of greenhouse gas emissions may be greater for seaweed production than for production of the alternative product (Figure 1b). Stated simply, when considered from the point of view of nutritional function, producing alternative foods which have a greater variety and quantity of essential nutrients per unit mass than seaweed, might result in more efficient use of limited resources whilst contributing less to some environmental problems.

As compared to the above scenario, it is perhaps more difficult to visualise how the sustainability of IMTA might function within the context of full value chains. Food value chains can be characterised by a network of unit activities each fulfilling a function that is necessary for the production, supply and consumption of food items. These unit activities include the acquisition of raw materials needed for energy production, the production of infrastructure and other inputs, agricultural activities, processing of foods and their distribution and retail, as well as consumer product use and disposal of wastes. Food value chains are typically complex and variable, but can be represented basically as belonging to several key activities (Fig. 2). The operations involved in the growing and harvesting of IMTA products constitute only one of a variety of units which make up the value chain. Although assessing the environmental impacts generated by the products up until the point of the farm-gate certainly is necessary, it is valuable to broaden the assessment to include impacts generated by the products at points further along the value chain. For food security, effective provision of nutrients depends upon efficient systems of food processing and preservation, as well as distribution, storage and methods of consumption. In the hypothetical system above, the production of nutritional function

[6] Developing units of comparison for food products is challenging because food fulfils a variety of changeable functions relating not only to nutritional necessity, but values of hedonic utility such as taste and texture.

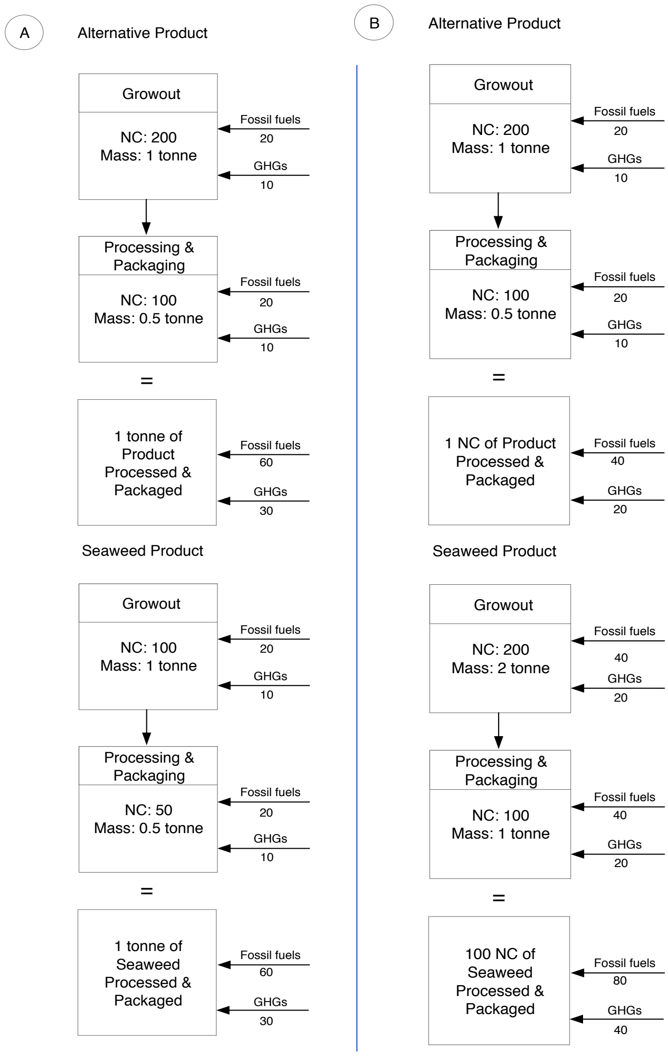


Figure 2.1. Diagram depicting a hypothetical comparison between seaweed and an alternative food product with a greater nutritional content (NC) per unit of mass. The alternative product has the same quantity of greenhouse gas emissions (GHGs) when compared on the basis of mass (A), but when compared upon the basis of nutritional content (B) seaweed has comparatively greater greenhouse gas emissions.

through growing seaweed released a greater quantity of greenhouses gas emissions than did the production of the equivalent quantity of nutritional function of an alternative product. This assessment can now be extended to include other units of the value chain. The quantity of greenhouse gasses emitted by activities associated with production, processing and packaging may be identical for seaweed and the alternative product when compared on the basis of mass. However, when nutritional function is the unit of comparison, greenhouse gases emitted by seaweed processing is

higher because its mass delivers proportionally less nutritional function than that of the alternative product. Conversely, it might also be the case that although the production of seaweed at the aquaculture stage generates more greenhouse gas emissions than does the alternative product, the processing of the alternative product is associated with a much greater production of emissions, and when considered from a value chain perspective, seaweed uses less energy and emits less greenhouse gases overall (Figure 1b) This method of assessing environmental impacts in relation to food security can be extended to all of the major contributing unit activities of the full value chain.

Life cycle assessment can be used to quantify the contributions of the value chain to a variety of environmental impacts. This approach can identify cases of environmental problem shifting, a term used to describe situations where efforts to reduce contributions toward one environmental impact result in increased contributions to another, either from the same unit activity or from another location of the value chain. Although these scenarios are entirely hypothetical, they illustrate the importance of widening the scope of assessment when considering the sustainability of IMTA and its contribution to food security. Although research does show that the function of bioremediation can be attributed to components of IMTA such as seaweed production during the growout phase, this environmental attribute is not by itself sufficient to determine that IMTA is a more sustainable alternative to monoculture production. An IMTA system may have less impacts associated with nutrient release than does a monoculture system, but it is yet to be fully determined if this environmental benefit comes at the cost of increasing contributions towards other environmental impacts relative to those of monoculture when compared using an appropriate unit of comparison. To understand the contribution that IMTA may have towards the development of a sustainable supply of nutritional function, multiple impacts generated by other components activities of the value chain must be assessed.

The concept of increased production per unit feed applied to the system emerges from the improved efficiency of use of nutrient flows through the production of additional crops (Chopin et al. 2012). Viewed from a value chain perspective, the nutrients that enter IMTA systems as feed, originate from the acquisition of fertilisers and their application to the agricultural production of feed ingredients. There is an ongoing discussion regarding the efficiency and sustainability of using agricultural crop and pelagic fisheries production to feed the growth of fish (Carter and Hauler 2000; Boyd et al. 2015). Indeed, LCA approaches show that agriculture could be the biggest contributor to the environmental impacts of farmed salmonid value chains (Pelletier et al. 2009). Producing multiple products from a common nutrient flow is central to the IMTA concept, and if achieved could contribute to improved resource efficiency associated with the production of ingredients for aquaculture feed. Whereas for the achievement of bioremediation it has been argued that the direct uptake of the fed species nutrient waste is not always necessary, it is essential for the purpose of demonstrating the common usage of a nutrient flow by the co-cultured species. In open-water IMTA, true integration based upon a shared nutrient source must result in a measurable increase in quality or quantity of the extractive crop relative to that which can be achieved in monocultures within the same location. Without this, there is no tangible increase in the efficiency of feed use. Fortunately, although not ubiquitous, there is some evidence of improved extractive crop performance. The quantification of improvement can be used as a basis to perform life cycle assessments that will help determine the significance of these improvements within the context of efficient resource use and potential trade-offs with other environmental impacts.

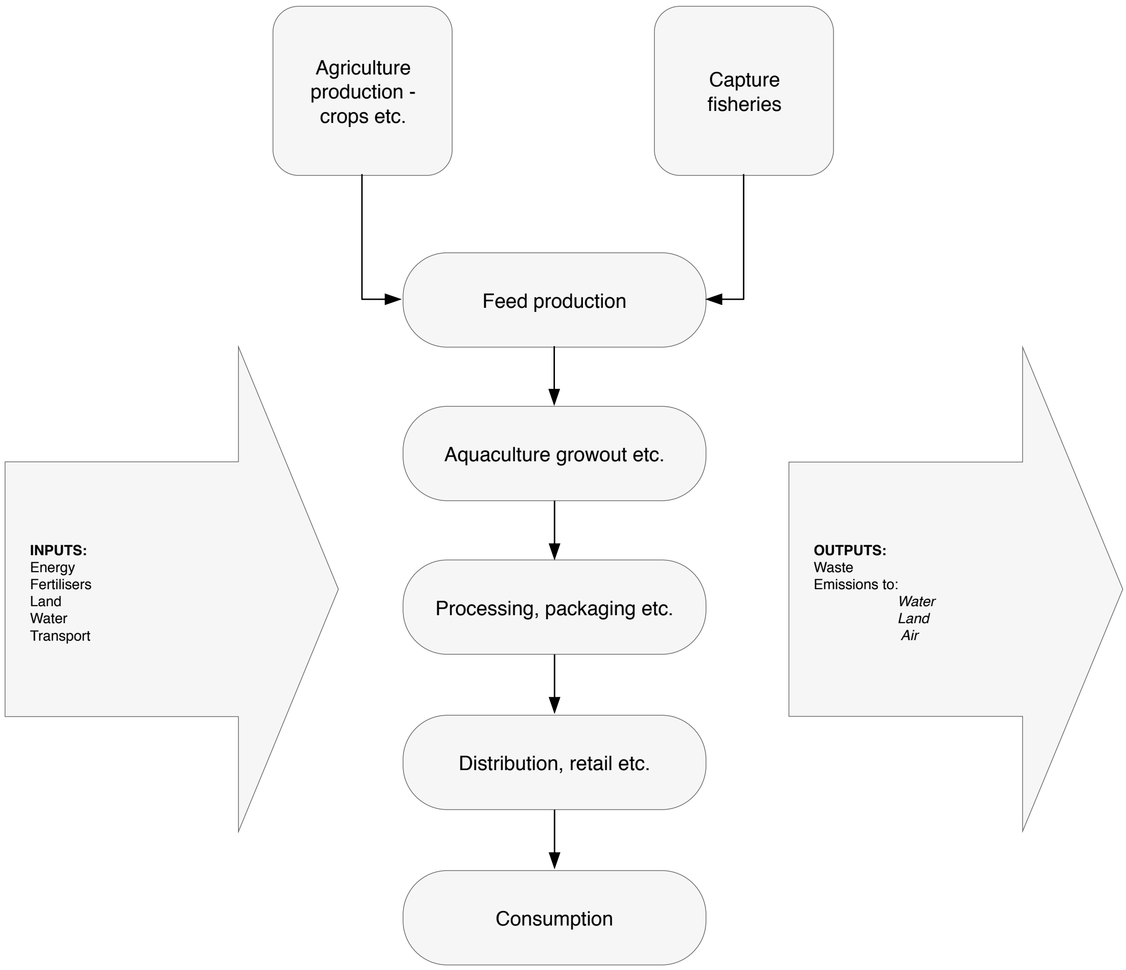


Figure 2.2. A schematic representation of the basic value chain of intensive aquaculture production.

Life cycle assessments do not always produce definite answers, rather they provide a framework for visualising complex scenarios where impacts can occur across multiple temporal and geographical locations. This entails collaborative interaction between industries and organisations to create the large datasets needed for a broad scope of analysis that can provide opportunities to identify trade-offs between different production options and environmental impacts. Through these processes, LCA contributes to the scientific and social foundations needed to deliver solutions to the multiple interacting challenges that define the issues of sustainability and food security. The strategies developed to improve the sustainability of food production will be challenged by conflicting environmental priorities when efforts to reduce one environmental impact can lead to increases in another. Clearly, the issue of greenhouse gas emissions is significantly present throughout the discussion and policy surrounding sustainability, and industrial activity faces global pressure to reduce these emissions. The contribution of food production to climate change has begun to receive particular attention (Garnett 2013). Consequently, any discussion of the importance of a reduction in nutrient releases into coastal environments achieved through adopting IMTA approaches must be balanced against consideration of the global warming potential posed by greenhouse gases that will be released by the activities to which the function of bioremediation is attributed. Of course, viewing sustainability from this widened perspective does not imply that localised impacts should be overlooked. Clearly, nutrient releases into coastal environments is an important issue requiring responsible management that cannot be disregarded by giving precedence to other environmental impacts that may occur at a wider-scale and receive more attention. Management solutions to localised, as well as regional and global scale impacts must coexist with minimal antagonism. However, the drive to develop sustainable solutions for nutrient emission reduction must necessarily consider emissions to other areas of concern such as climate change. The value of bioremediation offered by IMTA will be case specific. In some coastal areas of China with high inputs of nutrients from fish farming, urban economies and agricultural run-off (Li & Daler 2004), the cultivation of extractive species, seaweed in particular, appears to be a necessary component for preventing hypernutrification, and the products of these systems have established market applications. Arguably, in regions elsewhere, appropriate ecosystem management and legislation should be applied to prevent such a situation from developing. Agricultural practices and urban and industrial waste treatment are areas of the economy with potential nutrient management improvements that may reduce emissions to coastal zones (Gren and Limburg 2011). In aquaculture, reduction in nutrient releases and improvements in resource use have focused on efficient feeding practices, improvement to feed digestibility and ingredient use, and the siting of production zones in waters with good dispersive capacities. It is with a variety of options at the scale of watershed management that IMTA must compete. The merits of IMTA as part of an integrated ecosystem management strategy will depend upon the beneficial impacts of its bioremediation service, the demand for its contribution to human nutrition (or demand for other functions), and its relative contribution to environmental impacts such as climate change.

The possible environmental benefits of IMTA are not limited to food provision or bioremediation. Cultivation of extractive species might provide important habitats for a variety of species (Chopin et al. 2012) and so may serve important roles in maintaining local biodiversity. IMTA may also support positive adaptation to climate change. The cultivation of seaweed has been suggested as a method for carbon sequestration and is being investigated for its potential to provide a feedstock for biofuel production (Kerrison et al. 2015). Adopting a wider scope of analysis is particularly useful for this scenario. Regardless of its origin, carbon extracted by seaweed is likely to be released at a stage associated with the seaweeds utilisation as fuel, and so any sequestration within seaweed biomass will be short-term. A portion of carbon released by seaweed during growth might enter carbon pools for periods of time sufficient to provide sequestration (Hughes et al. 2012). This may appear to offer carbon neutral or carbon negative solutions to energy use, but evidence from life cycle assessments show that the economic activities required to produce and process the crop contribute to environmental impacts through the use of non-renewable energy (e.g. Alvarado-Morales. 2013). Without significant technological advances, it is doubtful that seaweed can be used to produce carbon neutral biofuels with a positive energy return on energy invested (Roberts et al. 2015). For reductions in carbon, the calcified shells of bivalve molluscs may offer longer term sequestration than does seaweed (Jiang et al. 2015) although this topic has received little attention. In general, the cultivation of aquatic species has yet to be demonstrated to provide a significant role in tackling anthropogenic carbon emissions.

2.5. Conclusions

Open-water marine IMTA has been promoted as a sustainable approach to aquaculture that can improve economic and environmental performance resulting from the shared use of nutrients, increasing the quantity of biomass produced per unit of feed input whilst simultaneously achieving bioremediation. Numerous studies have been published within the last 20 years, and a combination of evidence from growth trials, bioremediation experiments and economic performance studies have supported the frequent conclusion that IMTA offers a more sustainable alternative to the intensive monoculture of finfish. However, there is limited evidence that improved growth, economic performance and bioremediation occur simultaneously within one integrated system linked by common nutrient flows. Examples of fully functional open-water IMTA are mainly restricted to some Asian systems which do demonstrate bioremediation and the profitable production of crops from different trophic niches, supported by a demand for their products for a variety of economic applications. These large, bay-scale systems are characterised by high quantities of biomass, often in areas with low water exchange and high nutrient levels resulting from fed aquaculture and various other sources. Such circumstances, within which IMTA fulfils demonstrable and crucial roles, are not typical of the conditions within which intensive aquaculture operates and where extractive species cultivation is not imminently vital to prevent system collapse. It is difficult to conceive that these large systems can currently represent a sustainable, or even acceptable approach to aquaculture expansion in western nations.

Outside of these examples, there is evidence of bioremediation by both seaweeds and bivalves. For seaweed, the direct uptake of dissolved nutrients from aquaculture is not necessary to demonstrate successful nutrient reduction. Regardless of the nutrient source, the removal of nutrients through the harvesting of seaweed may balance the input from fed aquaculture. This could be beneficial for aquaculture zones with high nutrient inputs, and may allow the expansion of finfish producers without coincident nutrient enrichment. However, direct nutrient uptake is a necessary precursor for improved productivity per unit of feed. Quantifiable improvements in growth, or of an important qualitative attribute must be evident and must result from the availability of nutrient releases from fed aquaculture. Despite some examples of improved growth or condition, there is a general paucity of evidence demonstrating consistent improvements in extractive species production that can be attributed to nutrient emissions in western IMTA. To clearly demonstrate that IMTA represents a more efficient use of nutrient resources than does monoculture, evidence of consistent improvement over several cultivation cycles is required. These improvements must be sufficient to offset any increased energy or financial input.

Enhanced economic stability through the production of multiple crops is a frequently stated benefit of IMTA. However, there is insufficient evidence to conclude that IMTA offers a more financially stable alternative to monoculture approaches used by high value finfish producers. Finfish producers operate within established markets and economic structures. The available published studies provide basic economic analyses that can only function as preliminary investigations. It is not intended to be critical of this research, preliminary investigations are important and the absence of commercially operating systems makes economic analysis a difficult task. However, the current state of economic research cannot realistically encourage industry to incorporate the production of secondary crops within the established business structures of high value finfish producers. Advancements in this area of research will likely take time and will probably occur alongside IMTA pilot projects. Additional uncertainty surrounding the economic viability of IMTA is presented by a lack of demand for the products of IMTA in western economies. Global population growth and environmental change may provide increased demand for the products of IMTA as food commodities, but a role for IMTA in meeting increased demand for sustainable production of nutrition cannot be assumed. The ability of IMTA to contribute to the demand for sustainable production of nutrition is dependent upon an efficient production of nutrients important for human health, relative to alternative forms of production. This potential needs to be explored under a variety of scenarios and from variety of perspectives.

Although commonly defined as a sustainable approach to open-water marine aquaculture, the sustainability of IMTA has only been investigated within a limited context that does not provide the broader foundation upon which discussions of sustainability must necessarily be based. The environmental consequences of IMTA have mainly focused upon bioremediation occurring at the site of IMTA production. However, the sum of this research does not provide an insight into how bioremediation may be achieved at the expense of increased contributions to other areas of environmental impacts. The sustainability of IMTA must be analysed within the context of the sustainability of the full value chain upon which it depends. The commonly used tool to provide a framework for such analysis is life cycle assessment. These studies are essential to inform discussion of the importance of IMTA within the context of global environmental change and population growth.

In the absence of dialogues informed by lifecycle assessments as well as other methods of environmental impact assessment and strong socio-economic analysis, it is premature to define open-water marine IMTA as being a sustainable method of aquatic production and a more environmentally friendly alternative to monoculture. There remains a need for quantifiable, tangible improvements occurring consistently and simultaneously within one system. Without this, improved economic and environmental performance resulting from the shared use of nutrients within one integrated system, remains an abstract concept. Without this evidence, the common place adoption of IMTA by mainstream finfish producers is a prospect that is neither realistic or desirable. As a consequence, it is currently misleading to award the products of IMTA with labels that are likely to convey an ideal of improved sustainability to the consumer.

The conclusions of this article may seem critical. However, they do represent arguments based upon the perspective life cycle thinking that is a crucial element of sustainability analysis, but which have received little representation within the published literature coverage of IMTA. In defence, there is perhaps not a more practically manifest provision of support for the above arguments than absence of deliberate adoption of IMTA principles by western finfish producers. Regardless of the commonly negative portrayal provided by media, modern producers of marine finfish do value issues of sustainability and corporate social responsibility. The reason why IMTA has not been adopted, is because its multiple, very attractive benefits are not sufficiently demonstrated in practice.

To avoid a pessimistic conclusion, it is perhaps more reasonable to promote an attitude to the possibilities of IMTA that is cautiously optimistic. Much needed technological innovations have often arisen from unexpected sources, and it is important to fund research that aims to provide sustainable solutions to global problems. The principles of bioremediation through the cultivation of nutrient extracting species may deliver important functions within future integrated coastal zone management strategies. If population growth drives the expansion of intensive finfish cultivation, nutrient removal options may be needed. These are situations that must be understood from life cycle perspectives that allow strategies for the reduction of localised emissions and impacts to be considered within the context of global-scale environmental problems. The necessary application of life cycle assessment provides an opportunity to explore sustainable management solutions through analysis based upon the same holistic, ecosystem-based approach that has been a core ideal of the IMTA concept.

Chapter 3: The development of the Chilean Salmon Aquaculture Sector

3.1. Introduction

This review of the Chilean salmonid industry focuses upon the events that have shaped its development, its interaction with society and the environment, and the efforts that have been made to address the many challenges it has come to face. Amongst the leading salmon producing nations, Chile's experience is unique. The contradictory and controversial development of Chile's economic system has perhaps no better reflection than what can be seen though the rise of the country's industrial production of Atlantic salmon, from its beginnings as a private sector and state supported entrepreneurial experiment, to its positioning within the globalised world market as a leading producer of a global super-commodity. It is impossible to understand the industry as it is today in Chile, without considering its evolution within the political and social context which it occurred.

This review is based on upon research published in academic articles and industry and governmental reports. However, it is also largely a product of information gained from my own experience of working alongside the Chilean salmonid industry, and from many conversations and meetings that are too numerous to reference. It is an account of my own interpretation of the industry, and I have written it because I consider this to be an important part of what I learned throughout my research.

3.2. Production and Export

The production of salmonids is a major activity within the Chilean economy, producing a total of 883,102 tonnes in 2015 (Sernapesca 2016). Chile produces and exports more Coho salmon (*Oncorhynchus kisutch*) and rainbow trout (*Oncorhynchus mykiss*) than any other producing nation (154,109 and 107,109 tonnes production in 2015, of Coho and trout respectively). However, with a production of 621,884 tonnes, it is Atlantic salmon (*Salmo salar*) that has by far the greatest share of production, accounting for 68 % of Chilean salmonid exports, compared to and 21 % for Coho salmon and 11 % for trout (Subpesca 2016). Amongst the world's total harvest of farmed Atlantic salmon[7], Chile produces the second greatest quantity, followed by Scotland and Canada, and with Norway producing the most. As with the majority of Chilean produce, the greater part of its salmonid

[7] Produced with the intention of supplying human food chains, rather than for stocking "wild" fisheries.

production is destined for export, with the United States of America and Japan receiving greater quantities of this produce than other importing countries, both in terms of production weight and economic value (Salmonchile 2016).

3.3. The origins of Chilean salmon production

Salmon and trout are not native to Chile, their introduction being in the late 19th and early 20th centuries. Although there was some previous salmon and trout cultivation activities, such as those related to the stocking of rivers and lakes, an aquaculture sector capable of producing salmonids for export as a food product, has its principal origins within the 1970s. After a brief period of socialist governance, a rapid political shift in 1973 and ensuing neo-liberal reforms promoted development of private industry, and opened access to world markets. Partially in response to the country's economic reliance upon copper exportation, the new Chilean government sought to diversify the economy by encouraging specific emergent and expanding industries. Support to the aquaculture sector was provided by the formation of the Subsecretaría de Pesca y Acuicultura (here on referred to as 'Subpesca') in 1976, and the Servicio Nacional de Pesca y Acuicultura (Sernapesca) in 1978. Initiatives by these organisations and efforts by private enterprise, combined with the cooperation of foreign organisations, helped to establish the knowledge and technical capacity required to intensively farm salmonids at industrial scales. The Patagonian fjordic systems of Southern Chile, protected from rough weather and with a cool, temperate climate, provided the ideal natural environment for rearing salmon in large, net-pens, and the ample supply of freshwater in lakes and from rivers, provided conditions required for the production of smolts.

During a period of economic recovery following the severe financial crises of 1982-1983, the Chilean salmonid industry entered a period of expansion with increasing investment activity from the private sector. There was an increase in the number of established businesses (UN 2006), including producers of salmon and fish feed, businesses involved with infrastructure production, providers of transportation, and administrative services. This period of positive growth included the formation of the Association of Salmon and Trout Producers of Chile in 1986, which later became SalmonChile A.G., in 2002. The association created the Institututo Tecnológico de Salmon (INTESAL) in 1995, an organisation intended to help coordinate scientific research and technological developments necessary for the industry's development. The suitability of Chile's natural resources, its openness to foreign investment, a legislature that promoted industrial expansion, and the availability of a workforce in both rural and urban areas, helped encourage the establishment of operations by major salmon producers from abroad. A fall in market price of salmon occurred towards the end of the 1980s and continued into the 1990s, creating a financial pressure which proved too great for the continuation of many smaller businesses. This led to mergers and acquisitions, predominantly through investment by the larger, foreign companies (UN 2006). Some businesses moved towards vertical integration, operating such facilities as those for producing their own feed, eggs, smolts and salmon, or processed products such as fillets. This process of consolidation enabled the industry to develop throughout 1990s and beyond, into a mature and efficient business sector, enabling salmonids produced in Chile to be competitive on the world market.

3.4. Diseases, and the outbreak of Infectious Salmon Anaemia

Throughout the 1990s, and continuing into the 21st century, a variety of diseases have been problematic within the industry. Bacterial kidney disease (BKD), salmonid rickettsial septicaemia (SRS), infectious pancreatic necrosis (IPN) have all emerged within Chile resulting in economic loss. Additionally, sea-lice (*Caligus sp.*), remain a significant problem in the industry. In July 2007, a case of infectious salmon anaemia (ISA) was reported, and the virus spread rapidly causing severe declines in the production of Atlantic salmon, and significant economic loss. Although this was the first reported outbreak of ISA in Chile, it is likely that the disease had been present for some time previously. There had been a number of cases of unexplained mortality, and anecdotal evidence suggests that there was some suspicion of the disease being present, based upon partial laboratory tests (Alvial et al. 2014). Additionally, the virus detected in the first reported case featured mutations not present in some later cases, suggesting the virus has been present long enough to develop into different strains (Alvial et al. 2014).

It is should not be surprising that an outbreak of ISA occurred in Chile, and with such devastating results. Although legislation existed, covering issues such as biosecurity and site location, its suitability and the capacity for its enforcement was outpaced by the industry's rapid expansion. Resultantly, growing facilities were placed within close proximity to one another, and there was little appropriate application of biosecurity measures, facilitating the spread of pathogens. There was also (and sometimes still is), a tendency towards high stocking densities, which can increase stress, making fish more vulnerable to infection. Possible routes of transmission, such as ship ballast water, smolt movements, egg importation, and mortality management, were not subject to appropriate controls (Alvial et al. 2014). At the time, the appropriate technology for adequate detection of the ISA virus was not commonly employed. In other words, producers were often not looking for presence of the disease. All of this meant that the risk of ISA being transmitted from abroad, the risk of its subsequent proliferation throughout the industry, and the ability for it to remain undetected for at least some period of time, where entirely real. Perhaps it may be argued that it is the benefit of hindsight which makes such a conclusion obvious. However, even though the processes of transmission might not be fully understood, previous occurrences of outbreaks in Norway, Canada and Scotland, which resulted in major losses to productivity, should have served as sufficient warning that an eventual outbreak was possible, and would have devastating consequences if it occurred. In general, it was the failure of the industry at large to pre-empt and prevent the outbreak, a failure that was made possible due to state supported, rapid economic growth, combined with a weak framework for regulating the control of disease. However, it would be unfair to suggest that the entire industry behaved with an attitude of irreverence towards risk. INTESAL was clearly aware that disease was a significant issue within the industry, and it coordinated activities towards improving standards of production. These efforts included the creation of an integrated management system, the Sistema Integrado de Gestión (SIGES), which was developed along with collaboration from several industry members. Unfortunately, appreciation of the benefits of obtaining a voluntary SIGES certification was not industry wide, with many producers being focused upon short-term profit. Inevitably, the foresight and great efforts of INTESAL were insufficient to prevent a major epidemic.

The number of reported cases of ISA peaked towards the end of 2008, and the next two years witnessed a dramatic decrease in the production of Atlantic salmon, resulting in a production of 123,233 tonnes in 2010, a reduction of 68 % compared to 2008 (Sernapesca 2009; Sernapesca 2011). The economic loss resulted in an estimated 50 % of workers directly and indirectly employed by the aquaculture industry, becoming unemployed (Alvial et al. 2014). Signs of recovery were evident in 2011, through an increase in production. In 2012, production had returned to levels of previous years unaffected by ISA, and production in subsequent years exceeded levels before the outbreak occurred. However, the outbreak had given rise to serious questioning about the industry's future, and provided fuel to those already opposed to the farming of salmon.

3.5. The evolution of organised concern for issues relating to society and the environment

Throughout the larger part of 1980s, the aquaculture industry proliferated in the absence of restrictions that protected labour and the environment, whilst enjoying state promotion of the sectors growth. Despite the Chilean economy being based largely upon neoliberal policies that favoured free market principles, there was government imposed oppression[8] of those who opposed its policies. This oppression effectively stifled public contribution to discourse, including that relating to the issues of aquaculture expansion. The process of democratisation began effectively at the start of the 1990s, establishing a new phase within Chile where citizens became free to organise alliances aimed at the defence of self-interests and the promotion of regulatory change. This meant that the salmonid production industry, which was accustomed to acting with little opposition from government, media or civil society, was now potentially open to criticism from a variety of sources.

Despite the new arrival of democracy, there was no immediate abandoning of the imperative to economically develop and expand successful industry such as aquaculture. To an extent, the industry still enjoyed significant freedoms. It was clear that the industry's success had its benefits, bringing opportunities for economic development to rural areas which had previously experienced little investment. The establishment of the Region de Los Lagos, and Puerto Montt in particular, as the central hub of the industry's activity, had reversed the previous gradual decline of the region's economy (Barton and Fløysand 2010). There was also little initial response from public, or otherwise nongovernmental organisation. After eighteen years of an authoritarian rule which throttled civil liberties, the organisational capacities of public society were understandably poor. This was particularly true of communities in areas where salmon farms were located, especially the Chiloé archipelago, which has a societal history largely featuring in economic and cultural insularity, previous to its transformation into the site of Chile's first major expansion of salmon producing sea-farms. The attentions of nongovernmental organisation were mainly focused upon industry such as mining and forestry (Barton and Fløysand 2010), perhaps because it was easy to associate these sectors with an obviously visual alteration to natural landscapes, and due their history of boom and bust of a level that had not yet been seen in aquaculture. The principle locations of the salmon industry being in the country's South, it was not geographically placed to be within the vicinity of immediate concerns held by Chile's urban populations living mainly in the region of Santiago and other major cities in the North.

Although the transition to democracy witnessed no initial upsurge in organised concern for the potentially undesirable consequences of salmonid production, there was a gradual introduction of change that brought about new measures for managing environmental and social aspects of the industry's activity. Subsequent governments introduced policies aimed at environmental management and the protection of workers' rights. This included the passing of the fisheries and

[8] The oppression was sometimes brutal. Tens of thousands of individuals were either arrested, tortured, murdered, or otherwise disappeared.

aquaculture legislation in 1991, the Environment Law in 1994, and introduction of an environmental impact assessment system in 1997 (Fuentes and Engler 2016). The creation of INTESAL in 1995 provided the sector with an organised means for coordinating research, including that focused upon disease prevention and the environment. Importantly, INTESAL provided a platform capable of representing acknowledgements from within the sector, that a laissez-faire attitude towards salmonid production and its environmental and social impacts, could not continue.

The latter half the 1990s was a period when salmon production in Chile began to receive increased attention from actors outside the country. As a major producer of Atlantic salmon, the Chilean industry became a recipient of the criticism being directed at the worlds salmon production sector in general, and which featured as part of the emerging international debate surrounding intensive aquaculture production practices. However, the voice of nongovernmental organisations from within Chile did not have a particularly audible contribution to the debate unit the next decade. In 2006, organised protests against labour conditions began with a high-profile strike of workers at Mainstream Chile S.A.,[9] and was followed by the further striking of workers from other salmon production companies. This action was accompanied by the creation of the Labour and Environmental Observatory of Chiloé (OLACH), formed through the collaboration of several NGOs, including Oxfam and the Chilean Fundación Terram, as well as the Central Unitaria de Trabajadores de Chile (CUT), the national federation of worker's unions (Barton and Fløysand 2010). These events are significant in that they demonstrate an emerging capacity for organisation among the lower-income workforce, empowered through new alliances with non-governmental organisations capable of harnessing support from sympathetic audiences, both within Chile and throughout the international arena. Discussion about the practices of the Chilean salmon industry was no longer limited to Chile itself, it was now part of an international conversation among diverse actors. The industry had now to face the consequences of an increasingly obvious reality, that the processes of economic liberalisation and globalisation which led to the industry's success, also entail the globalisation of previously regionalised debate. The presence of NGOS provided a platform for organised concern, publicising perspectives opposed to the industry's practices, and raising awareness of the impacts associated Chilean salmon production (e.g. Pinto et al. 2007, Van Gelderen 2008). However, the ISA outbreak of 2007 is the event that most obviously lends support to claims of industrial malpractice and ineffective regulation. The rapidly spreading disease epidemic came to be viewed by many as evidence of the industry's unsustainability. The outbreak brought critical, international attention to the Chilean industry,

[9] At the time, and still currently, a major player in the Chilean salmon industry, Mainstream now has the name, Cermaq Chile S.A.

included allegations of excessive use of antibiotics (Barrionuevo 2008; Barrionuevo 2009), which were not entirely unfounded. Sensational stories pertaining to food production sell well, and a dramatically reinforced message was presented to the international consumer. The bleak illustration of Chilean salmon farming being drawn by those opposing its practices, was one of an industry that abuses the rights of its working poor, one which has intensified and expanded its production with little respect for nature, and an industry which is riddled with disease, treated with vast amounts of chemicals which not only pollute the environment, but endanger human health.

3.6. ISA: response, recovery, and the rise of a new governance

The first reported case of ISA occurred in July 2007 (Sernapesca 2008a). Generally, the immediate response was not ideal. The capacity for a rapid, well-coordinated, strategic reaction was not possible, owing to a lack of adequate, pre-planned contingency plans, as well as the ineffectually of the current regulatory system. A reported wait of several or more weeks between individual outbreaks being suspected and being confirmed, and further delays until infected stocks were removed, may have facilitated the diseases transition[10] (Mardones et al. 2009). However, despite any inadequacies, as the severity of the crises became clear, organised measures began to evolve. INTESAL played a leading role in the response. Their interaction with government and industry led to the initial, albeit voluntary implementation of measures, aimed at detecting outbreaks, removing infected stocks, and for controlling the spread of ISA. Governmental regulations aimed at controlling the disease came from limited contingency measures enacted by Sernapesca. Following its initial response, INTESAL introduced a more specific set of biosecurity measures to be implemented by the member companies of SalmonChile. Eventually in 2008, more than a year after the first reported outbreak, Sernapesca enacted a biosecurity strategy aimed specifically at monitoring and the controlling ISA virus (Sernapesca 2008b).

The severity of the crises was the driver for change. Regulations pertaining to environmental impacts and the control of disease already existed previous to the ISA outbreak. The General Fisheries and Aquaculture Act of 1989 provided a legal basis for regulatory control of the sector, and through a process of subsequent modification, had gradually evolved throughout the following two decades (Subpesca 2017). However, there was a generally weak capacity for the effective implementation of regulation, at least with regard to biosecurity and environmental impact mitigation (See Fuentes

[10] Mardones, et al. (2009) estimated from a of sample farms, that it took a mean of 9 weeks for the disease to be confirmed after is a suspected outbreak was first reported, and calculated a range from 1, to as much as 51 weeks.

Olmos and Engler 2016, for a review of the regulatory framework). Governance of the aquaculture sector was clearly poor, a situation which became all the more obvious with the spread of ISA. The outbreak provided a shock, catalysing progress towards a new regulatory governance that not only recognises, but enforces the imperative of disease prevention, and the need for an effective management of interactions with the environment upon which the industry depends. This was manifested in the reformation of the General Fisheries and Aquaculture Act, which strengthened and consolidated the legislation required for a more highly regulated industry (Fuentes Olmos and Engler 2016). Previously poorly controlled practices, such as the importation and disinfection of eggs, movements of live-fish, and issues relating to the handling of mortality and its treatment through ensiling, are now all subject to enforced, regulatory controls that are subject to modification when necessary (e.g. Sernapesca 2011; Sernapesca 2015). Importantly, changes have been made that aim to regulate the location fish growing sea-sites. This includes a minimum permitted distance between farms, and the introduction of 'Area Management Zones,' into which growing sites are integrated, allowing the coordination of activities, such as harvesting, mandatory fallowing, the application of chemical therapeutants and other disease management practices. These management zones are intended be located within 'Authorised Areas for Aquaculture' (AAA), that are designated with consideration being paid to environmental and social impacts, as well as issues relating to disease prevention (Alvial 2015)[11]. Changes have also been made to the process of granting licenses for new growing sites. Licenses should only be granted for areas within an AAA, and may be accepted dependent upon preliminary environmental impact assessments that evaluate benthic conditions, as well as potential impacts upon the natural environment and local culture, and an assessment of issues relating to biosecurity must also be performed. Licences for sea based growing sites, previously granted for an unlimited amount of time, are now restricted to a period of 25 years. The granting of licences to estuary based growing facilities has been restricted, and there have been no further licences granted to smolt production facilities located within freshwater lakes.

These developments in law and regulation were essential components in the recovery from ISA. But also essential, has been the role of the financial and banking sector. Akin to improved and enforced regulation, the action taken by the banking and finance sector has both enabled recovery and helped enforce the maintenance of responsible practice. At the peak of the crisis, billons of USD were owed to banks by the salmon farming industry (Alvial et al. 2014). As a response to the collapsing production, banks had the option of initiating bankruptcy proceedings with insolvent companies. Considering the

[11] Zoning had previously been introduced, but it the legal basis for their enforcement was subject to challenge (Fuentes Olmos and Engler 2016).

importance of salmon production to the Chilean economy, and the feasible prospect of its recovery, the banks decided to provide continued support to the industry, but upon new terms. Loans were renegotiated, extensions and other provisions where generally given, but there was no forgiveness of debt, and collateral increased. Importantly, the lenders required that indebted companies used practices which complied with the new regulations as well as those recommended by INTESAL.

3.7. The emerging sustainable Chilean salmon

The government, the financial sector, and organisations within the industry itself, have all had an important part to play in the rise of the new governance of Chilean Atlantic salmon production. From an optimistic point of view, their efforts appear to have moved the industry towards a greater economic, environmental and societal sustainably. However, this emerging sustainability has yet several hurdles to face.

The Industry is now largely recovered, and once more there has been a focus upon expansion. As an emerging consumer of Chilean farmed salmon, Brazil opens the prospect for growth of a new market place (SalmonChile 2013). If favourable market conditions can be maintained (at time of writing various restrictions on global supply are forcing increased prices for salmon) expansion certainly seems a possibility when considering the suitability of the fjords and channels of Chile's coastal South. It is likely that many new sites will be located in areas further south of the region of Los Lagos, where the bulk of salmonid farming takes place. Aysen is a region where salmon farming already takes place, and it will likely be the location of many new sites if the industry continues to expand. For salmon farmers, the journey South is sometimes seen as an opportunity to reduce the likelihood of contact with vectors of infectious disease (anonymous industry members in Aysen, personal communication 2013). For local communities, an intensified presence of aquaculture will bring development. This will likely be challenged by opposition from stakeholders, who may argue that development results in the industrialisation of rural areas, bringing unnecessary and unwanted societal change, and that the benefits of previous developments have not been adequately conferred to the rural poor (e.g. Fløysand and Barton 2014). Such arguments find support in the experiences of the Chiloé archipelago, one of Chile's major salmon producing regions (Box 1.) In Aysen, as well as other regions, any future development of rural areas must be a participatory process based upon meaningful communication with stakeholders, with the aim of maintaining relationships that accommodate the views and concerns of affected communities. Regulations pertaining to the granting of new aquaculture licences, as well as for the construction of infrastructures on-land, do require an assessment of impacts upon local communities. However, these assessments, which are usually conducted privately, with a report being submitted to Sernapesca for its consideration, do appear to be somewhat both basic and brief (authors personal observation). The accuracy and thoroughness of these assessments depends upon the expertise of those responsible for their production, as well as upon the standard of work that the authorities demand. The outcome depends upon those responsible for making decisions based upon the reports. In other words, the required process of assessment, reporting, and decisions making, does not by itself guarantee that societal impacts are adequately represented when new licenses are granted. For this reason, the applicability and effectiveness of this procedure is subject to regular review. However, it is important that impact assessments do not come to be an action of little functional purpose other than to satisfy an administrative requirement.

Box 1. Social impacts of the salmon industry on the Chiloé archipelago.
As a culturally sensitive area of historical importance, the islands of Chiloé provide the most frequently invoked depiction of a rural society to which the industry has brought dramatic change. Until the introduction of industrialised aquaculture, the highly ruralised population had existed in relative isolation since the influencing events of European colonisation. It was a distinct society, characterised by a dependence upon subsistence farming, an economy where exchange of goods was often with without money being transferred, and with males being seasonally itinerant workers, providing occasions when women commanded roles usually held by men (Leon 2015; Ramírez and Ruben 2015). The establishing of Chiloé as a centre of industrially produced salmon destined for a world market, was a process that paid little consideration to cultural sensitivity. The role of the local population, which was one of significance within their own isolated economy, is being changed to that of a workforce, which provides labour and other services to the production of a good for an international market (Daughters, 2016). Such general disregard for sustainable development risks the industry being perceived as one which operates to the tune of its own profit, and at a cost to the local community (e.g. Claude et al. 2000). It was protests against working conditions in Chiloé that led to industry’s first major labour strike, and attracted the attentions of international NGOs. Feelings of antipathy towards salmon farming in Chiloé have not been prevented by reforms following the ISA outbreak. As recently as 2016, conflict erupted and the islands of Chiloé became a site of popular unrest, with local communities blaming the salmon industry for an algal bloom which threatened the local fishing industry.

The regulation of environmental impacts has also been improved. However, effectively, many of the changes related to environmental aspects of salmon production are based upon measures implemented to prevent disease, rather than a dedicated focus upon impacts to the environment itself. A good example of this is the implementation of zoning, which is largely the result of the need for disease management, but which could also facilitate improved performance in terms of environmental impacts. If zoning successfully prevents high concentrations of cage farming activities being placed within areas of unsuitable hydrographic characteristics, then this, along with the introduction of mandatory fallowing, should go some way to preventing nutrient enrichment of water bodies and benthic sediments. The issuing of new licences for aquaculture production requires consideration of environmental aspects, including assessments of the benthic sediments. However, as with the assessing of social impacts, it is important that their application remains to be relevant, and that regulations are monitored and modified, supported by an appropriate knowledge of science. Clearly, monitoring and maintaining the effectiveness and applicability of societal and environmental regulation is a governmental responsibility. But this should not detract from one very important point; that it is the responsibility of the industry to inform and support any necessary regulatory change, as part of the commitment to social sustainability, which is expected by the international consumer. Indeed, much of the efforts to improve environmental performance, as well as other aspects of sustainability, have come from the within the industry itself. Clean Production Agreements (APL's) are a national, government funded initiative, intended to improve the sustainability of various Chilean industries, but are developed with significant participation of the private sector. INTESAL and SalmonChile have provided an important contribution towards establishing APL's for the salmon industry (Consejo Nacional de Producción de Limpia 2002; Consejo Nacional de Producción de Limpia 2010). As with governmental regulation of the industry, the APL's coverage of environmental aspects of salmon production focuses largely upon disease prevention. Despite this, the APL's do appear to offer a route through which the environmental impacts of production can be subject to further management. The research activities of INTESAL may provide opportunities to add modifications the APL's, which increase the demand for environmental responsibility among the members of SalmonChile. Industry led efforts to demonstrate a genuine commitment towards improved sustainability include achieving internationally recognised certifications, awarded to industry that complies with a standard of 'good-practice.' Significantly, INTESAL has worked towards aligning the practices of its member companies with those of various standards. This has included harmonizing the practices of several salmon producers with those of the internationally recognised standard GLOBAL G.A.P., as well as the standards for Best Aquaculture Practice (BAP), developed by the Global Aquaculture Alliance (SalmonChile 2017) In general, these ensure that that certified products comply with set standards for animal welfare, hygiene, food safety, and environmental and societal sustainability, throughout production and processing.

3.8. Conclusions and Opinions

The development of Chilean salmon farming is a story of success and rapid growth, an accomplishment made possible by the availability of highly suitable natural resources, combined with a unique economic and political set of circumstances. However, because of these same set of circumstances, it is an industry which has struggled with itself, and so it is also a story of short-sightedness and malpractice. Without being interpreted within an appropriate context, this may seem an emphatically critical statement. Though, when considering the industry's development in its wider political and social context, it becomes rational to argue that the persistence of fallacious behaviour became established as an inevitable result of a military dictatorship that encouraged economic development whilst simultaneously oppressed the civil freedoms necessary to help drive a responsible, and sustainably functioning industry. The industry has had to come to terms with a newly arrived democracy and the societal changes this entailed, as well as the responsibilities that international market success requires. In effect, within a short space of time, an industry unaccustomed to opposition, came to be confronted with an increasingly intense criticism from a variety of actors, on both a national and international platform. Considering the challenges faced, the industry's ability to maintain a leading position among the world's producers of salmon, is an impressive display of adaptation.

The industry's survival is, quite possibly, owing to the existence of SalmonChile and INTESAL, the presence of foreign companies and foreign investment, and the willingness of the financial sector to support domestic growth. SalmonChile and INTESAL help to promote responsibility and self-regulation within the industry, and foreign companies bring experience, as well as the financial capacity to respond to change. The financial sector offered conditional support to Chilean owned companies when the industry was at the point of collapse. However, although not reflective of the industry at large, an unfortunate attitude of defiance towards regulatory compliance, and a distrust of governmental departments, can still sometimes be observed among smaller producers (authors personal observations). Although this is unsurprising within the context of political history, it is not good for the continued survival of the industry. The progressive attitude exhibited by SalmonChile and INTESAL, must be reflected by all producing companies. Smaller, regional producers, must compete with the international experience and expertise afforded by larger, foreign companies. Smaller producers may also struggle to absorb the financial cost of tighter regulation, and of the improved practices associated with certified standards, such as lower stocking densities. In the long-run, it may be that these smaller producers are outcompeted by larger companies, both in terms of quality of management and ability to finance improvements in performance. Whether or not this is a desirable outcome can be, and is, debated. But from a pragmatic point of view, it may be that the continued adaptive capacity of the Chile Salmon industry as part of an international market, requires that the industry is composed of international actors, in the form of large companies with operations not necessarily being exclusive to Chile. The Chilean industry cannot be sustained if it functions as part of a resource periphery where the standards of core countries do not apply. Farmed Atlantic salmon from Chile is a product that must compete internationally, not only in terms of taste and texture, but in terms of embedded values, such as sustainability and acceptable practice. Without any doubt, this requires a visible, progressive industry-wide attitude towards issues of sustainability, and the demonstration of corporate, social and environmental responsibility through a mature commitment to self-regulation.

Chapter 4: Methodology

4.1. Introduction

This chapter is separated into two main parts, although there is much crossover between the themes of each. Part 1 describes the basic principles and methodology of life cycle assessment. Much of the literature that covers methodological aspects of Life Cycle Assessment (LCA), is sufficiently technical to make it somewhat inaccessible to those new to the field. To a great extent, as with most academic literature, this is unavoidable, but it is a problem in the expanding field of LCA, which has experienced a growth in popularity that is attracting participants and stakeholders with little or no experience of what LCA entails, nor of the results that it can be reasonably expected to provide. For many, the process of becoming acquainted with LCA methodology is a frustrating experience. More participants within the area of sustainable aquaculture research are becoming aware of the presence of LCA, and would likely benefit from an understanding of it principles. But when asked to provide introductory information, I find it difficult to point towards a suitable primer of the subject that provides accurate, yet easy to understand information. I have written part 1 with this in mind, and I hope that aquaculture researchers with an interest in learning about LCA will find this to be a useful introductory text, that will lay a foundation of basic understanding necessary to meaningfully interpret LCA research. It has been written so that it can be read within one sitting, whilst also serving as an accessible collection of definitions to be referred to if necessary.

Part 2 serves as a description of the methodology as it has been applied to the current study. It describes only the major aspects of the methodology used in this study. It does not provide an exhaustive account of each methodological issue that has been encountered, or each decision that has been taken, as this would be an impossible task within the context of this thesis as it is intended to be presented.

4.2. Part 1

4.2.1. A brief explanation of Life Cycle Assessment (LCA)

In general, Life Cycle Assessment is a methodology for quantifying contributions to a variety of environmental impacts made by an economic product or service. Typically, the contributions generated by the life-cycle of the product are accounted for. This life-cycle includes the activities related to the extraction and processing of required raw materials and energy, the production of the product, its use by the consumer, and the final disposal or reuse of the product or its components. This view of the product life-cycle is often referred to as being 'cradle to tomb.' However, assessments of partial life-cycles are also common. In the case of agriculture, a partial life-cycle may include activities relating to the extraction and generation of materials and energy used for the production of fertilisers and other necessary inputs, as well as the growing of crops, with the assessment ending at their harvest. Agricultural LCA's of this type are sometimes referred to as 'cradle to farm gate' assessments. In LCA, the various activities in the life cycle of a product are separated into individual processes that have inputs and outputs, and that are linked by flows of energy or materials. These 'unit processes' can be aggregated into larger process that represent stages of the product life cycle. The processes and stages included in the assessment are those required to fulfil the function of the product or service, which is defined by a 'functional unit.' This functional unit may be defined as a simple quantity of a product (e.g. 1 kg of potatoes) although the act of defining functional units is often far from simple. Contributions to environmental impacts can be viewed at the level of individual processes, as well as for each stage of the life-cycle, and as a total from all the stages of the life cycle that is required to produce the functional unit. This allows the environmental impacts to be compared between different process or stages to identify 'hotspots,' and to identify areas were reductions in impacts can be achieved, perhaps though the use of alternative procedures or technologies. In some circumstances, the profile of environmental impacts for the production of a functional unit can be compared to those of an alternative functional unit. As an example, the impact profile for 1kg of a particular brand of potatoes may be compared to the profile of a competing brand of potatoes.

The environmental impacts included in the assessment can vary depending upon the goal of the study and the methodology used to model the impacts. Most commonly, the impacts are modelled as occurring at global scales. This appears intuitive for impact *categories* such as global warming, but impacts more usually considered within regionalised contexts, such as eutrophication, are also frequently modelled as occurring at a global scale. Some efforts have been made to develop models for local scale impacts (e.g. Ford et al. 2012), but although interesting, this approach is not without its challenges. It is also important to consider that impact categories are modelled as being potential impacts, rather than absolute. This is because contributions (e.g. CO_2 emissions) to impacts are not *apportioned between* the different impacts towards which they may contribute, rather their *full quantity* is 'double counted' as contributing to *each impact separately*. The impact category 'global warming potential' describes a potential because a portion of the CO_2 emissions that are quantified as contributing towards it, may, in reality, enter another ecological compartment and contribute towards an environmental impact other than global warming.

4.2.2. The basics of Life Cycle Assessment Methodology

This section builds upon the information presented in the previous section. Whilst avoiding the level of detail found in the major texts explaining LCA methodology, the following section provides an overview of the technical aspects of LCA methodology that should provide the reader with sufficient knowledge for the understanding of the remainder of this work.

Life Cycle Assessment methodology has been standardised by the International Standards Organisation. The details of this methodology are described briefly in ISO 14040 (ISO:14040 2006) and with more detail in ISO 14044 (ISO:14044 2006). The procedure for completing an LCA is defined by these standards as consisting of four separate phases. The procedure is rightly described as an iterative process, because it is repeated several times. This repetition is necessary because the four separate phases are somewhat interdependent, each potentially having consequences for any previous phase, and for those to follow. The basics of each of these four phases, '**Goal and Scope Definition**,' '**Inventory Analysis**,' **Impact Assessment**,' and '**Interpretation**,' are described below.

4.2.3. Goal and Scope Definition

Upon initiation of an LCA, the first task is to define the goal of study. According to ISO 14044, the goal definition should state, with as little ambiguity as possible, the intended application of the study and its intended audience, identifying stakeholders involved with the project, and organisations or groups responsible for initiating the project, as well as those who will produce the LCA. Following this definition, the scope of the study should be defined by detailing choices regarding fundamental aspects of the methodology, which must be consistent with the study goal. This is necessary as it will influence how data is collected, and how the results will be calculated and communicated. The technical information and methodological choices that should be detailed within the scope are described below, along with explanations and definitions when appropriate.

4.2.3.1. Functional Unit

Functional units are important to understand because they are crucial components of our ability to understand the sustainability of products and product systems. The functional unit provides a reference unit for which the inputs and outputs (and therefore impacts) are quantified. This unit quantifies the ‘function(s)’ being delivered by the system. Of relevance to, and often confused with the functional unit, is the **reference flow**, which quantifies the outputs of the system that delivers the functional unit. To explain this further, a system that produces 60 W lightbulbs can be considered. The function of a light bulb is to deliver light, and so the functional unit may be defined as the delivery of 850 lumens (lm) of light for 1000 hours. If one 60 W light bulb delivers 1000 hours of 850 lm, then the reference flow could be one 60 W light bulb. Different products / references flows that deliver the function of 1000 hours of 850 lm, can be compared using LCA, as long as the same methodology is applied to the LCA of each product. Although some studies have attempted to do so, comparing products that deliver different functions, for example a comparison of lightbulbs with television sets, is contrary to rules of ISO 14044, and due to the complexities and challenges of LCA methodology, will likely produce meaningless results unless very large levels of uncertainty are considered acceptable. Defining the function of food products is particularly challenging because food products are invariably multifunctional. Food items are characterised by complex nutritional, as well as hedonistic functions, making it difficult to compare food items that initially may appear to be functionally equivalent. The issue of product multifunctionality is an important theme throughout this work.

4.2.3.2. Systems Boundaries.

This **system boundary** describes the boundary between the economic processes being studied and other economic process or the natural environment. **Economic processes** are those which involve human activity. As an example, the boundary between the **economic system** and the **environmental system** might be the mining of iron ore. The production of iron ore deposits is a natural process and is part of the environmental system, but the removal of these deposits removal requires human intervention and is therefore considered in terms of LCA methodology, as being an economic process within the economic system. The general details of these boundaries, as well as temporal and geographical boundaries, must be described in the scope.

4.2.3.3. Allocation

An understanding of product function is crucial for an understanding of product sustainability. Similarly, an understanding of how ‘**co-products**’ are defined will also be advantageous. Economic processes are commonly multifunctional. Viewed basically, **allocation** describes how the inputs and outputs of a process are allocated between its co-products. This will in turn, influence the impacts associated with each co-product. Issues of allocation have been, and still are, very much debated, because of their influence upon the final results of an LCA. Products may appear to be more or less sustainable depending upon the allocation procedure used within the study. The material differences between different products can mean that entirely different allocation issues must be dealt with, which is one reasons why comparisons between products with different functions are contentious. In general, the method for performing allocation must be consistent throughout the study. ISO 14044 states that issues of allocation must be dealt with using the following stepwise procedure:

1. Whenever it is possible, allocation must be avoided by dividing the process into sub-process with individual products, or to expand the system being studied to include the additional functions of each co-product (in other words, do an LCA of all the co-products, with their respective functions all being combined into one functional unit).

2. When this cannot be done, allocation should be performed using underlying physical relationships between the co-products.

3. When physical relationships cannot be used, allocation should be performed using other relationships between the products, with such a relationship being, for example, economic value of the co-products.

In practice, this procedure can be difficult to follow. It is often not possible to physically divide multifunctional processes into sub-processes without resorting to allocation, and expanding the system to include the functions of all co-products can significantly change the goal of the study, whilst being complicated and prohibitively resource intensive. The precedence given to allocation based upon underlying physical relationships is not universally accepted, with arguments having been offered that support preferences for allocation based upon economic value (e.g. Pelletier and Tydemers 2011; Weinzettel 2012). It is important to recognise that allocation decisions may, perhaps sometimes unavoidably, be reflective of subjective values. Thusly, the results of LCAs may contain an element of bias, occurring as a consequence of attitudes held by individuals or organizations.

4.2.3.4. Impact categories

Impact categories represent areas of environmental concern to which the inputs and outputs of the modelled system have the potential to contribute. Examples of possible impact categories are 'climate change,' 'freshwater eutrophication' and 'ocean acidification.' During the scope definition, the impact categories that will be assessed must be identified, along with a description of the chosen methodology for their calculation.

A range of other aspects should be detailed within the scope. These include descriptions of how the data will be collected, the type, quality and quantity of data needed, situations where assumptions and value based decisions will be made, and the limitations of the study.

4.2.4. Life Cycle Inventory (LCI[12]) Analysis

During the inventory analysis phase, detailed flow diagrams are made of the system that is to be studied. Using this method of visualising the system, the boundary of product system (with other economic, as well as natural systems) is defined in detail. The flow diagrams detail individual unit processes, and their flows between them.

The data for each unit process is then collected. In reality, data collection is usually the most time and resource intensive activity of an LCA. Forethought combined with careful planning is essential for successfully collecting the necessary data, which is greatly facilitated when performed by individuals with detailed knowledge of the systems being analysed. Data is usually broadly defined into two main categories. **Foreground data** is used to describe data required to model the major processes being assessed, referred to as **foreground processes**. In detailed LCAs, efforts are usually made to ensure foreground data is primary data collected from the actual economic processes being modelled, relative to the appropriate temporal and geographic location. Literature data is frequently used when such primary data is unavailable. **Background data** is used to describe data required to model **background processes,** which refers to those process which supply goods or services to the foreground processes. This is often sourced from databases that supply process data for life cycle assessments, such as the ecoinvent V.3. database (Wernet et al. 2013). Literature data is often used when process data is not otherwise available, and data from different but related processes may be used to represent processes when no data is available at all.

The ecoinvent database deserves particular mention, because it is the most widely used database in LCA. Ecoinvent provides life cycle inventory data for thousands of products, and it is typically used as a main source of background data. Originally intended for use in the assessment of Swiss economic production, it quickly came to be used as an inventory of European products, and now provides data with a more international coverage. Although not without issues, the ecoinvent process models are intended to be structured upon a consistent use of methodology. Users can also access the database

[14 & 15] Initials commonly used within LCA literature, but which can be confusing when placed within jargon heavy text. Presented here for informative purposes only, and will not be used throughout the following work.

in a rawer format, which enables the LCA practitioner to alter process models according to their own methodological choices. This is particularly useful when the methodology used across the ecoinvent database differs from that being used for a particular project. For a more comprehensive overview of version 3 of the database (the current version at the time of writing), see Wernet et al. (2016).

The output of the life cycle inventory analysis phase is a detailed model of the product life cycle being assessed, along with an inventory of what are referred to as '**elementary flows**'. These elementary flows represent the inputs and outputs that have the potential to contribute to environmental impacts. In other words, the inventory of elementary flows is an inventory of emissions *to* the environment (e.g. gaseous emissions to air, or nutrient emissions to water) and extractions *from* the environment (e.g. the mining of fossil fuels). More precisely, and importantly, they are to be understood as material or energy flows that are extracted from the environment and which have had no prior human (viz. economic) transformation, or material and energy flows entering the environment without any subsequent human transformation.

4.2.5. Life Cycle Impact Assessment (LCIA[15])

Environmental impacts are classified into **impact categories** that represent areas of environmental concern, such as climate change, or fossil fuel depletion. During the impact assessment phase, the inventory of elementary flows, compiled during the inventory analysis, are qualitatively assigned to each impact category and their quantitative contribution to each impact category is then calculated. Respectfully, these two steps are referred to as '**classification**' and '**characterisation.**' Classification can be easily understood as the process of assigning each elementary flow (input from, or output to nature) to the impact categories towards which they may contribute. During characterisation, the quantity of each elementary flow that might contribute to an impact category, is multiplied by a **characterisation factor** that converts it into the common quantitative unit of the **category indicator** of each impact category. The category indicator is a quantitative representation of the impact category. For the impact category 'climate change' the category indicator could be time-integrated radiative forcing as represented by global warming potential 100 (GWP100). Each elementary flow has its own characterisation factor that converts it to the common unit of the category indicator. Contributions to GWP100, are expressed as carbon dioxide equivalents (CO_2eq), and the relevant elementary flows are converted to CO_2eq by their respective characterisation factors.

It is important to recognise that if an elementary flow has the potential to contribute towards more than one impact category, it is counted as contributing its full amount to each one individually. So, if a particular substance has the potential to contribute towards both climate change and ozone depletion, 100% of the substance will be counted as contributing towards climate change, and 100% will also be counted as contributing towards ozone layer depletion. For this reason, the output of the impact assessment describes *potential impacts*, rather than definitive results.

ISO 14044 describes three optional steps which may proceed characterisation. **Normalisation** is a step where the impact category results are normalised to a reference value for each particular impact. The normalisation step can facilitate the communication of impact results by providing a context that allows the intended audience to visualise the magnitude of the potential impacts relative to those of an appropriate reference. Commonly, impacts are normalised to the average yearly impacts of an individual citizen of a particular region, but of course, it could also be possible to normalise the results with those of an alternative product. **Grouping** is a step whereby the impact categories are assigned to into groups, either through sorting by nominal characteristics, or through ranking. Impacts may be sorted according to the spatial scale at which they occur (e.g. local, regional and global), or perhaps according to the nature of their origin (e.g. sorting of impacts into those arising through emissions and those arising through resource extraction). The ordinal or hierarchical ranking of impacts (e.g. grouping of impacts into ranks of low, medium or high priority) is performed based upon value-based choices. Consequently, ranking presents results that are reflective of individual or group ideals. A common form of grouping is the sorting of impacts into categories of 'damage,' such as damage to human health, damage to ecosystems and damage in the form of resource depletion. However, as with ranking, these groups might also be seen to be separated based upon subjective division, at least in part. Methods of calculating damage may also make use of **weighting**. Weighting ascribes numerical 'weighting' factors to the category indicators (that often have been normalised), and by which the category indicators are multiplied. Weighting factors are unavoidably based upon subjectively held values. They might be used to aggregate impacts into a single score, with a single score being calculated for each life-cycle stage which can then be compared. Single scores might also be used to compare one product with another, or may be used to set reduction targets for efforts to reduce the overall impacts of a particular product. The weighting factors may be arrived at through various methods, such as through stakeholder panel discussions. Weighting factors are sometimes used internally by organisations, but, as appropriately stated by ISO 14044, they cannot be used to assert differences in performance with competing products when these assertions are to be communicated externally.

4.2.6. Interpretation

Interpretation is the phase in which the results of the impact assessment phase are interpreted and described within an appropriate context. ISO 14044 places the activities of the interpretation phase into three main steps, to be implemented reiteratively. All of these steps need to be performed within all complete LCAs, but with varying extent depending upon the scale of the project. Some of these steps may seem pedantic, and do involve some tedious work, but they are included not only as essential components of the interpretation of impacts results, they help to add efficiency to the organisation of the LCA project.

Firstly, is the **identification of significant issues** relating to the previous phases, and that need to be considered when interpreting the results. Paying due consideration to these issues, the second step is to perform an **evaluation** of the ***completeness***, ***sensitivity*** and ***consistency*** of the work. As noted by Guinèe (2002), to perform these evaluations in the order of completeness, sensitivity and consistency, is to an extent, illogical. I personally do not believe that the implication they must be performed in this specific order is deliberate, and I think it is both acceptable and recommendable to depart from the protocol of ISO 14044, by proceeding with the evaluations in the order of: consistency, completeness, and sensitivity. The consistency evaluation is a check to ensure that the methods, any assumptions, and the sources and type of data are consistent with the study goal and scope. As these should have been previously determined within the goal and scope phase itself, this step may seem pedantic, but it is necessary because the, quite usually, large amount of data being managed and the number of individual decisions that need to be made, can easily lead to error. Once the study is confirmed as being consistent with the goal and scope, the completeness evaluation checks that all necessary data is included, and that all necessary processes have been modelled without error. The difference between the consistency check and the completeness check is probably ambiguous to those without previous experience to LCA. Basically explained, it can be said that once the consistency evaluation checks that the correct processes have been modelled, the completeness evaluation checks detailed technical aspects of these process (e.g. that the input of material in a process is equivalent to that which processes or wasted, or in other words, the input and outflows add up). If the scope of the study so requires, the completeness check may be performed by a variety of LCA and technical experts. There seems to be little point in performing detailed, resource intensive technical checks, if the processes that have been modelled do not conform with the goal and scope (e.g. the wrong processes have been modelled), and so the completeness evaluation should be performed first. The final evaluation checks the sensitivity of the impact assessment results to changes within the data, assumptions, or methodological choices. A common check is to assess the sensitivity of the results to changes in the choice of co-product allocation factor (e.g. how the results differ when economic value is used as the basic of allocation, as compared to when product mass the basis of allocation). The sensitivity check is always valuable, but it is absolutely crucial when the results will be used to assert differences between one product or another, or otherwise used to inform public decision making. Related to sensitivity analysis is **uncertainty analysis**, whereby the uncertainty ranges for some data types are used to calculate the error ranges of the results. Part 2 of this chapter contains describes a method for calculating the uncertainty of the inventory data that is used to model the processes, and the contribution of this process inventory data to the impact assessment results. Another valuable evaluation is a contribution analysis, which assesses the proportional contribution of all flows and process to the different impact categories, thereby revealing which process should be predominantly focused upon (because their respective contributions are greatest) for uncertainty or further error checks. Contribution analysis is also a useful tool for presenting the results of the impact assessment, as it provides a method to describe impacts for individual processes within the product life-cycle. It is logical to perform the sensitivity checks following the consistency and completeness checks, so to avoid devoting resources analysing irrelevant, incorrect, or incomplete processes.

The final main step of the interpretation phase is the establishment of **conclusions and recommendations**. This step includes a description of the final model and its results. During the first iteration of the LCA procedure, preliminary conclusions will be drawn, which are then explored further during the subsequent iteration. Additional to the drawing of conclusions is the making of recommendations (e.g. recommended product choices) if it is required as part of the study goal, but otherwise, it is likely obvious that the conclusions will be discussed within the context of their limitations and applicability.

4.3. PART 2

4.3.1. Goal definition

The goal of this study is to investigate the life cycle impacts associated with open-water, marine, integrated multi-trophic aquaculture (IMTA) systems. This will be done by using ISO 144044 (2006) compliant methodology. It will compare the impacts of IMTA production with those of monocultures, but it does not intend to provide comparative assertions to be disclosed to the public. It will focus upon the production of Atlantic salmon (*Salmo salar*), the Chilean blue mussel (*Mytilus chilensis*), and the giant kelp (*Macrocystis pyrifera*), within monocultures and IMTA. The aquaculture systems under study are all in Chile. The results are intended to be presented both within this thesis, as well as publication within appropriate scientific journals. The results should be of relevance to anyone with an interest in the sustainability of aquaculture production, and of particular relevance to those with an interest in IMTA or Chilean aquaculture production.

4.3.2. Scope

4.3.2.1. Functional unit

A variety of functional units will be analysed for the assessment of IMTA systems. More details are provided in Chapter 9, and section 4.3.2.7. of this chapter. For assessment of feed, salmon production, kelp production and mussel production, the functional unit is described as 1 kg of product mass (wet weight).

4.3.2.2. System temporal boundary

The temporal boundary was set as a three-year period, from 2010 to 2013, and data was collected for the full extent of this period when possible. When it was not possible to collect data for the full extent of this period, efforts were made to collect data from within this period. Ideally, 5 years would have been used as a length of time, being appropriate to avoid collecting data reflective of a temporary fluctuation. However, this was deemed inappropriate due the outbreak of infectious salmon anaemia (ISA) that occurred in 2007, that resulted in a major reduction in the production of Chilean farmed Atlantic salmon, causing severe disturbance to the operation of the salmon aquaculture sector.

4.3.2.3. System geographical boundary

The geographical boundary was set as the country in which the specific processes occurred. This meant setting a boundary that was potentially global in outlook, but that in effect, focused mainly upon North and South America. All the aquaculture systems are in Chile, and the agriculture systems are in Chile, Argentina, and the USA.

4.3.2.4. System technological boundary and boundary between studied system and other economic systems

Some boundaries are based upon co-product allocation. For example, the salmon grow-out system produces both salmon and ensiled salmon mortality. The silage was originally considered as a co-product, with its share of the grow-out processes to be attributed based upon allocation. Other boundaries not related to co-product allocation were usually more obvious as being process involved with other economic systems. It is important to note, as with many LCA studies, there is some inconsistent inclusion of infrastructure processes, although this should not have significant influence upon the results. Background processes are mainly from the ecoinvent V.3 database (Wernet et al. 2016), in which there is some inconsistency relating to infrastructure processes.

4.3.2.5. Co-product allocation

Co-product allocation has been based upon economic value (Guinée et al. 2004). The method used to calculate the allocation of process inputs and outputs between co-products is demonstrated in the following example, where it is assumed that allocation must be performed between two co-products, *'co-product a'* and *'co-product b:'*

Eq.4.1.

$$Economic\ value\ of\ coproductx = economic\ value\ per\ quantity\ of\ coproductx\ X\ coproductx\ quantity$$

Eq.4.2

$$Allocation\ factor\ coproducta(or\ b) = \frac{economic\ value\ coproducta(or\ b)}{economic\ value\ coproducta(or\ b) + economic\ value\ coproduct(or\ b)}$$

4.3.2.6. Impact categories

Throughout all assessments, the CML baseline impact assessment method has been used, unless specifically stated otherwise. This impact assessment includes impact categories that are recommended and described in Guinée (2002). These impact categories are shown in Table 4.1., alongside their respective type of elementary flow, category indicator (the effect by the impact category is characterised), and the common unit of equivalence by which the potential of each elementary flow to contribute to a particular indicator, is standardised. Further information can be found at https://www.universiteitleiden.nl/en/research/research-output/science/cml-ia-characterisation-factors, and at https://www.pre-sustainability.com/download/DatabaseManualMethods.pdf. The latter source provides a basic, but useful explanation of this method.

Table 4.1. Impact categories of the CML baseline method, developed by the Leiden University Institute of Environmental sciences (CML). It shows the elementary flows, indicator, and indicator result unit for each impact category. For example, elementary flows of any nitrogenous and phosphoric nutrient to air, soil and water, contribute to the impact category eutrophication. Nitrogenous phosphoric flows are converted to the common indicator unit of phosphate equivalents, measured as kilograms. For the impact category ozone layer depletion, the elementary flows contributing to this impact are a number of gasses which result in the breakdown of ozone. The potential for each of these gasses to result in this effect is standardised to the common unit of kilograms of CFC-11 equivalents.

Impact category	Elementary flows	Indicator	Indicator result unit
Abiotic depletion (elements)	Elements	Depletion of ultimate reserves	kg antimony eq.
Abiotic depletion (fossil fuels)	Fossil fuels	Depletion of ultimate reserves	MJ
Global warming potential 100	Greenhouse gasses to air	Infrared radiative forcing (W/m_2)	kg CO_2 eq.
Ozone layer depletion	Gasses	Ozone breakdown	kg CFC-11 eq.
Human toxicity	To air, soil, water	Acceptable daily intake/predicted daily intake	kg 1,4 dichlorobenzene eq.
Freshwater ecotoxicity	To air, soil, water	Predicted environmental conc./predicted no-effect conc.	kg 1,4 dichlorobenzene eq.
Marine ecotoxicology	To air, soil, water	Predicted environmental conc./predicted no-effect conc.	kg 1,4 dichlorobenzene eq.
Terestrial ecotoxicology	To air, soil, water	Predicted environmental conc./predicted no-effect conc.	kg 1,4 dichlorobenzene eq.
Photochemical ozone creation	VOCs etc to air	Tropospheric ozone formation	kg ethylene eq.
Acidfication	To air	Deposition/acidification critical load	kg SO_2 eq.
Eutrophication	To air, soil, water	Nutrification potential of N and P	kg PO_4 eq.

4.3.2.7. Construction of IMTA scenario models

To explore the environmental impact profile of IMTA, a variety of scenarios have been developed, which are to be analysed using LCA. Assessing a variety of scenarios is worthwhile, because the specific species of the co-cultured crops and their respective production quantity, are obviously not uniform across marine, open-water IMTA systems. The assessed scenarios feature salmon production, integrated with a cultivation of nutrient extracting biomass, this biomass being either mussels or kelp, or both. For each scenario, the relative production quantity of each co-product is adjusted to three different ratios. Each ratio is determined by its ability to achieve a particular bioremediation efficiency, these efficiencies being measured as the proportion of either N or P emissions removed by the nutrient extracting crop. Further descriptions of these scenarios are to be found in chapter 9, section 9.3.2, including Table 9.1 and Table 9.2. A nutrient mass balance model, upon which the calculation of bioremediation efficiency is based, is described in Chapter 9, section 9.2.2. The inventory data describing the co-cultivated species, these being salmon, kelp and mussels, are described in chapter 6, chapter 7, and chapter 8, respectfully. The IMTA scenarios can be modelled within Simapro software, by combining the inventory of the respective co-products, according to the calculated ratios. As an example, consider an IMTA system consisting of salmon, kelp and mussels, combined at a weight ratio of 1:2:3. The data inventories which each separately describe the production of either 1 kg of salmon, 1 kg of kelp, or 1 kg mussels, are the economic inputs which complete the inventory data describing IMTA, and they are included at a rate of 1 kg of salmon, 2 kg of kelp, and 3 kg of mussels.

Combined at these weights, this represents the 6 kg of biomass produced in the IMTA system. If 1 only kg of biomass produced in the IMTA system is to be assessed, the weight ratio of the co-products is maintained, although their inclusion rate is now 0.167 kg, 0.33 kg, and 0.5 kg, for salmon, kelp, and mussels, respectively.

The potential impacts of the above scenarios are assessed using three functional units, these being product mass, product mass-adjusted protein content, and product mass-adjusted economic value. Further details can be found in Chapter 9, section 9.3.3., and their respective appropriateness and their influences upon the outcome of results, are discussed throughout this same chapter.

4.3.3. Data collection.

4.3.3.1. Data collection for foreground processes

Data for foreground processes was collected as primary data directly from industry contacts. When this was not possible, either primary data from similar processes was used, or data was obtained from literature sources, depending upon the availability and suitability of each. Sometimes, data was obtained from the ecoinvent database, or the Agri-footprint database (Vellinga et al. 2013) and was usually modified to better represent the process being modelled. As processes usually involve a variety inputs and outputs, some foreground processes were modelled using both data from primary sources and data from literature or databases. Data from databases or literature ranged in 'quality,' which was determined using the protocol described in section 4.3.5. When appropriate, collected data were used as inputs to models with outputs that serve as inputs to foreground processes.

4.3.3.2. Data collection for background processes

Data for background processes came mostly from the ecoinvent V.3. database (Wernet et al. 2016) On the many occasions when no suitable process was available within this database, an attempt was made to create a process, either entirely from literature data, or else by modifying an existing process using literature data.

4.3.3.3. Methodology for collecting data from industry (foreground processes).

Data collection from industry was the most time and resource intensive phase of the LCA. The majority of data was collected from Chile, between 2012-2014, a period during which I resided in the city of Puerto Montt, Chile. Puerto Montt was a suitable location, not only because it is the site of the i-mar

Centro de Investigacion y Dessarollo de Recursos de Ambientes Costeros, but also due to its position as being the site of the majority of administrative activities, processing centres, and service providers for the Chilean aquaculture industry. Possessing only a primitive knowledge of the Spanish language, and with permanent access to a translator being impractical, engaging with the Chilean industry for the purpose of data collection required that I became as proficient as possible in the speaking and writing of Spanish. Email and telephone communication, presentations, meetings, and documents such as surveys, were, for the most part, all conducted in Spanish.

Prior to collection, a variety of industry members and organisations in the appropriate countries, were identified as possible providers. These where contacted through email or telephone. In some cases, a presentation was delivered to industry member organisations, to attract interest. There were some positive responses, although in general, the response was poor. The main strategy of securing provisions of data was based upon personal introductions made to specific members of industry by existing contacts. This strategy was particularly important in Chile, where a mistrust of researchers and members of external organisations in general, was clearly apparent. This was commonly compounded by a lack of awareness of the advantages to industry attained through engaging with environmental research, which is often perceived as threatening rather than potentially useful. For this reason, the majority of effort was applied to obtaining data from the major industrial organisations, where such mistrust was generally absent. Following introductions and discussions, this data was usually provided based upon a personal agreement to provide an LCA in return. Collection of these data where usually preceded by a visit to the relevant facility, were as much information as possible was gathered through interviews and observation. Visits were payed to grow-out sites, hatchery and smolting facilities, feed production factories, and product processing facilities. After producing flow diagrams representative of the appropriate processes, the relevant data was acquired. In some cases, access was granted to company databases, which provided an appropriate format for data visualisation and collection. Access was also sometimes granted to company documentation, such as official environmental impact assessments. In other cases, surveys were created as a format for data collection, each being specifically tailored to the relevant organisation and the processes under study. The construction of surveys was done with care. Surveys that are too demanding of the respondent easily result in a failure to collect data. Furthermore, special attention was paid to the wording of the document, as ambiguous sentences elicit responses of varying relevance. As the majority of these surveys were written in the Chilean dialect of the Spanish language, efforts were made to have the documents proof read for accuracy before they were disseminated. It was also important that the survey was visually minimalistic to avoid demotivating the respondent. In general, the surveys were designed to require as little effort as possible on the part of the individual(s) responsible for their completion, whilst being capable of eliciting the precise information necessary and in the correct format. An example of a data collection survey is shown in Figure 4.1.

4.3.4. Information about data collection for specific foreground process

Energía

Insumos energéticos	*En estas columnas queremos conocer el consumo de energía y, si es posible, asignar los insumos de energía entre los diferentes usos. Esto es importante para entender las emisiones al ambiente porque los distintos combustibles y los motores tienen diferentes emisiones. Es posible que no todas las columnas sean aplicables.*								*Fuente de los datos*		
Tipo de energía	*Tipo (ej, gasolina 93/ bunker n° 6)*	*Consumo total en el último ciclo completo*	*Unidad*	*Unidad alternativa*	*Utilizada para barcos*	*Motores estacionarios*	*Ensilaje*	*Otros motores estacionarios*	*Datos exactos (derivados directamente de archivos)*	*Datos indirectos (basados en un cálculo)*	*Datos estimados*
Gasolina			l		%	%	%	%			
Petróleo			l		%	%	%	%			
Gas natural licuado			l		%	%	%	%			
Otro gas			l		%	%	%	%			
Aceite			l		%	%	%	%			
Aceite			l		%	%	%	%			
Electricidad de red			kWh		%	%	%	%			

Figure 4 1. Example of a data collection table. This table is typical of those used within this study for the collection of data describing energy use on a fish farm. It is designed to allow the correspondent to enter a variety of data in a chosen format of their convenience, whilst remaining accessible for use within LCA. It provides a brief description of the required data and the reason for its importance. It prompts the correspondent to indicate the source of data, permitting a choice of either estimated data, data derived from calculations, and exact data from records or measurements. This latter information is used within the calculation of qualitative uncertainty

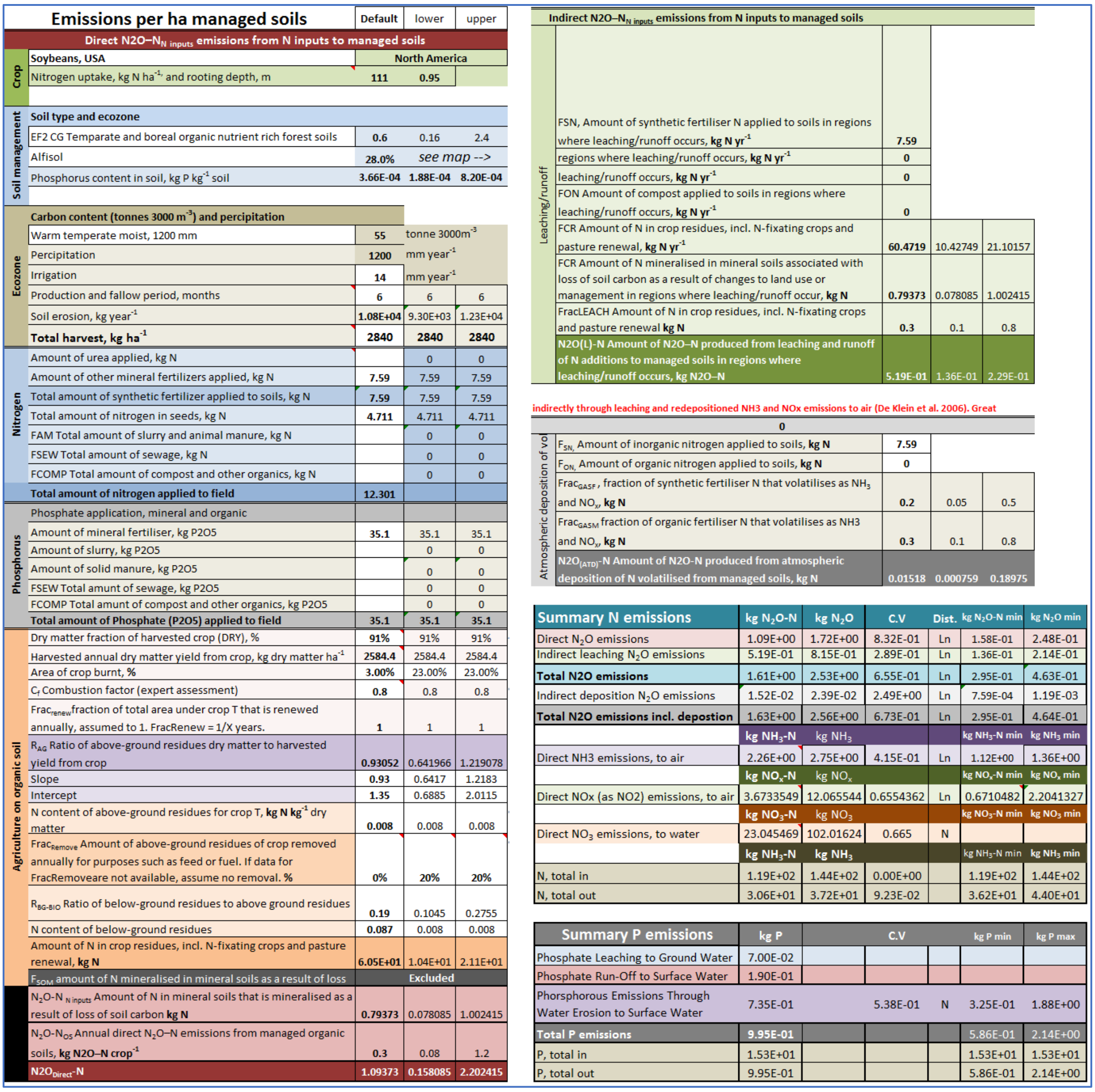
4.3.4.1. Data collection from the agricultural production of crops

It was predicted that agriculture would have a dominate contribution towards the total environmental impacts associated with salmon farming. This is apparent from previous studies, such as Pelletier et al. (2009). If economic value can be considered an acceptable indicator (e.g. Pelletier and Tyedmers 2011), this dominant contribution is perhaps implied by the financial cost of supplying feed to salmon grow-out facilities, of which the products of agriculture are major ingredients. Despite the expected prominence of agricultural activities, obtaining primary data directly from producers of agricultural products was an unrealistic prospect. This is due to the quantity of data required, and the limitations of project resources, such as time. For this reason, literature sources were used to collect data for the agricultural production of feed ingredients. Attempts were made to ensure that literature data was representative of the appropriate crop production method, country, and specific region of production, as well as being representative of a time period sufficient to avoid obtaining values that reflect periodic fluctuation. In some cases, agricultural process where available in the ecoinvent V.3. database, and these where modified to improve their representation of the processes being modelled. Data was collected describing:

- Energy use
- Seed production
- Production and yields
- Nutrient use
- Soil types and climatic conditions
- Storage
- Transport

For inputs regarding energy use and production values for Chilean agriculture, statistics made available by the Instituto Nacional de Estadisticas and the Oficina de Estudios y Politicas Agrarias were useful, as were data available through publications by a group of researchers from the Universidad de Talca (e.g. Iriarte et al. 2010). For Argentinian agriculture, statistics made available by the Instituto de Technología Agropecuaria were useful for providing values for production and yield. Occasionally, when data were considered to be important, but were either unavailable or presented ambiguously, the relevant organisation was contacted to obtain the correct values. When the regional location of crop production was unclear, efforts were made to trace the supply through information made available by ingredients processors and feed millers.

Agricultural soil emissions associated with the application of nutrients to crops (e.g. nutrient leaching, runoff, and gaseous emissions to air), were calculated using a model kindly provided by Patrik Henriksson (Figure 4.2). For emissions from nitrogen inputs, this model is based upon the methodology described by the Intergovernmental Panel on Climate Change, Guidelines for National Greenhouse Gas Inventories (IPCC 2006). The model requires a variety of inputs, such as information describing crop cultivation practices, nutrient status of the crop, the soil type and climatic variables. These input values were sourced from appropriate literature sources. The model gives the following outputs: Indirect and Indirect emissions of N2O to air, NH_3 and NO_2 emissions to air, NO_3^- emissions to water, and emissions of P to ground water and surface water. In some cases, the model is capable of generating information regarding uncertainty, such as minimum and maximum values, and a coefficient of variation. When available, the coefficient of variation was used as a basic uncertainty value, which, along with uncertainty values generated by a pedigree matrix, contributed to an overall uncertainty estimate.



Emissions per ha managed soils	Default	lower	upper
Direct N2O–N$_{N\ inputs}$ emissions from N inputs to managed soils			
Crop			
Soybeans, USA	North America		
Nitrogen uptake, kg N ha^{-1}, and rooting depth, m	111	0.95	
Soil management			
Soil type and ecozone			
EF2 CG Temparate and boreal organic nutrient rich forest soils	0.6	0.16	2.4
Alfisol	28.0%	*see map -->*	
Phosphorus content in soil, kg P kg^{-1} soil	3.66E-04	1.88E-04	8.20E-04
Ecozone			
Carbon content (tonnes 3000 m^{-3}) and percipitation			
Warm temperate moist, 1200 mm	55	tonne 3000m^{-3}	
Percipitation	1200	mm year^{-1}	
Irrigation	14	mm year^{-1}	
Production and fallow period, months	6	6	6
Soil erosion, kg year^{-1}	1.08E+04	9.30E+03	1.23E+04
Total harvest, kg ha^{-1}	2840	2840	2840
Nitrogen			
Amount of urea applied, kg N		0	0
Amount of other mineral fertilizers applied, kg N	7.59	7.59	7.59
Total amount of synthetic fertilizer applied to soils, kg N	7.59	7.59	7.59
Total amount of nitrogen in seeds, kg N	4.711	4.711	4.711
FAM Total amount of slurry and animal manure, kg N		0	0
FSEW Total amount of sewage, kg N		0	0
FCOMP Total amount of compost and other organics, kg N		0	0
Total amount of nitrogen applied to field	12.301		
Phosphorus			
Phosphate application, mineral and organic			
Amount of mineral fertiliser, kg P2O5	35.1	35.1	35.1
Amount of slurry, kg P2O5		0	0
Amount of solid manure, kg P2O5		0	0
FSEW Total amunt of sewage, kg P2O5		0	0
FCOMP Total amunt of compost and other organics, kg P2O5		0	0
Total amount of Phosphate (P2O5) applied to field	35.1	35.1	35.1
Agriculture on organic soil			
Dry matter fraction of harvested crop (DRY), %	91%	91%	91%
Harvested annual dry matter yield from crop, kg dry matter ha^{-1}	2584.4	2584.4	2584.4
Area of crop burnt, **%**	3.00%	23.00%	23.00%
C_f Combustion factor (expert assessment)	0.8	0.8	0.8
$Frac_{renew}$ fraction of total area under crop T that is renewed annually, assumed to 1. FracRenew = 1/X years.	1	1	1
R_{AG} Ratio of above-ground residues dry matter to harvested yield from crop	0.93052	0.641966	1.219078
Slope	0.93	0.6417	1.2183
Intercept	1.35	0.6885	2.0115
N content of above-ground residues for crop T, **kg N kg^{-1}** dry matter	0.008	0.008	0.008
$Frac_{Remove}$ Amount of above-ground residues of crop removed annually for purposes such as feed or fuel. If data for FracRemoveare not available, assume no removal. **%**	0%	20%	20%
$R_{BG\text{-}BIO}$ Ratio of below-ground residues to above ground residues	0.19	0.1045	0.2755
N content of below-ground residues	0.087	0.008	0.008
Amount of N in crop residues, incl. N-fixating crops and pasture renewal, **kg N**	6.05E+01	1.04E+01	2.11E+01
F_{SOM} amount of N mineralised in mineral soils as a result of loss	Excluded		
$N_2O\text{-}N_{N\ inputs}$ Amount of N in mineral soils that is mineralised as a result of loss of soil carbon **kg N**	0.79373	0.078085	1.002415
$N_2O\text{-}N_{OS}$ Annual direct N_2O–N emissions from managed organic soils, **kg N2O–N crop^{-1}**	0.3	0.08	1.2
$N2O_{Direct}$-N	1.09373	0.158085	2.202415

Indirect N2O–N$_{N\ inputs}$ emissions from N inputs to managed soils			
Leaching/runoff			
FSN, Amount of synthetic fertiliser N applied to soils in regions where leaching/runoff occurs, **kg N yr^{-1}**	7.59		
regions where leaching/runoff occurs, **kg N yr^{-1}**	0		
leaching/runoff occurs, **kg N yr^{-1}**	0		
FON Amount of compost applied to soils in regions where leaching/runoff occurs, **kg N yr^{-1}**	0		
FCR Amount of N in crop residues, incl. N-fixating crops and pasture renewal, **kg N yr^{-1}**	60.4719	10.42749	21.10157
FCR Amount of N mineralised in mineral soils associated with loss of soil carbon as a result of changes to land use or management in regions where leaching/runoff occur, **kg N**	0.79373	0.078085	1.002415
FracLEACH Amount of N in crop residues, incl. N-fixating crops and pasture renewal **kg N**	0.3	0.1	0.8
N2O(L)-N Amount of N2O–N produced from leaching and runoff of N additions to managed soils in regions where leaching/runoff occurs, kg N2O–N	5.19E-01	1.36E-01	2.29E-01

indirectly through leaching and redepositioned NH3 and NOx emissions to air (De Klein et al. 2006). Great

0			
Atmospheric deposition of vol			
F_{SN}, Amount of inorganic nitrogen applied to soils, **kg N**	7.59		
F_{ON}, Amount of organic nitrogen applied to soils, **kg N**	0		
$Frac_{GASF}$, fraction of synthetic fertiliser N that volatilises as NH_3 and NO_x, **kg N**	0.2	0.05	0.5
$Frac_{GASM}$ fraction of organic fertiliser N that volatilises as NH3 and NO_x, **kg N**	0.3	0.1	0.8
$N2O_{(ATD)}$-N Amount of N2O-N produced from atmospheric deposition of N volatilised from managed soils, kg N	0.01518	0.000759	0.18975

Summary N emissions	kg N_2O-N	kg N_2O	C.V	Dist.	kg N_2O-N min	kg N_2O min
Direct N_2O emissions	1.09E+00	1.72E+00	8.32E-01	Ln	1.58E-01	2.48E-01
Indirect leaching N_2O emissions	5.19E-01	8.15E-01	2.89E-01	Ln	1.36E-01	2.14E-01
Total N2O emissions	1.61E+00	2.53E+00	6.55E-01	Ln	2.95E-01	4.63E-01
Indirect deposition N_2O emissions	1.52E-02	2.39E-02	2.49E+00	Ln	7.59E-04	1.19E-03
Total N2O emissions incl. depostion	1.63E+00	2.56E+00	6.73E-01	Ln	2.95E-01	4.64E-01
	kg NH_3-N	kg NH_3			kg NH_3-N min	kg NH_3 min
Direct NH3 emissions, to air	2.26E+00	2.75E+00	4.15E-01	Ln	1.12E+00	1.36E+00
	kg NO_x-N	kg NO_x			kg NO_x-N min	kg NO_x min
Direct NOx (as NO2) emissions, to air	3.6733549	12.065544	0.6554362	Ln	0.6710482	2.2041327
	kg NO_3-N	kg NO_3			kg NO_3-N min	kg NO_3 min
Direct NO_3 emissions, to water	23.045469	102.01624	0.665	N		
	kg NH_3-N	kg NH_3			kg NH_3-N min	kg NH_3 min
N, total in	1.19E+02	1.44E+02	0.00E+00		1.19E+02	1.44E+02
N, total out	3.06E+01	3.72E+01	9.23E-02		3.62E+01	4.40E+01

Summary P emissions	kg P		C.V		kg P min	kg P max
Phosphate Leaching to Ground Water	7.00E-02					
Phosphate Run-Off to Surface Water	1.90E-01					
Phorsphorous Emissions Through Water Erosion to Surface Water	7.35E-01		5.38E-01	N	3.25E-01	1.88E+00
Total P emissions	9.95E-01				5.86E-01	2.14E+00
P, total in	1.53E+01				1.53E+01	1.53E+01
P, total out	9.95E-01				5.86E-01	2.14E+00

Figure 4.2. The interface of the model used to calculate emissions from nutrients applied to manage soils. The model was constructed and kindly supplied by Patrik Henriksson. Emissions from nitrogen inputs are calculated based upon IPCC Guidelines for National Greenhouse Gas Inventories methodology (2006).

4.3.4.2. Data collection for anchoveta fisheries, fish meal, and fish oil production.

Pelagic fisheries supply facilities that reduce fish to fish-meal and fish-oil. In general, there appears not to have been a source of primary data available for many published LCAs of animal production, which have relied on old data unlikely to be reflective of actual production of conditions. Fortunately, this scenario was avoided, as data for Peruvian anchoveta fisheries and reduction facilities was kindly supplied by Angel Avadi, who collected the data for LCAs relating to Peruvian aquaculture (e.g. Avadi et al. 2015). When possible, this data was modified to reflect production in Chile. Data provided by salmonid feed producers made it possible to identify specific reduction plants in Chile.

4.3.4.3. Data collection for poultry production

Data for poultry agriculture was obtained from a variety of published literature sources, covering poultry breeding (including the breeding and rearing of parents for egg laying), egg and chick production, and broiler production. The formulation for poultry feed was modelled using the data published by Pelletier (2008) which was chosen as it consisted of primary data and data based upon expert opinion. The inputs for the poultry feed involved the modelling of agricultural products from the USA.

4.3.4.4. Data collection from feed ingredient milling and other processing

In some cases, crops or animal co-products are subject to processing steps prior to their delivery to the salmon feed mill. For example, poultry ingredients are separated within multifunctional processing steps, producing products such as bone and feather meal, and processed animal protein. Another example is oil from oil seeds (such as soya), which is extracted during a milling process. Data was taken from the most appropriate sources available. For poultry processing, a PhD thesis by Ramirez provided detailed process data (Ramirez 2012). In the case of oil extraction, it was possible to locate the mill or regional area where the extraction takes place, as well as the particular extraction process used. This allowed the correct type of oil extraction process to be modelled, avoiding the use of a generic or otherwise alternative process data, for what is a major feed ingredient.

4.3.4.5. Data collection from salmonid feed milling and production

Data for feed milling was obtained from two major feed producers in Chile (references to these industrial sources of data are omitted to avoid breach of confidentiality). The data collected describes aspects relating to:

- Ingredients (formulation)
- Nutritional profile
- Energy use
- Waste
- Ingredient supplier
- Transport types and distances for major ingredients

Laboratory reports detailing biological and chemical characteristics of 'waste' sludge[13] and waste water produced by the feed production facilities was sourced from the Servicio de Evaluación Ambiental (SEA), which is a Chilean governmental body responsible for activities relating to

[13] This sludge should not necessarily be considered a waste. It has potential economic applications, and so can be viewed as a coproduct.

environmental legislation. As the locations of ingredient suppliers and previous processing was provided, crop ingredients could be traced to their respective production regions.

4.3.4.6. Data collection for salmonid rearing

Primary data was collected from various contacts in industry. One large salmonid producer provided detailed data for the inputs and outputs from two land-based salmon-smolt production facilities, as well as approximately 10 grow-out facilities. Data was obtained from another producer describing the production of smolts in a freshwater lake, and additional information describing the lake production of salmon smolts was obtained from environmental impact assessments made available through the Servicio de Evaluación Ambiental (SEA). Detailed production data for the production of Chilean Atlantic Salmon during the period of 2010-2013 was provided by the Instituto Tecnológico de Salmón (INTESAL), which is the technological and scientific research and information arm of the Chilean salmonid industry association, SalmonChile.

For emissions associated with fish metabolism (solid and dissolved forms of nitrogen, phosphorous, and carbon) a mass balance, spreadsheet model was produced. For this purpose, relevant values relating to salmon physiology and feed proximate composition was taken from literature sources and primary data provided by feed companies. The average eFCR was taken for the production of Chilean Atlantic salmon across the three-year reference period, calculated using the data supplied by INTESAL. Undoubtedly, this model could be improved (and in the future, should be improved). More specific physiological data from tissue samples of farmed Atlantic salmon, as well as data from feeds of various proximate compositions, could be applied to models, perhaps using components such as a growth-temperature co-efficient (Ferreira et al. 2008). However, developing such a model for the current project would be an emphasis that disproportionate to the overall data collection effort.

4.3.4.6. Data collection for mussel growing

Information about the general practices and industrial organisation of Chilean mussel farming was obtained through meeting with the regional manager of a major producer. The majority of data was collected through a collaborative effort with AVS Chile, an organisation that conducts research of the Chilean aquaculture industry. AVS was conducting a project to investigate life-cycle impacts associated with Chilean mussel farming, following a well organised strategy to encourage stakeholder participation from across the mussel farming sector, achieved through hosting a series of seminars with top producers and other interested groups. My participation in the data collection effort began after these seminars had been held. Subsequently, we visited the facilities of the data providers to gain a more detailed understanding of the various production processes, which was used to develop data collection surveys. The approach enabled collection of good quality, primary data from within an industry that could be categorised generally as being resistant to providing information. Detailed primary data was collected for 4 seed production facilities (mussel seed collection farms), and 3 major grow-out sites, as well as other important processes.

4.3.4.7. Data collection for seaweed growing

The i-mar Centro de Investigacion y Dessarollo de Recursos de Ambientes Costeros conducts research and development relating to the production of seaweed, in particular, that of *Macrocystis pyrifera*, as well as its use for a variety of post harvesting applications (a general overview of these research activities has been detailed by Buschmann et al. 2014). Data was collected from the (quite significant) facilities operated by i-mar. This included a land-based facility for the production of seedlings, as well as a 20 ha grow-out facility, producing 200 t ha^{-1}. This is, at the time of writing (and as far as I am aware), the largest grow-out facility of *M.pyrifera* in existence, and likely the largest kelp farm outside of Asia. As the grow-out facility was situated within close proximity to a salmonid grow-out facility, both situated within a bay, this could be considered a multi-trophic aquaculture system. Detailed data was collected for all relevant aspects of the production of *M.pyrifera*, including physiological data.

4.3.4.8. Data collection for transportation

The unit for transport input values was tonne kilometres (tkm), calculated as distance multiplied by weight of the product being transported. When possible, primary data was obtained detailing the distance products are transported as well as the type of transportation used. The majority of these cases related to ingredients delivered to salmonid feed mills, and for transportation relating to seaweed production. When these data were not available, information relating to the geographical location of production sites, canal, river and sea ports, and likely transportation routes and types, was sourced from contacts in industry, published reports, and internet websites of various producers. This information was used to measure distances for the expected or estimated routes, making use of Google Earth imagery software (e.g. Google, Digital Globe©2015) and Google Maps (e.g. Google, Map Data©2015). For feed ingredients, many of the major producers where known, as was transport distance and type, but information relating to the supply of agricultural products to these producers was unknown. Reports produced by these producers, as well as information available on their websites, were used to provide information that either identified, or was used to trace or estimate the locations of their suppliers, from seaports, to any previous processing and storage, through to the districts of agricultural production. The distance between such points was then measured using the imagery software referenced above, and the transport type was split between possible modes (e.g.

ship, rail, truck) using reports detailing statics for the transport modal shares of agricultural products (e.g. López 2012, was used for Argentinian grains and for maize from USA). In the case of Argentina and the USA, the estimated agricultural production region was vast, and so distances were measured from the city closest to the centre of these regions. Although this method can clearly be improved upon, it was not possible to do so due to resources limitations.

4.3.5. Uncertainty

4.3.5.1. The calculation of uncertainty in the Ecoinvent database.

Uncertainty in the results of life cycle assessments can come from a variety sources. For data values, quantitative uncertainty may be inherent (for example, the variation surrounding a mean value being used to quantity a specific input), whilst decisions taken by the LCA practitioner can also be a source of qualitative uncertainty. Methodological choices regarding the use of characterisation models, allocation factors and system boundary settings are common sources of practitioner influenced uncertainty. To minimise these latter sources of uncertainty, category rules are being developed to increase the consistency of methodology applied both within and across LCA models for specific products. Other sources of practitioner influenced uncertainty arise from the choice of data itself. In addition to quantitative uncertainty inherent within the chosen data, qualitative uncertainty arises from the suitability of the chosen data for quantifying specific inputs or outputs. This suitability can be influenced by qualitative attributes such as data age, the type of data chosen (for example precise measurements, estimates etc.) and the similarity between the process from which the data is taken and the process for which the data is intended to represent. The Numeral Unit Spread Assessment Pedigree (NUSAP) approach was introduced by Funtowicz and Ravetz (see Funtowicz and Ravetz 1993), as a method to analyse and describe both quantitative and qualitative dimensions of data uncertainty. In LCA, the NUSAP approach is used to provide data quality indicators which are expressed numerically to provide a quantitative assessment of qualitative uncertainty, based upon methods described by Weidema (1998). The employment of these methods within ecoinvent V.2 are outlined by Frischknecht et al. (2005). In the majority of cases, values for inputs and outputs in the ecoinvent databases are geometric means of a calculated, or assumed, lognormal distributions. The value used to describe the overall uncertainty is the square of the geometric standard deviation (95% interval). This is calculated using seven uncertainty factors described in Frischknecht et al. (2005), as being '*expressed as a contribution to the square of the geometric standard deviation*,' which appears to imply they are equivalent to squared geometric standard deviations. Six of the uncertainty factors are based upon those described by Weidema (1998) and are described in Table 4.2. Another uncertainty factor describes 'basic uncertainty,' which appears to represent variability and stochastic error of the mean values that quantify inputs and outputs. A variety of basic uncertainty values are provided for different categories of inputs and outputs, and are described as being '*based on expert judgements*.' Although it is somewhat unclear, if these are equivalent to standard deviations, the basic uncertainty can be calculated when sample data are available. Using these uncertainty factors, the overall uncertainty value (σ_g^2) is calculated using the following equation:

Eq.4.3

$$\sigma_g^2 = \exp\sqrt{[\ln(U_1)]^2 + [\ln(U_2)]^2 + [\ln(U_3)]^2 + [\ln(U_4)]^2 + [\ln(U_5)]^2 + [\ln(U_6)]^2 + [\ln(U_b)]^2}$$

where U_1 to U_6 are the uncertainty factors for 'reliability,' 'completeness,' 'temporal correlation,' 'geographic correlation,' 'technological correlation' and 'sample size' respectively, and *Ub* represents 'basic uncertainty.'

The release of ecoinvent V.3 introduced some changes in how uncertainty is calculated, as described in Weidema et al. (2013). The pedigree matrix has been changed to that of Weidema (1998), albeit with some slight moderation. These changes include an extra quality option for both the indicators '*geographical correlation*' and '*further technological correlation*' (Table 4.3.), and the data quality indicator '*sample size*,' has been removed. Furthermore, the uncertainty factors of the data quality indicators (Table 4.3.) as well as the uncertainty factors for basic uncertainty (not shown) are expressed as variances of the underlying normal distribution of the lognormal distribution. The overall variance is calculated as:

$$\sigma^2 = \sum_{n=1}^{6} \sigma_n^2$$

Eq.4.4

Table 4.2. Pedigree matrix as used in ecoinvent V.2 (Frischknecht et al., 2005).

Indicator	Indicator score		Uncertainty factor
Reliability	1	Verified data based on measurements	1
	2	Verified data partly based on assumptions OR non-verified data based on measurements	1.05
	3	3. Non-verified data partly based on qualified estimates	1.1
	4	Qualified estimate (e.g. by industrial expert); data derived from theoretical information (stoichiometry, enthalpy, etc.)	1.2
	5	Non-qualified estimate	1.5
Completeness	1	Representative data from all sites relevant for the market considered over an adequate period to even out normal fluctuations	1
	2	Representative data from >50% of the sites relevant for the market considered, over an adequate period to even out normal fluctuations	1.02
	3	Representative data from only some sites (<<50%) relevant for the market considered or >50% of sites but from shorter periods	1.05
	4	Representative data from only one site relevant for the market considered OR some sites but from shorter periods	1.1
	5	Representativeness unknown or data from a small number of sites and from shorter periods	1.2
Temporal correlation	1	Less than 3 years of difference to the time period of the dataset	1
	2	Less than 6 years of difference to the time period of the dataset	1.03
	3	Less than 10 years of difference to the time period of the dataset	1.1
	4	Less than 15 years of difference to the time period of the dataset	1.2
	5	Age of data unknown or more than 15 years of difference to the time period of the dataset	1.5
Geographical correlation	1	Data from area under study	1
	2	Average data from larger area in which the area under study is included	1.01
	3	Data from smaller area than area under study, or from similar area	1.02
	5	Data from unknown OR distinctly different area	1.1
Further technological correlation	1	Data from enterprises, processes and materials under study	1
	3	Data on related processes or materials but same technology, OR Data from processes and materials under study but from different technology	1.2
	4	Data on related processes or materials but different technology, OR data on laboratory scale processes and same technology	1.5
	5	Data on related processes or materials but on laboratory scale of different technology	2
Sample size	1	n = >100, continuous measurement, balance of purchased products	1
	2	n = >20	1.02
	3	n = > 10, aggregated figure in environmental report	1.05
	4	n = >3	1.1
	5	Unknown	1.2

Table 4.3. Default basic uncertainty values (U_b) used in ecoinvent V.2 (Frischknecht et al., 2005).

Input / output group	c	p	a
c=combustion emissions, p=process emissions, a=agricultural emissions			
Demand of:			
Thermal energy, electricity, semi-finished products, working material, waste treatment services	1.05	1.05	1.05
Transport services (tkm)	2	2	2
Infrastructure	3	3	3
resources:			
Primary energy carriers, metals, salts	1.05	1.05	1.05
Land use, occupation	1.5	1.5	1.5
Land use, transformation	2	2	2
Pollutants emitted to water:			
BOD, COD, DOC, TOC, inorganic compounds (NH_4, PO_4, NO_3, Cl, Na etc)		1.5	
Individual hydrocarbons, PAH		3	
Heavy metals		5	1.8
Pesticides			1.5
NO_3, PO_4			1.5
Pollutants emitted to soil:			
Oil, hydrocarbon total		1.5	
Heavy metals		1.5	1.5
Pesticides			1.45
Pollutants emitted to air:			
CO_2	1.05	1.05	
SO_2	1.05		
NMVOC total	1.5		
NO_X, N_2O	1.5		1.4
CH_4, NH_3	1.5		1.2
individual hydrocarbons	1.5	2	
PM>10	1.5	1.5	
PM10	2	2	
PM2.5	3	3	
Polycyclic aromatic hydrocarbons (PAH)		3	
CO, heavy metals		5	
Inorganic emissions, others		1.5	
Radionuclides (e.g. Radon-222)		3	

4.3.5.2. The calculation of Uncertainty using SimaPro Software

The calculation of uncertainty in the ecoinvent v3 database as it is employed within SimaPro 8.2. operates slightly differently. The options of quality levels for each data quality indicator within the pedigree matrix are the same as those described in Weidema et al. (2013). However, the uncertainty factors of the data quality indicators (Table 4.4) and for basic uncertainties (not shown) are expressed using the same form described in Frischknecht et al. (2005). It appears that these uncertainty factors are converted from variance of the underlying normal distribution as presented by Weidema et al.

(2013), to squares of the geometric standard deviation using the following formula (or an equivalent calculation):

$$\sigma_g^2 = \exp((Variance\ of\ logtransformed\ data)^{0.5})^2 \qquad \text{Eq.4.5}$$

Table 4.4. Pedigree matrix as used in ecoinvent V3., with uncertainty factors being expressed as variances of the underlying normal distribution of the lognormal distribution (Weidema et al. 2016).

Indicator	Indicator score		Uncertainty factor
Reliability	1	Verified data based on measurements	0
	2	Verified data partly based on assumptions or non-verified data based on measurements	0.0006
	3	Non-verified data partly based on qualified estimates	0.002
	4	Qualified estimate (e.g. by industrial expert)	0.008
	5	Non-qualified estimate	0.04
Completeness	1	Representative data from all sites relevant for the market considered, over an adequate period to even out normal fluctuations	0
	2	Representative data from >50% of the sites relevant for the market considered, over an adequate period to even out normal fluctuations	0.0001
	3	Representative data from only some sites (<<50%) relevant for the market considered or >50% of sites but from shorter periods	0.0006
	4	Representative data from only one site relevant for the market considered or some sites but from shorter periods	0.002
	5	Representativeness unknown or data from a small number of sites and from shorter periods	0.008
Temporal correlation	1	Less than 3 years of difference to the time period of the dataset	0
	2	Less than 6 years of difference to the time period of the dataset	0.0002
	3	Less than 10 years of difference to the time period of the dataset	0.002
	4	Less than 15 years of difference to the time period of the dataset	0.008
	5	Age of data unknown or more than 15 years of difference to the time period of the dataset	0.04
Geographical correlation	1	Data from area under study	0
	2	Average data from larger area in which the area under study is included	0.000025
	3	Data from area with similar production conditions	0.0001
	4	Data from area with slightly similar production conditions	0.0006
	5	Data from unknown OR distinctly different area	0.002
Further technological correlation	1	Data from enterprises, processes and materials under study	0
	2	Data from processes and materials under study (i.e. identical technology) but from different enterprises	0.0006
	3	Data from processes and materials under study but from different technology	0.008
	4	Data on related pro-cesses or materials	0.04
	5	Data on related pro-cesses on laboratory scale or from different technology	0.12

The uncertainty values entered into SimaPro describe the uncertainty for individual inputs and outputs (i.e. inventory flows). The values are used as inputs to Monte Carlo Analysis which is used to describe uncertainty in the calculated results. Monte Carlo Analysis in SimaPro can be used to generate confidence intervals that describe the uncertainty from within the inventory for each impact category of the impact assessment. The results of the Monte Carlo analysis viewed as a probability distribution for each impact category result and for each substance that contributes to the category. Uncertainty can also be compared between products or unit processes. Thus, Monte Carlo analysis is used to provide a detailed exploration of the how uncertainty within the inventory contributes to the results of the impact assessment. However, it must be made clear that this does not describe or include the uncertainty associated with the characterisation models that are used to calculate the impacts of the modelled inventory.

4.3.5.3. Horizontal averaging of unit process data

When collecting data for life cycle inventories, practitioners may use a value from a single data source (e.g. the mean of sample of values taken from a single unit process), or a value representing a number of values taken from multiple sources (e.g. the mean of the means of individual samples taken from a variety of examples of the unit process). Using SimaPro, it is possible to add an uncertainty value to inputs and outputs using a built-in function that uses the pedigree matrix detailed in Table 4.4. This allows the user to select a basic uncertainty value and select the appropriate quality level for each data quality indicator. This is useful when an input quantifying value for which the uncertainty is being calculated, is representative of a single unit process (single data source). However, its use can be problematic when the value being entered is representative of a variety of data sources. In this study, when literature values representing unit processes are being used to build the life cycle inventories, efforts have been made to collect data from a number of literature sources. This decision is based on the assumption that using a greater number of data sources can improve the accuracy of the quantifying process values, or, in other words, reduce the associated uncertainty. This seems an intuitive approach. For example, if the LCI data is being collected to represent the operation of a process at a national level, data from one example of that process within the country will not be as representative as data taken from all examples of the process (assuming there is sufficient variation nationwide). However, if, for example, a collection of means, each from different literature sources, are horizontally averaged to produce a mean value to be used as the quantifying value for a process input, this averaging itself will introduce further uncertainty in the form of spread. As the horizontal averaging of values introduces further uncertainty, and the application of the pedigree matrix to the mean value this averaging produces, is problematic. In this study, the generating uncertainty

Table 4.5. Pedigree matrix and uncertainty factors as used in Simapro 8.2. (Goedkoop et al., 2016).

Indicator		Indicator score	Uncertainty factor
Reliability	1	Verified data based on measurements	1
	2	Verified data partly based on assumptions OR non-verified data based on measurements	1.05
	3	Non-verified data partly based on qualified estimates	1.1
	4	Qualified estimate (e.g. by industrial expert); data derived from theoretical information (stoichiometry, enthalpy, etc.)	1.2
	5	Non-qualified estimate	1.5
Completeness	1	Representative data from all sites relevant for the market considered over an adequate period to even out normal fluctuations	1
	2	Representative data from >50% of the sites relevant for the market considered, over an adequate period to even out normal fluctuations	1.02
	3	Representative data from only some sites (<<50%) relevant for the market considered or >50% of sites but from shorter periods	1.05
	4	Representative data from only one site relevant for the market considered OR some sites but from shorter periods	1.1
	5	Representativeness unknown or data from a small number of sites and from shorter periods	1.2
Temporal correlation	1	Less than 3 years of difference to the time period of the dataset	1
	2	Less than 6 years of difference to the time period of the dataset	1.03
	3	Less than 10 years of difference to the time period of the dataset	1.1
	4	Less than 15 years of difference to the time period of the dataset	1.2
	5	age of data unknown or more than 15 years of difference to the time period of the dataset	1.5
Geographical correlation	1	Data from area under study	1
	2	Average data from larger area in which the area under study is included	1.001
	3	Data from smaller area than area under study, or from similar area	1
	4	Data from area with slightly similar production conditions	1.05
	5	Data from unknown OR distinctly different area	1.1
Further technological correlation	1	Data from enterprises, processes and materials under study	1
	2	Data from processes and materials under study (i.e. identical technology) but from different enterprises	1.05
	3	Data on related processesor materials but same technology, OR Data from processes and materials under study but from different technology	1.2
	4	Data on related processes or materials but different technology, OR data on laboratory scale processes and same technology	1.5
	5	Data on related processes or materials but on laboratory scale of different technology	2

estimates for horizontal averaging of data from different sources has been performed following the method developed by Henriksson et al (2014). The approach and calculation steps used are described in detail below, as the use of this method is a significant and somewhat novel component of this study, intended to improve the quality of life cycle assessments of aquaculture production systems.

Input Name	Unit											
			ln(x̄gi/x̄g)^2	0.15	ln(x̄gi/x̄g)^2	0.01	ln(x̄gi/x̄g)^2	0.09	ln(x̄gi/x̄g)^2		ln(x̄gi/x̄g)^2	
x̄g(wt)	**4.62478197**	$\bar{x}_g i$	8		5		4					
$\bar{x}g$	5.428835233	$\sigma_g{}^u$	1.05		1.05		1.05		1.05		1.05	
$\sigma_g{}^u$	1.05	Reliability	1. Verified	1.00	4. Qualifie	1.20	1. Verified	1.00	1. Verified	1.00	1. Verified	1.00
$\sigma_g{}^r$	1.009950494	Completeness	5. Represe	1.20	1. Represe	1.00	2. Represe	1.02	1. Represe	1.00	1. Represe	1.00
$\sigma_g{}^s$	1.424504764	Temporal correlation	1. Less tha	1.00	1. Less tha	1.00	1. Less tha	1.00	1. Less tha	1.00	2. Less tha	1.03
		Geographical correlation	1. Data fro	1.00	1. Data fro	1.00	1. Data fro	1.00	1. Data fro	1.00	1. Data fro	1.00
$(\sigma_g{}^o)^2$	**2.043408085**	Further technical correlation	1. Data fro	1.00	1. Data fro	1.00	1. Data fro	1.00	1. Data fro	1.00	1. Data fro	1.00
		$\sigma_g{}^r$	1.095445115		1.095445115		1.009950494					
		$\sigma_g{}^{ur}$	1.108930645		1.108930645		1.05104478					
		wi	93.53865259		93.53865259		403.4672146					
		wilnx̄i	194.50816		150.5446538		559.3243245					

Figure 4.3. Spreadsheet built for the horizontal averaging of data points, producing a weighted geometric mean and overall uncertainty value suitable to be used alongside the ecoinvent V3. database within SimaPro 8 software, based upon the protocol described by Henriksson et al. (2014). In this example, values from 3 hypothetical sources have been entered. The empty white cells are those into which the user enters a value. Values ($\bar{x}_g{}^i$) are entered from each separate source, each describing a particular common input or output of an activity. The geometric standard deviation ($\sigma_g{}^u$) of each of these values is entered in the cells below. Below each of these cells are a further 5 white cells, each of which is a drop-down list, allowing the user to select the correct qualitative level for the specific value. For example, an $\bar{x}_g{}^i$ value of 8 has a $\sigma_g{}^u$ of 1.05, and is assigned the level 1, 5, 1, 1, 1, for the quality indicators reliability, completeness, temporal correlation, geographic correlation, and technical correlation. The calculated weighted mean ($\bar{x}_{g(wt)}$) of the three $\bar{x}_g{}^i$ values 8, 5 and 4, calculated using their respective standard deviation and quality scores, is an output of the spreadsheet, and in this case is 4.625. Another crucial output, the uncertainty value, $(\sigma_g^o)^2$, is 2.044.

The method developed by Henriksson et al. (2014) uses a weighting procedure that produces a weighted average of values that are each from a different data source. The method also generates an overall dispersion value (σ^o) that is used to describe the uncertainty associated with the weighted mean. Weighting of the mean is done using a weighting factor, calculated using a value for representativeness (σ^r) and inherent uncertainty (σ^u). The value for representativeness is obtained from the uncertainty factors of the pedigree matrix, which is applied to each of the data points that are to be averaged. The inherent uncertainty for each is the standard deviation for each mean value from the individual data sources. The application of the pedigree matrix to each data point is intended to avoid the problems associated with applying the pedigree matrix to one value obtained from the horizontal averaging of a sample of data values from different sources, as described above. The pedigree matrix used will depend upon the intended application, and supplementary material by Henriksson et al. (2014) provide spreadsheets to be used for calculating arithmetic weighted means and uncertainty values using the pedigree matrices described by Frischknecht et al. (2005) and Weidema et al. (2013). However, as the calculation of uncertainty in in SimaPro uses a pedigree matrix blending features of both Frischknecht et al. (2005) and Weidema (2016), and as the majority of quantifying values for each input within a process are assumed to be from a lognormal distribution (unless otherwise apparent), in this study the protocol developed by Henriksson et al. (2014) was modified to calculate the weighted ***geometric*** mean using the pedigree matrix as it features in SimaPro. This was used to build a spreadsheet for the horizontal averaging of data points to produce a weighted geometric mean and overall uncertainty value suitable to be used alongside the ecoinvent v3. database within SimaPro 8 software (Figure 4.3.). The calculation procedure is now described below. For calculating weighted means using the pedigrees of Frischknecht et al. (2005) and Weidema et al. (2016), as well as example calculations, description of equations, and background information for the methods development, please refer to Henriksson et al. (2014).

Each mean value ($\overline{x}_g^{\,i}$) from the literature sources was assumed to be of a lognormal distribution unless otherwise stated. The values of inherent uncertainty for each data point were entered as geometric standard deviations ($\sigma_g{}^u$). When this value was not supplied by the data source, or the source contained insufficient data to calculate the value, a default value was used. The default values used were those provided as default basic uncertainty values within SimaPro, the most common of which is 1.05. Representativeness ($\sigma_g{}^r$) was calculated as the sum of squared uncertainty factors (Eq.4.6), with the uncertainty factors being those provided by the pedigree matrix available in Simapro. The weighting factor (w) was then calculated using equation 4.7. For each data point from each data source, a value for $\sigma_g{}^u$ *and* $\sigma_g{}^r$ and w are generated. The weighted mean was then calculated using equation 4.8.

$$\sigma_g^r = \sqrt{\exp\sqrt{[\ln(U_1)]^2 + [\ln(U_2)]^2 + [\ln(U_3)]^2 + [\ln(U_4)]^2 + [\ln(U_5)]^2 + [\ln(U_6)]^2}} \qquad \text{Eq.4.6}$$

$$w = \frac{1}{\ln(\sigma^{u+r})^2} \qquad \text{Eq.4.7}$$

$$\overline{x}_{(wt)} = \frac{1}{\sum w_i} \sum w_i \, \overline{x}_i \qquad \text{Eq.4.8}$$

Overall dispersion was calculated using equation 4.10 and its output is the square of the geometric standard deviation ($\sigma_g{}^o$), the input value for uncertainty in Simapro when lognormal distributions are assumed. As recommended by Henriksson et al. (2014), the lowest reported inherent uncertainty and representativeness are used in the calculation of overall dispersion, as their calculation in for each data source contributes towards the weighting factors. Spread of the data values was calculated as the geometric standard deviation of the entered data points using equation 4.9, where x_g is the geometric mean of the data points.

$$\sigma_g^s = \exp\left(\sqrt{\frac{\sum\left(\ln\frac{x_i}{x_g}\right)^2}{n-1}}\right) \qquad \text{Eq.4.9}$$

$$\sigma_g^{o^2} = \exp\sqrt{\left[\ln\left(\sigma_g^{u^2}\right)\right]^2 + \left[\ln\left(\sigma_g^{s^2}\right)\right]^2 + \left[\ln\left(\sigma_g^{r^2}\right)\right]^2} \qquad \text{Eq.4.10}$$

Chapter 5: Life Cycle Assessment of Salmonid Feed Production

5.1. Introduction

This chapter presents the life cycle assessment of salmon feed. In general, the requirement for formulated, quality feed carries the majority share of financial cost and environmental burdens associated with intensively farmed salmon. Among the ingredients of salmon feed, agricultural products can be expected to contribute significantly to these environmental impacts. More specifically, it can be anticipated that processes associated with the growing phase of these agricultural and animal crops will be accountable for a large share of impacts (e.g. Pelletier et al. 2009). For these reasons, it is important that LCAs of intensively reared high value finfish, such as salmon, are supported by a foundation of good agriculture process models. This requirement presents challenges, and perhaps also, opportunities for researchers wishing to develop comprehensive LCAs for salmon aquaculture systems. These challenges arise from the variety of feed formulations used throughout the industry, as well as the variety of formulations used within an individual feed product. This latter source of variety is due to a fluctuation of the type and quantity of ingredients used to provide a standard proximate composition (nutritional profile and other contents such as moisture), and it occurs when the specific formulation of a given feed product is calculated based upon changeable prices of several competing, alternative ingredients. Additionally, within an individual salmon farming system, fish will be reared using different feed types at different phases of the cultivation cycle. Different feed types are available for the different juvenile stages, for smolts, and for fish being grown at sea. Some feeds may be medicated with chemotherapeutics, and fish given 'finishing diets' which improve certain qualitative aspects of the fish product, such as factors related to the fatty acid content of the meat. Alternative salmon feeds may contain various levels of the natural carotenoid pigment astaxanthin, which colours the salmon's flesh to the appropriate shades of pink required by the consumer. As a consequence of variation both within specific feed types and among the alternative feed types used in different enterprises and regions, researchers can be faced by the difficulty of obtaining a sufficient quantity and quality of data from producers of feed. They are also challenged by the difficulty of utilising data within what is likely to be a static model, but which is representative of a dynamic process. This latter problem is particularly encountered by those performing LCAs of salmon reared over several geographic regions. Of course, problems of data representativeness in LCA are not unique to salmon feed production. However, when considering LCAs of intensively reared finfish like salmon, such issues as they relate to feed production ought to be given some priority. If an LCA compares the production of salmon in one country with that of another, assumptions regarding the feed ingredients used in those respective countries may well lead to misleading results. Methods for dealing with uncertainty can, and should be, applied when modelling feed production processes. However, efforts need to be made to reduce this uncertainty through the acquisition of representative data and through sensible modelling approaches.

Another challenge faced by those wishing to perform LCAs of salmon feed relate to the production of its ingredients. Due to their significant contribution towards the environmental impacts of farmed salmon, effort must be made to obtain good quality and representative data. Agricultural systems have variable inputs, methods, and cultivation environments, and are frequently part of logistically complex networks. In general, obtaining good quality, comprehensive data from primary agricultural processes requires considerable effort and resources. This challenge is further compounded when the regions or even nations of the source of agriculture production are not known. Consequently, the common circumstance of researchers working with limited resources, has necessitated an unfortunate reliance upon data from published literature and experimental scale projects that do not adequately represent the real situation. Concurrently, there has been an absence of available data describing capture fishery production. Recent progress has been made in this area and more recent and process specific data has been obtained for capture both fishery production and reduction of fish into oil and meal (Avadi et al. 2014; Avadi et al. 2015). However, this new data is limited in its geographical scope, and so further advances are still required. Hopefully, improving the future availability of quality data relating to capture fisheries and reduction processes should be an achievable objective.

Accurate data describing the types and quantities of vegetable oil ingredients in compound feed require is another important requirement. Previous LCA research suggests that the contribution of vegetable oils towards the impacts of salmon feed, may be quite considerable (e.g. Pelletier et al. 2009). Various vegetable oils may be used as a feed ingredient, and their inclusion can be subject to regional, as well as feed type specific variation. Considering that vegetables oils are expected to have a non-negligible contribution to the impacts of feed production, and that these contributions are likely to differ across different types of oil, efforts should be made to maintain accuracy in the inventory data describing both their production and their inclusion as an ingredient. As it relates to the LCA of salmonid feed described below, this consideration is deemed important, because inaccuracies may be reflected not only in the environmental profile of feed, but also in that of salmon production, thus inappropriately influencing the comparative performance of salmon monoculture and IMTA systems.

5.2. Goal and Scope - Brief Definition

The goal is to produce a good quality LCA of salmonid feed production (Functional unit: 1 kg of feed at mill-gate) and the production of its requisite ingredients, within the conditions of the above-mentioned limitations. As significant resources and efforts were required to collected data representative of Chilean salmon rearing, mussel growing, and seaweed cultivation, the collection of primary agriculture and capture fishery data could not reasonably be included within the scope if the study. For this reason, like previous studies, data was collected from secondary sources, but with efforts made to improve the efficacy of gathering of these data and their utilisation within the LCA. This improvement effort was made using a methodological approach by which data from a variety of sources are averaged for each individual process flow (rather than using an individual data point from only one source), combined with an advanced method of propagating the uncertainty around the data point resulting from the averaging process. Agricultural data was collected in accordance with the data collection methodology specifically developed for the agricultural production of crops and poultry, as detailed in Chapter 4.

The attainment of primary data from leading feed producers was a set objective, and efforts to this end have been successful. However, due to the above-mentioned limitations regarding variability both within and between available feed types, data was collected from a common, widely used feed type, the formulation of which was calculated as the average inclusion of ingredients overtime. The averaging of feed ingredients was calculated by the data provider, and reduces the variation[14] of ingredient inclusion rates within the feed type. Through using data representative of a widely used feed type from a major producer, at least some variation between feed alternative types will hopefully be reduced, although this reduction is clearly assumed and cannot be quantified. The data provided included ingredient suppliers, their location, and the distance travelled by the modes of transport used in their delivery to the feed production facility. The regions of agricultural production were estimated based upon the location of the ingredient suppliers and any information regarding their activities that is publicly obtainable.

[14] As this calculation was performed by the producer using commercially sensitive data, no quantification of the reduction in variation is possible within this study.

5.3. Inventory

The major process required for the production of salmon feed are identified in the flowchart below (Figure 5.1). Issues and information relating to the inventory of these processes are described within this section. The inventory data for the production of a generic formula salmon feed is shown in Table 5.1.

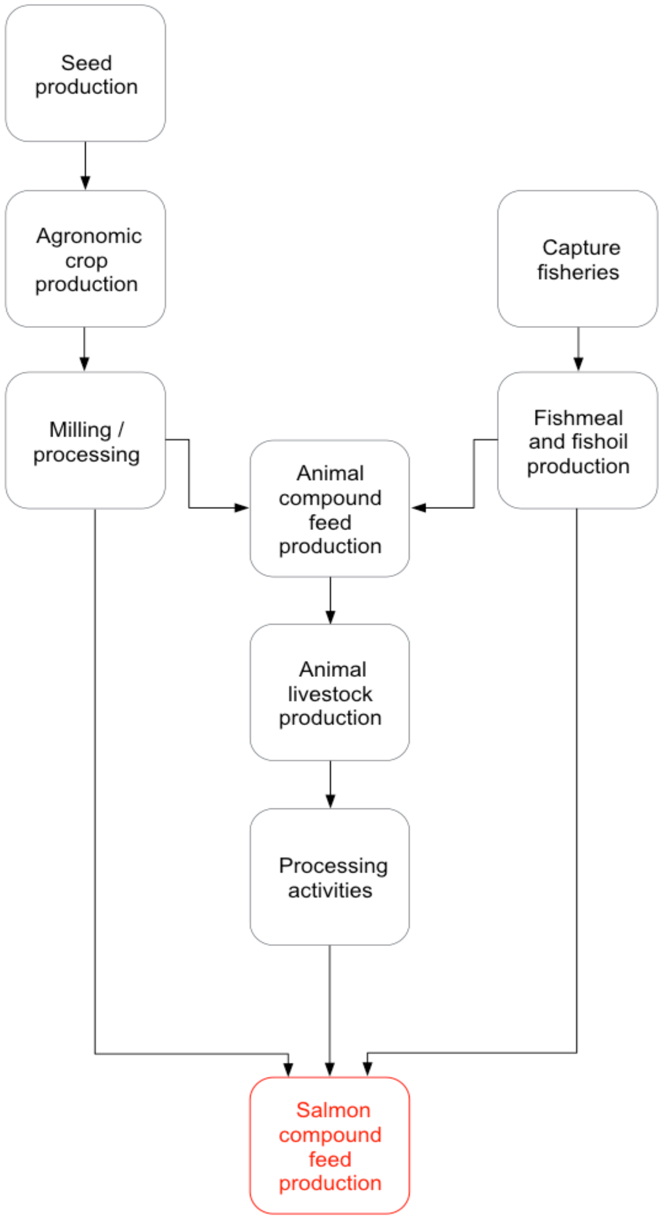


Figure 5.1. Flowchart depicting the major upstream processes required to supply products to the salmon feed production process. For the benefit of easy viewing, inputs such as energy carriers, transport and fertiliser, and outputs such a co-products and emissions to environmental compartments, are not depicted.

Table 5.1. Inventory data describing the production of 1 kg of generic formula salmon feed, available at the mill-gate.

Outputs		**Value**	**Unit**		
Salmon compound feed		1	kg		
Inputs		**Value**	**Unit**	**Distribution**	**SD^2**
Materials/fuels					
Wheat {CL} at farm gate		0.0000207	ha	Lognormal	1.11
Rape oil, crude {CL}		0.092	kg	Lognormal	1.11
Sunflower oil, crude, {AR}		0.092	kg	Lognormal	1.11
Sunflower meal, from oil extraction, {AR}		0.10421397	kg	Lognormal	1.11
Maize gluten meal {US}		0.078	kg	Lognormal	1.11
Rape meal {CL}		0.05	kg	Lognormal	1.11
Hydrolysed feather meal {US}		0.118	kg	Lognormal	1.11
Poultry processed animal protein {US}		0.16	kg	Lognormal	1.11
Tap water {RoW}| market for | Alloc Def, U		0.01778603	kg	Lognormal	1.11
Fish oil, anchoveta reduction {PE}		0.073	kg	Lognormal	1.11
Fish meal, anchoveta reduction {CL}		0.08	kg	Lognormal	1.11
Generic Forklift truck, Liquefied Natural Gas		0.000843195	l	Lognormal	1.11
Transport of feed ingredients for Salmon feed		1	kg	Undefined	
Electricity/heat					
Electricity, medium voltage {CL}| market for | Alloc Def, U		0.063	kWh	Lognormal	1.11
Heavy fuel oil, burned in refinery furnace {RoW}		0.6873808	MJ	Lognormal	1.11
Emissions	**Compartment**	**Value**	**Unit**	**Distribution**	**SD^2**
..to water					
Nitrogen	ocean	0.993024	mg	Lognormal	1.22
BOD5, Biological Oxygen Demand	ocean	13.27104	mg	Lognormal	1.22
Phosphorus	ocean	0.237312	mg	Lognormal	1.22
Oils, biogenic	ocean	1.65888	mg	Lognormal	1.22
Suspended solids, unspecified	ocean	9.51552	mg	Lognormal	1.22

5.3.1. Production of feed ingredients derived from agronomic crop systems

The average feed formulation consisted of the following ingredients derived from the production of agronomic crops.

- Whole wheat
- Rapeseed meal
- Sunflower meal
- Maize gluten meal
- Vegetable oil

Wheat is obtained from three suppliers in Chile, and wheat was assumed to be grown in this country. Maize gluten is supplied from a variety of producers in the United States of America. It is likely that the agricultural production of maize also comes from the same country, as the production of maize gluten takes place as part of vertically integrated organisations controlling most aspects of production, including agriculture.

Vegetable oil was obtained from two suppliers, with each individually supplying 50% of the total vegetable oil used. Details of the suppliers indicate that the vegetable oil is a 50:50 blend of crude sunflower-oil and crude rapeseed-oil. The rape-oil is produced in Chile, and the sunflower oil is produced in Argentina. The agricultural production of rapeseed and sunflower seed was assumed to be grown in the same country as where its milling takes place (i.e. Chile and Argentina respectively). Clearly, it is entirely possible that each mill variably derives its crop inputs from a variety of locations. However, both the mills in Chile and in Argentina region are located in their nations region with the highest production of the relevant crop type. This is considered to be further justification for selecting these countries for agricultural production. The production of rape-meal and sunflower-meal also occur in Chile and Argentina respectively, and so these same countries have been chosen for agricultural production.

5.3.1.2. Agricultural production of agronomic crops

The production of all crop types is modelled according to a standardised framework, as depicted by the flow diagram below (Figure 5.2). This was used to define the basic structure of all modelled

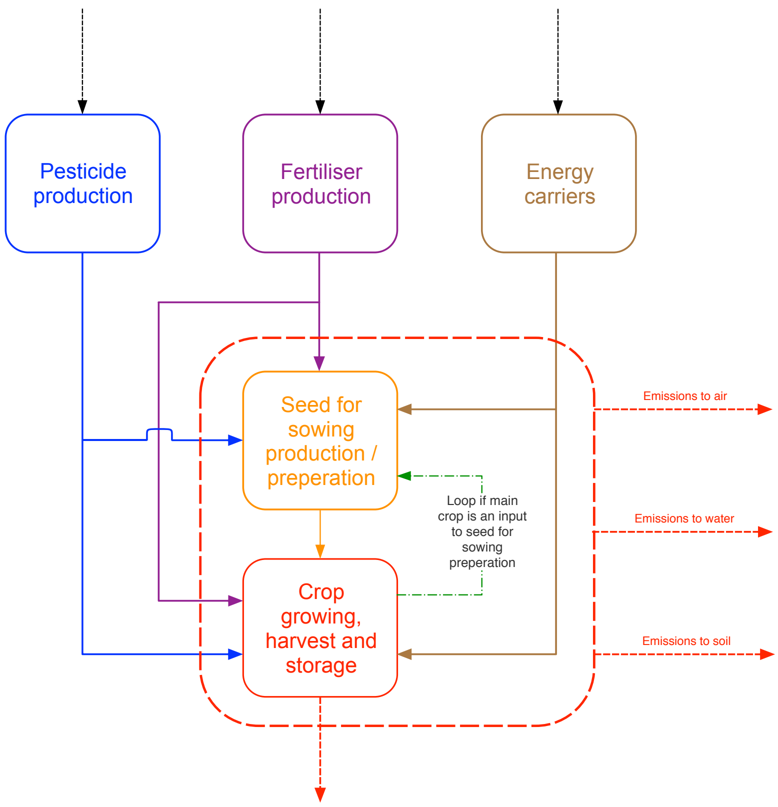


Figure 5.2. Standard model framework for agronomic crop production. The red, dashed-lined box delineates foreground processes for which efforts were made to collect and compile process data. The ecoinvent database V.3. is used for processes outside of this box.

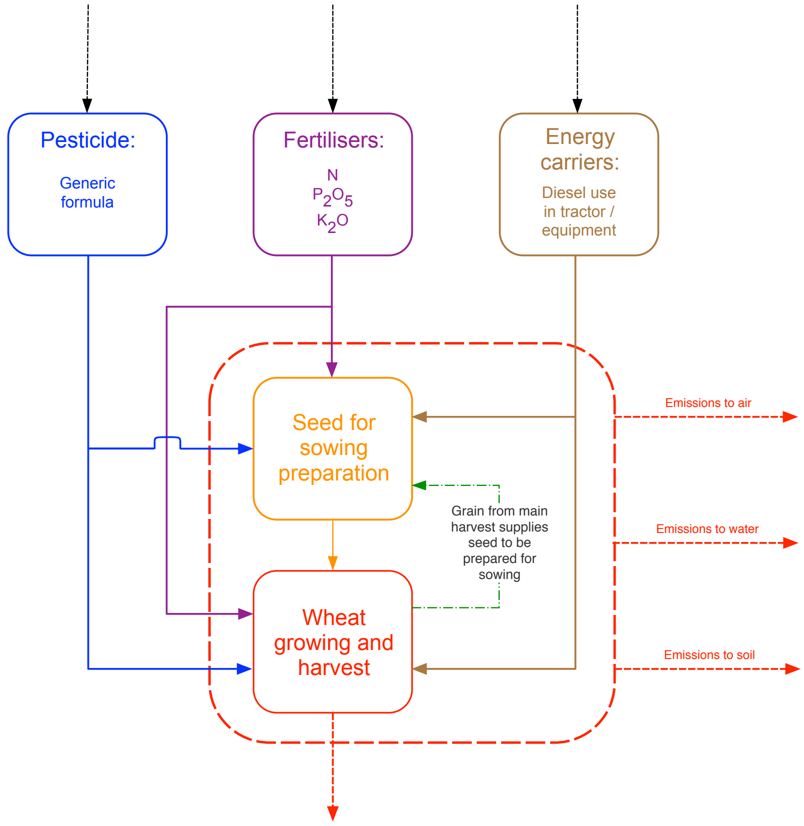


Figure 5 3. Major input and output flows for the production of wheat grain, modelled using the standard framework depicted in Figure 5.2.

agriculture systems, and provided the required consistency across the types of inputs and outputs for each crop system.

A flow diagram depicting the agricultural production of wheat is shown in Figure 5.3, and it can be seen that its structure correlates to that of the standardised system boundaries defined by Figure 5.2. The inventory data for this process can be seen in Table 5.2. An input to wheat is 'wheat seed for sowing.' This input was modelled by modifying the processes 'Wheat seed, Swiss integrated production, for sowing,' which was obtained from the ecoinvent V.3. database. This provided data for inputs such as application of a chemical herbicide to the harvested seed that will be used for sowing, and electricity use. The reason behind using a processes from ecoinvent is thus: 1) Data from Chile (and other countries) for the preparation of wheat into seed for sowing, is difficult to find 2) The input flow of wheat seed for sowing is approximately 3% of the final yield of wheat by mass, and so should not account for significant proportion of impacts towards the impacts of wheat grain 3) Within the context of salmon production, the production of seed for sowing should not be a significant input flow by mass. This same reasoning was applied to the treatment of seed for sowing for other crop types. The ecoinvent process 'seed for sowing' is assigned pedigree scores upon the basis of the processes being used to reflect Swiss production. Therefore, the uncertainty scores have been modified to reflect the quality of the data within the context of its use for describing Chilean production. The `wheat seed for sowing´ process requires an input of 'wheat seed,' as the process only describes the preparation of wheat seed into a product that is suitable for its intend use as a seed for sowing. The ´wheat seed' input is the process describing the cultivation of wheat to produce seed that will be treated in preparation for sowing. As there is (according to ecoinvent) no difference between the production of wheat grain and the production of wheat seed for sowing, the 'wheat seed' input is the same process as the wheat grain process. In this respect, the structure of my own ´seed for sowing' model is identical to that of the structure employed by ecoinvent. However, whereas the process provided by the ecoinvent database uses a generic global market model for the 'wheat seed' input, my own model describing 'Chilean wheat grain production' was used as the input for seed. In effect, this creates a loop, but looping was avoided by creating a second, physically separate but identical process to that of 'Chilean wheat grain production,' differing only in the name given to the process. Here, it is worth mentioning the case of sunflower seed production. The cultivation of 'sunflower seed for sowing' is considered to be the same procedure as cultivating the main crop. However, when sunflower is grown for producing seeds for sowing, the yield is appreciably lower than conventional sunflower seed production. This was easily accounted for within the model through manipulation of the input value for 'sunflower seed' into the process 'sunflower seed for sowing.' In this case, the yield for conventional seed production is 3151 kg/ha, whereas the yield from the production of seed for sowing is 2000 kg. Therefore, instead of 1 kg of sunflower seed being the input value required for an output of 1 kg of seed prepared for sowing, the input is 1.576 kg (3151 kg ÷ 2000 kg). Manipulation of the input value in this way accounts for the difference in yield, whilst maintaining the correct flow of mass balance.

Table 5.2. Inventory data for the production of 1 kg of wheat grain in Chile, available at the farm-gate.

Product output	**Value**	**Unit**		
Wheat {CL} at farm gate	1	ha		
Inputs	**Value**	**Unit**	**Distribution**	**SD^2**
Materials/fuels				
Urea, as N {GLO}| market for	105.113	kg	Lognormal	1.624
Phosphate fertiliser, as P2O5 {GLO}| market for	116.14	kg	Lognormal	1.14
Potassium fertiliser, as K2O {GLO}| market for	67.1	kg	Lognormal	1.47
Transport, tractor and trailer, agricultural	3323.394	tkm	Lognormal	2.837
Pesticide, unspecified {GLO}| market for	17.07	kg	Lognormal	1.14
Wheat seed, for sowing {CL}| production	168.947	kg	Lognormal	1.176
Transport, freight, lorry 16-32 metric ton, EURO5 {RoW}	16.895	tkm	Lognormal	2.24
Emissions	**Value**	**Unit**	**Distribution**	**SD^2**
..to air				
Dinitrogen monoxide	12.9	kg	Lognormal	2.802
Ammonia	5.12	kg	Lognormal	2.631
Nitrogen dioxide	61.51	kg	Lognormal	2.799
..to water				
Nitrate	309.53	kg	Lognormal	1.58
Phosphorus	12.8	kg	Normal	5.811
Phosphorus	0.226	kg	Lognormal	1.58

5.3.1.3 Milling of feed ingredients derived from agronomic crop systems

The agronomic crops products are milled, or otherwise processed, to become the products as they are included within the salmon feed. The exception is wheat, which enters the feed production process without prior milling. The milling company and its location was provided as part of the data describing the feed production process, supplied by the manufacturer. Using information available on the internet, it was possible to learn some information about the milling technique used within the mill.

Both the sunflower-oil and rapeseed-oil used to make the salmon feed are supplied by the mill as crude oil, with no subsequent refining taking place after crude oil production. The production of crude sunflower oil and crude rapeseed oil produces sunflower-meal and rapeseed-oil, respectively. The ecoinvent V.3. database provides process data for the production of rapeseed-oil and rape-meal. Some data was found for the production of rapeseed oil, but these were limited, and there were insufficient data sources to generate a process using the horizontal averaging technique described in Chapter 4. For this reason, the process provided by the ecoinvent V.3. has been modified to represent Chilean production. This was done by replacing electricity inputs with electricity produced in Chile, and by using my own agricultural rapeseed production process. The value for the input quantity of rapeseed was modified to reflect differences in rapeseed moisture content between my own process for the production of rapeseed and the process provided by ecoinvent. The inputs and outputs of the milling process have been allocated between crude-oil and meal based upon their weight adjusted economic value (economic value was the price averaged over 5 years). The prices for these calculations were retrieved from https://www.fao.org/economic/est/prices. The ecoinvent database provides no process data (at the time of writing) for the production of sunflower-oil and meal. Limited data for the production of sunflower-oil and meal is provided by Spinelli et al. (2013), but as with rapeseed oil, an insufficient quantity of data was available for the horizontal averaging of data. As there was a lack of data defining the process for sunflower oil and meal production, and the data provided by Spinelli et al. (2013), appears to be based upon various literature sources, the ecoinvent V.3. process for rapeseed milling was adapted to represent the production of sunflower-oil and meal in Argentina. This was done by using the production ratio of sunflower-oil to meal, and the input quantity of sunflower seed, provided by an FAO document describing some technical aspects of the production of crude sunflower oil (Punda et al. 2010). Allocation was performed between the two co-products using the same method as for rapeseed milling, but with price data available from http://www.indexmundi.com/commodities (IndexMundi 2017).

Table 5.3. Inventory data describing the production of crude rapeseed-oil and the production of crude sunflower-oil. Both are modelled as being a product of seed milling, which produces both meal and oil. Inputs and outputs are assigned between meal and oil using mass-adjusted economic allocation. For rapeseed oil, ecoinvent V.3 (Wernet et al. 2016) has been used, but modified to reflect Chilean production. This ecoinvent V.3. process for rapeseed oil was also used to describe sunflower oil production, but was modified to reflect Argentinian production, and was further modified by allocating inputs and outputs between the sunflower ratio of oil to meal provided by Punda et al. 2010.

Data describing the processes involved in the milling of maize in the U.S.A. can be found in the Agri-footprint® LCA database, created by Blonk-Consultants. Usefully, Blonk Consultants provide a report

			Rapeseed	Sunflower
Products		**Unit**	**Value**	**Value**
Crude-oil		kg	1	1
Resources from 'nature'	**Compartment**	**Unit**	**Value**	**Value**
Water	in water	m3	5.30E-06	1.40E-05
Water, cooling	in water	m3	2.11E-05	5.60E-05
Carbon dioxide, in air	in air	kg	7.21E-01	1.91E+00
Materials/fuels		**Unit**	**Value**	**Value**
Seed input		ha	0.00022	0.00146
Hexane		kg	0.00015	0.00039
Activated bentonite		kg	0.00032	0.00083
Phosphoric acid, industrial grade		kg	0.00028	0.00075
Oil mill		p	0.00000	0.00000
Transport		**Unit**	**Value**	**Value**
Transport, freight, lorry 16-32 metric ton, EURO3		tkm	0.810	0.878
Transport, freight train		tkm		0.134
Transport, freight, inland waterways, barge		tkm		0.021
Electricity/heat		**Unit**	**Value**	**Value**
Electricity,		kWh	0.047	0.123
Heat, district or industrial, natural gas		MJ	0.095	0.252
Emissions	**Compartment**	**Unit**	**Value**	**Value**
..to air				
Water	high. pop.	m3	1.1E-05	2.8E-05
Carbon dioxide, biogenic	high. pop.	kg	7.3E-01	1.9E+00
Hexane	high. pop.	kg	1.5E-04	3.9E-04
..to water				
Water	-	m3	1.59E-05	4.22E-05
Waste to treatment		**Unit**	**Value**	**Value**
Wastewater		m3	3.64E-07	9.62E-07

detailing process data for the wet-milling of maize (van Zeist et al., 2012). This data was used for the process describing the production of maize gluten meal. Some errors within the flow of mass balance have been corrected, and the data quality pedigree scores for the generation of uncertainty values have been selected based upon information provided in the report, and based upon the literature sources (van Zesit et al., 2012) which provided the data for the Blonk report and Agri-footprint® processes. The agricultural production process that provides maize as an input to the milling process developed by Blonk-Consultants and features in the Agri-footprint® model, was replaced by my own process describing maize cultivation, that was developed for this purpose. Allocation between co-products was performed using price data available from www.fao.org/economic/est/prices (FAO 2017).

5.3.2. Production of feed ingredients derived from the agricultural production of animals

The average feed formulation consisted of the following ingredients derived from the agricultural production of animals:

- Feather meal
- Bird viscera meal

Both of these products are likely to be derived from poultry production. Data provided by the salmon feed manufacturer indicate that the products are obtained from, and produced in, the USA. The supplier of the poultry products controls all aspects of production, from the production of poultry feed, to the breeding and rearing of chickens, their processing, and the transport of products to a port for subsequent shipment.

5.3.2.1. Agricultural production of poultry

The modelled poultry agricultural system consists of two main stages:

- The breeding of chickens, producing chicks weighing 4 grams per individual. This process also produces chicken parents for slaughter and chicken eggs for human consumption.
- The rearing of chicks, producing broilers.

Both of these stages require a feed input. A variety of literature sources are available describing the ingredients used for poultry feed production. However, without being subject to prior manipulation, these data cannot be combined and subjected to the horizontal averaging procedure. The variety of alterative ingredients being used, and differing inclusion rates of each ingredient, creates an unbalanced feed formulation from a nutritional and mass balance perspective. For this reason, the generic poultry formulation described by Pelletier (2008) has been used. The generic poultry formulation consists of the following ingredients:

- Maize
- Soybean-meal
- Poultry ‘by-product’ meal
- Poultry fat
- Atlantic Menhaden (*Brevoortia tyrannus*) fishmeal
- Salt
- Limestone

Maize, soybean-meal, and menhaden fishmeal are all produced in the USA, and have been modelled accordingly. Soybean agricultural production has been modelled as described in Chapter 4 (also see above, 5.3.1.2). Data describing the production of soybean-meal is provided by the ecoinvent V.3. database, and has been modified by using my own process describing the agricultural production of soybean, for the input of soybean into the milling process. Two processes, one describing a menhaden capture fishery, and another describing the reduction of this fish into meal and oil, have been developed by combining data provided by Pelletier (2006), with the process data describing an anchoveta reduction fishery, which was used as of this current study. The production of processed poultry animal protein (used to represent poultry ‘by-product’ meal) and poultry rendered fat, are described below. The inventory for the generic chicken feed is shown in Table 5.4.

Table 5.4. Inventory data for the production of 1 kg of generic chicken feed, available at the farm gate. The inventory is based upon that provided by Pelletier (2006).

Product output	**Value**	**Unit**		
Generic Chicken Feed, at farm gate {U.S.}	1	kg		
Inputs	**Value**	**Unit**	**Distribution**	**SD^2**
Materials/fuels				
Maize grain, at farm-gate, U.S.	7.5148E-05	ha	Lognormal	1.13
Soybean meal {US}	0.2	kg	Lognormal	1.13
Poultry processed animal protein {US}	0.025	kg	Lognormal	1.13
Poultry rendered fat {US}	0.025	kg	Lognormal	1.13
Fishmeal, menhaden {US}	0.025	kg	Lognormal	1.13
Limestone, crushed, washed {GLO}| market for	0.0125	kg	Lognormal	1.13
Sodium chloride, powder {GLO}| market for	0.0125	kg	Lognormal	1.13
Transport of feed ingredients for chicken feed (incl. uncertainty)	1	kg	-	-
Electricity/heat				
Electricity | market for	0.137	MJ	Lognormal	1.13
Heat, natural gas {US} at boiler modulating >100kW	0.28224425	MJ	Lognormal	1.13
Emissions	**Value**	**Unit**	**Distribution**	**SD^2**
..to water				
Phosphorus	0.00344828	mg	Lognormal	1.52
Nitrogen dioxide	0.0012069	mg	Lognormal	1.52
Nitrate	30	mg	Lognormal	1.52
COD, Chemical Oxygen Demand	8.96551724	mg	Lognormal	1.52

5.3.2.2. Poultry processing

Only limited data describing poultry processing could be found. The most comprehensive data for these processes was available from Ramirez (2012). This study provides process data for the conversion of broilers into various co-products, and for the subsequent conversion of some of these co-products into rendered and hydrolysed products. The data describes processes within the UK, and was adapted to be representative of production in the USA, by using country specific inputs (such as energy produced in the USA) and through selecting appropriate data quality pedigree scores. The product ‘processed poultry animal protein’ was used to represent the salmon feed ingredient, ‘bird viscera meal.’ The product ‘hydrolysed feather meal’ was used to represent the salmon feed ingredient ´feather meal.’ As inputs to the generic chicken feed formula, ‘processed poultry animal protein’ has been used to represent ‘poultry by-product meal,’ and ‘rendered poultry fat’ was used to represent ´poultry fat.’

5.3.3. Capture fisheries and the reduction of fish into meal and oil.

Primary data regarding capture fishery was kindly provided by Angel Avadi. These data cover processes relating to the fishing activities of a Peruvian fishing fleet, which captures Peruvian anchoveta (*Engraulis ringens*), and land-based activities taking place at the harbour. It also provides data for the reduction of anchoveta to fishmeal and oil. As the reduction facility for the modelled feed diet is situated at a localtion in Chile close the border with Peru, and because the fish used by reduction facilities in Chile come from both of these countries, the use of these data was considered to be acceptable. This decision was made easier by the lack of up-to-date, representative data from South American capture fisheries. Efforts to obtain data from Chilean capture fisheries only yielded basic data from some small, artisanal fishing boats which are not used to capture small pelagic species.

5.3.4. Other ingredients

Additional to ingredients derived from agronomic and animal production systems, there are three further ingredients in the salmon feed formula. These are:

- Vitamin and mineral premix
- Water
- Reprocessed item

The vitamin-mineral premix is added to salmon feed to complete the nutrition profile of the diet. There is a lack of available information detailing the precise ingredients of such mixes, and there are a range of premixes and possible ingredients that may be used. Considering that a process describing the production of such a product will be time consuming to compile and create, and that the premix has the lowest percentage contribution to the diet formulation in terms of mass (<2%), the vitamin/mineral premix was not included in the LCA. It is considered unlikely that this will have anything more than a marginal influence upon the results of the feed model (or other models of which salmon feed in an input).

The source of this 'reprocessed item', and, indeed, what this input is actually supposed to be, is uncertain. However, information supplied with the data provided by the feed manufacturer, suggests that this ingredient is actually particles of feed or 'fines' generated by the feed production process itself. Only a small amount of this product is included as an ingredient. Rather than create a loop, or otherwise omit the ingredient from the feed production process entirely, the small quantitative contribution made to the mass balance of the feed product has been represented by increasing the amount of the vegetable oil included in the formula, by the mass of the 'reprocessed item' (50% of this mass is added to rapeseed oil, and 50 % is added to the sunflower oil).

5.4. Impact Assessment

As can be seen in Figure 5.4, the 'vegetable oil' ingredient (rapeseed-oil and sunflower-oil) dominates all but one of the impact categories, the exception being 'ozone layer depletion,' to which feather meal provides the largest of contributions. Sunflower oil provides the largest share of the contributions from vegetable oil towards each impact category, with one exception being 'eutrophication,' for which it is rapeseed oil that has the largest share. Immediately, we are presented with some interesting results that have potential implications for the sustainability of salmonid feed formulations. In terms of contributions, the production of feather meal[15] follows in second place behind the production of vegetable oil, for most of the impact categories. For the impact category 'eutrophication,' the contributions from feather meal are surpassed by those from wheat production, and wheat has quite a visible contribution to some other categories, such as 'abiotic depletion' and 'marine ecotoxicology'.

[15] It is important to remember that when the contributions from the production of feather meal are being discussed, in addition to the contributions from the production process of feather meal itself, the contributions are also the sum of those from all processes which occur upstream of the actual process of feather meal production, and which supply functions (products or services) required for the production of feather meal. This same situation is true for all other process included in the production of salmon feed, such as sunflower oil.

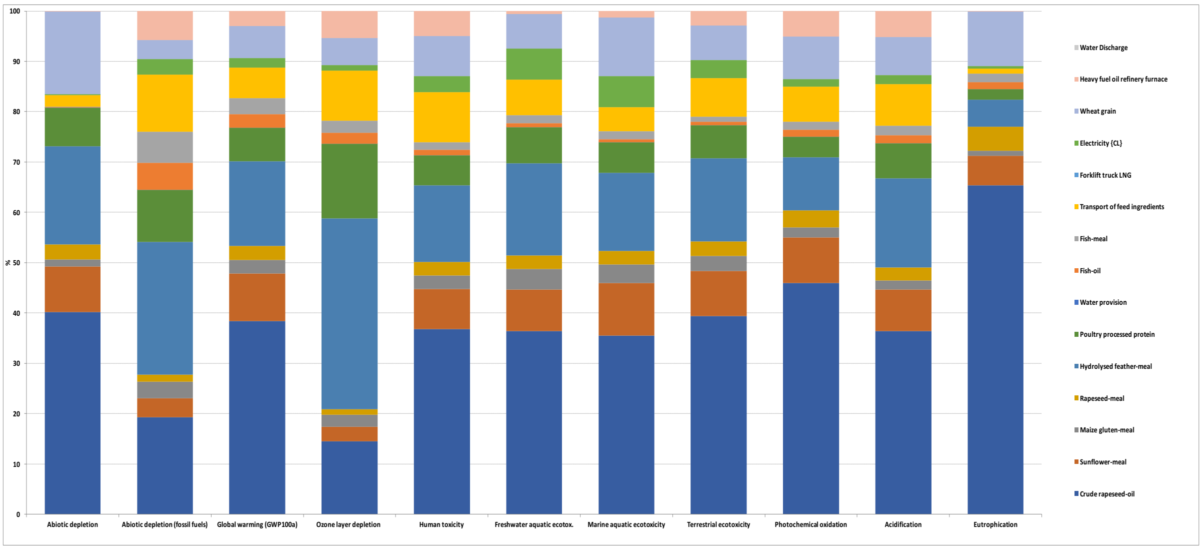


Figure 5.4. Characterisation model results for the generic formula salmon feed, showing the contribution of each process to each impact category, expressed as a percentage of the total contribution from all processes with a given impact category, calculated using the CML-IA-baseline method V3.03.

Grouping processes into ingredient types, as has already been done by grouping rapeseed oil and sunflower oil into the ingredient type 'vegetable oil' is a useful way to analyse the environmental impact profile of salmon feed. Figure 5.5. shows the characterisation model results for salmon feed, with the different ingredients having been grouped based upon the source type of the ingredient. These types are 'agronomic crop agriculture,' 'animal agriculture,' and 'capture fisheries.' Other inputs, such as energy carriers, or water, and which are not ingredients of the feed formula, have not been put into any groups, although their contributions are part of the assessment. By looking at the chart, it can be clearly seen that 'agronomic crop agriculture' has the largest contribution to all impact categories, apart from 'ozone layer depletion,' to which 'animal agriculture' has the largest contribution (accounting for 49.9 % of contributions to this category). If both agronomic crop production and animal agriculture production are considered together, they by far account for the majority of contributions to all impact categories. Based upon these results, it can be said that the relative contribution of agriculture towards the impacts of salmonid feed production, are considerable.

Interestingly, the transportation of feed ingredients from the farm gate up until the feed production facility (their combined contributions are considered rather than analysing the contribution of transport of ingredient separately), has a greater contribution than the production of capture fishery ingredients, towards the majority of impact categories. In the category 'fossil fuel depletion,' the contributions from capture fishery ingredients slightly exceed those from ingredient transport, whereas they considerably exceed those of ingredient transport in the category 'eutrophication.' Towards the category 'global warming potential' (GWP100), both capture fishery ingredients and transportation of ingredients offer the same contribution (contributing 4.7 % each). In the case of some impacts, the burning of heavy fuel within the feed production facility has a greater contribution than capture fishery ingredients, as does sometimes the use of electricity.

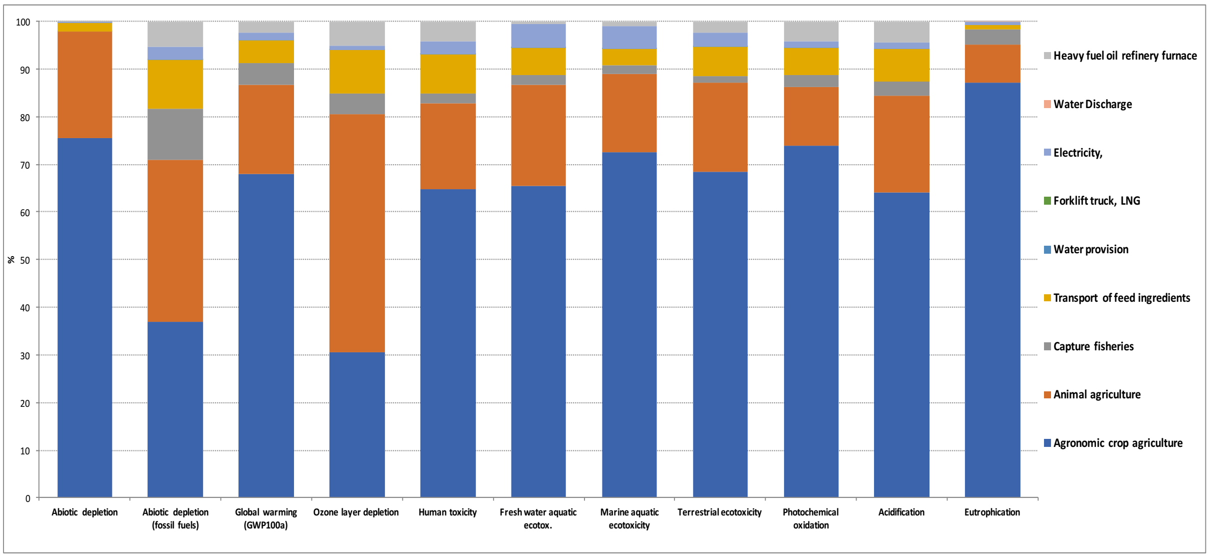


Figure 5.5. Characterisation model results for the generic formula salmon feed, showing the contribution of the ingredient groups 'agronomic crop agriculture,' 'animal agriculture,' and 'capture fisheries,' with inputs and outputs not in these categories remaining separate. Calculated using the CML-IA-baseline method V3.03.

5.4.2. Agronomic crop agriculture

Due to the significant contribution of agricultural products towards the environmental impacts of salmon feed, it seems sensible to determine which particular ingredients account for the majority of these contributions. The impacts of ingredients derived from agronomic crop production are shown in Figure 5.6, where they are presented as percentage of the total contributions towards impacts delivered by ingredients from agronomic crop production only. From this set of pie charts, it is clear that crude sunflower oil has an obviously greater contribution towards impacts than all other ingredients from agronomic crops, when they are considered individually. This is true for all impact categories, except for eutrophication, towards which the alternative vegetable oil, crude rapeseed oil, has the greatest share of contributions, delivering 8 % more contributions than crude sunflower oil. Notably, crude sunflower oil accounts for 52 % of contributions to GWP100 and 52 % of those to marine aquatic ecotoxicology. In other words, for both of these impact categories, crude sunflower oil is accountable for more contributions than all of the other ingredients from agronomic crops combined. For both photochemical oxidation and acidification, crude sunflower oil accounts for a 50 % share of contributions, and for all other impact categories apart from eutrophication, it accounts for between 41 % and 49 % of contributions.

5.4.2.1. Vegetable-oils

The significant contribution of crude sunflower-oil towards the combined impacts of agronomic crop derived feed ingredients, as well as its significant contribution to the production of feed itself, raises important questions that must be explored. These questions are:

- 'what processes associated with the production of crude sunflower oil (and also of crude rapeseed oil) produce the majority of contributions towards impacts?'
- 'What is (or are) the source(s) of the differences in the relative quantities of contributions to impacts between crude sunflower oil and crude rapeseed oil?'

'Are these differences realistic or accurate?'

When observing the impact characterisation profile of crude sunflower oil production and crude rapeseed oil production, it is very clear that the significant majority of contributions towards impacts come from the input of the seed crop to the milling and extraction process (data not shown). For crude sunflower oil, the input of sunflower seed accounts for between 77.5 % and 99.2 % of contributions across all impacts. For crude rapeseed oil, the input of rapeseed accounts for between 62.6 % and 99.1 % of contributions across all impacts. The same data was used to describe the milling and extraction process for both sunflower and rapeseed oil, albeit with differences in oil and meal extraction ratio between the two oil products (oil and meal being the co-products of the milling and extraction process), the allocation values derived from the mass adjusted economic value of oil and meal (because the economic value of sunflower oil and rapeseed oil differs), and differences between the amount of crop / seed input required to produce the two oil types. To see if the likely cause is the differences in the combined effect of the oil : meal extraction ratio and the mass adjusted economic allocation factors, 50 % of contributions were allocated to each co-product (oil and meal) for both crude sunflower oil and crude rapeseed oil. This eliminates any causative effect that may occur as a result of differences between the extraction ratio and allocation factors. The characterised impact profile was then compared between the equal amount of produced sunflower oil and rapeseed oil. As was the case before any changes to allocation were made, the contributions from crude sunflower oil are observably higher than from oil from rapeseed, towards all impacts are from the category 'eutrophication,' and the differences between them follow the same general pattern (data not shown). These results permit the easy conclusion, that the source of differences in contributions between the two oil products are highly unlikely to be associated with the differences in the oil : meal

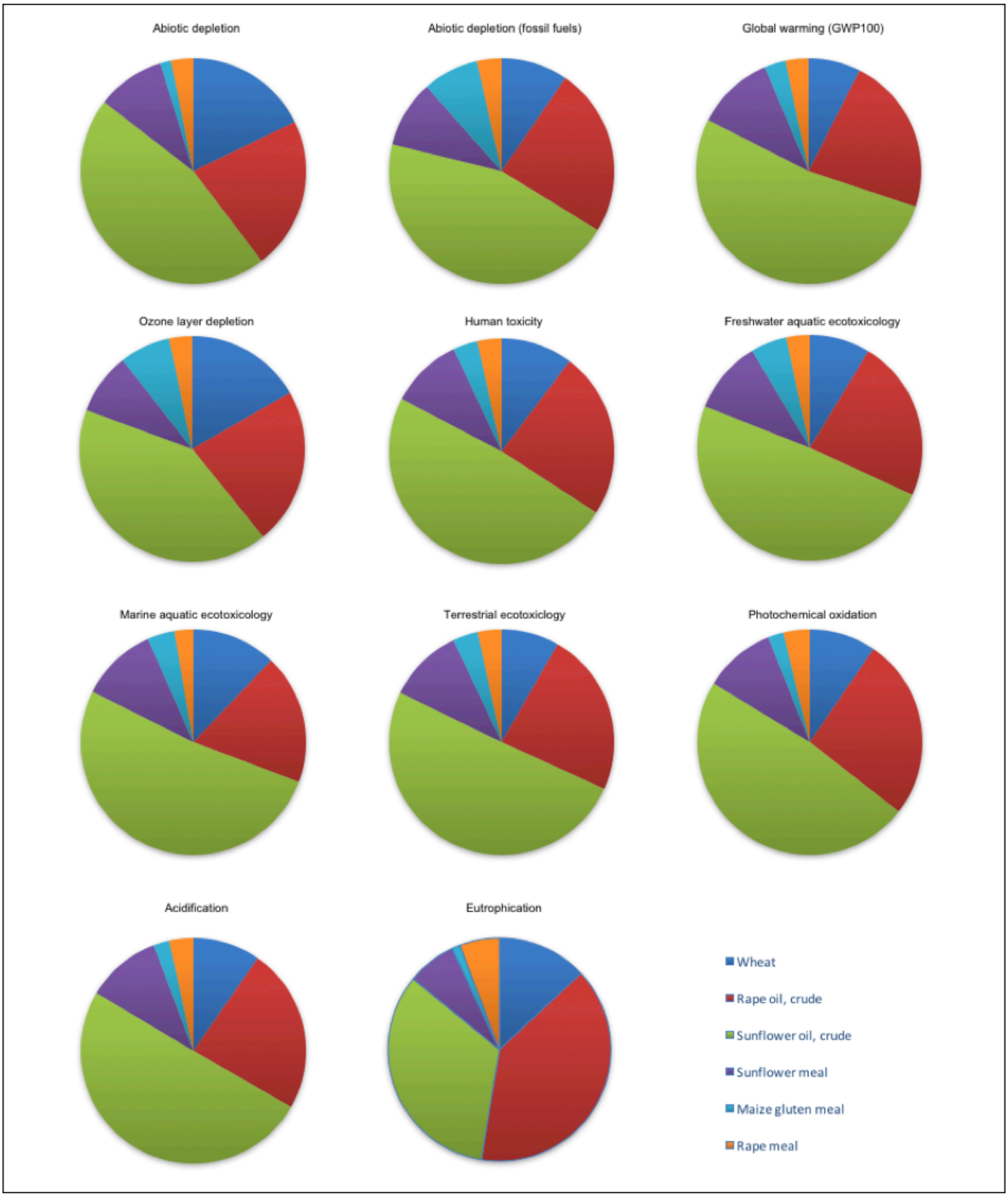


Figure 5.6. The percentage contribution of each individual agronomic crop ingredient towards the total contributions of all agronomic crop ingredients to each impact category. Calculated using Calculated using the CML-IA-baseline method, V3.03.

extraction ratio and the respective allocation factors between oil from sunflower and oil from rapeseed. The more likely source of differences in the relative contributions is the different quantities of seed inputs to each oil type. The sunflower seed input value for sunflower oil is larger than the input of seed for rapeseed oil (Table. 5.3). To further understand the consequences of the difference in seed input, sunflower oil was given the same input quantity of sunflower seed, as the rapeseed input to rapeseed oil.[16] The results are interesting, with the general pattern of differences between the relative contributions of both oil types across each impact category being markedly different (Figure 5.7). With both sunflower and rapeseed oil now having the same quantity of seed input required to produce 1 kg of oil, rapeseed oil has a greater contribution to all impacts than does sunflower oil, apart from the categories GWP100 and marine aquatic ecotoxicology. Interestingly, the contribution of rapeseed oil

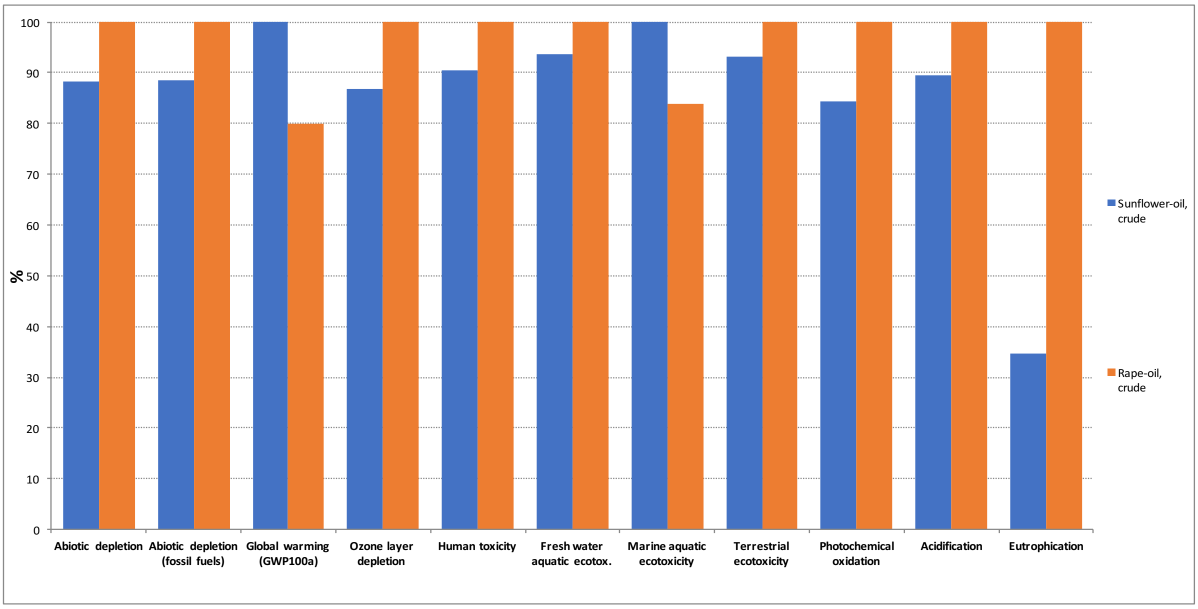


Figure 5.7. Comparison between the production of an equal quantity of crude sunflower-oil and crude rapeseed-oil, when the quantity of seed input to the extraction process is the same for both products. Calculated using the CML-IA-baseline method, V3.03.

to the impact category eutrophication, is still higher than that of sunflower oil (it was also higher before the change to seed input was made, (see Figure 5.6), except the relative difference in contributions is greater still.

This strongly suggests that differences between the agricultural production of rapeseed and that of sunflower seed, may be somewhat responsible for the notable difference in contributions between the two oil products. For this reason, the impacts of sunflower seed available at the farm gate (i.e. the

[16] Technically, this results in a mass balance error, as the extractable oil and meal content are different between the equivalent quantity of sunflower seed and rapeseed. However, the analysis is still appropriate to test the sensitivity of results to the input quantity of seed from agricultural production.

agricultural production of sunflower seed) have been compared to those of the same quantity of rapeseed available at the farm gate (Figure 5.8). In every case except eutrophication, the contributions to impacts are higher for sunflower seed than rapeseed (between 45.7 % and 63.9 % greater across impact categories).

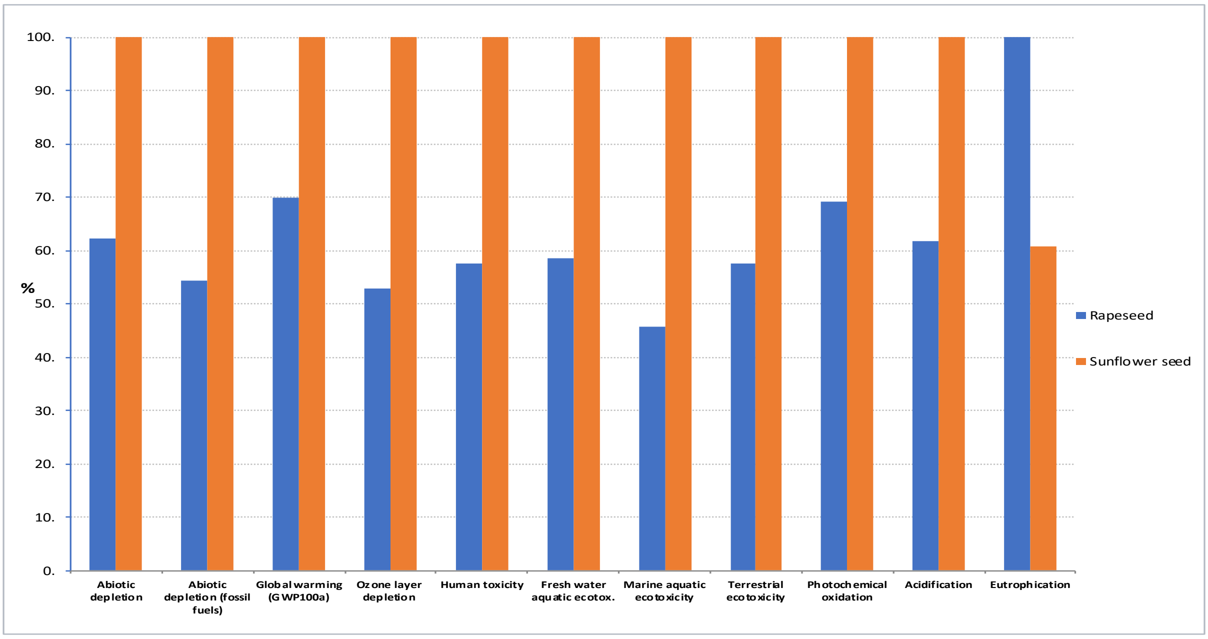


Figure 5.8. Comparison between the production impacts of 1 kg of sunflower seed and 1kg of rapeseed. It shows the lowest contribution between the two seed types as a percentage of the highest contribution to each category. Calculated using the CML-IA-baseline method, V3.03.

Finally, it is important to know if the differences in contributions between rapeseed production and sunflower seed production are reflective of the real-world situation. Investigating this question could easily turn into a detailed and time-consuming procedure that somewhat exceeds the scope of this thesis. However, it is possible to shed some light upon the answers rather than completely avoiding the question. The uncertainty of the data that describes the processes required for production of rapeseed as it is available at the farm gate can be compared to those for the production of sunflower seed. To do this, the Monte Carlo method has been used. For each product, a series of simulation runs are performed, with each run calculating a characterised impact assessment for each product using a randomly selected data point from the range of possible values determined by the uncertainty score (e.g. $\sigma_g{}^{\wedge 2}$), for each value of the inputs and emissions of a process (the generation of these uncertainty values is described in Chapter 4. In this analysis, 1000 runs have been performed, and the amount of times (expressed as a percentage of the total number of runs) that rapeseed has higher contributions than does sunflower seed, and also the amount of times that sunflower seed has higher contributions than does rapeseed, is recorded for each impact category, and is presented in Figure 5.9. It is important to note, that for any given characterised impact category, the closer towards an equal frequency of each product having greater contributions than the other, the less reliable is the characterisation result. In other words, if the Monte Carlo method returns a result of sunflower seed as having a greater contribution towards a particular impact for 50 % of the total simulation runs, and so, logically, also returns a result of rapeseed as having a greater contribution for the other 50 % of runs, the results of the original characterised impact assessment will be very unreliable. This is because the probability of one product having greater contributions than the alternative is 0.5, the same chance of 'heads' landing face up when a coin is flipped. The chart in Figure 5.9 clearly shows that for all impacts except eutrophication, sunflower seed having a greater contribution than does rapeseed is the result for the vast majority of runs. For the impact category eutrophication, rapeseed has the largest amount of contributions for almost every run (98.9 % of 1000 runs). These results are of some encouragement, because not only do they mirror the profile of the original characterised impact assessment, they do so whilst delivering this pattern with a frequency of ≥ 95 % for 8 out of 10 impacts,

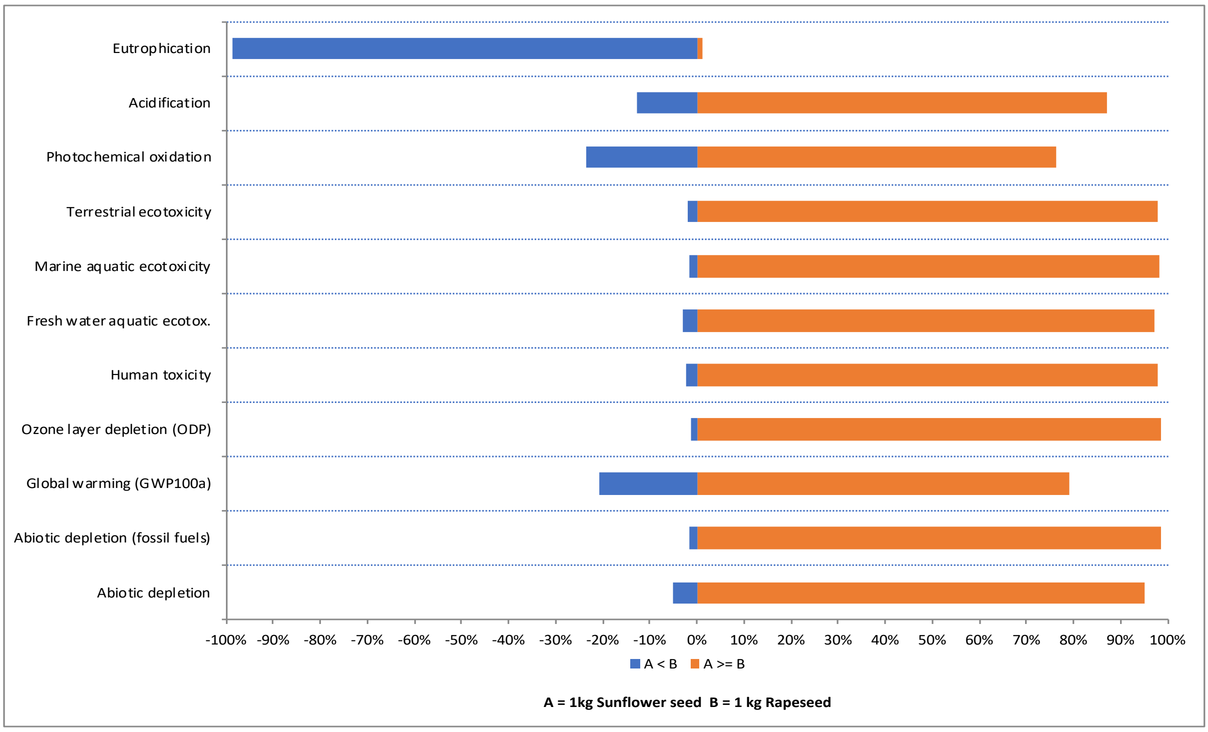


Figure 5.9. Monte Carlo based uncertainty analysis of the comparison between the impacts of 1 kg of sunflower seed at farm gate (A) and the impacts of 1 kg of rapeseed at farm gate (B). A total of 1000 runs have been performed, with each calculating the characterised impact assessment of both products. The chart shows the frequency of A ≥ B and A < B, for each impact category.

indicating a significant difference. Those categories falling below the 95 % level of significance (i.e. acidification, photochemical oxidation, GWP) still return a result of sunflower seed having the most contributions in 76 % to 86 % of runs, although the difference cannot be said to be highly significant.

Despite the results of the Monte Carlo simulation being encouraging, there is at least one further source of possible uncertainty that may reduce the reliability of the characterised impact assessment results. This could be described as 'uncertainty within the uncertainty,' and it is an issue that I feel is particularly challenging to the handling of uncertainty in LCA. It is possible that the uncertainty values generated by the combination of pedigree matrix scores and basic uncertainty values, do not meaningfully capture the uncertainty of the mean value used to describe an input or output. If the uncertainty values assigned to each input or output value are very unequal in their accuracy, the results of the Monte Carlo simulation itself will be very uncertain. The problem is that, currently, there is no method developed specifically for analysing or reducing this source of uncertainty. However, thoughtful use of the pedigree matrix methods of propagating uncertainty suggests that there is indeed a real scope for the existence 'uncertainty within the uncertainty.' This is discussed with more detail in Chapter 10 'final discussion and conclusions'.

In the very specific case of sunflower oil and rapeseed oil, it must be acknowledged that some, quite large, assumptions have been made. Crude sunflower oil has been assumed to have been extracted using the same process as crude rapeseed oil, and the choice of describing the vegetable oil ingredient to of the feed formula as being a 50 : 50 blend of rapeseed oil and sunflower oil, is also an assumption. Pelletier et al. (2009), do not include sunflower oil as an ingredient to Chilean salmon feed production, rather they include soybean oil in addition to rapeseed. Anecdotal observation (authors observations), suggests that rapeseed is a common ingredient to salmon feed, and is used in Chilean production[17]. Rather than assume the use of a vegetable oil (sunflower) with a higher impact profile than an alternative product (rapeseed), it is perhaps more justifiable to assume that only crude rapeseed oil is used. As it is my understanding that rapeseed is a common ingredient, whereas I am less sure of the frequency of sunflower oil (or soybean oil) being used, I am certainly more comfortable with the choice of assuming rapeseed oil as being the only vegetable oil ingredient. Although I acknowledge this is hardly a scientifically derived choice, it is an unfortunate reality that assumptions within LCA are sometimes made on an almost arbitrary basis when a lack of information is available. The large amount of data required, and the large number of decisions that must be made, means that such circumstances are not entirely uncommon. From this point onwards, rapeseed crude oil will be assumed to be the only vegetable oil ingredient. The consequent results of the impact assessment will now be discussed, within the context of the effects of replacing crude sunflower oil with crude

[17] In reality, it may well be that the vegetable oil used is a variable mixture of rapeseed, sunflower, and soybean oil, that changes as a result of price and availability.

rapeseed oil. This type of analysis is similar to those undertaken during detailed life cycle assessments within an industrial setting, although this current analysis is more basic to avoid an excessive deviation from the scope of the project.

5.4.3. Salmon Feed with crude rapeseed-oil as the only vegetable-oil ingredient

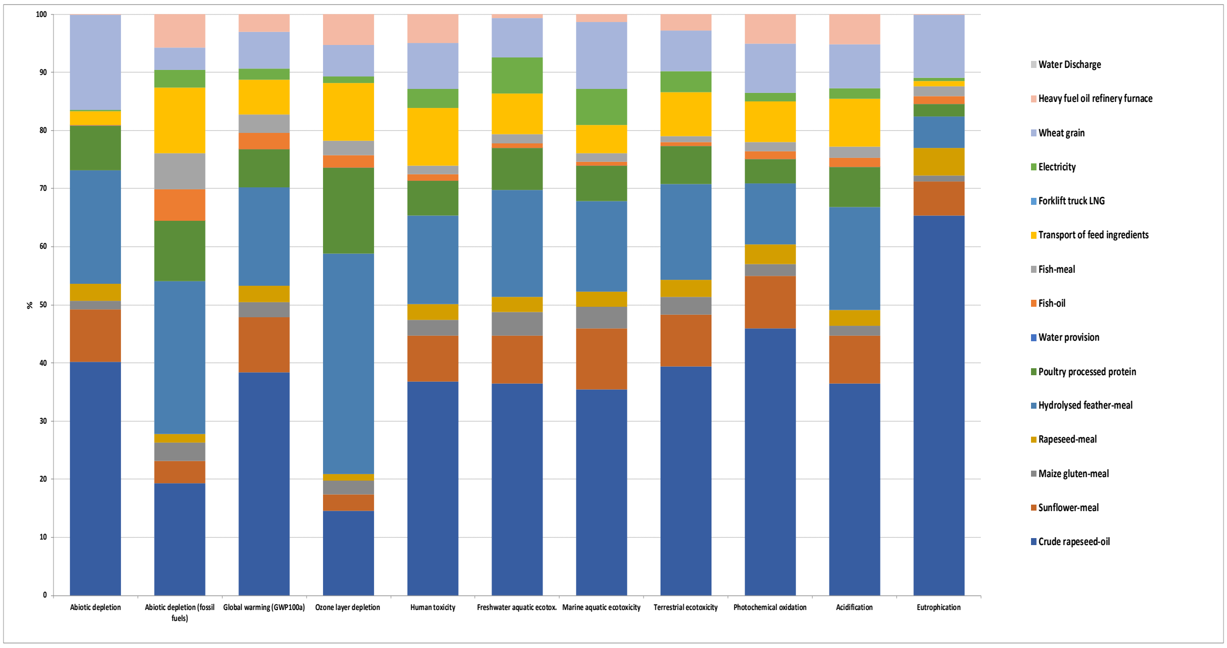


Figure 5.10. Characterisation model results for 1 kg of generic formula salmon feed, with crude rapeseed-oil as the only vegetable oil used, showing the contribution of each input and output for towards each impacts category, calculated using the CML-IA-baseline method V3.03.

The characterised impact assessment for salmon feed with crude rapeseed-oil as the sole vegetable-oil is shown in Figure 5.10. The general profile of contributions of each ingredient towards the impacts is similar to that of the impact assessment for salmon when the vegetable oil is a blend of both rapeseed and sunflower crude oil. However, as a consequence of inherent limitations to the amount of information these charts can convey, in addition to a complex presentation of the information that is contained, comparison between the two profiles is difficult without presenting the data in different ways. The total contributions towards each impact category from salmon feed with sunflower oil and feed without sunflower oil, as well as the differences between them, are presented Table 5.5. This shows that the contributions are lower when rapeseed is the only vegetable oil ingredient, apart from contributions to the category ‘eutrophication’. This is to be expected, because rapeseed oil has, according to this current model, lower contributions than does sunflower oil, towards all impact categories except eutrophication (Figure 5.8).

The information is presented differently in Figure 5.11. It shows the difference between the total contributions from feed both with and without sunflower oil, by displaying the lowest contribution from the feed types as a percentage of the highest contribution to each category. To give an example, feed including sunflower oil as an ingredient has a greater contribution towards global warming than does feed absent in sunflower oil (2.49 and 1.99 kg CO_2 eq. respectively). In the chart, the comparison is made by presenting the contribution from feed without sunflower-oil as a percentage of the contribution from feed with sunflower oil (79.9 % and 100 % respectively).

Table 5.5. The quantity of contributions from feed with including sunflower oil and from feed not including sunflower oil, towards each impact category, and the change in contribution that occurs as a result of using rapeseed as the only vegetable oil, with each change being expressed as a percentage of the total contribution from feed with both sunflower and rapeseed oil.

Impact category	Unit	Total (Feed incl. sunflower oil)	Total (Feed not incl. Sunflower oil)	Difference in total impact (%)
Abiotic depletion	kg Sb eq	7.25E-06	5.94E-06	-18.071
Abiotic depletion (fossil fuels)	MJ	17.592	16.283	-7.439
Global warming (GWP100a)	kg CO2 eq	2.494	1.993	-20.069
Ozone layer depletion	kg CFC-11 eq	2.24E-07	2.11E-07	-5.551
Human toxicity	kg 1,4-DB eq	0.5	0.421	-15.811
Fresh water aquatic ecotox.	kg 1,4-DB eq	0.272	0.226	-16.794
Marine aquatic ecotoxicity	kg 1,4-DB eq	1175.89	894.73	-23.91
Terrestrial ecotoxicity	kg 1,4-DB eq	0.003	0.002	-18.297
Photochemical oxidation	kg C2H4 eq	0.001	0.001	-16.342
Acidification	kg SO2 eq	0.019	0.015	-17.003
Eutrophication	kg PO4--- eq	0.0166	0.0175	5.444

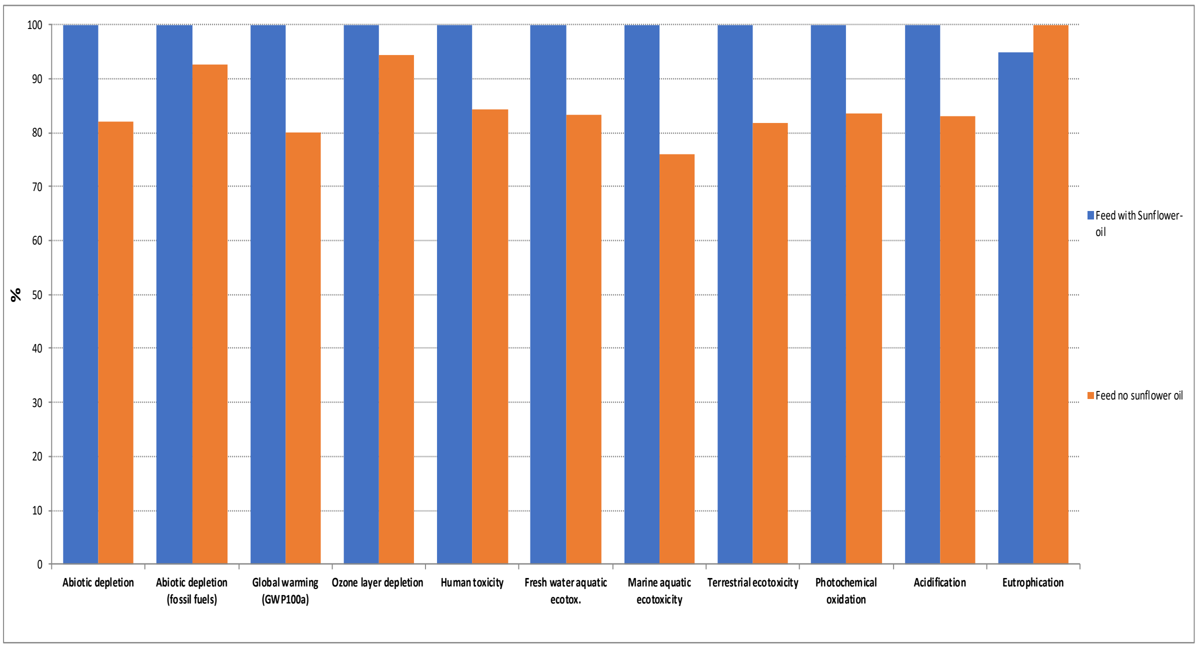


Figure 5.11. Comparison between total potential contributions from feed with and feed without sunflower oil. calculated using the CML-IA-baseline method V3.03.

Table 5.6. The change in contribution from each input and output as a result of eliminating the inclusion of crude sunflower-oil and doubling the inclusion rate of crude rapeseed-oil. Expressed as the difference in percentage contribution towards a specified impact category, from feed not including sunflower oil, relative to the contribution from feed including sunflower-oil.

Impact category	**Difference in percentage contribution from specified input/output of feed not incl. Sunflower oil**														
	Rapeseed oil, crude	Sunflower meal	Maize gluten meal	Rapeseed meal	Feather meal	Poultry meal	Water	Fish oil	Fish meal	Transport of	Wheat	Forklift truck	Electricity	Steam boiler	Water Discharge
Abiotic depletion	+23.746	+1.625	+0.259	+0.530	+3.532	+1.387	+9.03E-05	+0.014	+0.014	+0.494	+2.970	+4.02E-13	+0.028	+0.014	na.
Abiotic depletion (fossil fuels)	+10.369	+0.285	+0.239	+0.105	+1.962	+0.771	+5.91E-05	+0.402	+0.459	+1.111	+0.284	+1.87E-11	+0.228	+0.426	na.
Global warming (GWP100a)	+23.078	+1.900	+0.521	+0.563	+3.386	+1.330	+1.16E-04	+0.545	+0.635	+1.323	+1.273	+2.31E-11	+0.386	+0.606	na.
Ozone layer depletion (ODP)	+7.686	+0.159	+0.131	+0.059	+2.104	+0.827	+7.94E-05	+0.118	+0.136	+0.795	+0.301	+5.08E-12	+0.058	+0.297	na.
Human toxicity	+21.290	+1.259	+0.427	+0.424	+2.415	+0.948	+1.80E-04	+0.164	+0.243	+1.771	+1.255	+9.51E-12	+0.503	+0.785	na.
Fresh water aquatic ecotox.	+21.289	+1.381	+0.683	+0.447	+3.080	+1.210	+8.85E-04	+0.130	+0.271	+1.355	+1.147	+6.38E-12	+1.033	+0.104	na.
Marine aquatic ecotoxicity	+21.991	+2.508	+0.888	+0.619	+3.718	+1.460	+4.57E-04	+0.149	+0.372	+1.212	+2.764	+8.07E-12	+1.485	+0.320	na.
Terrestrial ecotoxicity	+23.281	+1.646	+0.552	+0.526	+3.026	+1.188	+6.63E-04	+0.130	+0.180	+1.536	+1.271	+5.05E-12	+0.655	+0.521	na.
Photochemical oxidation	+26.733	+1.482	+0.323	+0.548	+1.725	+0.678	+8.14E-05	+0.222	+0.269	+1.197	+1.382	+1.69E-11	+0.241	+0.827	na.
Acidification	+21.305	+1.403	+0.300	+0.452	+3.010	+1.182	+6.73E-05	+0.262	+0.327	+1.455	+1.270	+7.97E-12	+0.314	+0.888	na.
Eutrophication	+30.914	-0.319	-0.054	-0.260	-0.293	-0.115	-6.79E-06	-0.076	-0.090	-0.040	-0.587	-9.13E-14	-0.030	-0.007	-3.56E-04

It is also possible to compare the differences between the two scenarios from the perspective of their relative inputs and outputs. Table 5.6. shows how the contribution from each input and output changes as a result of eliminating the inclusion of sunflower oil. By obvious necessity, the actual *unit quantity of a contribution* towards the total of an impact category (e.g. the quantity of emission of CO_2 equivalents contributing to global warming potential), will double for crude rapeseed oil, simply because the quantity of rapeseed oil has been doubled in the absence of sunflower oil. Also by necessity, the unit quantity of each contribution type will stay the same for all other inputs and outputs, because their rate of inclusion per unit quantity of feed produced remains unchanged[18]. What does change is the relative contribution of all inputs and outputs *as a proportion of the total contributions* towards each impact. If an emission from a particular ingredient remains exactly the same for both feed with and without sunflower oil, its proportional contribution to any impact category will still change because the total quantity of emissions towards each impact has changed. As a consequence of using only rapeseed oil to fulfil the required quantity of vegetable oil, the contributions of all inputs and outputs increases as a proportion of the total contributions towards each impact category. Predictably, the exception is eutrophication, to which the proportion of contributions are lower (although the proportion contributions of rapeseed oil are higher, because the quantity of rapeseed has doubled).

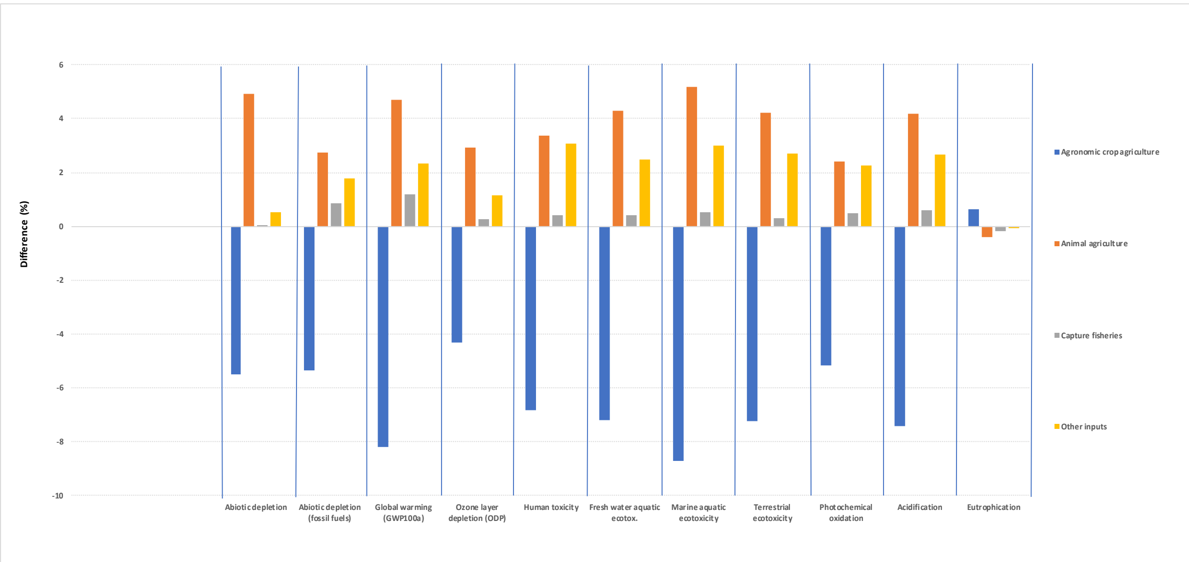


Figure 5.12. The change in contributions of the grouped input types to the total of each impact category when crude sunflower-oil is replaced by rapeseed-oil. Expressed as the percentage difference, relative to feed including sunflower-oil.

[18] The only exception to this rule is that of contributions associated with transport. The 'quantity' of the transport from oil mill to feed mill doubles as the quantity of rapeseed oil being used has doubled. The quantity of transport for sunflower oil is removed when sunflower oil as an ingredient is also removed. However, the contributions from transport do not remain unchanged, because the quantity of transport inputs differ between those required for delivering rapeseed oil and those for sunflower oil.

As a last comparison, the inputs and outputs have been grouped by type. This is probably easier to digest than when comparing each input and output separately, and it has been presented as a bar chart in Figure 5.12. As is to be expected, the largest difference in proportional contributions is exhibited by ingredients derived from agronomic crops, because this is the category into which vegetable-oil has been grouped.

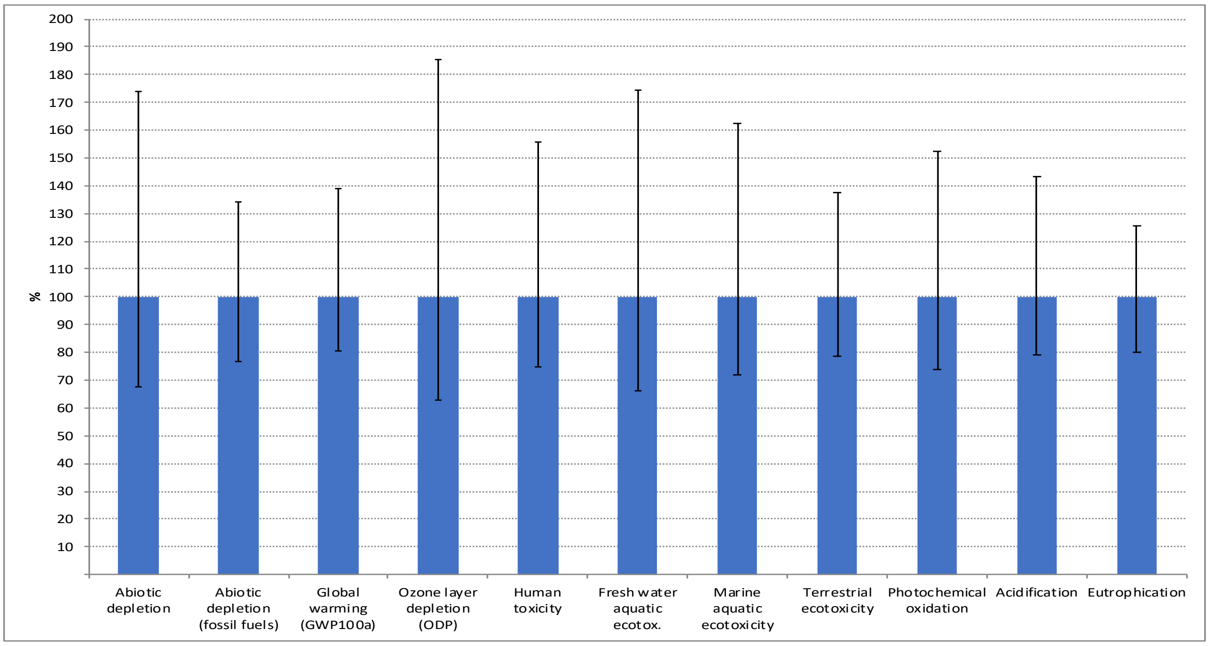


Figure 5.13. Monte Carlo uncertainty analysis of the characterised impact assessment results for salmon feed with rapeseed-oil as the only vegetable oil. The error bars show the upper and lower values of the 95 % confidence interval.

Finally, an uncertainty analysis has been performed upon the characterisation results of salmon feed with crude rapeseed-oil only, using the Monte Carlo technique. The results of this assessment are shown as a bar chart in Figure 5.13, with errors bars representing the uncertainty as the 95 % confidence interval for the total value (i.e. the total contributions) of each impact category. As can be realised from this chart, the uncertainty ranges are quite large, with the three largest 95 % confidence intervals being for the categories 'ozone layer depletion,' freshwater ecotoxicology' and 'abiotic depletion.' Large uncertainty ranges are common in LCA. There are a large number of input values, many, if not the majority, of which, are derived from sources which do not perfectly describe the intended input or output. The various sources of uncertainty that are measured and expressed numerically (providing uncertainty values as a basis for Monte Carlo analysis), combined with the large number of upstream processes (and therefore values) that are required for the eventual production of salmon feed (or any other product being analysed), makes possible the potential for large uncertainty ranges. It is preferable that efforts should be made to measure and report uncertainty surrounding the results of an LCA, but it should be done so with the goal of reducing such uncertainty when the results are deemed important. Reducing uncertainty for a large project, such as the current assessment, will likely be a resource consuming task, usually taking place as part of the iterative process innate to LCA.

5.4.4. Fish-oil compared to crude-rapeseed-oil

The result of vegetable oil having a greater contribution to the impacts of the feed formulation than does fish-oil appears important. It is certainly important if it is a realistic outcome, especially when considering the negative attention that the use of fishery ingredients has attracted (e.g. Naylor et al. 2009), and efforts to replace fish-oil with terrestrial plant-based alternatives. For this reason, rapeseed oil and fish oil have been compared on the basis of equal-weight (Figure 5.14). The comparison between these two oil types is an imperfect one. Both oils have a different nutritional profile, and, as such, do not offer alternative functions if substituted on the basis of weight (see section 5.5. Discussion and Conclusions), although it is still useful to make the comparison.

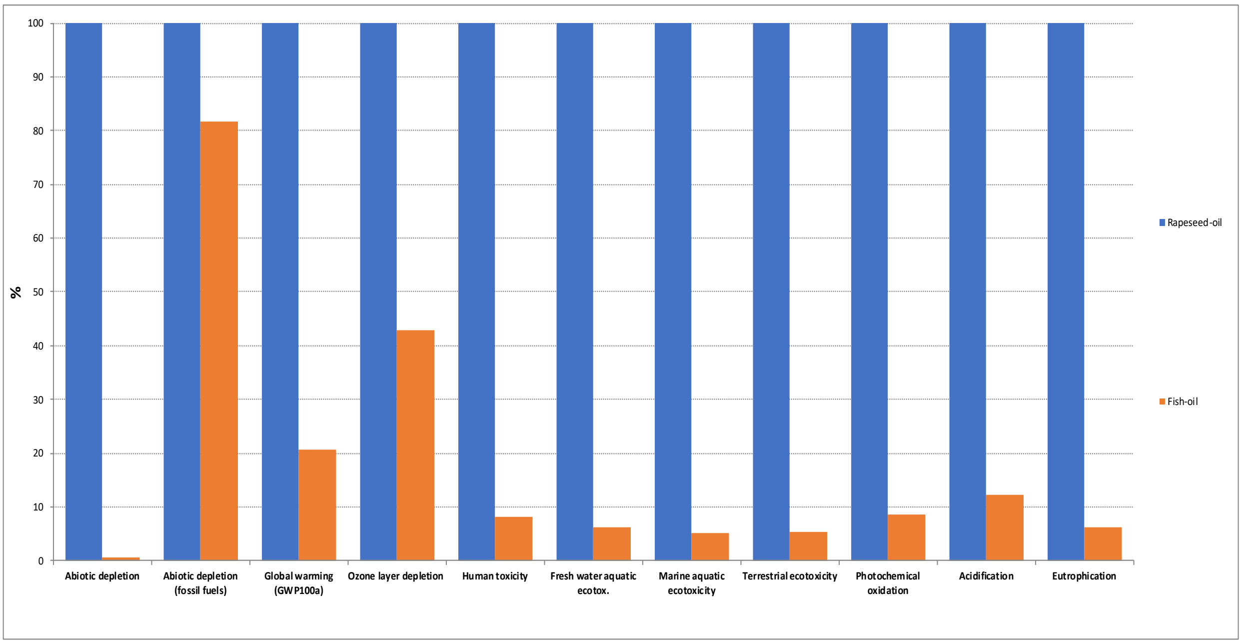


Figure 5.14. Comparison between the contributions from 1 kg of rapeseed-oil and those from 1 kg of fish-oil, with the lowest scoring product being presented as the percentage of contributions from the highest scoring product, for each individual impact category. calculated using the CML-IA-baseline method V3.03.

As can be seen from Figure 5.14, the difference is quite remarkable, with the contributions from fish-oil being less than 13 % of those from rapeseed oil, towards all but three categories. Apart from fossil-fuel depletion, the contributions of fish-oil towards the remaining categories are less than 50 % of rapeseed-oil. According to my own knowledge, this is the only study showing such a dramatic difference between fish oil-and a vegetable-oil), so it is important to analyse the uncertainty surrounding this outcome. The results of this uncertainty analysis (Figure 5.15) suggest a highly significant difference between both products, with rapeseed oil having the highest impacts. Only the result for the category fossil-fuel depletion (displayed upon the graph as ‘abiotic - fossil fuel’) falls below the 95 % range of confidence, with 76 % of Monte Carlo runs returning a result of rapeseed oil having the largest contribution. However, the uncertainty test must be interpreted within the context of its limitations, which are discussed in the following section.

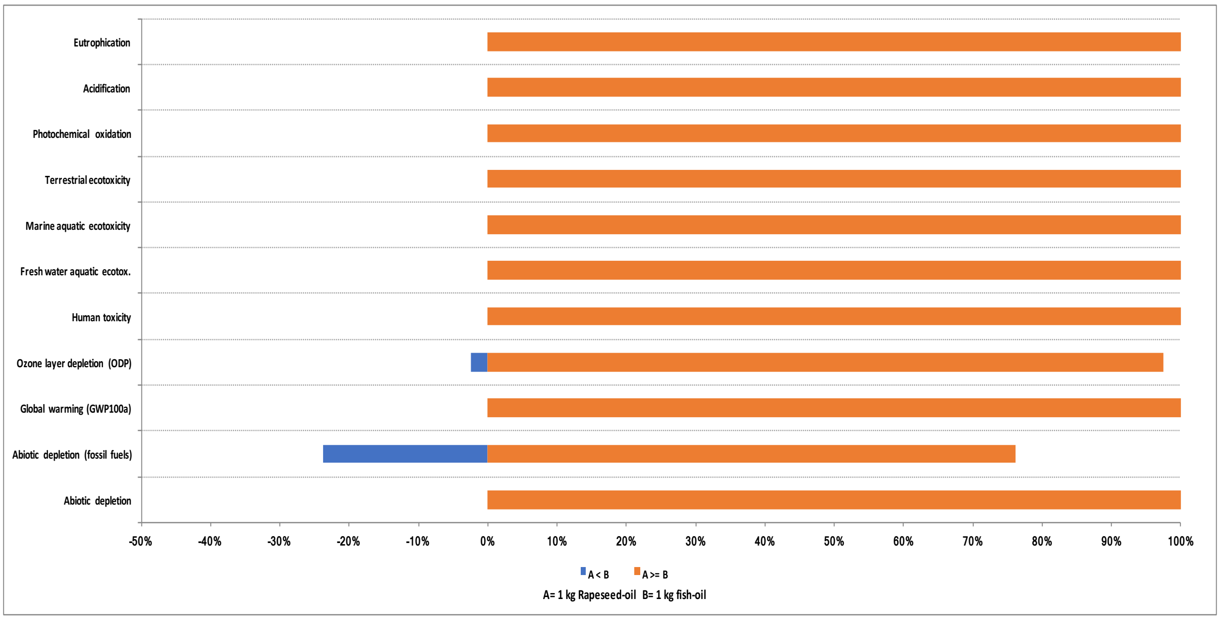


Figure 5.15. Monte Carlo uncertainty analysis of the comparison between the characterised impact assessment for rapeseed-oil and fish-oil.

5.5. Discussion and Conclusions

The work in this chapter describes a life cycle assessment of a typical salmon feed produced in the Chilean salmon aquaculture industry. Various interesting issues worthy of discussion have become apparent at different stages of assessment.

The dominant contribution of vegetable oil towards all but one of the impact categories deserves attention. This is a result carrying important implications for the development of feed formulations when vegetable oil is being used as a substitute, and perhaps as a more sustainable alternative (Erasmus 2009; Nasopoulou and Zabetakis 2012) to fish-oil. The salmon farming industry has been criticised for its reliance upon capture fisheries for the provision of important nutritional components (including polyunsaturated, and highly unsaturated fatty acids). Perhaps the most notorious of such criticisms is that which formed an important part of the high-profile publication by Naylor et al. (2000). Regardless of whether or not this publication was accurate in its assumptions, the media attention by which it was surrounded helped to bring the issue of using '*wild fish to feed farmed fish'* to the attention of consumers. Assuming for a moment, that consumer driven change is a reality within global food systems, it could be possible that this attention helped drive research efforts to find sustainable alternatives to fish oil and meal, but the persistence of the theme is undoubtedly also due to its resonance throughout the salmon farming industry, which has generally accepted that a heavy dependence upon ingredients from capture fisheries is problematic. The increasing trend towards using vegetable oil within salmonid feeds might not be a direct result of this situation, as there are various considerations involved within the development of feed formulations. However, the sustainability of employing high inclusion rates of vegetable oil within salmon feed, and especially the use of vegetable oil as even a partial replacement for capture fishery ingredients, is brought into question by the results of this LCA. Crude rapeseed oil having a much greater potential impact in every category than does fish-oil when compared on the basis of equal weight (Figure 5.14), is a potentially poignant result that raises serious questions about the sustainability of replacing fish oil with vegetable-oil alternatives. If the result is reflective of the real situation, the conclusion would be that although reductions in the use of fish-oil are desirable from the perspective of reducing fishing pressure upon wild fish stocks, doing so through substitution with vegetable-oils results in a much greater contribution towards a variety of environmental impacts. This suggestion of a need to more closely examine the relative impacts of fish-oil and vegetable-oils is not entirely unexpected. Although some previous LCA research concludes that marine ingredients contribute more to the impacts of aquafeed than do ingredients from non-animal agricultural crops (Pelletier and Tyedmers 2007), other LCA research has found fish-oil to have lower impacts towards some categories, including GWP and acidification potential (Newton and Little, in press). Results of another LCA found that potential impacts of salmon feed increased for 6 out of 8 impact categories when the inclusion rate of fish meal and oil where reduced through partial substitution with plant derived ingredients, such as rapeseed-oil (Boissy et al. 2011). Considering this later assessment, it is somewhat unsurprising that feed with higher quantities of fish meal and oil presented higher contributions towards the use of net primary production. Another relevant study using LCA to investigate salmonid feeds performed a comparison between rapeseed oil and fish-oil, which showed rapeseed-oil had a much greater potential for impacts to eutrophication, GWP and cumulative energy demand, although fish-oil had a significantly larger potential for contributing to some other categories (Smárason et al. 2017). Of course, these different studies are not directly comparable, but they do support the suggestion that vegetable-oil are not necessarily a sustainable option to oil derived from capture fisheries.

Based upon indications from previous publications, and the quite striking results obtained within this current study, it can certainly be concluded that the respective environmental impacts of vegetable oils and fish-oils constitute an area requiring further attention. To do this, improvements to the assessments can be made by improving the quality of data that they are based upon. In this study, data describing agricultural processes have been collected entirely from secondary sources. For LCAs that are performed with the intention of making meaningful comparisons, this really isn't good enough. The quality of data would likely be improved if primary data can be gathered that represents the majority of agricultural production of the region or particular economy being assessed. Information such as the specific fertilizer types being used, and the doses at which they are applied, should be combined with robust models describing emissions from agricultural land to soils, water and air for the regions of interest. This will require participation of agricultural producers and a willingness towards transparency that is not easily attainable. Efforts to improve the assessments are essentially efforts to improve the accuracy of the results, that is, they are analogous to reducing the ranges of uncertainty. Although the uncertainty assessment suggests that the outcomes of the comparison between fish-oil and crude rapeseed oil are reliable (Figure 5.15), this uncertainty assessment itself is limited to the efficacy of the methods that have been used to propagate uncertainty values. The issue of '*uncertainty within the uncertainty*' is discussed in the final chapter of this thesis, but it merits some consideration here. A lack of data describing capture fisheries has typically been a problem for LCAs describing aquafeeds containing fish oil. Although primary data are used in this study (and are most likely more relevant than those from the alternative sources available), their 'representativeness' in relation to the processes they have been used to describe, is itself uncertain, owing to a lack of information. Additionally, the 'completeness' is not known to the author, and there may be other inputs, outputs and emissions that are not described by the data. There is a general need for an improved availability of data describing capture fisheries and fish-oil reduction in the field of salmon aquaculture LCA. However, it is fair to state here, that the need to reduce uncertainty is not unique to this study, as such ranges have been the outcome of other aquaculture LCAs that have reported upon uncertainty (e.g. Roberts et al., 2015). Finally, the comparison between crude rapeseed-oil (or other vegetable-oils) and fish-oil is, to an extent, flawed, as vegetable- oil is not a complete substitute for fish-oil, with different oils having different nutritional profiles. The comparison has been made based upon an equal quantity of oil, both of which deliver different functions when understood from the perspective of nutrition. A frequent mistake in LCA is to compare two or more products on the basis of a functional unit described as a mass or other quantity. When considering food products, and using the correct interpretation of LCA methodology, the reference flow would be the quantity, mass or otherwise, that is required for the fulfilment of the functional unit. The functional unit may be a designated unit of nutrition (as discussed in Chapter 9), but quite how this should, or could be done, is a problem inherent to the majority of food based LCAs. The sensible approach to dealing with the issue of functional comparability should, by necessity, be considered as part of any future improvements to the comparison.

The various limitations described above are also relevant when considering the comparison between rapeseed-oil and sunflower-oil. However, with regards to confidence in the results of this comparison, it is encouraging that other studies have also found that the agricultural production of sunflower grain has a greater contribution towards a variety of impacts than does the production of rapeseed (Roberts et al. 2015, Newton and Little, in press).

As a final conclusion that does not focus on issues of data uncertainty, it can be said that there appears to be real potential for reducing the environmental impacts of compound feeds for salmon production. This statement is at least applicable to those feeds similar in formula to the generic Chilean diet analysed in this study. Some reductions may be achieved through seeking better alternatives to poultry-based ingredients, or through seeking animal reduction processes with reduced impacts. The careful selection of agronomic products may also lead to improvement. Grains from some global regions may have differing environmental impacts than those from others (e.g. Boisy et al., 2011). Using cropping systems that maximize yields in relation to fertilizer use may be one way to reduce nutrient related impacts, but, of course, these options need to be available. Alterative, competing grain types may also differ in their impacts. It seems likely, that the impact profile of alternative vegetable-oil will differ, perhaps in some cases, significantly. Therefore, selecting an oil with lower impacts overall, whether this oil be a straight oil or a specific blend, could yield improvements. Finally, and most controversially, there could be a need to reevaluate the value of replacing fish-oil with vegetable-oil alternatives, and in general, the value of replacing marine ingredients with those of a terrestrial plant-based origin. Considering concerns about the sustainability of marine fish stocks, this will be a difficult conversation.

Chapter 6: Life Cycle Assessment of Atlantic salmon (*Salmo salar)* Available at the Farm-Gate.

6.1. Introduction

6.1.1.

This chapter describes the life cycle assessment of Chilean Atlantic salmon (*Salmo salar*). The chapter follows the same structure as the previous, but with less attention given to the various modelling changes that have been made before the final model framework was arrived upon.

This chapter begins with a short description of the Chilean salmon industry. The development of the Chilean salmon farming industry is discussed within a sociological, political and environmental context in Chapter 3. What follows is a short, basic technical description of how Chilean salmon is farmed. The information it contains was obtained from specific interviews with industry members, as well as from my own personal observations during many visits to various facilities and fish production sites.

6.1.2. Producing farmed salmon in Chile.

In Chile, salmon is farmed using the same system that is standard throughout most of the industry's global production. However, there are some peculiarities that are not necessarily common practice in all producing nations, and so Chilean production does have some distinction in terms of its technical practices.

Of course, a salmon breeding operation is essential to provide significant quantities of eggs. The eggs available in Chile are obtained both from breeding facilities in Chile itself, as well as those imported from other nations (~20-30 %). The eggs are incubated within a land-based facility, hatching to become alevins, fry and then parr. Smolting is either induced artificially, or allowed to occur naturally. This may take place at the same facility, or else takes place at another land-based facility. Smolting may also take place in estuary, or lake-based smolting systems. Although smolting is still performed within lakes, due to the possible environmental effects of nutrient and chemical discharges, no new licences have been issued for the authorising of lake-based smolting since 1990 (Sernapesca, personal communication, 2016). After smotification, the fish are transferred to net-pens for fattening (grow-out). The transporting of smolts (indeed all live-salmon regardless of life-stage), takes place using specially adapted trucks with tanks, and using well-boats for transportation at sea. In some instances, trucks are loaded onto a ferry for delivering smolts to the grow-out site.

Grow-out occurs in bays and throughout much of the Patagonian fjordic system, usually taking between 16-18 months to complete. The fish are fed, as they are at all stages, using industrially produced feed containing various ingredients. The feed is applied the net-pens using a mechanised system. For grow-out facilities sited within close proximity to land, the feed may be stored within large silos, from which the feed is extracted and blown through floating, high-density polyethylene pipes, into the pens containing fish. More commonly, grow-out sites are located offshore, in which case feed is stored within floating feed barges with storage wells, to which the feeding pipes are connected. It is much less common that feeding is performed by hand. Offshore, manual feeding makes little sense, requiring a number of personal which need to be transported and housed. Close to land, manual feeding is somewhat more common, and can be seen on lake-based smolting facilities. As the extent of salmon grow-out faculties penetrates deep into the Patagonian fjords, personnel are often housed on floating facilities, where they typically live for 'shifts' of approximately 2 weeks. These housing facilities are built upon large floating, concrete platforms, the structure often including the feed storage wells, which are located below the water surface level, underneath the living quarters.

Figure 6.1. A typical Salmon grow-out facility in Chile. The rearing cages, or net-pens, can be seen in the foreground. Photograph is authors own.

Fish mortality is commonly treated through ensiling, a process by which mechanically minced mortality is mixed with formic acid, producing a thick slurry. The process requires electrically powered infrastructure, such as pumping equipment. Electricity is usually produced in a diesel-powered generator, which supplies energy for the ensiling process, as well as that needed for powering the mechanised feeding system, and also for the living facilities. Small boats equipped with diesel, or petrol powered outboard motors are used to ferry staff around the various grow-out structures during the course of their work activities. The fish are harvested at various sizes, often determined by market demand and customer specification. Most commonly, live salmon are transported via wellboat to the slaughtering facility (which is sometimes part of a joint processing facility) where they may be held for a short period of time in holding cages, before being slaughtered. Alternatively, fish may be slaughtered at the grow-out site by workgroups, with the fish being killed either manually or mechanically, packed into ice bins, and then transported to a processing facility by boat, and / or, road. At the processing facility, various products may be produced, including fresh or frozen whole fillets, and value-added products. The products are packaged, and then transported until they reach their market destination in Latin America, USA, the European Union, and Japan, as well as other Asian nations.

Figure 6.2. The same grow-out facility as in Figure 6.1. The floating platform with living quarters and working facilities can be seen in the foreground. It houses two feed silos, which extend downwards beneath the water line. Photograph is authors own.

6.2. Goal and Scope - Brief Definition

6.2.1.

The goal is to produce an LCA of Atlantic salmon production, standardised to a functional unit of 1 kg of live Atlantic salmon, available at the farm gate. This functional unit has been chosen so to avoid the process stages which take place, beginning with slaughter, or with transportation to the slaughtering facility. The boundary for the assessment has been placed at the farm gate because the model will be used as part of an LCA assessment of integrated multi-trophic aquaculture systems that does not consider post-cultivation processing activities.

The intention was to collect as much data as possible describing salmon farming activities within Chile, with the objective of creating the most comprehensive, representative LCA of Chilean salmon farming that has been produced so far. Despite evidence that capital goods might contribute somewhat significantly towards the impacts of agricultural production systems (Frischknecht et al. 2007), infrastructure (e.g. net-pens and feeding equipment) is not part if this assessment in order to keep the scope of the study manageable (although data describing infrastructure has actually been obtained).

When allocation is unavoidable or undesirable, mass-adjusted economic allocation is used. The specific allocation issue encountered through the modelling of mortality treatment is described below.

As there is some variation of the practices used for the different stages of salmon production in Chile, a preliminary LCA has been produced to identify the stages of production upon which data collection efforts should be focused.

6.2.2. Mortality and ensiling.

In Chile, it is common in current practice for mortality to be ensiled. In grow-out systems distant from on-shore operations, this takes place on floating platforms. In grow-out systems located in close proximity to on-shore operations, the enisling may take place on-land. In either case, the ensiling processes are very similar. The silage can be sold as forage-feed for cattle. It could be considered that silage is the co-product rather than mortality alone. Alternatively, the ensiling process may be seen as a waste-management process (which may be more in-line with the methodology employed in the ecoinvent V.3 database). The appropriate approach can be decided upon by determining the economic value, if any, of salmon mortality and the silage it is used to produced. Although mortality is generally reported by the industry as representing a financial cost to salmon production (e.g. Marine Harvest 2017), a very basic valuation has been performed to determine if any positive financial value can be given to the mortality (Table 6.1). Attempts to obtain the cost of producing silage, as well as the price received upon sale, failed. The only primary data obtained from industry was a from a farm manager who said the selling price is ‘negligible’ (anonymous industry member, personal communication). A literature search retrieved limited data from a report about salmon co-products, published by the FAO (Ramírez 2007), and this was used in the valuation. An estimate value for the price of salmon feed of 2 USD per kg of salmon harvested was obtained from the research director of a major Chilean salmon producer (anonymous industry member, personal communication). Encouragingly, this is highly similar to the value of 1.96 USD per kg of salmon, reported in the Marine Harvest salmon farming industry handbook (Marine Harvest 2017). An average price value of 5.94 USD per kg of 'gutted-fish' (assumedly equivalent to HOG[19]) was obtained by calculating the average of values reported for 5 consecutive years in the Marine Harvest integrated annual report (Marine Harvest 2016). It is my experience that price values reported by the Chilean industry are given per fillet FOB, and although, with effort, it may be possible to find values for Chilean whole salmon, the basic nature of the valuation does not make it worthwhile. The total cost of producing salmon[20] was also obtained by calculating the 5-year average of values given in this same report (Marine Harvest 2016). The value for net revenue has been calculated using the values for price and cost. As the values given in the report are per kg of salmon gutted-weight equivalent, they may negate any value held by the removed viscera. The value of silage and of mortality has been calculated using the above-mentioned data, combined with data describing the weight of ingredients (e.g. acids) of the salmon silage (data not shown).

Table 6.1. Basic valuation of mortality derived from the price of silage and the cost of its production. The value of silage per kg works out to be the same as for mortality per kg (calculated based upon the weight of the ingredients of salmon silage). All prices and costs are given in USD.

Cost of producing silage / kg ✿	**Cost of feeding fish / kg growth**	**Selling price of silage / kg**
0.06	2	0.1
Value of mortality / kg	**Value of lost opportunity to harvest live / kg**	**Balance of mortality value and lost opportunity / kg**
-1.96	-1.82	-3.78

✿ This value does not include the cost incurred through feeding mortality whilst living.

If the valuation presented in Table 6.1 is assumed to be correct, then it can be said that the procedure of managing mortality through ensiling is performed at a financial loss. Even if salmon silage can be sold at a price that covers the cost of its production, it is highly unlikely that any resulting net revenue could rival that which would have been obtained had the fish survived. According to the Marine Harvest industry handbook (Marine Harvest 2017), mortality incurs a cost of 0.03 USD out of a total

[19] Head-on gutted.
[20] Unfortunately, the value for the cost of production may include the cost of mortality management, but it is not stated either way within the report.

cost of 5.58 USD in their Chilean operations. This value is much lower than the estimated costs shown in Table 6.1, but it still supports the argument that mortality is not currently a valuable co-product of Chilean production. For this reason, there seems to be little point in attempting to model mortality as an economically allocable co-product. Rather, it can be considered as a waste-product with a possible economic application. Allocation is possible using allocation factors other than one based on economic value. With mass-based allocation, or allocation performed using other physical properties such as energy content, allocation between the two products is possible. Arguably, this is an instance where physical allocation factors might be considered as superior to an economic alternative. In this case, economic allocation is not possible due to the co-products negative value, whereas allocation based upon, say, mass-adjusted protein content, enables allocation to the co-product in a way that reflects its industrial application. The contrasting argument is that the principle goal of the cultivation system is to produce economically valuable salmon, and that mortality is an undesirable waste incurring a cost, and so should be modelled as such. Regardless of one's opinions on this matter, mass-adjusted economic allocation is the method being used in this assessment, and so a decision needs to be made as to how the waste treatment process should be modelled. Two possibilities exist. Either the ensiling process can be modelled as a process taking place outside the boundaries of the salmon production process, or the inputs of the ensiling process can be included in the inventory of salmon production. Although the ensiling of mortality is not physically essential to salmon production, it nevertheless does take place in common practice. For this reason, ensiling will be modelled as part of the salmon production process.

6.3. Preliminary LCA of Atlantic salmon production in Chile.

6.3.1. Introduction to the preliminary LCA.

The purposes of this preliminary LCA is to identify which stages of production should be focused upon, a decision based upon the contribution of each production stage towards the impact categories. The preliminary LCA was a produced as part of a project in which I worked on calculating the life cycle impacts of Chilean salmon production, for INTESAL – Instituto Technológico del Salmón – the technological and scientific research and development branch of SalmonChile, the latter being the major supportive, non-governmental association for the Chilean salmon industry. Fortunately, both INTESAL and SalmonChile realise the importance of understanding the life cycle impacts of production, not only to demonstrate a willingness towards corporate social and environmental responsibility, but to gather information needed to increase their ability to manage the industry in terms of its environmental impacts. The preliminary LCA is performed with a wider range of assumptions than would normally be acceptable, and the process data is much less complete and representative compared to the other LCAs in this study. The impact assessment that was used is the ReCiPe 2008 midpoint indicator method for classification and characterisation, which features additional impact categories.

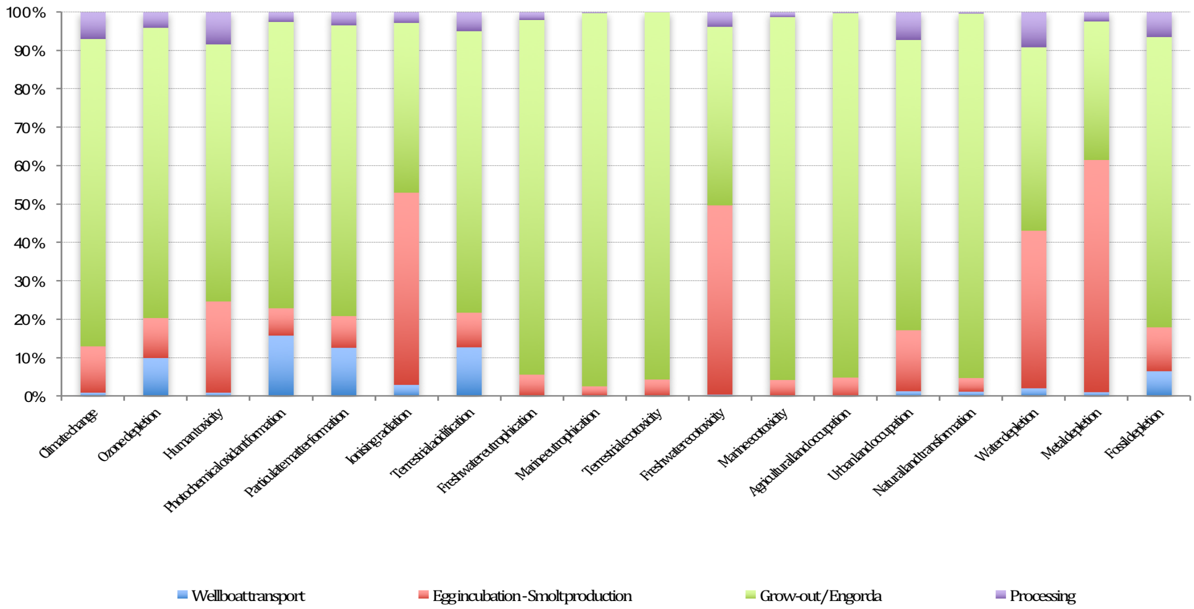


Figure 6.3. Characterised preliminary impact assessment of the production of 1 kg of salmon fillet. Calculated using the ReCiPe 2008 midpoint indicator method.

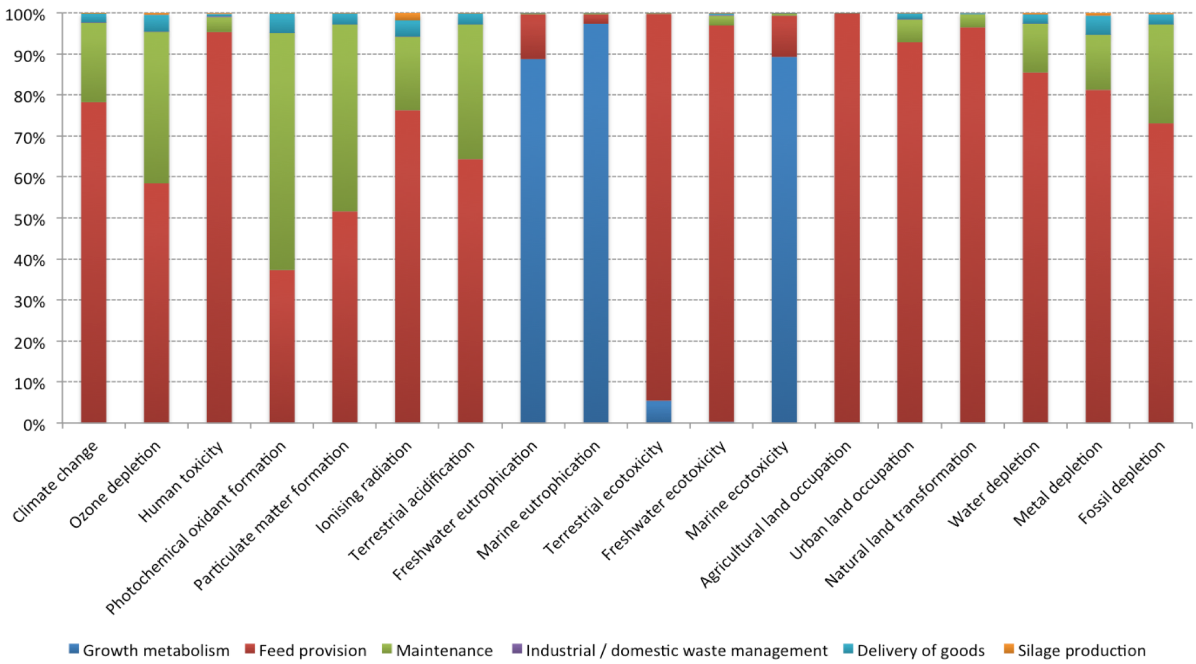


Figure 6.4. Characterised preliminary impact assessment of the production of 1 kg of Atlantic salmon, available at the farm gate. Calculated using the ReCiPe 2008 midpoint indicator method.

Two impact assessments are shown. One shows the contribution of the major stages that contribute towards the production of 1 kg of salmon fillet, and so also includes the slaughtering and processing stage. This was done as part of the INTESAL project, but its results were used to inform the method of progression for the data collection process, and so are relevant here. The other assessment shows the contribution of different processes towards the grow-out production of 1 kg of live weight salmon at the farm gate, although it does not include the production or transportation of smolts, which is covered by the assessment of 1 kg of fillet. The modelled smolt production phase (including egg incubation) is a land-based system, as land-based systems are much more common in Chile than smolt production systems that include an open-water phase.

Figure 6.3 suggests that both well-boat transportation and smolt production have non-negligible contributions towards the impacts of producing fillets, and so, should also have significant contributions towards the production of live weight salmon at the grow-out farm-gate. Figure 6.4 shows that the process 'maintenance' has obvious contributions towards many of the impact categories. This process includes electricity generation and boat use. As is to be expected, both feed and emissions from fish metabolism (nitrogen, phosphorous and carbon) have significant contributions. The process 'delivery of goods,' includes the transportation of feed and fuel to the grow-out site. Its contribution is less noticeable than those pre-mentioned, but it is still enough to warrant inclusion in the final LCA of salmon. The management of industrial waste, such as plastic, wood, and paper etc., does not feature significantly and so data collection efforts did not focus on this aspect of production. The contribution of mortality ensiling is visible for the categories 'ionising radiation and metal depletion.' Despite having relatively minor contributions overall, it will be included in the final salmon LCA because it is a co-product of salmon production.

6.4. Inventory

6.4.1. Grow-out production

The major processes for which inventory data has been collected are shown in the flowchart below (Figure 6.5). Compound feed production is covered in the previous chapter (chapter 5). The ecoinvent database V.3 is the source of data describing background processes. Grow-out production biomass data was provided by INTESAL, and covers the production of Atlantic salmon within the grow-out systems of the salmon producing members of Salmonchile A.G. (approximately 95% of Chilean salmon production). The data describes production, yields, mortality, biological and economic feed conversion ratios (bFCR and eFCR respectively), and smolt inputs. The FCR values give the quantity of feed used per unit quantity of fish growth, and allow the quantity of waste feed to be accurately derived (eFCR-bFCR). A basic mass balance model was constructed, using the overall average values for bFCR and eFCR, as well as values describing feed nutrient composition obtained from feed producers and literature sources, and values describing the retention and ejection of nutrients by fish, obtained from literature. This model was used to produce the values for dissolved and solid-bound emissions of nitrogen, phosphorous and carbon, to the receiving water body.

Data detailing the use of petrol and diesel was obtained from 10 grow-out facilities located within Patagonian Chile, and operated by a major Chilean salmon producer (anonymous by request of the producer). Site visits provided information about how the fuel was used. Biomass data was also supplied for these grow-out sites, and the uses of fuel by each site was averaged to the production of 1 kg of live weight salmon at the farm gate of each site. Transport distances and modes for the different inputs and outputs were also supplied by the same producer. The inventoried data is shown in Table 6.2

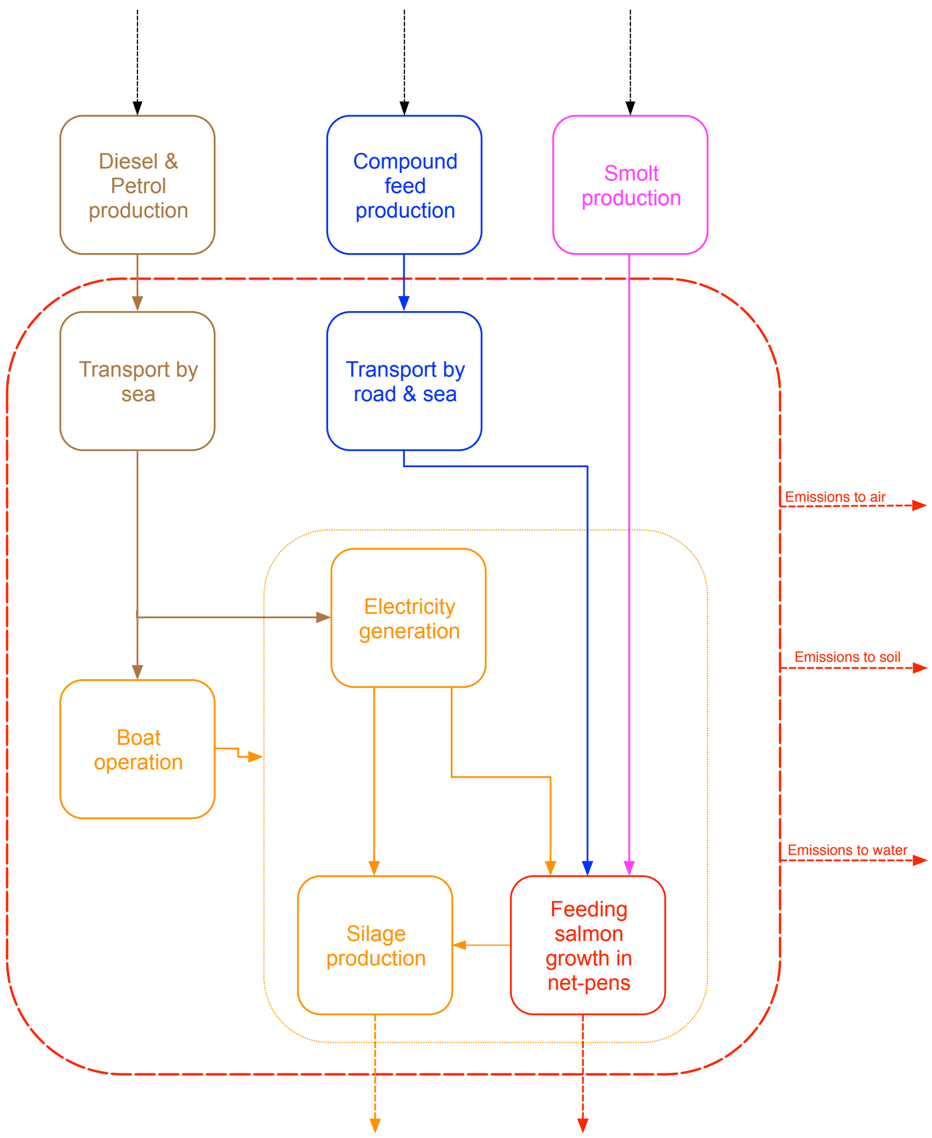


Figure 6.5. Flow chart depicting the major process required to produce farmed Atlantic salmon.

Table 6.2. Inventory data describing the production of 1 kg of Atlantic salmon available at the farm-gate. The data points in the 'value' column are geometric means of lognormal distributions of populations collated using data from a variety of anonymous data providers and also from data provided by Intesal.

6.4.2. Smolt production

Product outputs		Value	Unit	Allocation	
Atlantic salmon		1	kg	100%	
Salmon silage		0.02999	kg	0%	
Inputs		**Value**	**Unit**	**Distribution**	**SD^2**
Materials/fuels					
Feed		1.323	kg	Lognormal	1.05
Salmon smolt, production		0.035	kg	Lognormal	1.05
Treatment of mortality | ensiling		0.029	kg	Lognormal	1.07
Diesel, burned in electric generating set, 18.5kW		0.795	MJ	Lognormal	1.09
Outboard-motor boat; petrol burned in		0.002	kg	Lognormal	1.09
Transport					
Freight, lorry, >32 tonne, Euro5 (for feed)		0.085	tkm	Lognormal	2.01
Freight, sea, transoceanic ship (for feed)		0.356	tkm	Lognormal	2.01
Freight, sea, transoceanic ship (for fuel)		0.853	tkm	Lognormal	2.01
Freight, sea, transoceanic ship (wellboat)		0.004	tkm	Lognormal	2.01
Emissions	**Compartment**	**Value**	**Unit**	**Distribution**	**SD^2**
..to water					
Carbon	ocean	0.538	kg	Lognormal	1.51
Nitrogen, organic bound	ocean	0.013	kg	Lognormal	1.51
Nitrogen	ocean	0.034	kg	Lognormal	1.51
Phosphorus	ocean	0.010	kg	Lognormal	1.51

The major processes required for the production of smolts at a land-based system are shown in the flowchart depicted in Figure 6.6. Chemical inputs (not shown in the flowchart), such as fungicides and antibacterials, do not feature as part of the LCA. They usually consist of one or more chemical compounds in an aqueous solution. Data describing production of these chemicals are not always easy to obtain, and the use of these compounds are not well described by characterisation models. To avoid further effort being required for what was an already demanding project, the use of chemicals has not been included within the system boundaries. However, the use of salt has been included because it is a major input which can easily be modelled. Salt is used during smotification, and as well as a therapeutic additive to the previous life-stages.

The data was collected from the main smolt production facility of a major Chilean salmon producer (anonymous at the request of the producer). I visited a number of such faculties in different areas of

Chile, and the facility from which I collected data can be described as a typical example, although there are, unavoidably, particular differences between all of them[21]. Very good quality data was collected from this site, describing biomass, fuel use, salt use, feed use, sludge production, sludge chemical characteristics, inlet water quality, outlet water quality, transport types and distances, as well as various other aspects of production.

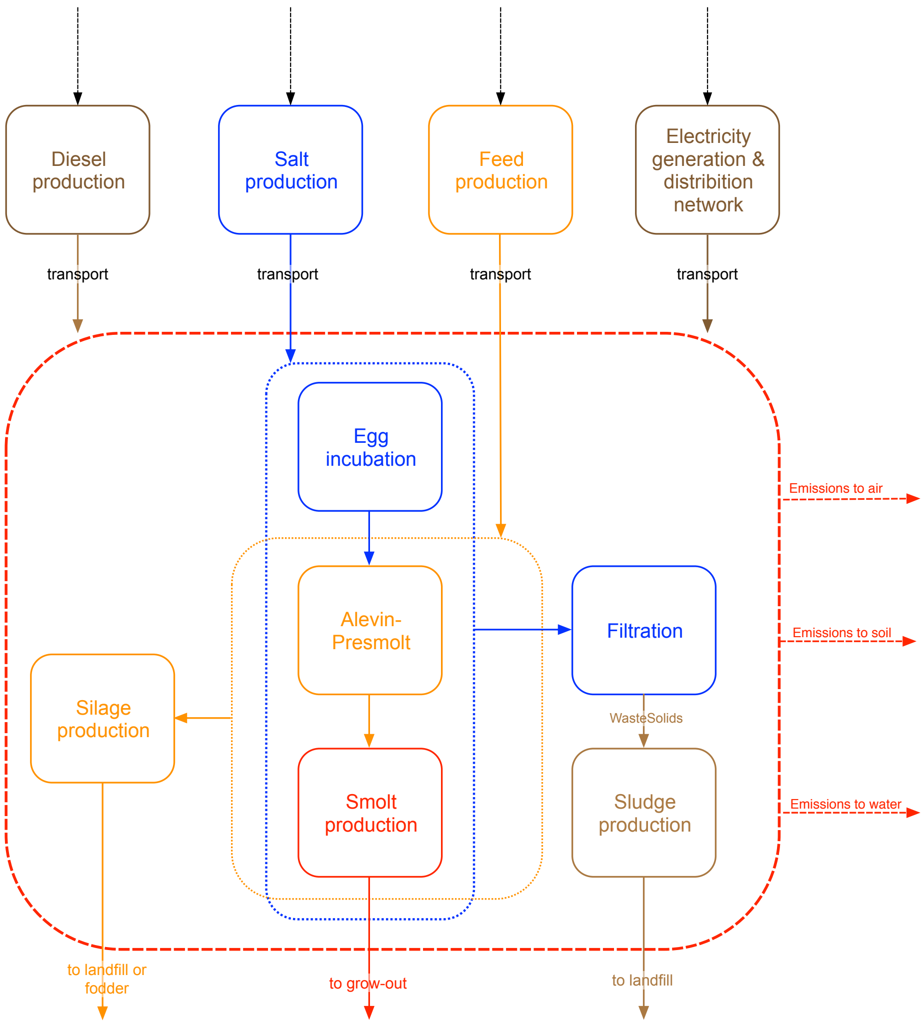


Figure 6.6. The major input and output flows included in the modelled smolt-production system. Egg production is not included in the assessment, and so is not shown in the flowchart. The blue dotted line delineates those processes which receive and discharge water, as well as salt. Water exiting these processes is filtered, removing sludge which is processed before being removed and delivered to a landfill site. The orange dotted line delineates the processes which receive feed as an input, and from which mortality is an input to the silage production process.

[21] One remarkable peculiarity of the data providing facility is its location directly upon a major seismic fault and its alarmingly close proximity to one of the most dangerous volcanoes in Chile.

The eggs are incubated, and the resulting juveniles are reared in large, circular tanks, with a circular motion of flow. Very well oxygenated water enters the facility from a point of a river that is raised above the level of the rest of the site. After use, the water re-enters the river at a location slightly below the elevation of the production units. As such, it both enters and leaves the system via gravity. Despite this, much pumping is required. Before the water enters the production units, it is treated using ultraviolet radiation to destroy unwanted pathogens. Water leaving the production units travels through drum filters at various locations across the site, before travelling towards the outlet. The sludge which this filtration produces, travels via suction towards the sludge treatment facility. Fish mortality is placed manually into a grinder, and the resulting mince is removed via suction and travels through pipes towards a silage production facility.

Table 6.3. Inventory data describing land-based production of 1 kg of salmon-smolts, available at the farm gate. Data points in the 'value' column are geometric means of lognormally distributed primary data populations, collected from an anonymous data source.

Product output		**Value**	**Unit**	**Allocation**	
Salmon smolt		1	kg	100%	
Salmon silage		0.0846	kg	0%	
Inputs		**Value**	**Unit**	**Distribution**	**SD^2**
Materials/fuels					
Salmon Feed		1.134	kg	Lognormal	1.11
Sodium chloride, powder {GLO}| market for		4.129	kg	Lognormal	1.12
Processes					
Treatment of mortality | ensiling		0.082	kg	Lognormal	1.12
Transport					
Transport, freight, lorry >32 tonne Euro3 (for feed)		0.113	tkm	Lognormal	2.03
Transport, freight, lorry >32 tonne Euro3 (for salt)		0.413	tkm	Lognormal	2.03
Transport, freight, lorry >32 tonne Euro3 (for fuel)		0.0098	tkm	Lognormal	2.03
Electricity/heat					
Electricity, medium voltage {CL}| market for		3.714	kWh	Lognormal	1.12
Diesel, burned in electric generating set, 18.5kW {GLO}		4.168	MJ	Lognormal	1.15
Emissions	**Compartment**	**Value**	**Unit**	**Distribution**	**SD^2**
..to water					
Carbon	river	0.2824	kg	Lognormal	1.52
Nitrogen	river	0.0353	kg	Lognormal	1.52
Nitrogen, organic bound	river	0.0021	kg	Lognormal	1.52
Phosphorus	river	0.0034	kg	Lognormal	1.52
Waste to treatment		**Value**	**Unit**	**Distribution**	**SD^2**
Sludge		0.0019	m^3	Lognormal	1.12

The site is divided into two main production areas. The first houses the units for egg hatching and the rearing of fry until the pre-smolt life phase is reached, and the second area is designated principally for the smotification process, and thus, the production of smolts. Not all fish produced in first area are sent to the smotification units, with certain batches being destined for smotification elsewhere.

Because the data was provided separately for the two production areas, it was possible to account for those fish batches that left the site without undergoing smotification. The collected data covers a time period of 2 years, which is sufficient to cover the full extent of production variations, such as the variable use of photoperiod manipulation. Inventory data for the smolt production stage is shown in Table 6.3.

6.5. Impact Assessment

6.5.1. Smolt production

The results of the characterised impact assessment of land-based smolt production can be seen in Figure 6.7 and Table 6.4. Overall, feed production has a significant contribution (between 12.48 % and 36.72 % across all impacts). However, whereas the production of feed is coming to be recognised as being responsible for the greatest contributions towards the global scale environmental impacts of salmon farming (e.g. Pelletier et al. 2009), it does not have an overwhelming contribution towards the land-based smolt production facility being analysed in this current assessment. It can be seen from Figure 6.7 and Table 6.4, that the provision of electricity from the Chilean electricity network, and the provision of salt, rival those contributions from feed. The use of diesel for electricity generation holds the position of 4th place in terms of its overall contribution across impacts, although for some impact categories, this process exceeds the contributions from network electricity and salt provision. In reality, diesel is not only used for electricity generation, and is used to power other mechanical equipment. The diesel-powered electricity generator is used here, as a process intended to represent those other forms of equipment which combust diesel and release emissions. The contributions of fossil fuels, electricity, and salt inputs rival, and, in most cases (when their contributions are summed), surpass those from feed, because production of fish on-land requires significant inputs to deliver the necessary conditions that would otherwise have been provided by the 'natural' environment in open-water systems. This later proposition is supported by an LCA comparison between smolt-production on-land, with lake based smolting (data not shown). However, it is currently difficult to see how the production of smolts could be completed without the use of some land-based production, as required by egg incubation and the subsequent rearing of fry. Notwithstanding, the demand for inputs to land-based production explains why feed does not absolutely dominate any of the impact categories. Carbon, nitrogen and phosphorous emissions from fish are the major contributors towards eutrophication, despite a portion of these nutrients being removed through filtering (the emissions of these removed nutrients occur during the land phase of sludge treatment). Sludge production /

treatment, nor the ensiling of fish have a significant contribution to any of the impacts, and the transportation of feed, diesel and salt also has a minor contribution.

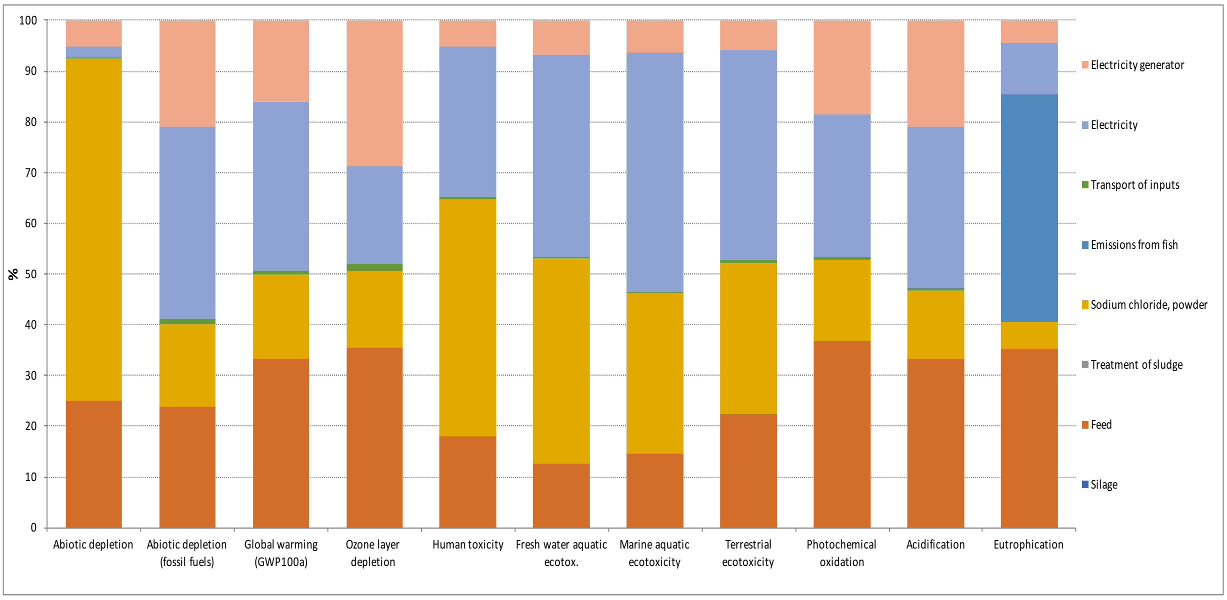


Figure 6.7. Characterised impact assessment of the production of 1 kg of salmon-smolts in a land-based production facility. Calculated using the CML-IA-baseline method V3.03.

Table 6.4. Percentage contribution of each foreground process required for the production of 1 kg of salmon-smolts, towards the total contributions of each impact category.

Impact category	% Contribution to total							
	Feed	Emissions from fish	Salt	Electricity generator✿	Electricity network	Treatment of sludge	Mortality ensilement	Transport✪
Abiotic depletion	24.957	0	67.459	5.192	2.030	0.0002	0.059	0.303
Abiotic depletion (fossil fuels)	23.793	0	16.128	21.003	37.863	0.0008	0.162	1.050
Global warming (GWP100a)	33.355	0	16.406	16.061	33.339	0.0005	0.109	0.728
Ozone layer depletion	35.345	0	15.181	28.716	19.216	0.0010	0.148	1.393
Human toxicity	17.888	0	46.630	5.172	29.569	0.0005	0.097	0.643
Fresh water aquatic ecotox.	12.483	0	40.380	6.732	39.931	0.0003	0.086	0.388
Marine aquatic ecotoxicity	14.594	0	31.581	6.413	47.107	0.0002	0.093	0.212
Terrestrial ecotoxicity	22.320	0	29.738	5.731	41.529	0.0004	0.086	0.596
Photochemical oxidation	36.717	0	16.072	18.435	28.186	0.0003	0.130	0.460
Acidification	33.313	0	13.251	20.847	32.008	0.0004	0.085	0.496
Eutrophication	35.340	44.826	5.177	4.394	10.088	0.0469	0.020	0.108

✿Diesel powered electricity generator. Used as a representative process for all mechanical processes that use diesel as an input.

✪Transport of salt, feed and fuel.

6.5.2. Salmon grow-out production

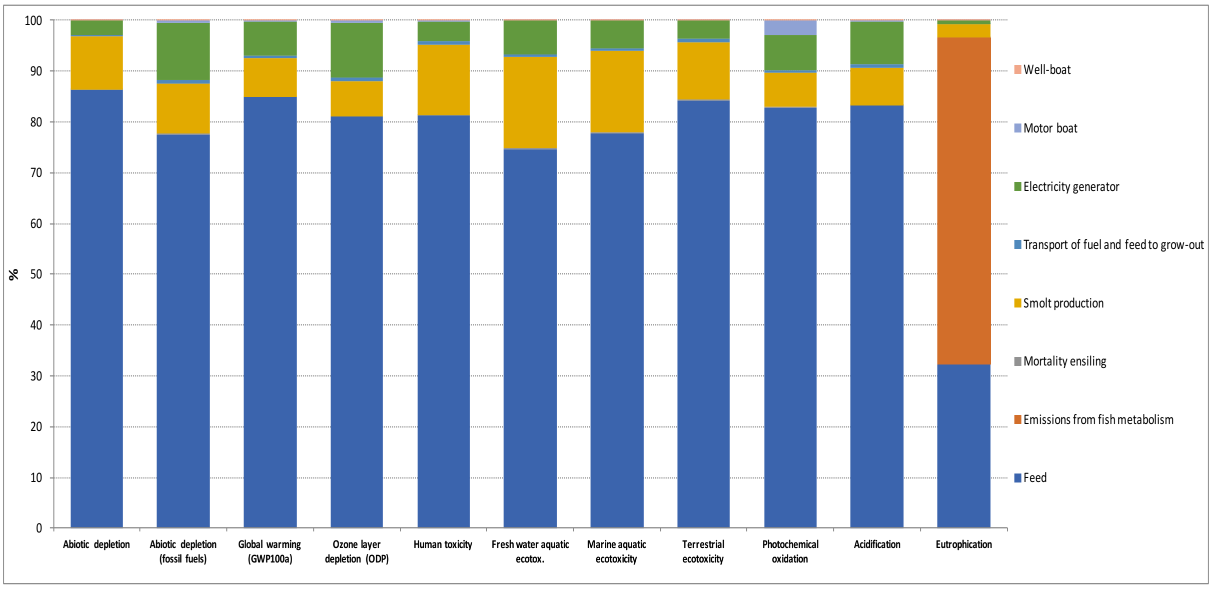


Figure 6.8. Characterised impact assessment of the grow-out production of 1 kg of Atlantic salmon. Calculated using the CML-IA-baseline method V3.03.

The results of the characterised impact assessment can be seen in Figure 6.8 and Table 6.5. The provision of feed contributes the most to all impacts except eutrophication, towards which emissions from fish metabolism has the largest contribution (64.19 %). The contributions from feed are those contributions from the production of feed ingredients, as well as the feed-milling process, which are described in Chapter 5. In addition to providing the majority of contributions towards 10 out of the 11 impact categories, the contributions of feed are very high relative to those of other inputs (between 74.64 % and 86.35 % of the totals). This result also appears to have some consistency with results of other studies. As an example for comparison, Newton and Little (in press) found the contributions of feed to be between 80.5 % and 99.9 % of the total of several impact categories, excluding eutrophication. Momentarily disregarding eutrophication, the production of electricity in a diesel-powered electricity generator, and the production of smolts are the major 'non-feed' contributors towards impacts. Generators are typically used to power a variety of processes within Chilean grow-out systems, such as feeding systems. It is easier to view the contributions of the inputs and outputs other than feed, by removing the provision of feed from the assessment. In Figure 6.9, production of feed has been removed, although the transport associated with the delivery of feed is still included. The contributions from electricity generation and smolt production are now clearly visible. The contribution from the operation of the outboard motorboat is also visible on this chart, with its largest contribution being towards photochemical oxidation (28.26 % of contributions when feed production is excluded, but 3.14 % when feed production is included). Both electricity generation and operation of the motor boat may be considered together because they are both input processes that rely upon fossil fuels for their operation, and they are the two process which cover the majority operational activity taking place at the grow-out site (whereas transport processes deliver to the grow-out site, but are not part of it, and smolt production takes place elsewhere). The production of silage also takes place at the grow-out, although its contribution towards impacts is negligible.

Table 6.5. Percentage contribution of each foreground process required for the grow-out production of 1 kg of Atlantic salmon, towards the total contributions of each impact category.

Impact category	% contribution to total							
	Feed	Emissions from fish	Mortality ensiling	Smolt production	Well-boat	Electricity generator	Motor boat	Transport of fuel and feed
Abiotic depletion	86.35	0	0.062	10.431	0.0002	2.935	0.015	0.207
Abiotic depletion (fossil fuels)	77.504	0	0.159	9.820	0.003	11.179	0.443	0.893
Global warming (GWP100a)	84.804	0	0.083	7.665	0.002	6.672	0.266	0.507
Ozone layer depletion	80.993	0	0.102	6.908	0.002	10.752	0.420	0.823
Human toxicity	81.30	0	0.133	13.702	0.002	3.841	0.236	0.786
Fresh water aquatic ecotox.	74.64	0	0.154	18.027	0.001	6.578	0.054	0.547
Marine aquatic ecotoxicity	77.796	0	0.149	16.072	0.001	5.586	0.051	0.345
Terrestrial ecotoxicity	84.285	0	0.097	11.384	0.002	3.536	0.079	0.616
Photochemical oxidation	82.855	0	0.088	6.803	0.003	6.798	2.921	0.532
Acidification	83.123	0	0.064	7.523	0.004	8.499	0.174	0.613
Eutrophication	32.349	64.188	0.006	2.760	0.0002	0.657	0.013	0.028

The delivery of feed and fuel to the grow-out sites is another visible contributor when feed provision is removed (Figure 6.9). When transportation of these goods is analysed separately (data not shown), the transportation of feed has greater contributions than does the transportation of fuel. Differences in the quantity of contributions between the delivery of feed to the grow-out site and the and delivery of fuel, are due to there being more feed transported than there is fuel, and also because the delivery process for feed contains some transport by road, whereas the delivery of fuel does not. As 1 tkm of road truck transport has significantly greater contributions towards impacts than does 1 tkm of transportation by ocean freight, the inclusion of road transportation in the feed delivery process also contributes to the difference.

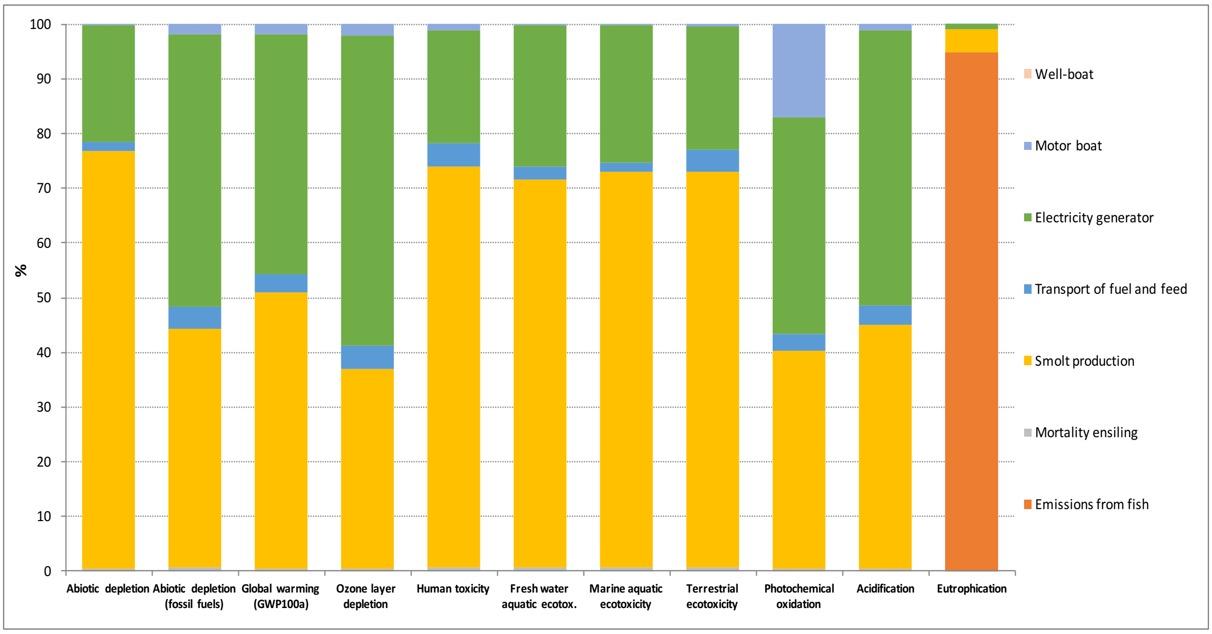


Figure 6.9. Characterised impact assessment of the grow-out production of 1 kg of Atlantic salmon, when feed production has been removed from the analysis. Calculated using the CML-IA-baseline method V3.03.

When considering the relative contributions of smolt production and feed production towards the impacts of the salmon grow-out phase, it is important to acknowledge that feed production is also included as an input to smolt production. It is useful to be able to view the total combined contributions from the feed produced for all stages of cultivation. Figure 6.10. shows the characterised impact assessment for 1 kg of live weight salmon at the farm gate, with all processes grouped into 3 product stages. The three stages are 'feed production,' which is the combined contributions of feed from both smolt and grow-out production; 'smolt production,' representing the contribution from the 0.0352 kg of smolts (amount required to produce 1 kg of salmon), excluding feed input; and 'grow-out production,' the combined contributions from the grow-out phase, without any feed input. Of course, feed production accounts for the majority of contributions, and across all impacts excluding eutrophication, it accounts for between 76.8 % and 89 % of contributions. Somewhat obviously, the proportional contribution of feed towards each impact is increased by splitting production into these stages. However, the actual increase in contributions towards each impact is marginal; producing 0.0352 kg of smolt requires 0.0399 kg of feed, which is only 3.014 % of the 1.323 kg of feed input to the grow-out phase.

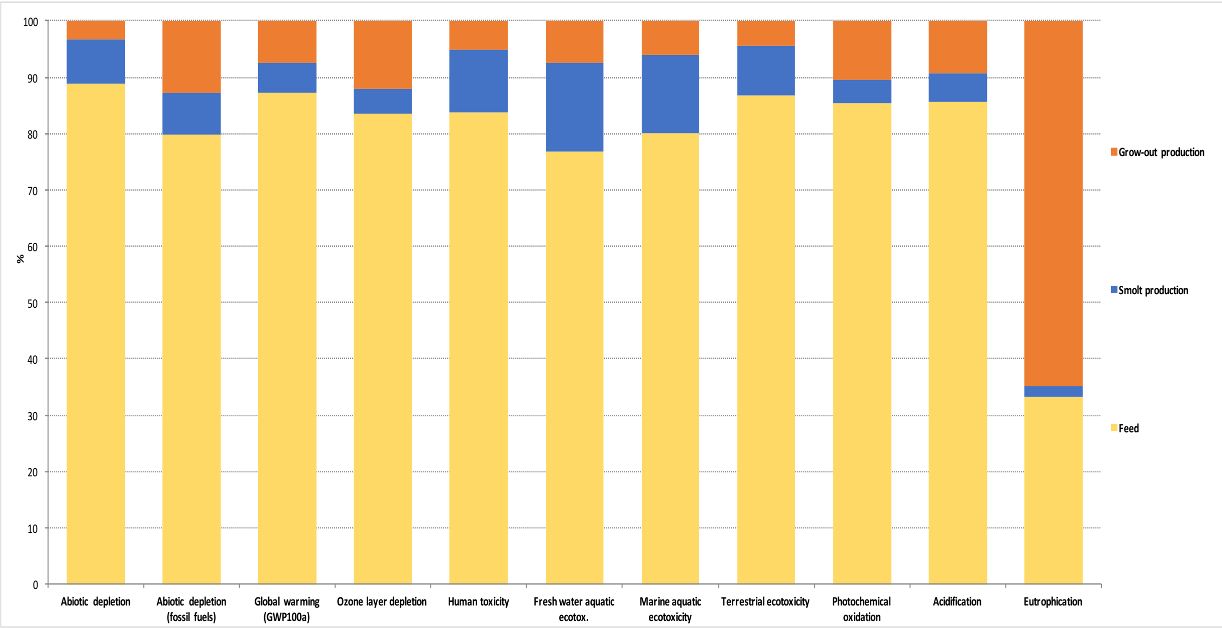


Figure 6.10. Characterised impact assessment for 1 kg of salmon, with all processes grouped into 3 product stages. Calculated using the CML-IA-baseline method V3.03.

Interesting results are found in the relative impacts of smolt production and grow-out production, which are discussed in this paragraph. Towards the categories 'abiotic depletion,' 'human ecotoxicity,' 'marine aquatic ecotoxicity' and 'terrestrial ecotoxicity,' contributions from smolt production are greater than those from the grow-out phase. This result is more noteworthy when considering that a comparison between the respective contributions from each of two stages, is a comparison between the production of 0.0352 kg of smolts, and 0.9648 kg of grow-out production (these two values = 1 kg of salmon in total). In other words, smolt production contributes more than the grow-out stage towards the total of each of the previously specified impact categories, even though the production quantity of smolts is only 3.64 % of the production quantity from the grow-out. From a superficial glance, it may seem surprising that smolt production has the higher contributions towards 'marine ecotoxicology,' an impact which might be assumed to be more associated with marine grow-out systems than from land-based farming systems. It must be remembered that emissions from fish metabolism, such as dissolved nitrogen, are not modelled as contributing towards this category, but are modelled as contributing towards eutrophication, a category for which grow-out production does have the highest relative share. Through exploring the detailed life cycle inventory data (data not shown) it is possible to identify the processes of each product stage that contribute the most to individual impact categories, as well as the process emissions of the individual substances that are the actual contributions. For the smolt production stage, it is the supply of network electricity or, more specifically, the generation of electricity and the processes involved with the mining of hard-coal that this requires, that account for the majority of contributions (55.2 %) towards marine ecotoxicology.

To a lesser extent, processes required for the provision of sodium chloride (common salt), these mainly being chemical processing activities, also account for a portion (37 %) of smolt productions contributions towards this same impact category. For grow-out production, it is the diesel-powered electric generator that is the process responsible for the majority of contributions (91.2 %), to marine ecotoxicity. It is mostly processes involved with the production of the generator itself, rather than the provision and combustion of diesel, that are responsible for the majority of these, through the release of emissions to water, such as beryllium (Be). These results are important, because they help to illustrate the point that the impacts of salmon farming upon the marine environment are not always a result of those aspects of production that commonly receive the most criticism.

Table 6.6. The percentage contribution of each product stage towards the total contributions of salmon production per impact category**.**

Impact category	% contribution to total		
	Feed production	Smolt production	Grow-out production
Abiotic depletion	88.953	7.828	3.219
Abiotic depletion (fossil fuels)	79.840	7.484	12.676
Global warming (GWP100a)	87.361	5.108	7.531
Ozone layer depletion	83.434	4.467	12.099
Human toxicity	83.751	11.251	4.998
Fresh water aquatic ecotox.	76.891	15.776	7.333
Marine aquatic ecotoxicity	80.142	13.726	6.132
Terrestrial ecotoxicity	86.827	8.843	4.330
Photochemical oxidation	85.353	4.305	10.342
Acidification	85.629	5.017	9.354
Eutrophication	33.324	1.784	64.891

6.6. Uncertainty Analysis

The results of the Monte Carlo analysis shown in Figure 6.11 suggest that there is some significant uncertainty within the smolt production inventory data. Most notable, is the very large 95 % confidence interval for the impact category 'abiotic depletion.' Much of this uncertainty may be attributable to the ecoinvent process for the production of 'sodium chloride powder,' which contains some inputs which have been assigned high estimate uncertainty values. For example, the infrastructure input describing the supply of a chemical factory in which the sodium chloride (NaCl) is assumed to be pressed, is assigned a σ_g95 (σ^2 of lognormal distribution) of 4.63, with other inputs all having an σ_g95 greater than 2. The 95 % confidence interval is particularly large for 'abiotic depletion' and 'human toxicity', as they are also for these two impact categories in the smolt production analysis. The ecoinvent transportation process used as an input to smolt production contains some high estimate uncertainty values, and 95 % confidence intervals for these processes are particularly high for 'ozone layer depletion,' 'freshwater aquatic ecotoxicity,' and 'marine aquatic ecotoxicology' (data not shown), which may account for the high levels of uncertainty displayed in Figure 6.11.

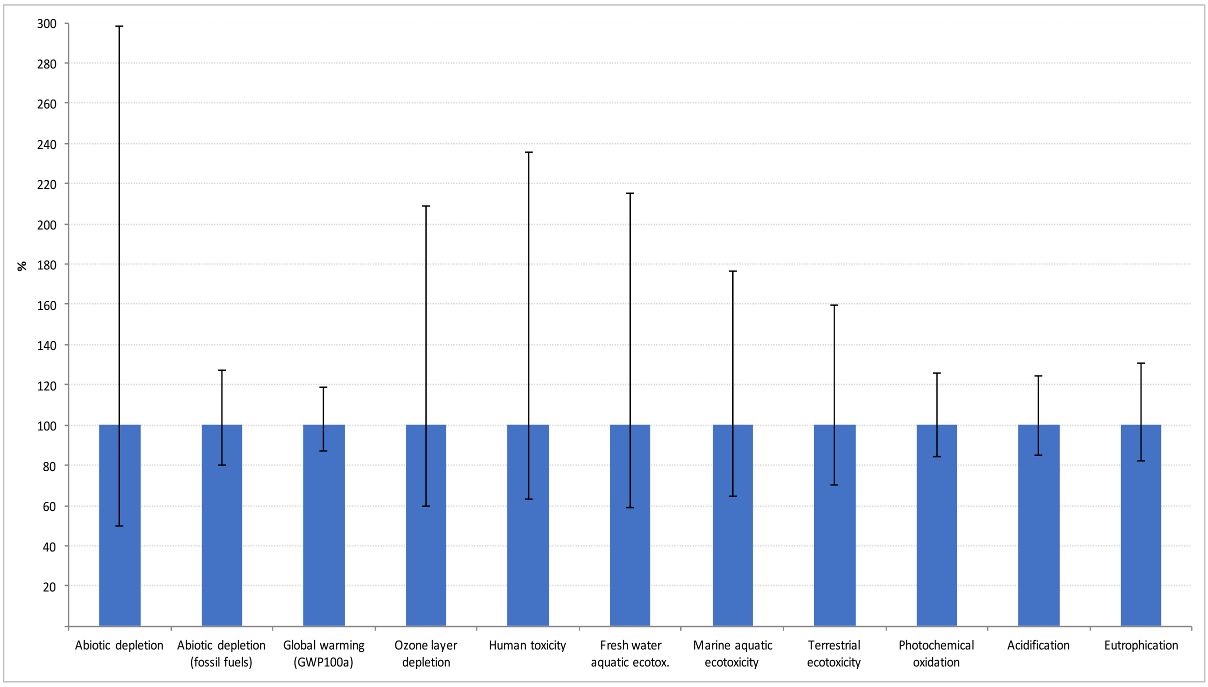


Figure 6.11. Uncertainty ranges (95 % confidence interval) for the impact assessment of the production of 1 kg of salmon smolts. Calculated using the Monte Carlo assessment method, with 1000 test runs.

The 95 % confidence intervals depicted in Figure 6.12 suggest a level of uncertainty within the inventory data of grow-out production that is much lower than what has been found for smolt production. Uncertainty within the results of LCAs are most frequently high, and in comparison, to what is generally expected, the uncertainty estimates for grow-out production are within a range that is currently acceptable. The highest uncertainty range is found in the category 'ozone layer depletion.' The source of uncertainty may be from ecoinvent V.3 processes describing the diesel-powered electricity generator, and transportation processes such as transoceanic ship sea freight (used to represent well-boat transportation), which have high uncertainty ranges for this category (data not shown).

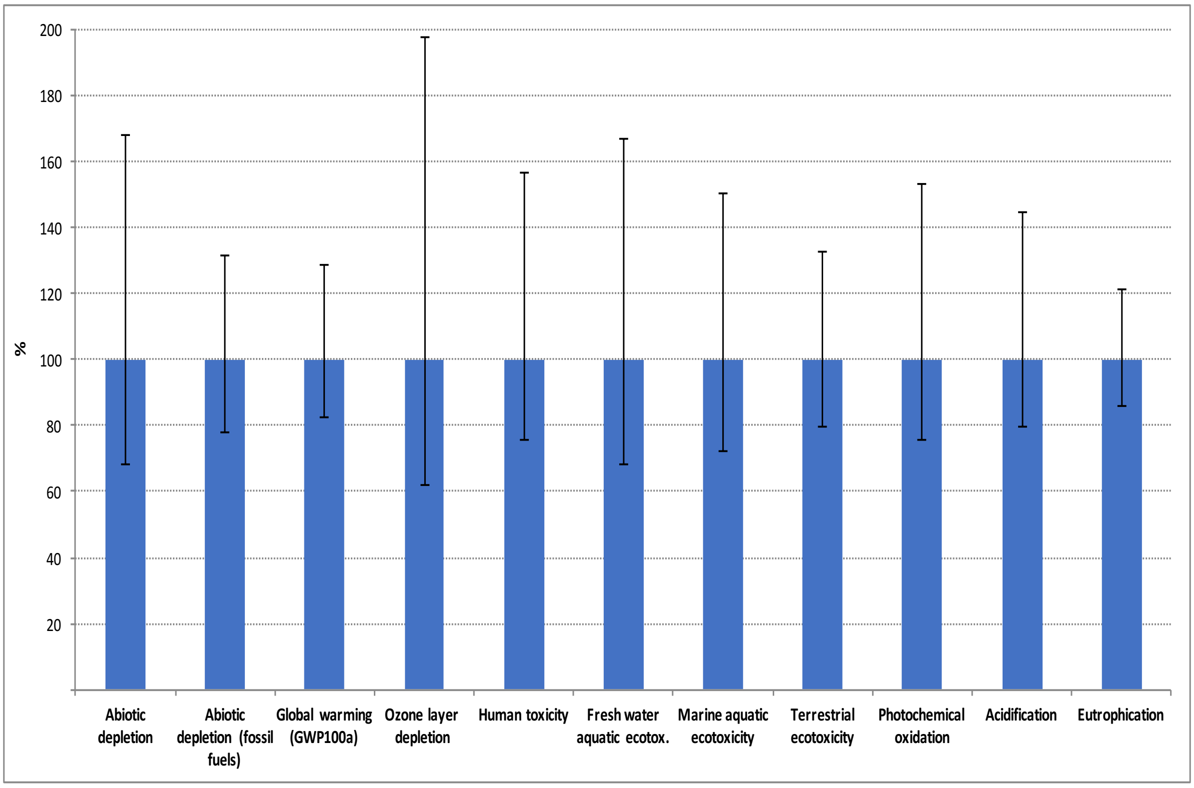


Figure 6.12. Uncertainty ranges (95 % confidence interval) for the impact assessment of the grow-out production of 1 kg of Atlantic salmon. Calculated using the Monte Carlo assessment method, with 1000 test runs.

6.7. Discussion and conclusions

The majority of contributions towards the global scale impacts of cultivated, live weight salmon, as it is available at the farm gate (e.g. pre-slaughter), are from the production of feed. More precisely, as concluded in Chapter 5, these contributions are from the production of ingredients, especially the agricultural production of ingredients. This majority contribution of feed is predictable. It is no coincidence that the provision of feed usually accounts for the majority of financial costs of salmon grow-out phases, as the price of feed, to some extent, is reflective of the quantity of inputs the production of salmon feed requires. Although it would be presumptuous to assume that economic price always reflects the relative impacts of any given product, or that such reflections are accurate, there is some logic in using price as an indicator for predicting which inputs are likely to be the bearers of the majority of burdens. High market prices of an input are, to some extent, evident of high levels of economic activity, thus providing increased opportunity for the production of contributions, although correlations can be distorted by market mechanisms such as supply and demand. In any case, feed is commonly the highest economic cost of producing farmed salmon, and is found here to present its highest potential environmental cost. Previous LCAs of salmon production have also found that feed provision is responsible for the majority of contributions towards all impacts modelled, apart from eutrophication (Pelletier et al. 2009; Boissy et al. 2011; Newton and Little in press). The metabolites of fish growth are found in this study to be responsible for the majority of contributions towards eutrophication, a finding that is also supported by previous studies. Ayer and Tyedmers (2009), Pelletier et al. (2009), and Newton and Little (in press) all found that emissions from fish metabolism accounted for the majority of contributions towards eutrophication, and Boissy et al. (2011) found that the majority are from 'farm running,' which, although ambiguous, appears to be emissions of nitrogen and phosphorous, as well as chemical oxygen demand.

The significant contribution of smolt production was predicted by the preliminary LCA (Figure 6.3), but it is a finding that is in contrast to previous studies (Ayer and Tyedmers 2009; Pelletier et al. 2009). That smolt production has a larger contribution towards certain impacts than does the grow-out production itself (Figure 6.10), is an unexpected finding. Much of the impacts of smolt production are due to contributions from the salt input, as well as the provision of energy carriers. The use of energy carriers is necessary on land-based smolt production facilities, which depend upon mechanical processes to operate the system. However, the high use of salt does appear questionable. Salt use in Scottish smolt production systems is negligible, with products such as the formaldehyde-based solution formalin being more commonly used for therapeutic purposes (Newton, personal communication). The quantity of salt used in the Chilean smolt production system of this study may seem strange, but, nevertheless, the inventory data was acquired directly from the records maintained by the producer, and so would seem to be accurate. Despite this, the contrasting and unexpected nature of the result does warrant further investigation. The data, however precise it may be, is only representative of one out of many such facilities in Chile. Future studies should strive to collect data from a more comprehensive range of facilities. In addition to any uncertainty surrounding the input quantity of salt, the ecoinvent V.3. process 'Sodium chloride powder (ROW) production,' contains some large uncertainty estimates, meaning that the magnitude of contributions from salt production may be significantly overestimated. It may also mean they are significantly underestimated, which would increase the overall contribution from smolt production to the impacts of live weight salmon at the farm-gate. For this reason, if the use of salt in smolt production systems is indeed wide spread throughout Chile, future LCAs should also aim to improve the quality of salt production inventory data.

The contribution of the wellboat transportation of smolts to the grow-out site has only a low contribution to the impact categories. This is in contrast to the findings of the preliminary LCA, but not particularly surprising. The results of the preliminary LCA are based upon many assumptions and incomplete data, and so are only a general guideline at best. Predictably, the impact of ensiling of mortality is only small, and so whether this process is designated as a waste production process, or is considered to be an allocable co-product, will have little influence on the overall results.

In general, excepting the contributions from smolt production, the results of the LCA for live weight salmon, available at the farm-gate, are in accordance with those from other LCA studies. The global production of salmon in net-pens is based upon a common system, and so a high comparability among studies is to be expected. For Chilean industry in particular, this current LCA provides a good analysis which can be used as a base upon which further investigation can be made. Chilean smolt production practices need further assessment, and post farm-gate activities also need to be analysed to identify potential areas where improvement in eco-efficiency is necessary and possible. The adoption of electricity generators with lower emissions is one improvement that should be within reach of most producers that are not suffering from financial difficulties. However, before this can happen, producers in Chile need to understand the value of lowering their carbon dioxide-equivalent emissions.

Finally, as with the results of aforementioned published LCAs of salmon farming, the results of the LCA can be used to challenge the prevalent criticisms the industry faces from its opponents whose arguments focus upon the open-nature of rearing salmon in net-pens. To support my argument, the website, www.farmedanddangerous.org, provides perfect material. When detailing the environmental impacts of salmon farming, the focus of this website is almost entirely upon the grow-out phase of production, and refers, predictably, to the damaged from discharges of faeces and waste-feed, as well as other problems associated with cage systems such as escapees. The progressive use of LCA to analyse the impacts of salmon farming suggests that such a narrow focus is misleading at best. As has been seen in this study, many of the contributions towards impacts upon the marine environment do not come from the grow-out stage. Furthermore, across the suite of impact categories included in this assessment, the majority of impacts are from stages of the value chain other than grow-out production. By focusing upon the potential impacts from grow-out production, the foundations are being laid to promote solutions aimed at eliminating these problems:

"*Closed containment technologies offer a major step forward in fish farming practices. Providing a physical barrier between wild and farmed fish, closed tanks can eliminate, or greatly reduce many of the negative impacts of out-dated net-cage salmon farming*" (www.farmedanddangerous.org).

Life cycle assessment suggests that this argument is entirely missing the point, promoting a case of environmental problem shifting, whereby through an attempt to eliminate impacts from the emissions of net-pens (as well as avoiding other possible, undesirable consequences), emissions are released from an alternative type of production that shifts the impacts from one environmental impact category, to a variety of others.

However, what is more important, is that the results of correctly performed LCA studies are used to inform a sensible direction for the development and improvement of the industry, from an environmental, and economic, point of view. The results of the LCA in this study, as well as those of others, should not merely be of interest to those working in the field of food production LCA. Rather, they should be used to provide a basic understanding of life cycle impacts of salmon farming to scientists who conduct research in the field of aquaculture sustainability, and to inform those involved in the legislation of the industry, by highlighting the dangers of focusing only upon the most immediately obvious stage of production.

Chapter 7: Life Cycle Assessment of Giant Kelp (*Macrocystis pyrifera*) Cultivation.

7.1. Introduction (Production of *Macrocystis pyrifera* in Chile)

The following descriptions are based upon information gathered through my own visits to facilities, and upon the data collected during these times.

The Región de los Lagos (Región X) in Southern Chile, is the site of what is, at the time of writing, the largest kelp farm in the Western hemisphere[22]. Located off the coast of Dalcahue, a small town on the island of Chiloé, 4000 tonnes (ww[23]) of *Macrocystis pyrifera* are harvested each year, equivalent to a yearly production of 200 tonnes (ww) per hectare. The farm is operated as part of a project led by Dr Alejandro Buschmann, of the Centro de Investigación y Desarrollo de Recursos y Ambientes Costeros (i~mar), which aims to develop the cultivation practices of *M.pyrifera*, and use the harvest as a substrate for the production of bioethanol. Somewhat coincidentally, the site is located in proximity to a net-pen, salmonid grow-out facility, and so may be under the influence of fish nutrient discharges.

In addition to the kelp grow-out site, a seed production facility is operated by i~mar, in an area called Metri. In this facility, zoospores are extracted from fertile sporophylls (spore bearing blades), collected from natural populations. A more detailed explanation of this process is described by Guiterrez et al. (2008). The released zoospores are allowed to settle upon ropes, 2mm in diameter, coiled quite tightly around PVC pipes, each with a diameter of approximately 10 cm. The pipes are kept submerged within seawater contained within glass tanks, and sporophyte fronds begin to develop upon the coiled rope, producing a 'seeded cartridge.' This method of growing sporophytes takes place in what is effectively a flow through system, but seawater entering the system must be modified before it is exposed to the juvenile kelp. The inflowing water is filtered and passes through a UV sterilising unit that kills pathogens. Various compounds are added for fertilisation, such as sodium nitrate, glycerophosphate, and vitamins. Eventually, the water is discharged into the sea. After an average period of 56 days, the new fronds are approximately 1 mm in length, and are ready for transplanting to the sea.

[22] Most probably, it is also the largest of all seaweed farms in the Western hemisphere, regardless of the species being cultivated.
[23] ww = wet-weight.

The seeded cartridges are kept in seawater whilst being transported by truck to the grow-out site. The grow-out infrastructure is a long-long line system, consisting of submerged, horizontal ropes upon which the kelp attach their holdfasts, growing in an upwards direction. The main cultivation ropes are maintained at approximately 3 metres depth below the sea surface, and are held in place by a system of anchors and floating buoys. Upon arrival, the small seeded cartridge ropes are unravelled from the pipes, and then wound around the main cultivation ropes of the grow-out system. Seeding the main ropes in this way results in an initial biomass density of 9 kg (ww)/ha. The grow-out cycle is completed within approximately six months, resulting in a total production of 2000 tonnes (ww) of kelp across the site, or 100 tonnes (ww)/ha, equivalent to a yield of 99.99 tonne (ww)/ha, or, roughly, 20 kg (ww)/m of main-line. The kelp is harvested by hoisting the main cultivation lines onto a wooden rack, and the fronts are removed manually, although mechanical alternatives are possible. Other than the occasional checking of fronds and infrastructure, there is no maintenance required throughout the grow-out period.

Figure 7.1. Alejandro Buschmann displaying cultivated *Macrocystis pyrifera*, Chiloé. Photograph is the authors own.

7.2. Goal and scope – Brief Definition

The goal is to produce a life cycle assessment of the production of the above described cultivation system of *Macrocystis pyrifera*. Privileged access to the full production system has been secured through collaboration with Dr Alejandro Buschmann. This provided an opportunity to collect detailed data for every process required in each stage of production. Surveys were constructed and distributed to farm staff, and supplemented through visits to the production facilities.

7.3. Inventory

The quantity of nitrogen phosphorous and carbon removed upon the harvesting of kelp has been based upon the tissue contents of these elements (Table 7.1). The inventory data for the production of one seeded cartridge is shown in Table 7.2. The inventory data for the production of 1 ha / yr^{-1} of *Macrocystis pyrifera* is shown in Table 7.3. This is equal to a production of 200 tonnes ha / yr^{-1}, and a yield of 199.82 ha / yr^{-1} (yield = total production weight – seed input weight).

Table 7.1. *Macrocystis pyrifera* tissue content of elemental carbon, nitrogen and phosphorous.

Nutrient element	***Unit***	***Tissue content***
Carbon	kg / kg (ww)	0.0924
Nitrogen	kg / kg (ww)	0.0046
Phosphorous	kg / kg (ww)	0.0004

Table 7.2. Inventory data for the production of 1 cartridge seeded with *Macrocystis pyrifera*. Process input values in the 'value' column primary data collected from *M.pyrifera* production facilities operated by i-mar.

Product output	**Value**	**Unit**	**Allocation**	
Macrocystis pyrifera seed cartridges	1	piece	100%	
Inputs	**Value**	**Unit**	**Distribution**	**SD^2**
Materials/fuels				
Polypropylene rope	0.01937	kg	Lognormal	1.05
Tap water for cleaning	1505.16	l	Lognormal	1.05
Motor-boat; petrol combustion	0.01100	kg	Lognormal	2
Chilean electricity network	13.7255	kwh	Lognormal	1.05
Transport				
Transport, lorry 7.5-16 tonne, EURO5	0.8289	tkm	Lognormal	2
Outputs to waste treatment				
Waste water	1505.16	l	Lognormal	1.07

Table 7.3. Inventory data for the production of 1 ha / year^{-1} Macrocystis pyrifera, available at the farm-gate. Process input values in the 'value' column primary data collected from *M.pyrifera* production facilities operated by i-mar.

Product output		**Value**	**Unit**	**Allocation**	
Macrocystis pyrifera		1	ha	100%	
Inputs		**Value**	**Unit**	**Distribution**	**SD^2**
Materials/fuels					
Seeded cartridges		200	piece	Lognormal	1.05
Motor boat; petrol combustion		306.6	kg	Lognormal	2
Infrastructure					
Polypropylene rope		168.69	kg	Lognormal	1.07
Polypropylene granulate		13.13	kg	Lognormal	1.05
Injection moulding		13.13	kg	Lognormal	1.07
Steel		2.28	kg	Lognormal	1.05
Steel forging		2.28	kg	Lognormal	1.07
Polystyrene (expandable)		12.41	kg	Lognormal	1.05
Polymer foaming		12.41	kg	Lognormal	1.07
Steel chains		580.16	kg	Lognormal	1.05
Transport					
Transport of infrastructure (truck)		111.12	tkm	Lognormal	2.01
Transport of infrastructure goods (sea)		2690.81	tkm	Lognormal	2.01
Transport of seed to growout (truck)		213.10	tkm	Lognormal	2
Emissions	**Compartment**	**Value**	**Unit**	**Distribution**	**SD^2**
..to water					
Carbon	ocean	-18478.34	kg	Lognormal	1.8
Nitrogen	ocean	-923.92	kg	Lognormal	1.5
Phosphorus	ocean	-83.99	kg	Lognormal	1.8

7.4. Impact assessment

7.4.1. Seed production

The characterised impact assessment for seed production is shown in Figure 7.2. The supply of electricity from the national network system has the greatest contribution across all impact categories, apart from abiotic depletion. Across all impact categories, electricity accounts for between 23 % and 82.38 % contributions (Table 7.4). The treatment of waste water from the system has significant contributions towards the categories abiotic depletion, terrestrial ecotoxicology, and eutrophication. The treatment of waste water has been modelled using a generic ecoinvent V.3. process describing wastewater production, because the modelling of waste treatment scenarios is a particularly difficult aspect of LCA, and requires a level of attention that would be disproportionate within the scope of the study. In rural parts of Chile, such as the site of the seed production facility, the treatment of waste water is often poor, and frequently absent, with wastewater being discharged directly into river systems and the sea. If this is the case at the seed production facility, impacts associated with aquatic environments, such freshwater and marine ecotoxicology, and eutrophication, are likely to be significantly under estimated. Other processes do not feature heavily within the assessment.

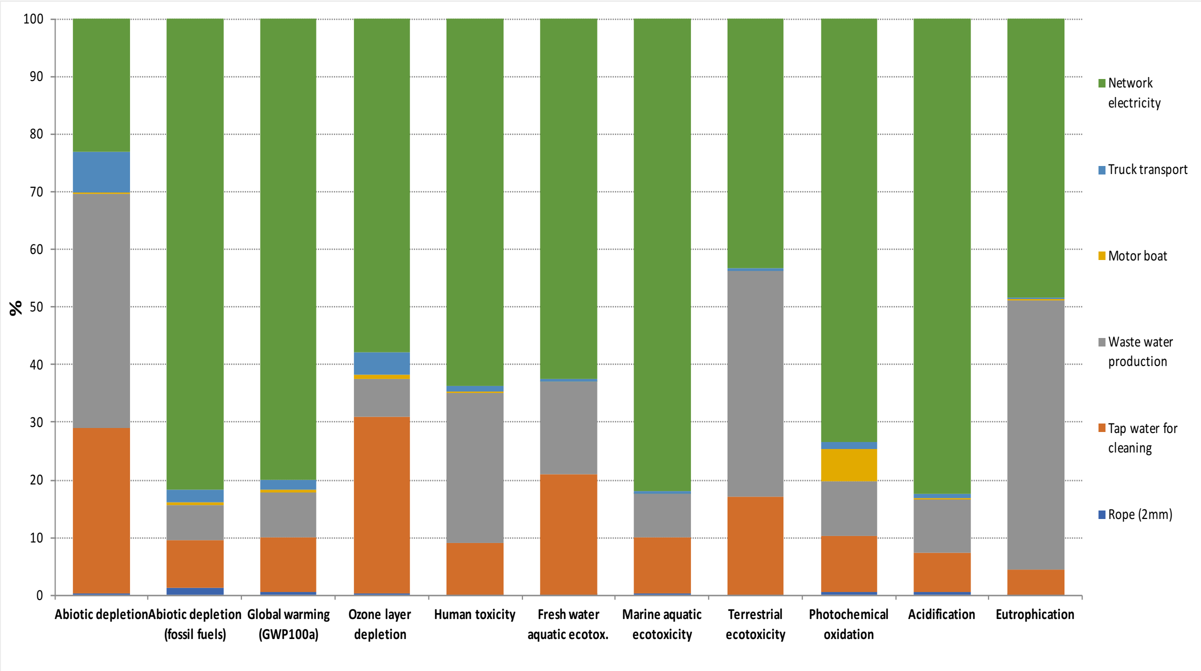


Figure 7.2. Characterised impact assessment results for the production of 1 cartridge seeded with *Macrocystis pyrifera*. Calculated using the CML-IA-baseline method V3.03.

Table 7.4 The contribution that each process required for the production of 1 seeded cartridge, provides towards the total contributions of each impact category.

Impact category	% contribution to total					
	Rope (2mm)	**Network electricity**	**Motor boat**	**Tap water**	**Truck transport**	**Waste water**
Abiotic depletion	0.318	23.100	0.085	28.638	7.088	40.771
Abiotic depletion (fossil fuels)	1.328	81.644	0.500	8.235	2.101	6.192
Global warming (GWP100a)	0.684	79.969	0.428	9.330	1.719	7.870
Ozone layer depletion	0.350	57.767	0.939	30.685	3.843	6.416
Human toxicity	0.207	63.711	0.191	8.878	0.915	26.099
Fresh water aquatic ecotox.	0.190	62.505	0.024	20.791	0.467	16.023
Marine aquatic ecotoxicity	0.303	82.013	0.028	9.803	0.409	7.444
Terrestrial ecotoxicity	0.208	43.151	0.037	16.974	0.519	39.111
Photochemical oxidation	0.622	73.473	5.751	9.623	1.074	9.457
Acidification	0.463	82.384	0.306	6.850	0.780	9.217
Eutrophication	0.121	48.346	0.112	4.268	0.309	46.845

7.4.2. Kelp grow-out production

The characterised impact assessment of the kelp grow-out phase is shown in Figure 7.3 and Table 7.5. Infrastructure is a major contributor towards impacts, although, as with other process, has little contribution towards eutrophication. Excluding eutrophication, infrastructure accounts for between 13.96 % and 88.59 % of contributions towards impacts. Operation of the diesel-powered motorboats has particularly large contributions towards ozone layer depletion (44.35 % of the total) and photochemical oxidation (76.37 % of the total). The contribution of boat operation towards these two categories originate from the burning of diesel in the outboard motor (data not shown). The production of seed also has a significant contribution across impacts. This is perhaps surprising considering its relatively insignificant contribution, in terms of growth, towards the final production of biomass. Towards all impacts except eutrophication, seed production accounts for between 9.44 %, and 48.63 % of total contributions. This is because of the relative intensity of water and energy use, and the quantity of wastewater requiring treatment, per unit weight of algae seed produce. Transportation of seed to the grow-out site is not a particularly efficient process, because only a small truck is used for this purpose, which means that two trips need to be made for each grow-out cycle, as well as two, unloaded return trips. Despite this, the transportation of seed has little contribution towards the overall impacts of production. The negative contribution towards eutrophication (-376.51 phosphate equivalents, data not shown), is quite obviously due to the uptake of nitrogen and phosphorous.

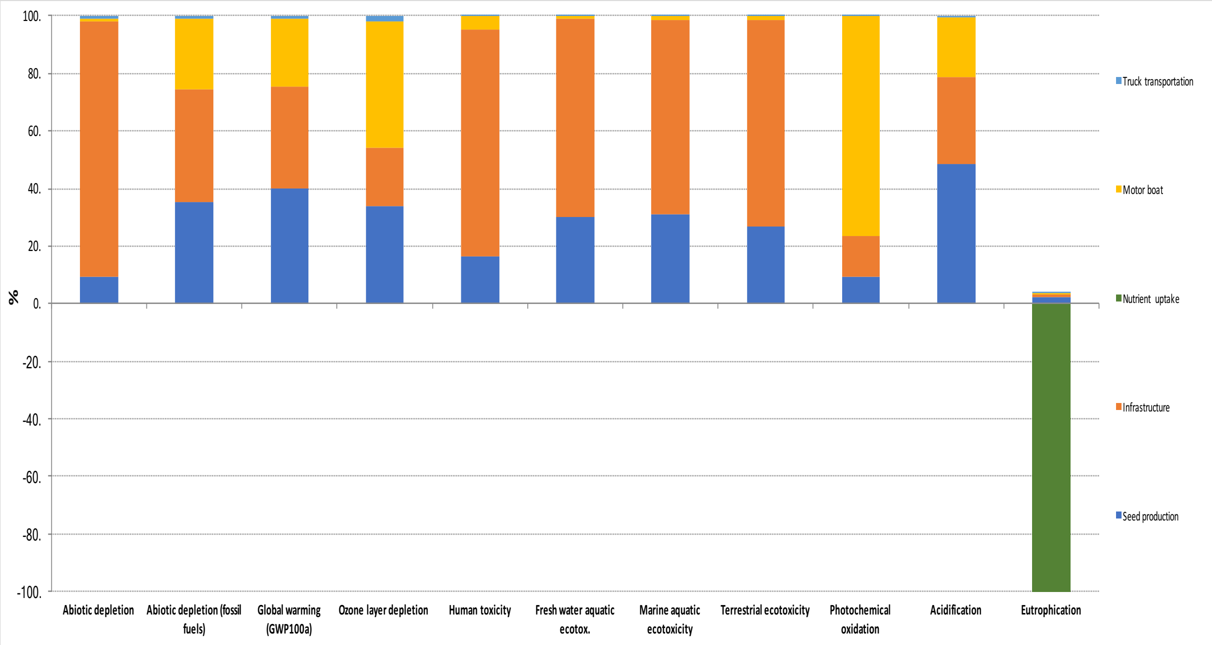


Figure 7.3. Characterised impact assessment results for the production of 1 ha / yr^{-1} of *Macrocystis pyrifera,* available at the farm-gate. Calculated using the CML-IA-baseline method V3.03.

Table 7.5. The contribution that processes required for the production 1 ha / yr^{-1} of *Macrocystis pyrifera,* provide towards the total contributions of each impact category.

Impact category	**% contribution to total**				
	Seed production	**Infrastructure**	**Nutrient uptake**	**Motor boat**	**Truck transport**
Abiotic depletion	9.435	88.588	0.000	1.118	0.860
Abiotic depletion (fossil fuels)	35.385	39.008	0.000	24.651	0.956
Global warming (GWP100a)	39.961	35.335	0.000	23.821	0.883
Ozone layer depletion	33.909	20.065	0.000	44.351	1.675
Human toxicity	16.425	79.014	0.000	4.367	0.193
Fresh water aquatic ecotox.	30.051	68.767	0.000	1.002	0.180
Marine aquatic ecotoxicity	30.977	67.630	0.000	1.230	0.163
Terrestrial ecotoxicity	26.726	71.723	0.000	1.373	0.178
Photochemical oxidation	9.533	13.964	0.000	76.371	0.132
Acidification	48.628	30.174	0.000	20.711	0.487
Eutrophication	2.296	1.082	-103.744	0.357	0.009

7.5. Uncertainty analysis

The uncertainty ranges produced by the Monte Carlo assessment are generally within those typical for LCA characterisation scores. Other than for ozone layer depletion and terrestrial ecotoxicology potentials, the uncertainty ranges are relatively low. This is due to the quality of primary data that has been used to create the inventories of foreground data. The higher ranges of ozone layer depletion and terrestrial ecotoxicology are mainly due to the uncertainty surrounding the transport distances for the delivery of infrastructure processes.

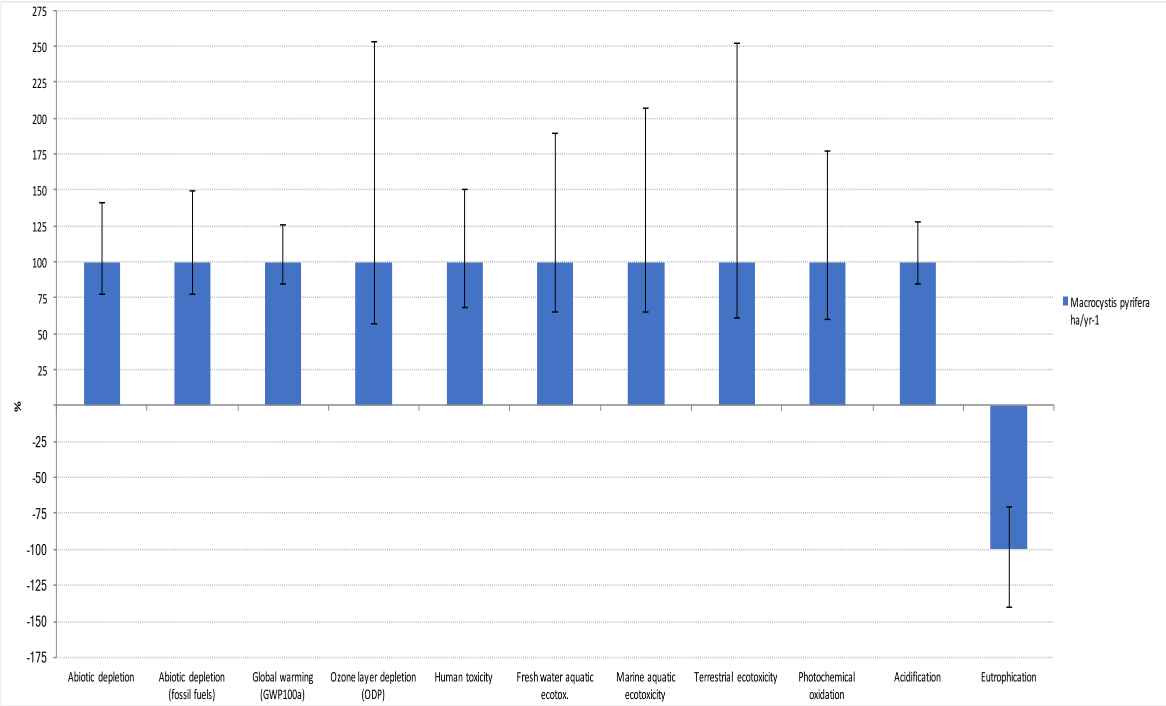


Figure 7.4. Uncertainty ranges of the impact assessment of 1 ha/yr^{-1} of cultivated *Macrocystis pyrifera*. Calculated using the Monte Carlo assessment method, with 1000 test runs.

Table 7.6. Outcome of the uncertainty analysis of 1 ha/yr^{-1} of cultivated *Macrocystis pyrifera*. SD= standard deviation; CV= central value; SEM= standard error of the mean.

Impact category	Unit	Mean	Median	SD	CV	2.5%	97.5%	SEM
Abiotic depletion	kg Sb eq	0.02	0.02	0.00	17.16	0.01	0.03	0.00
Abiotic depletion (fossil fuels)	MJ	74393.08	72207.75	13492.93	18.14	55754.62	107547.42	426.68
Acidification	kg SO2 eq	30.94	30.40	3.31	10.68	25.85	38.94	0.10
Eutrophication	kg PO4--- eq	-632.81	-627.78	112.36	-17.76	-883.25	-444.98	3.55
Fresh water aquatic ecotox.	kg 1,4-DB eq	3188.43	2978.80	1076.54	33.76	1929.75	5655.54	34.04
Global warming (GWP100a)	kg CO2 eq	5217.91	5155.91	525.69	10.07	4372.34	6457.54	16.62
Human toxicity	kg 1,4-DB eq	5477.20	5345.04	1130.83	20.65	3672.55	8059.06	35.76
Marine aquatic ecotoxicity	kg 1,4-DB eq	9.36E+06	8.70E+06	2.96E+06	3.16E+01	5.65E+06	1.80E+07	9.36E+04
Ozone layer depletion	kg CFC-11 eq	0.00	0.00	0.00	44.86	0.00	0.00	0.00
Photochemical oxidation	kg C2H4 eq	5.97	5.70	1.69	28.32	3.44	10.07	0.05
Terrestrial ecotoxicity	kg 1,4-DB eq	30.31	26.47	14.15	46.68	16.08	66.77	0.45

7.6. Discussion and conclusions

Over recent years various life-cycle assessments of seaweed production have been published. In contrast to this study, rather than focusing upon seaweed production itself, they have modelled seaweed cultivation as part of biofuel production scenarios. Perhaps the most relevant of these are those studies by Langois et al. (2012), Aitken et al. (2014), and Czyrnek-Delêtre et al. (2017). In particular, Aiken et al. (2014) provides an LCA of *Macrocystis pyrifera*, and indications from this publication suggest it is based, at least in principle, upon the same system as used in this study.

However, these studies do not provide the same level of detail for seed and grow-out production, and so it is difficult to make any meaningful comparisons between their results and those of produced in this LCA. Due to the nature of grow-out production, it can be assumed that, as with this study, provision of infrastructure and the operation of boats will have been major contributors to these production phases. However, it would have been particularly interesting to compare the contribution of seed production towards the impacts of grow-out production, between studies.

The significant contribution of the seed production phase towards the final harvested product, may be an issue of scale. The grow-out system is (at the time of writing), the largest cultivation of any kelp species within the western hemisphere, and the seed facility has been designed specifically for its supply. However, in comparison to what would be required for a commercially profitable business, it is still quite small. It is certainly comparatively tiny compared to the scale of open-water kelp aquaculture like that to be found in China (Ferreira et al. 2008). The seed site itself is only small, and the input intensity per unit of production is high. Assuming the rules of economies of scale apply, a properly designed, larger scale facility, should have an improved environmental profile.

At first glance, the negative contribution towards eutrophication (-376.51 phosphate equivalents, data not shown) may seem a desirable result. But if a cultivation is located within a previously well-functioning nutrient environment, the removal of nitrogen and phosphorus may cause problems. This is especially true if the cultivations are of a large scale, and in marine environments such as bays, with low tidal exchange. In this scenario, removal of large amounts of nitrogen, phosphorous and also carbon, could adversely affect ecosystem functioning. The practical response to this issue seems quite obvious. Kelp cultivation should be performed where there is a need to remove nutrients, or where the cultivation is expected to have a negligible impact upon its environment. Relevant to this theme, is a recent LCA that analyses end-use scenarios of cultivated kelp (*Saccharina latissima*), such as fertiliser production and it application (Seghetta et al. 2016). Based upon the principle of circular nutrient management (closed-loop management), this LCA shows that kelp cultivations could form part of a management system with a net-reduction in eutrophication.

Finally, it is worth mentioning the link between kelp carbon sequestration and climate change impacts. Some authors have proposed that large-scale cultivation of seaweed may reduce the carbon levels of surface water, to the extent that it results in a sequestration of atmospheric CO_2 (Hughes et al. 2012; Tang et al, 2011). As discussed in Roberts et al. (2015), this idea isn’t validated to the extent where it should be included within a realistic assessment of cultivated kelp. For this reason, it isn’t considered as part of this study.

Chapter 8: Life Cycle Assessment of cultivated, Chilean Blue Mussel (*Mytilus chilensis*).

8.1. Introduction

The cultivation of bivalve molluscs in an important sector of the Chilean aquaculture industry. *Mytilus chilensis* is by far the most commonly cultivated species of bivalve, and with a production of 300648 tonnes, it accounted for 97.97 % of the total quantity of bivalves produced in 2016 (Sernapesca 2017). Of this production, 62005 tonnes were destined for export, with a value of over 163 million USD FOB (Subpesca 2017). Over 99 % of recorded mussel production takes places within the Region de Los Lagos (Sernapesca 2017). The cultivation of *M.chilensis* ranges from large, commercial scale production for exportation, to very small scale production intended for sale within local markets (personal observation).

The cultivation cycle of *Mytilus chilensis* is quite simple. Mussel seed is usually collected by natural settlement of mussel spat upon drop-ropes suspended from horizontal mother ropes. Once the spat reach a suitable size, they are transplanted to a grow-out site, where they are placed in mesh bags surrounding drop ropes, to which they eventually attach, before the mesh bags break up and fall way. They are then allowed to grow for a period of approximately 18 months, before being harvested. The production of mussels at a commercial scale is a mechanised process, with specialised equipment being used for preparing the cultivation infrastructure and for harvesting the crop.

8.2. Goal and Scope

The goal of this study is to produce a life-cycle assessment of the production of 1 kg of harvested, whole (shell-on) *Mytilus chilensis*, as it is available at the farm gate in Chile. The methods used and system boundaries are those described in Chapter 4. The data collection phase of this study was completed as part of a collaboration with AVS Chile S.A. Stakeholder conferences where held, during which the purposes of collecting life cycle data was explained to interested companies. A variety of companies took part, and a variety of data were collected for the different phases of mussel production, including post-harvest processing (although this phase does not feature as part of this study). All of the companies providing data are commercial scale producers.

8.3. Inventory

The quantity of elemental carbon, nitrogen and phosphorous removed upon harvest is calculated by their quantity the mussel meat and shell, per wet-weight mussel (Table 8.1). The meat yield of 30 % and shell yield of 70 % is based upon Fuentes et al (2009). A moisture content of 80.25 % of total meat weight is assumed, based upon the central value of a range of values provided by Fuentes et al. (2009). Carbon and nitrogen content of tissue is based upon a central value of ranges provided Vernocchi et al. (2007). Carbon content is based upon the central value of a range provided by Stirling and Okumus (1998). Carbon content of shell is based upon the molecular weight of $CaCO_3$, assuming that 95 % of the shell is made of this compound.

Table 8.1. Carbon, nitrogen and phosphorous content of *Mytilus chilensis* tissue (meat), and carbon content of shell. Carbon content of is shell calculated based upon the molecular weight of carbon in $CaCO_3$. Nitrogen and Phosphorous content based upon the central value of ranges provided by Vernocchi et al. (2007). Carbon content is based upon the central value of the range provided by Stirling and Okumus (1998).

Nutrient element	***Unit***	***Tissue content***	***Shell content***
Carbon	kg / kg (ww)	0.0138	0.0798
Nitrogen	kg / kg (ww)	0.0047	-
Phosphorous	kg / kg (ww)	0.0007	-

Table 8.2. Process inventory data for the grow-out production 1 kg (wet weight) of harvested, whole *Mytilus chilensis*, available at the farm gate. Input values are the geometric means of populations collected from data provided by a variety of sources.

Product output		**Value**	**Unit**	**Allocation**	
Mytilus chilensis		1	kg	100%	
Inputs		**Value**	**Unit**	**Distribution**	**SD^2**
Materials/fuels					
Mussel seed production		0.1763	kg	Lognormal	1.07
Mussel working; petrol combustion		0.000403	kg	Lognormal	1.07
Outboard-motor boat; petrol combustion		0.004789	kg	Lognormal	1.07
Infrastructure					
Polypropylene rope		0.0116	kg	Lognormal	1.07
Concrete block		0.04298	kg	Lognormal	1.07
Cotton mesh bag		0.00342	kg	Lognormal	1.09
Steel		0.00011	kg	Lognormal	1.09
Polyethylene high density (bouy)		0.00350	kg	Lognormal	1.09
Transport					
Transport of Infrastructure goods (truck)		0.00154	tkm	Lognormal	2.05
Transport of seed to growout (truck)		0.042	tkm	Lognormal	2.01
Transport of seed to growout (sea)		0.030	tkm	Lognormal	2.01
Emissions	**Compartment**	**Value**	**Unit**	**Distribution**	**SD^2**
..to water					
Carbon	ocean	-0.09360	kg	Lognormal	1.53
Nitrogen	ocean	-0.00465	kg	Lognormal	1.53
Phosphorus	ocean	-0.00069	kg	Lognormal	1.53

8.4. Impact Assessment

8.4.1. Mussel seed production

The characterised impact assessment for mussel seed production is shown in Figure 8.1 and Table 8.3. Provision of infrastructure is the process with the largest contribution to all impact categories apart from photochemical oxidation. Across all categories, it contributes between 1.76 % (eutrophication) and 96.3 % (marine aquatic ecotoxicology) towards the total impact. The majority of infrastructure related impacts are from the production of polypropylene rope. Rope accounts for between 49 % and 82 % of the contributions from infrastructure across all categories (Table 8.4). The production of expanded polystyrene for use within buoys accounts for 43.4 % of the contributions from infrastructure towards its photochemical oxidation potential. Operation of the diesel-powered motorboat contributes significantly towards ozone layer depletion (46.2 % of the total) and photochemical oxidation (63.2 % of the total).

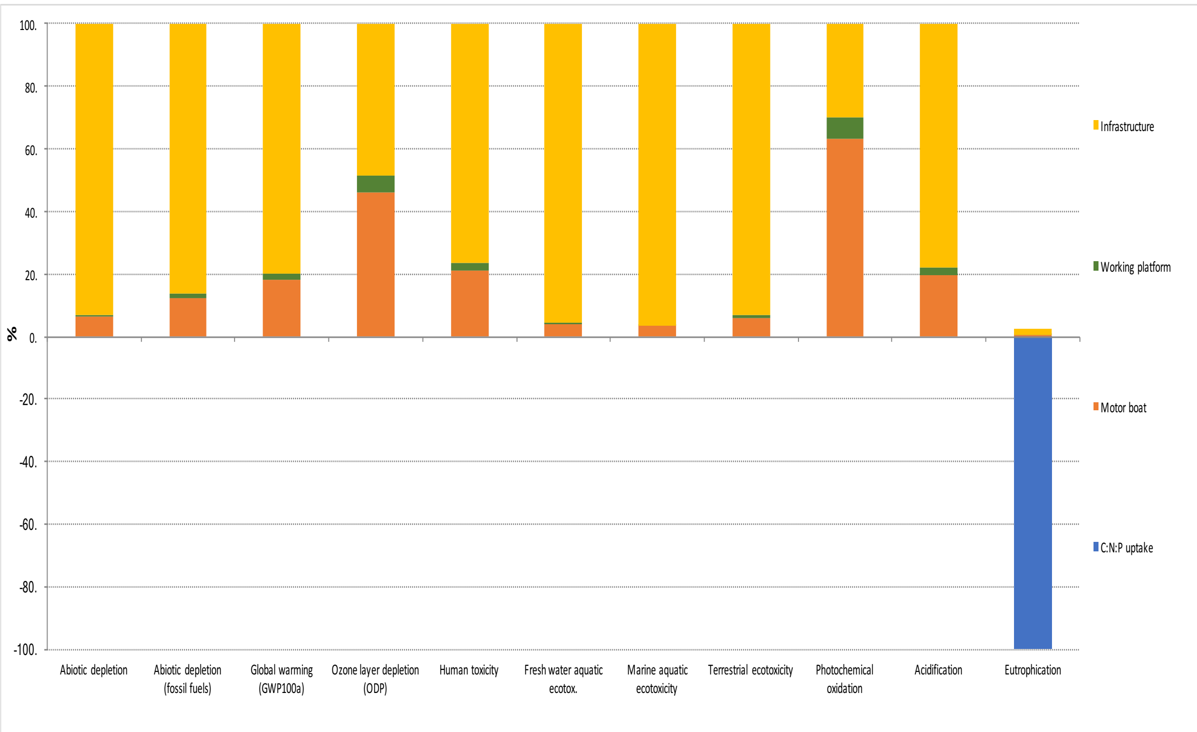


Figure 8.1. Characterised impact assessment of the production of 1 kg of mussel seed (*Mytilus chilensis*). Calculated using the CML-IA-baseline method V3.03.

Table 8.3. Contribution analysis for 1 kg of mussel seed (*M.chilensis*) available at the farm gate. Contributions expressed as a percentage of the contributions towards each impact category.

Impact category	% contribution to total			
	C:N:P uptake	Motor boat	Working platform	Infrastructure
Abiotic depletion	-	6.184	0.687	93.129
Abiotic depletion (fossil fuels)	-	12.333	1.370	86.296
Global warming (GWP100a)	-	18.231	2.026	79.744
Ozone layer depletion (ODP)	-	46.243	5.138	48.619
Human toxicity	-	21.243	2.360	76.397
Fresh water aquatic ecotox.	-	4.068	0.452	95.480
Marine aquatic ecotoxicity	-	3.301	0.367	96.332
Terrestrial ecotoxicity	-	6.020	0.669	93.311
Photochemical oxidation	-	63.237	7.026	29.736
Acidification	-	19.804	2.200	77.995
Eutrophication	-102.396	0.574	0.064	1.758

Table 8.4. Contribution of infrastructure components, to the total impacts of infrastructure that is required for the production of 1 kg of mussel seed.

Impact category	% contribution to total					
	Rope (PP)	Concrete Anchor	Steel	Buoy (HDPE)	Buoy (EPS)	Transport
Abiotic depletion	53.021	28.672	6.284	5.531	4.040	2.451
Abiotic depletion (fossil fuels)	80.929	1.576	0.099	7.188	9.941	0.266
Global warming (GWP100a)	77.905	4.689	0.233	6.158	10.609	0.407
Ozone layer depletion (ODP)	75.591	9.050	0.308	6.315	7.012	1.724
Human toxicity	64.163	12.213	6.325	6.565	10.144	0.590
Fresh water aquatic ecotox.	72.172	7.797	2.672	6.808	10.183	0.368
Marine aquatic ecotoxicity	77.636	4.630	1.069	5.753	10.694	0.218
Terrestrial ecotoxicity	77.601	8.342	4.238	4.385	5.031	0.402
Photochemical oxidation	49.043	2.242	0.319	4.838	43.383	0.176
Acidification	82.047	3.415	0.246	5.363	8.643	0.287
Eutrophication	75.328	6.699	1.059	7.605	8.908	0.400

8.4.2. Mussel grow-out production

The characterised impact assessment for mussel grow-out production is shown in Figure 8.2 and Table 8.5. As is the case for mussel seed production, the provision of infrastructure accounts for the majority of impacts towards all categories except photochemical oxidation. Across all impact categories it accounts for between 5.85 % and 99.5 % of the total contributions. The majority of the contributions from infrastructure provision, come from the production of the cotton mesh bags that are used to keep the mussel seed attached to the drop-ropes when they are first transplanted to the grow-out site. Across all impacts, the mesh bags account for between 37.2 % and 99 % of the total contribution from infrastructure (Table 8.6). In comparison, polypropylene rope is responsible for 0.527 % and 46.9 % of these contributions. Operation of the diesel-powered motor boats accounts for 52.9 % of contributions towards photochemical oxidation, but in general, its contributions are dwarfed by those from infrastructure.

The overall contributions 'towards' the eutrophication potential of mussels, have a negative value. Clearly, this is due to the removal of nitrogen and phosphorous upon the harvest of mussels. It is interesting that this uptake of nutrients is sufficient to more than compensate for the total amount of contributions towards eutrophication from the other processes, such as infrastructure provision. As the data show, a proportion of this uptake comes from the production of seed, and both seed production and grow-out production results in a net-reduction of phosphate equivalents (the standardised unit of emissions with a eutrophication potential).

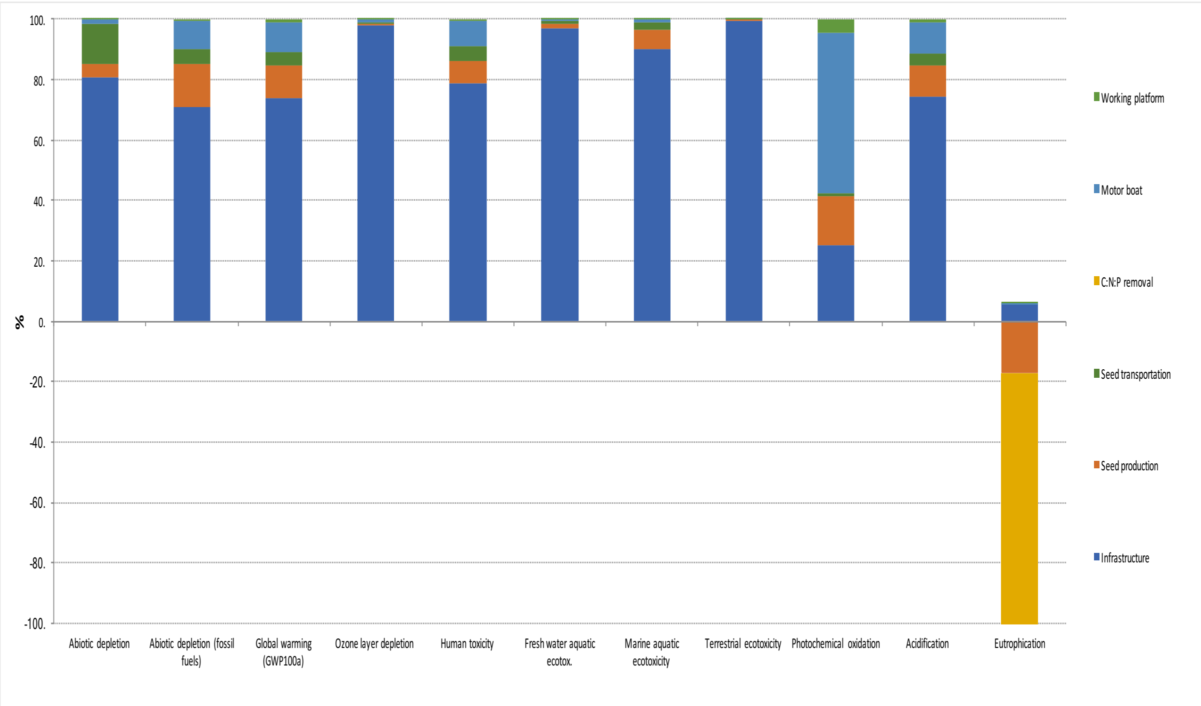


Figure 8.2. Characterised impact assessment of the production of 1 kg of whole *Mytilus chilensis*, available at the farm gate. Calculated using the CML-IA-baseline method V3.03.

Table 8.5. Contribution analysis for 1 kg of whole *M.chilensis* available at the farm gate. Contributions expressed as a percentage of the contributions towards each impact category.

Impact category	% contribution to total					
	Infrastructure	Seed production	Seed transportation	C:N:P uptake	Motor boat	Working platform
Abiotic depletion	80.789	4.295	13.416	-	1.383	0.116
Abiotic depletion (fossil fuels)	71.188	14.262	4.618	-	9.161	0.771
Global warming (GWP100a)	74.083	10.417	4.778	-	9.890	0.832
Ozone layer depletion (ODP)	97.994	0.419	0.495	-	1.008	0.085
Human toxicity	78.765	7.459	4.831	-	8.252	0.694
Fresh water aquatic ecotox.	96.883	1.810	0.892	-	0.383	0.032
Marine aquatic ecotoxicity	89.988	6.742	2.013	-	1.159	0.098
Terrestrial ecotoxicity	99.450	0.293	0.157	-	0.092	0.008
Photochemical oxidation	25.256	16.066	1.318	-	52.909	4.451
Acidification	74.160	10.355	3.908	-	10.679	0.898
Eutrophication	5.849	-18.441	0.196	-88.201	0.551	0.046

Table 8.6. Contribution of infrastructure components, to the total impacts of infrastructure that is required for the grow-out production of 1 kg of *M.chilensis*.

Impact category	% contribution to total					
	Rope (PP)	Concrete Anchor	Steel	Buoy (HDPE)	Cotton mesh	Transport
Abiotic depletion	8.804	7.550	1.525	3.105	78.407	0.609
Abiotic depletion (fossil fuels)	46.929	1.450	0.084	14.092	37.214	0.231
Global warming (GWP100a)	29.299	2.796	0.128	7.829	59.719	0.229
Ozone layer depletion (ODP)	0.527	0.100	0.003	0.149	99.204	0.018
Human toxicity	15.570	4.699	2.243	5.385	71.889	0.214
Fresh water aquatic ecotox.	4.318	0.740	0.234	1.377	93.299	0.033
Marine aquatic ecotoxicity	18.795	1.777	0.378	4.708	74.263	0.079
Terrestrial ecotoxicity	0.716	0.122	0.057	0.137	98.963	0.006
Photochemical oxidation	31.115	2.255	0.295	10.377	55.791	0.167
Acidification	29.968	1.978	0.131	6.622	61.144	0.157
Eutrophication	14.006	1.975	0.288	4.781	78.840	0.111

8.5. Uncertainty analysis

The results of the uncertainty analysis (Figure 8.3 and Table 8.7) show that the uncertainty is well within the range for what is common in LCA impact assessments. In fact, the majority of ranges are relatively low. These low levels of uncertainty are due to the quality of primary data used within the inventory.

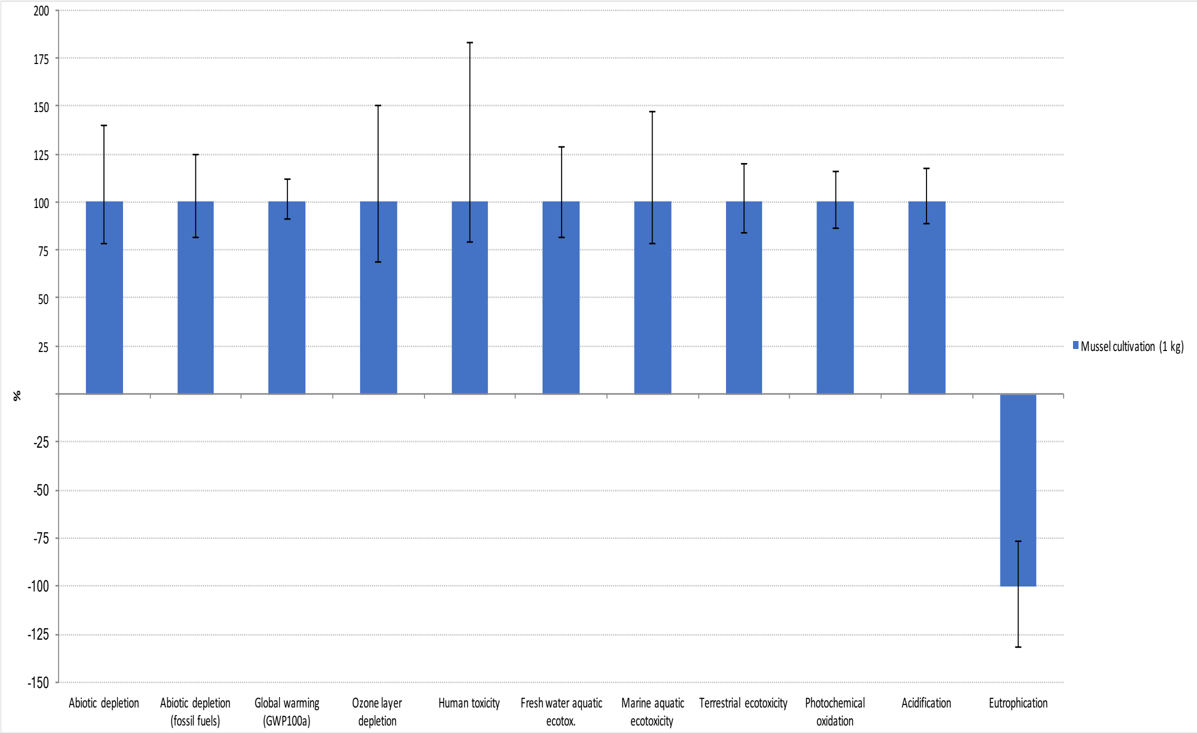


Figure 8.3. Uncertainty ranges of the impact assessment of 1 kg of cultivated *Mylilus chilensis*. Calculated using the Monte Carlo assessment method, with 1000 test runs.

Table 8.7. Outcome of the uncertainty analysis of 1 kg of cultivated *M.chilensis*. SD= standard deviation; CV= central value; SEM= standard error of the mean.

Impact category	Unit	Mean	Median	SD	CV	2.5%	97.5%	SEM
Abiotic depletion	kg Sb eq	2.35E-07	2.30E-07	3.70E-08	1.57E+01	1.79E-07	3.22E-07	1.17E-09
Abiotic depletion (fossil fuels)	MJ	3.15E+00	3.13E+00	3.52E-01	1.12E+01	2.55E+00	3.89E+00	1.11E-02
Acidification	kg SO2 eq	9.41E-04	9.30E-04	7.27E-05	7.72E+00	8.26E-04	1.09E-03	2.30E-06
Eutrophication	kg PO4--- eq	-3.78E-03	-3.74E-03	5.29E-04	-1.40E+01	-4.91E-03	-2.86E-03	1.67E-05
Fresh water aquatic ecotox.	kg 1,4-DB eq	1.33E-01	1.31E-01	1.72E-02	1.30E+01	1.06E-01	1.68E-01	5.45E-04
Global warming (GWP100a)	kg CO2 eq	1.97E-01	1.96E-01	1.06E-02	5.38E+00	1.78E-01	2.20E-01	3.34E-04
Human toxicity	kg 1,4-DB eq	4.68E-02	4.41E-02	1.18E-02	2.53E+01	3.49E-02	8.08E-02	3.74E-04
Marine aquatic ecotoxicity	kg 1,4-DB eq	1.59E+02	1.54E+02	2.80E+01	1.76E+01	1.21E+02	2.27E+02	8.86E-01
Ozone layer depletion	kg CFC-11 eq	3.41E-07	3.33E-07	6.96E-08	2.04E+01	2.29E-07	5.00E-07	2.20E-09
Photochemical oxidation	kg C2H4 eq	1.34E-04	1.33E-04	1.02E-05	7.65E+00	1.15E-04	1.54E-04	3.23E-07
Terrestrial ecotoxicity	kg 1,4-DB eq	7.06E-03	7.05E-03	6.47E-04	9.17E+00	5.91E-03	8.44E-03	2.05E-05

8.6. Discussion and conclusions

This is not the first time LCA has been used for calculating the potential global scale impacts of cultivated mussel production. A series of publications have included the life-cycle assessment of seafood products from *Mytilus galloprovincialis* cultivated in the Galician region of North West Spain (e.g. Irribaren et al. 2010a; Iribarren et al. 2010b; Lozano et al. 2010). However, these LCAs focus upon product end-use scenarios, such as fresh, frozen and canned-mussel production and consumption. As a result of this focus, they do not provide a detailed LCA analysis of seed production and grow-out production needed to be useful for comparison with this study. However, the thesis produced by Iribarren (2010), does provided a sufficiently detailed LCA of mussel grow-out production, useful for this purpose. As with this study, Iribarren (2010) finds that the majority of impacts come from the provision of capital goods (infrastructure), and that, in general, the contributions from operation processes (diesel combustion etc.) are, much lower. However, it is interesting that the contribution of cotton mesh bags towards these impacts does not appear to be of the same magnitude as is found in this study. This could be partly explainable by the rate of cotton use being 0.27 g / kg of mussel harvested, which is much lower than the 3.42 g / kg og mussels cultivated in Galicia, which are grown upon ropes attached to wooden rafts, and so the quantities and types of materials differ with those of this study.

The negative net-contribution of mussel production towards the category eutrophication, must be interpreted within the same context as discussed for the similar result for the production of kelp (previous chapter). That is, the net-removal of nutrients is not necessarily a good thing, and can lead to unwanted environmental impacts. The use of cultivated mussels as a substrate for fertiliser production has been assessed as a method of removing nutrients from the sea for their application to agricultural systems (Spångberg et al. 2013). This might prove beneficial within the context of closed loop, nutrient management systems, although much more research would be required to understand the potential environmental consequences of such a system.

Chapter 9: A Comparative Assessment Between the Life-Cycle Impacts of Chilean, Marine Open-Water, Integrated Multi-Trophic Aquaculture Systems, and the Monoculture of Atlantic Salmon.

9.1. Introduction

There are no examples of intentionally developed IMTA systems in the marine coastal waters of Chile, but there are numerous, although unquantified and mainly unrecorded occurrences, of species from different trophic levels being cultivated as monoculture, incidentally placed within close proximity to one another. Similar to the development of some IMTA systems in China, but of a much lesser scale, unintended IMTA systems have occurred in Chile through a historic lack of restriction upon the number of aquaculture sites that can be placed within a particular area (see Buschmann et al. 2009 and Chapter 3 of this thesis). Resultantly, examples of seaweeds cultivated next to mussel or salmonid farms, and mussel farms being placed close to salmonid cultivations, can be found throughout the Región de Los Lagos (Región X), and those regions further south where these species types are also cultivated. Within Región X alone, there are some notable examples. Within Metri Bay, the agarophytic rhodophyte, *Gracilaria chilensis*, is grown in a simple 'bottom culture.' Also in this same bay, the 'Steelhead' variant of rainbowtrout (*Onchorhynchus mykiss*) is reared within an on-growing facility. This example of IMTA is the same site as described in the research of Troell et al. (1997) and Abreu et al. (2009), although their work focused on experimental long-line cultivations of *G.chilensis*, which are not permanent features. More significantly, the 20 ha cultivation of the '*giant kelp*' *Macrocystis pyrifera* (described in Chapter 2, and in Buschmann et al. 2014), situated of the coast of Chiloé, is another example of seaweed being grown within proximity to the cage rearing of *O.mykiss*. In various locations, especially of the coast of Chiloé, cultivations of the Chilean blue mussel, *Mytilus chilensis*, can be found growing within a proximity to Salmonid grow-out sites that would not be permitted in salmon producing countries such as Canada and Scotland (personal observation). Coastal cultivations of salmonids and *M.chilensis* are very common in Southern Chile, with both being particularly concentrated within Región X, allowing for the possible of numerous, undocumented instances of co-cultivation and potential bioremediation.

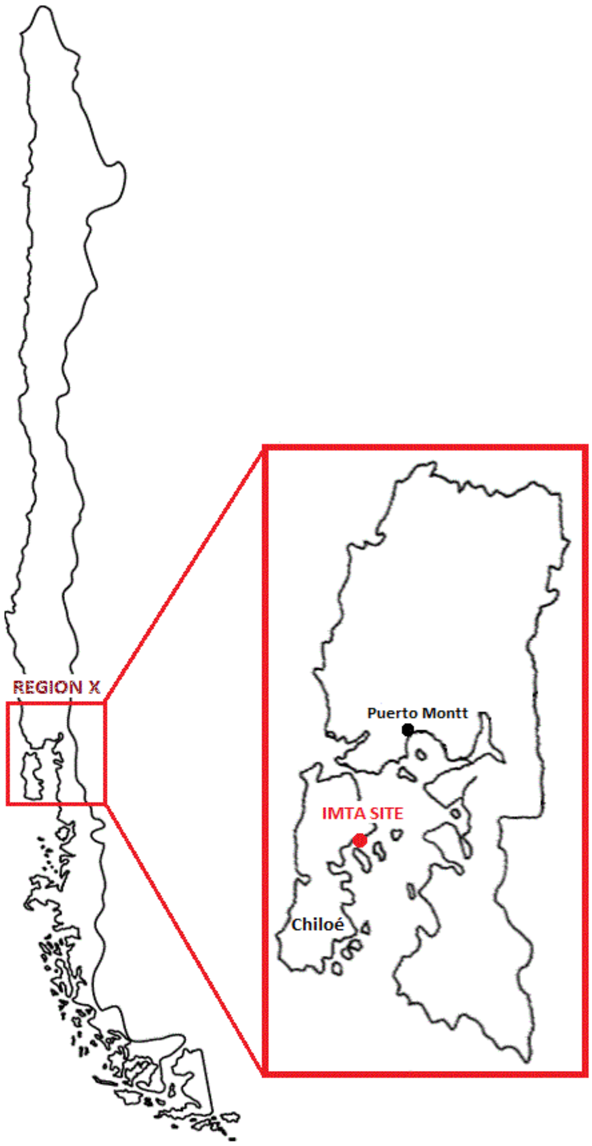


Figure 9.1. Map of Chile showing Región X, the central hub of Chilean aquaculture activity. The IMTA site, which consist of several salmonid farms and the grow-out site of *M.pyrifera* operated by i-mar, is located on the coast of the Chiloé peninsula.

Realising that the above scenario holds opportunities for studying and developing IMTA systems (e.g. Buschmann et al. 2009), research initiatives co-ordinated by the phycologists Dr Alejandro Buschmann and Dr Alfonso Gutierrez both once directors of the Centro de Investigación y desarollo en Recursos y Ambientes Marinos (i~mar), have focused upon various aspects of seaweed-salmonid integration. This research facility has been involved with, by far, the majority of publications within the subject of IMTA in Chile. Additional to investigating the potential for bioremediation both within open-water (e.g. Troell et al. 1997; Buschmann et al. 2008; Abreu et al. 2009) and land-based systems (e.g. Buschmann et al 1994; Buschmann et al 2001), research has also focused upon the development of seaweed production systems (e.g. Halling et al. 2005; Gutierrez et al. 2006), possible negative consequences of seaweed cultivation and upon various ways of utilising seaweed as a product, such as plant feed supplements (Buschmann et al. 2005), food for human consumption (Gutierrez et al. 2006), feed for abalone cultivations (Correa et al. 2016), and bioethanol production (Wargacki et al. 2012).

The need for an adequate market demand is a prerequisite for successful development of IMTA in any country (see Chapter 2). Demand for Chilean farmed salmon, mussels and seaweed already exists, and so IMTA in Chile may, in theory, be consolidated further through the coupling of already existing production units when it is physically and legally possible to do so. However, further increases in IMTA production will essentially be dependent upon, in addition to various other factors, the creation of an enlarged demand for its products. Toward this end, the development of bioethanol is a particularly notable aspect of research that takes place within i~mar facilities. The idea of growing *M.pyrifera* close to salmonid farming is being investigated as a way of utilising the nutrient emissions of salmon farming to produce a substrate, in the form as kelp, to be converted to ethanol. Additional to the already operative aforementioned pilot cultivation, a facility for ethanol production has been constructed. The production of *M.pyrifera* has been very successful, but despite commercial involvement and US government funding[24], the production of ethanol has not taken place, and BAL Biofuels S.A., the company leading the bioethanol research arm of the project, has mostly abandoned it prospects for using Chilean grown kelp to produce fuel. Although the use of IMTA grown *M.pyrifera* to produce biofuel may seem a perfect scenario for providing the demand needed to stimulate expansion of IMTA, this project, and others like it, are unlikely to be successful without a significant technological advance. The reasons for this are not the subject of this thesis, but instead have been discussed in my work produced as part of a recent report commissioned by the Scottish Aquaculture Research Forum (see Roberts et al 2015).

The potential environmental benefits of IMTA have been widely covered, and most research has been conducted towards this end (see Chapter 2). However, the potential undesirable environmental impacts of IMTA has received comparatively little attention. Studies focusing upon environmental impacts have focused mainly upon local scale effects (e.g. Buschmann et al. 2014), but little is known about the potential contributions of IMTA towards global scale impacts. The success of IMTA depends not only upon factors such as financial viability and the flexibility of regulation. It is essential that IMTA has an environmental impact profile that enables an acceptable balance between environmental costs and benefits. From this perspective, IMTA is not a proven concept.

[24] The research published by Wargacki et al. 2012, was supported by the U.S. government Department of Energy, and was coordinated by Santiago based BAL Biofuels S.A, with i~mar providing assistance as well as being responsible for the production of *M.pyrifera* biomass.

9.2. Goal and Scope – Brief Definition

The goal of this study is to produce a life cycle assessment of marine based, open-water integrated multi-trophic aquaculture systems in Chile. The results will be discussed within the context of trade-offs between the environmental costs and benefits of IMTA production, with the intention of providing a much needed contribution to our understanding of this subject.

The modelling of IMTA grow-out systems for the purposes of assessing their life-cycle impacts can be done in a variety of ways, based upon a variety of perspectives. As an example, it is possible to treat seaweed and mussels as secondary co-products of salmon production. This could be modelled by setting the functional unit as 1 kg of salmon, and then allocating a proportion of contributions to seaweed and mussels, using a selected allocation factor (e.g. mass-adjusted economic value). It can be predicted that this will increase the contributions of salmon production towards most impacts, although contributions towards eutrophication potential may decrease. By modelling the system in this way, some of the contributions from the co-production of kelp and mussels will be assigned to the functional unit, depending upon the numerical value of the allocation factor. Modelling the system in this way may seem counterintuitive to some observers, but it may be considered as intuitive if salmon production is considered to be the main economic product, the production of which necessitates the co-cultivation of the secondary products, mussels and kelp. On the surface at least, this appears to be an argument based upon subjective reasoning rather than any solid scientific or material basis. However, these dilemmas are the basis of dealing with the very frequent incidences of multifunctionality that occur as part of industrial production systems. There are various other ways of dealing with the multifunctional nature of IMTA, and a discussion of these could be a publication in its own right. Various options have been explored as part of this research, but rather than explain at length, I will describe only the model employed, and revisit the subject in the discussion and conclusions section, when relevant.

The assessment will be based upon various functional units describing three products, in the form as they are available at the farm-gate. These products are farmed Atlantic salmon (*Salmo salar*, farmed giant kelp (*Macrocystis pyrifera*), and farmed Chilean blue mussels (*Mytilus chilensis*), the life cycle assessments of which, are the subject of Chapter 6, Chapter 7, and Chapter 8, respectively. These chapters contain information pertaining to the collection of data, system boundaries, allocation decisions, data inventories and life-cycle impacts for each of the three products. Various IMTA scenarios are formulated based upon different ratios between salmon and kelp, salmon and mussels or all three of the products. The ratios are selected as a factor of their ability to achieve variable efficiency levels for the bioremediation of nitrogen (N) and phosphorous (P), emitted by an Atlantic salmon grow-out facility. The chosen efficiencies are 100 %, 50 %, 20 % of either N or P.

The scenarios will be analysed using four different functional units. Each of these functional units is defined a quantity of a different factor, these being mass (1 kg), protein content (100 kg), and economic value (100 US$). Using these functional units, the different scenarios will be compared to the quantity of salmon monoculture production required to fulfil the equivalent functional unit.

Bioremediation is calculated based upon the principle of a black box, mass balance model, introduced in Chapter 2, and detailed below (9.2.2). This model assumes that no direct uptake of emissions from salmon farming needs to be demonstrated, and that the balancing of nutrients is achieved upon harvest of *M.pyrifera* and / or, *M.chilensis*, which take up nutrients from the marine environment as part of their metabolic functioning throughout growth.

9.2.2. Nutrient mass balance model

There are different ways of measuring the efficiency of bioremediation, which is usually defined as the percentage of a specific nutrient (e.g. nitrogen) that is removed from a system by nutrient extracting species. Various studies have investigated bioremediation by concentrating on direct uptake of nutrient emissions, and with consideration being given to nutrient emissions from the extractive species themselves and other aspects of their physiology. This is especially true for bivalves, which have the potential to acquire nutrients from naturally occurring particular organic material, a portion of which, along with any captured solid wastes from fish farms, can be ejected as pseudofaeces. Mussels also release dissolved nutrients as part of their metabolic functioning, which, potentially, may then be available for extraction by co-cultivated seaweeds. Understanding these nutrient dynamics within IMTA systems is important. However, much of this relates to end-fate modelling and local scale impacts, which are not analysed as part of this study. As it relates to life-cycle assessment, end-fate modelling, if included in the system boundaries, would require its application to the multiple emissions from the very numerous processes that are inventoried within a life-cycle. This could easily become an unmanageable task, and may render the assessment inaccessible to many by producing results that require an expert knowledge in various different fields, if they are to be interpreted meaningfully. As has been argued in Chapter 2, for the purposes of reducing the emissions of nutrients from aquaculture, it may be more useful to understand bioremediation in terms of a black box, mass-balance model. This method of measuring bioremediation disregards the specific source of nutrients being taken up by the extractive species, and focuses on the quantity of nutrients removed when these species are harvested. The amount nutrients removed by kelp and mussels upon their harvest are quantified based upon the tissue contents of carbon, nitrogen and phosphorous, and in the case of mussels, the carbon contained within the shells is also accounted for. The method for quantifying the nutrient contents of kelp and mussels are described in Chapter 7 and 8 respectively. The basic mass balance model can be described by the following equation:

$$\textit{Bioremediation efficiency} = ((NC_{extractivespecies}) \div NE_{fedspecies}) \times 100 \qquad \text{Eq. 9.1}$$

Where $NC_{extractivespecies}$ denotes the N or P content of the selected extractive species, and $NE_{fedspecies}$ is the emission of either N or P by salmon grow-out cultivation.

9.3. Inventory

9.3.1. System boundaries

The system boundaries of the basic IMTA framework are depicted by the flowchart below (Figure 9.2.). Alongside the mass-balance method of determining bioremediation efficiency, it provides the basis of the IMTA scenarios being assessed.

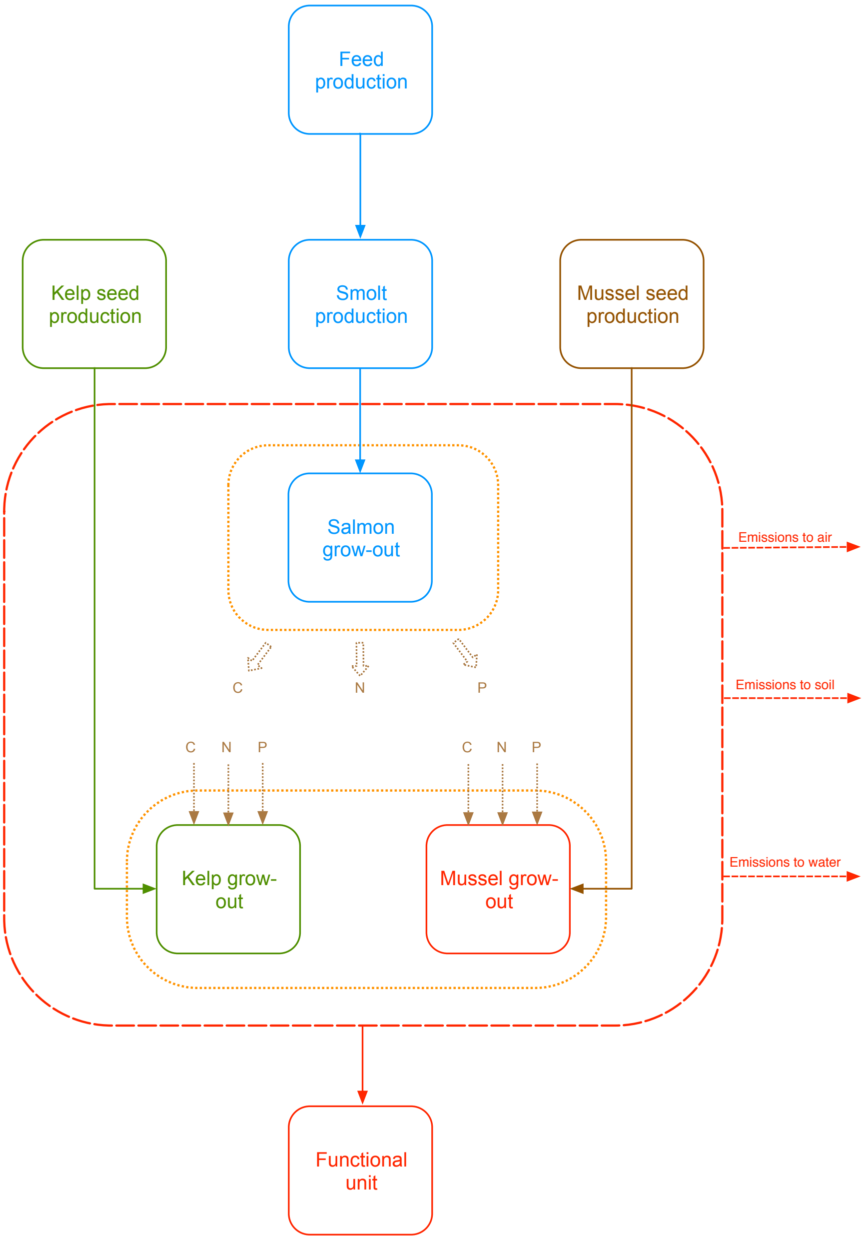


Figure 9.2. Flow chart depicting the major product phases required for IMTA production. The red dotted line delineates the IMTA grow-out phase.

9.3.2. IMTA scenarios

Using the mass balance method, the wet weight quantity of kelp or mussels required for the different removal efficiencies of nutrient emissions from the grow-out production of 1 kg of salmon, has been calculated (Table 9.1). For example, a 100 % efficiency for the bioremediation of the nitrogen released by 1 kg of salmon growth, is achievable through the co-cultivation of either 10.228 kg of kelp, or 10.160 of mussels.

Table 9.1. The quantity of either kelp or mussels required for the balancing of 100 %, 50 % and 20 % of the nitrogen and phosphorous metabolism emissions from the production of 1 kg of Atlantic salmon.

Bioremediation efficiency (%)	*M.pyrifera (kg) required for removal of...*			*M.chilensis (kg) required for removal of...*		
	..C	*..N*	*..P*	*..C*	*..N*	*..P*
100	5.819	10.228	24.479	5.744	10.160	14.804
50	2.909	5.114	12.240	2.872	5.080	7.402
20	1.164	2.046	4.896	1.149	2.032	2.961

Regardless of the functional unit, these ratios will remain the same when the IMTA system consists of either salmon and kelp, or salmon and mussels. Of course, the ratios are not maintained when both kelp and mussels are integrated with salmon. This later scenario has been modelled by assuming a roughly equal biomass production of kelp and mussels. As the nutrient contents of kelp and mussels are not the same, it is not possible to maintain an equal amount of kelp and mussel to perfectly match a specific removal efficiency of salmon emissions, although the values are close (Table 9.2).

Table. 9.2. The quantity of kelp or mussels required within a salmon: kelp : mussel IMTA system, for the balancing of 100 %, 50 % and 20 % of nitrogen and phosphorous emitted by the metabolism of 1 kg of Atlantic salmon.

Bioremediation efficiency (%)	*M.pyrifera (kg)*			*M.chilensis (kg)*		
	..C	*..N*	*..P*	*..C*	*..N*	*..P*
100	2.909	5.114	12.240	2.872	5.080	7.402
50	1.455	2.557	6.120	1.436	2.540	3.701
20	0.582	1.023	2.448	0.574	1.016	1.480

9.3.3. Functional units.

The protein content and economic value per wet weight of product are shown in Table 9.3. The values are derived from literature sources (Table 9.4), taking into account physiological variables when relevant.

Table 9.3. Quantity of protein per kg (wet weight) of product mass, and the economic value (USD) per kg (wet weight) product mass. Data values have been obtained from the values shown in Table 9.4.

	S.salar	*M.pyrifera*	*M.chilensis*
Protein (kg / kg)	0.189	0.026	0.029
Economic value (US$ / kg)	8.23	0.078	2.63

For each different combination of species, the total quantity of biomass required to fulfil the functional unit is calculated, whilst maintaining the weight-ratios between salmon and extractive species that are required to achieve the desired level of bioremediation efficiency. As an example, a functional unit of 100 kg protein is achieved through the cultivation of 219.813 kg salmon, integrated with 2248.285 kg of kelp, which is a combination that maintains the weight ratio of salmon to kelp required for 100 % bioremediation efficiency of nitrogen. This same functional unit is also achieved through a combination of 310.594 kg of salmon and 1588.394 kg of kelp, but the resulting efficiency of nitrogen bioremediation is 50 %.

Table 9.4. Literature sources used to provide data for the calculation of protein content and economic value per kg of product mass.

	S.salar	***M.pyrifera***	***M.chilensis***
Protein	Newton et al. (2014)	Buschmann et al. (2008)	Vernocchi et al. (2007)
Economic value	Undercurrent News (2017a)	Correa et al. (2016)	Undercurrent News (2017b)

9.4. Impact Assessment

9.4.1. Functional unit: Mass, 1 kg

9.4.1.1. IMTA- co-cultivation of salmon and kelp

Figure 9.2. shows the relative impact profiles for salmon monoculture, and three Salmon : kelp IMTA systems that achieve the nitrogen bioremediation efficiencies of 100 %, 50 % and 20 %. This chart clearly shows that when comparing systems using mass as a functional unit, each of the IMTA bioremediation scenarios has an obviously lower contribution to impacts than monoculture. Across all impacts, IMTA 'salmon : kelp 100 % N' has a contribution profile that is between 87.17 % and 95.12 % lower than those of salmon monoculture. The IMTA system with the greatest impacts is 'salmon : kelp 20 %,' which has a contribution profile that is between 64.27 % and 70.13 % lower than those of the monoculture. When considering the impact category 'eutrophication potential,' it is expectable that the IMTA system with a bioremediation efficiency of 100 % will have the lowest contribution among all system alternatives. This same pattern being apparent across all impact categories is easily explainable by the fact that kelp has lower impacts per category than does salmon monoculture, when using mass as a functional unit. Thus, as the quantity of kelp decreases with each drop in bioremediation efficiency, the proportion of co-cultivated salmon increases so to maintain an equivalent mass unit, and as a consequence, the contribution towards all impacts also increases.

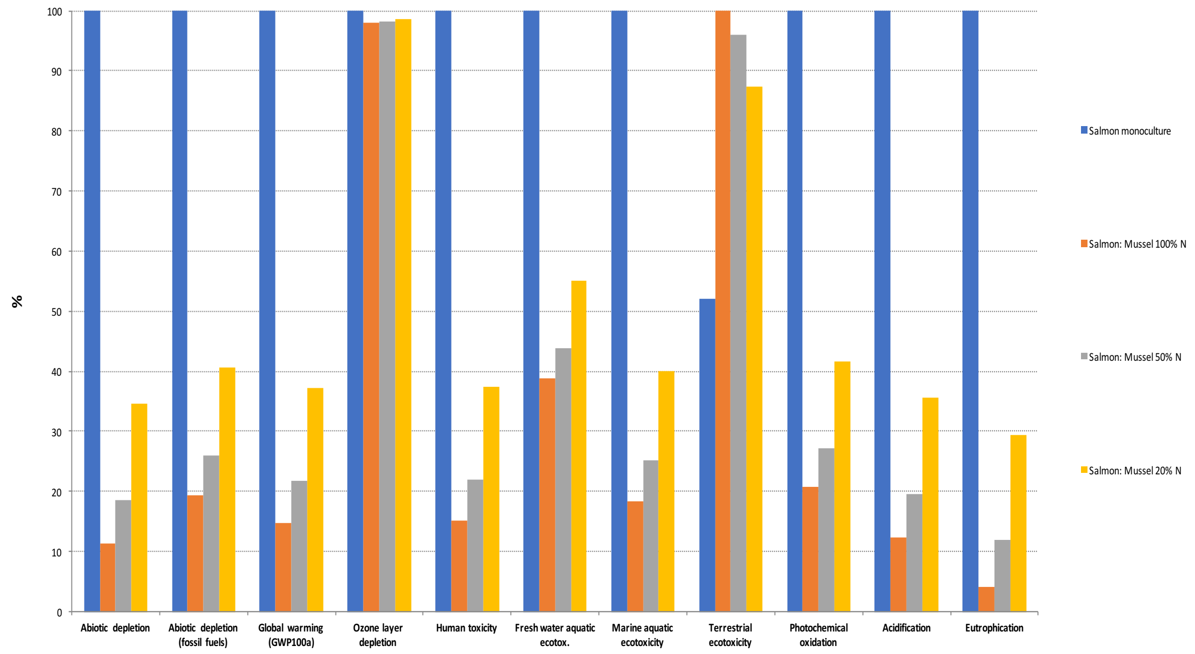


Figure 9.3. Comparison between salmon monoculture and salmon : kelp IMTA combined at the three ratios required for the balancing of 100 %, 50 %, and 20 % of N. Functional unit: 1 kg product mass. Calculated using the CML-IA-baseline method V3.03.

Contributions towards eutrophication potential are measured as phosphate equivalents, to which nitrogen compounds such as ammonia, and phosphorous compounds are converted during the characterisation process. This goes some way to explaining why the IMTA scenario with 100 % N bioremediation does not have zero contributions towards this category. Throughout the life cycle of the modelled products, there are opportunities for numerous sources of these emissions. This is especially true for salmon production, which depends upon the supply of agricultural crop ingredients that have been grown using the application of nitrogenous fertilisers. An integrated kelp cultivation which removes an amount of nitrogen equivalent to 100 % of that from salmon metabolic emissions, is insufficient to balance the emissions of phosphate equivalents from across those processes upstream of the salmon grow-out process (e.g. the leaching of fertilisers during crop production).

Another reason for the net positive contribution towards eutrophication potential, is that kelp has a higher content of nitrogen than it does of phosphorus. An outcome of the mass balance model is that the quantity of kelp required to remove the equivalent of 100 % of nitrogen from fish emissions, is insufficient for the total balancing of phosphorous emissions (Table 9.1). Balancing 100 % of nitrogen emissions through the harvesting of kelp will only remove the equivalent of 58.22 % of phosphorous emitted by fish. Therefore, the grow-out phase of this IMTA scenario still has a net positive contribution towards the eutrophication impact category.

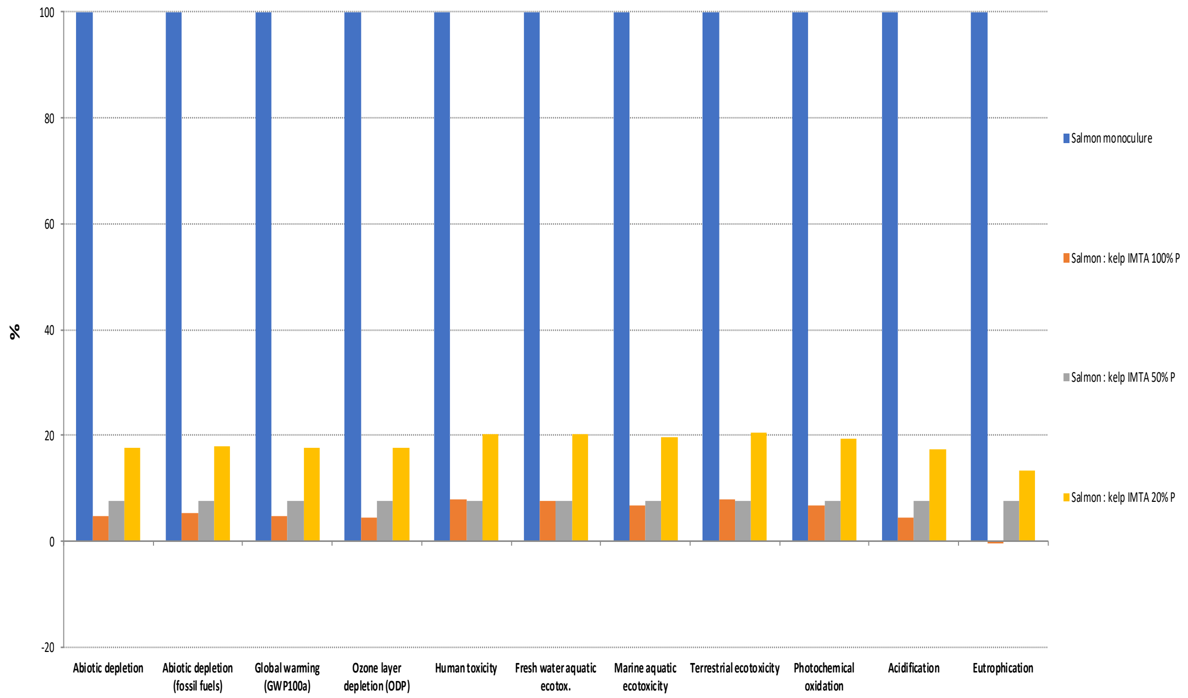


Figure 9.4. Comparison between salmon monoculture and salmon : kelp IMTA combined at the three ratios required for the balancing of 100 %, 50 %, and 20 % of N. Functional unit: 1 kg product mass. Calculated using the CML-IA-baseline method V3.03.

Only calibrating the ratio of salmon and kelp so that 100 % of phosphorous is removed, can result in a system that achieves complete removal of the equivalent quantity of both phosphorus and nitrogen (and also carbon), that is released by fish. Figure 9.4. shows the comparison between the potential impacts from salmon monoculture, and those of IMTA with salmon and kelp combined at ratios aimed at removing phosphorous at the three different levels of efficiency. For the IMTA system 'salmon : kelp 100 % P,' the amount of nitrogen and phosphorus removed through the harvesting of kelp, is a quantity that is greater than all emissions of phosphate equivalents from process across the entire modelled life-cycle. When considered from the point of view of localised emissions and impacts within the IMTA grow-out phase, this is not necessarily a good thing. Removing more nutrients from the system than what enters, may have undesirable environmental consequences. Marine systems with low-levels of nutrients can have reduced primary production, with knock on consequences for other organisms, extending beyond the immediate environment. However, the potential to effectively create such an oligotrophic system would likely require an IMTA system of a substantial size, located within a largely enclosed, marine bay.

For all other impact categories, the pattern of contributions is similar to those when nitrogen is the target nutrient for removal. In this respect, the main distinguishing feature is that contributions are of a lower magnitude when ratios are calculated to remove phosphorous. This is explainable by the fact that the ratio of kelp to salmon is higher, due to kelp having a lower content of phosphorus than nitrogen.

9.4.1.2. IMTA- co-cultivation of salmon and mussels

Figure 9.5. shows the comparison between salmon monoculture and IMTA systems composed of salmon and mussels at ratios required for the removal of phosphorous at the three different levels of efficiency. The pattern of results is different for those of salmon integrated with kelp. For 9 out of the 11 impact categories, the potential impacts of salmon monoculture are clearly higher than each of IMTA bioremediation scenarios. As expected, eutrophication is among these categories, and is the impact towards which IMTA has the least amount of contributions. However, the contributions towards 'ozone layer depletion, of the IMTA systems with a bioremediation efficiency of 100 %, 50 % and 20 %,' are 98.55 %, 98.19 %, and 98.03 %, respectively, of the total contributions from salmon monoculture. In other words, the contribution IMTA towards this impact category, are very similar to those from salmon-only production. Even more strikingly, the contributions of the three IMTA configurations towards 'terrestrial ecotoxicology' are between 40.34 % and 47.91 % greater than those from salmon monoculture. For both of these impacts, the major contributor is the production of cotton for the mesh-bags which are used to cover the mussel seed transplanted onto mussel ropes in the grow-out stage. A procedure for normalisation of results is not part of this study, but in this instance, it will help to provide information about the relative seriousness of IMTAs contribution towards these two impacts. The results of 'salmon : mussel 100 % N' have been normalised against the average yearly contributions of a European citizen of the 'EU 27+3, a population of 464,036,294. This is a commonly employed method of normalisation in LCA. On the resulting chart (not shown), the total contributions towards the two categories are barely visible, and the quantitative value of contributions for ozone layer depletion and terrestrial ecotoxicology are 3.32 E-14 and 5.78 E-14 respectively. In comparison, the normalised result is many times higher for the marine ecotoxicology impact category (99.08 % and 99.47 % higher than ozone layer depletion and terrestrial ecotoxicology, respectively). With consideration being paid to this result, it can be said that, although IMTA compares similarly to salmon monoculture in terms of contributions towards ozone layer depletion, and compares very poorly in terms of contributions to terrestrial ecotoxicology, the magnitude of these contributions is not high at all when compared to marine ecotoxicology, for which IMTA compares relatively well against salmon monoculture.

9.4.2. Functional unit: Protein, 100 kg

Using a functional unit of 100 kg of protein, the comparison between salmon monoculture and the three salmon : kelp IMTA calibrations is shown in Figure 9.6. for when phosphorous is the target of bioremediation, and in Table 9.7. for when the target nutrient is either nitrogen or phosphorous. The equivalent comparisons for salmon : mussel IMTA are shown in Figure 9.6 and Table 9.8. The ratios of

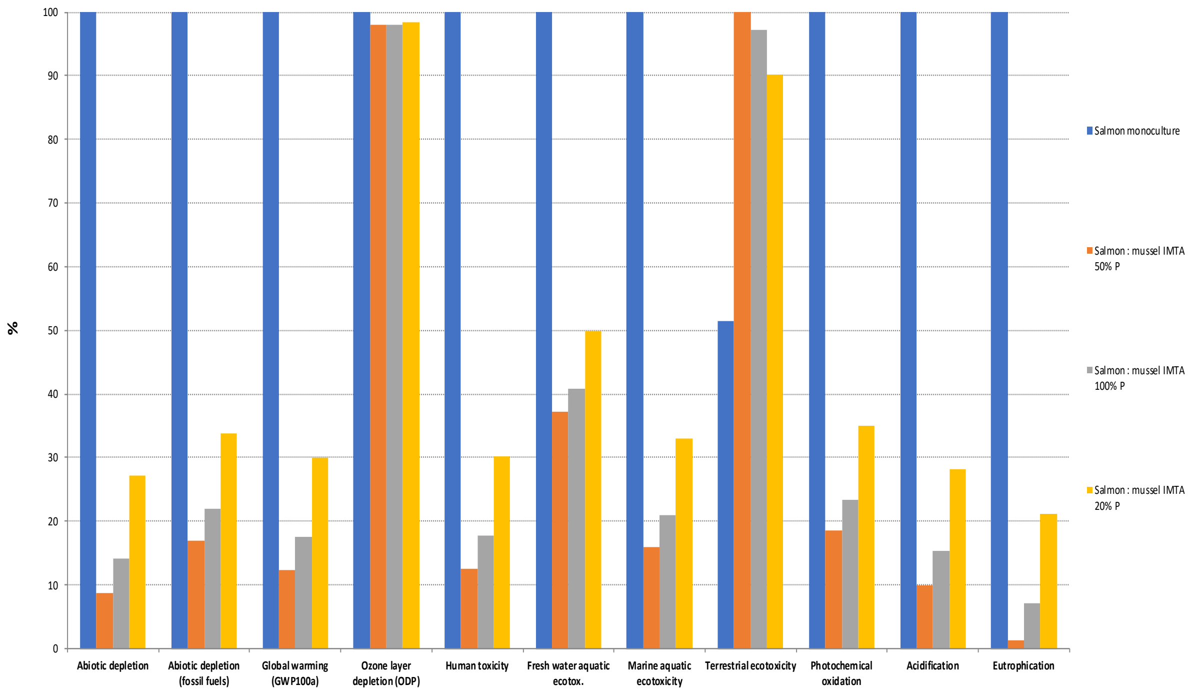


Figure 9.5. Comparison between salmon monoculture and the three combinations of salmon and mussel, with phosphorous being the target of bioremediation. The pattern of results is the same, and the same conclusions can be drawn as for when nitrogen is the target nutrient. The principal difference being, that for those impact categories which IMTA compares favourably to salmon monoculture, the IMTA calibrated for phosphorous removal provides lower contributions than IMTA calibrated for nitrogen removal, and for the impact category terrestrial ecotoxicology, the impacts when targeting phosphorous increase. Calculated using the CML-IA-baseline method V3.03.

salmon and mussels remains roughly similar as when using mass as the functional unit. A key difference of interest is how switching from a functional unit of product mass, to a function unit of protein content, changes the potential impacts from the IMTA systems relative to the potential impacts of salmon monoculture. These changes are shown in Table 9.5. for salmon : kelp IMTA and in Table 9.6. for salmon : mussel IMTA. In all cases, the contributions of IMTA relative to the contributions from salmon monoculture, are higher when the functional unit is mass-adjusted protein content, than when the functional unit is mass of product. This is because, from a purely numerical point of view, there is no distinction between a kilogram of salmon and a kilogram of kelp or mussels. In contrast, a kilogram of salmon has a higher protein content than kelp or mussels. The salmon : mussel IMTA systems fair particularly badly when using this choice of functional unit, because shell accounts for approximately 70 % of total mussel weight. As mussel shell is approximately 95 % calcium carbonate ($CaCO_3$), mussels compare poorly to salmon in terms of protein content per equal weight of harvested product. Thus, it is the IMTA system with the highest proportion of mussel cultivation (IMTA salmon : mussel 100 % P), that performs the most poorly against salmon monoculture, for all impact categories apart from eutrophication potential. To fulfil a functional unit of 100 kg protein through production in the IMTA system salmon : mussel 100 % P, a total harvest biomass of 2551.812 kg is required (161.47 kg of salmon + 2390.34 kg of mussels). In comparison, the salmon monoculture need only to produce a harvest of 529.1 kg. This means, that for every 1 kg of biomass produced in the salmon monoculture, 4.82 kg of biomass must be produced in 'IMTA salmon : mussel 100 %. P.' When using a functional unit of product mass, the IMTA system must produce 1 kg of biomass, likewise must the monoculture. When considering these two ratios, it become easy to understand how switching from a functional unit of product mass to one of protein production, creates a decline in the environmental performance of IMTA, relative to salmon monoculture. For IMTA salmon : mussel 100 % P, the contributions expressed as a proportion of those from salmon monoculture, increase between 5.1 % and 744.03 % across all impact categories, when the functional unit is switched from mass, to protein production. For IMTA salmon : kelp 100 % P, the increases are between 1.53 % and 38.93 %.

Table 9.5. The changes in contributions from salmon : kelp IMTA that occur when the functional unit is switched from 1 kg of mass, to 100 kg protein. Change in contribution is expressed relative to contributions when functional unit is defined by mass.

	Change in contributions from salmon : kelp IMTA relative to monocuture, when FU changes from mass to protein					
	100 % balancing of..		***50 % balancing of..***		***20 % balancing of..***	
Impact category	***..N***	***..P***	***..N***	***..P***	***..N***	***..P***
Abiotic depletion	+36.05	+23.71	+45.41	+34.36	+46.15	+44.93
Abiotic depletion (fossil fuels)	+37.15	+25.25	+46.41	+35.88	+46.46	+45.63
Global warming (GWP100a)	+35.44	+22.87	+44.87	+33.54	+45.98	+44.55
Ozone layer depletion	+35.02	+22.28	+44.48	+32.96	+45.86	+44.29
Human toxicity	+46.22	+37.86	+54.56	+48.26	+48.97	+51.32
Fresh water aquatic ecotox.	+46.08	+37.67	+54.43	+48.07	+48.93	+51.23
Marine aquatic ecotoxicity	+43.09	+33.51	+51.75	+43.99	+48.10	+49.35
Terrestrial ecotoxicity	+47.02	+38.98	+55.28	+49.35	+49.19	+51.82
Photochemical oxidation	+42.24	+32.33	+50.98	+42.83	+47.87	+48.82
Acidification	+34.75	+21.90	+44.24	+32.59	+45.79	+44.12
Eutrophication	+17.90	+1.53	+29.09	+9.57	+41.12	+33.54

Table 9.6. The changes in contributions from salmon : mussel IMTA that occur when the functional unit is switched from 1 kg of mass, to 100 kg protein. Change in contribution is expressed relative to contributions when functional unit is defined by mass.

	Change in contributions from salmon : mussel IMTA relative to monocuture, when FU changes from mass to protein					
	100 % balancing of..		***50 % balancing of..***		***20 % balancing of..***	
Impact category	***..N***	***..P***	***..N***	***..P***	***..N***	***..P***
Abiotic depletion	+37.93	+33.42	+44.89	+41.51	+45.47	+46.78
Abiotic depletion (fossil fuels)	+64.78	+64.89	+62.61	+64.19	+53.19	+58.09
Global warming (GWP100a)	+49.37	+46.82	+52.44	+51.17	+48.76	+51.60
Ozone layer depletion	+328.89	+374.54	+236.96	+287.32	+129.11	+169.37
Human toxicity	+50.60	+48.27	+53.25	+52.21	+49.11	+52.12
Fresh water aquatic ecotox.	+130.47	+141.92	+105.98	+119.69	+72.07	+85.77
Marine aquatic ecotoxicity	+61.75	+61.34	+60.61	+61.63	+52.32	+56.81
Terrestrial ecotoxicity	+644.04	+744.03	+445.01	+553.57	+219.70	+302.14
Photochemical oxidation	+69.67	+70.63	+65.84	+68.32	+54.59	+60.15
Acidification	+41.75	+37.89	+47.40	+44.73	+46.57	+48.39
Eutrophication	+13.78	+5.10	+28.94	+21.11	+38.53	+36.60

Table 9.7. The contributions of salmon : kelp IMTA with different bioremediation efficiencies: (100 %, 50 %, and 20 % of nitrogen and phosphorous), expressed as a percentage relative to the total impacts of salmon monoculture. For example, a contribution of -54.12 % means that the contribution quantity of IMTA is 54.12 % lower than the respective contribution quantity from salmon monoculture. Functional unit: 100 kg protein (mass-adjusted).

	Contributions of salmon : kelp IMTA, expressed as a percentage relative to contributions from salmon monoculture					
	100 % balancing of..		***50 % balancing of..***		***20 % balancing of..***	
Impact category	***..N***	***..P***	***..N***	***..P***	***..N***	***..P***
Abiotic depletion	-54.12	-71.38	-38.23	-58.08	-20.33	-37.26
Abiotic depletion (fossil fuels)	-52.71	-69.52	-37.24	-56.57	-19.80	-36.29
Global warming (GWP100a)	-54.89	-72.39	-38.78	-58.91	-20.62	-37.79
Ozone layer depletion	-55.43	-73.11	-39.16	-59.49	-20.82	-38.16
Human toxicity	-41.17	-54.30	-29.09	-44.19	-15.47	-28.34
Fresh water aquatic ecotox.	-41.35	-54.54	-29.21	-44.38	-15.53	-28.47
Marine aquatic ecotoxicity	-45.15	-59.55	-31.90	-48.46	-16.96	-31.09
Terrestrial ecotoxicity	-40.15	-52.96	-28.37	-43.09	-15.08	-27.64
Photochemical oxidation	-46.23	-60.98	-32.66	-49.62	-17.37	-31.83
Acidification	-55.77	-73.56	-39.40	-59.86	-20.95	-38.40
Eutrophication	-77.22	-101.85	-54.55	-82.88	-29.01	-53.16

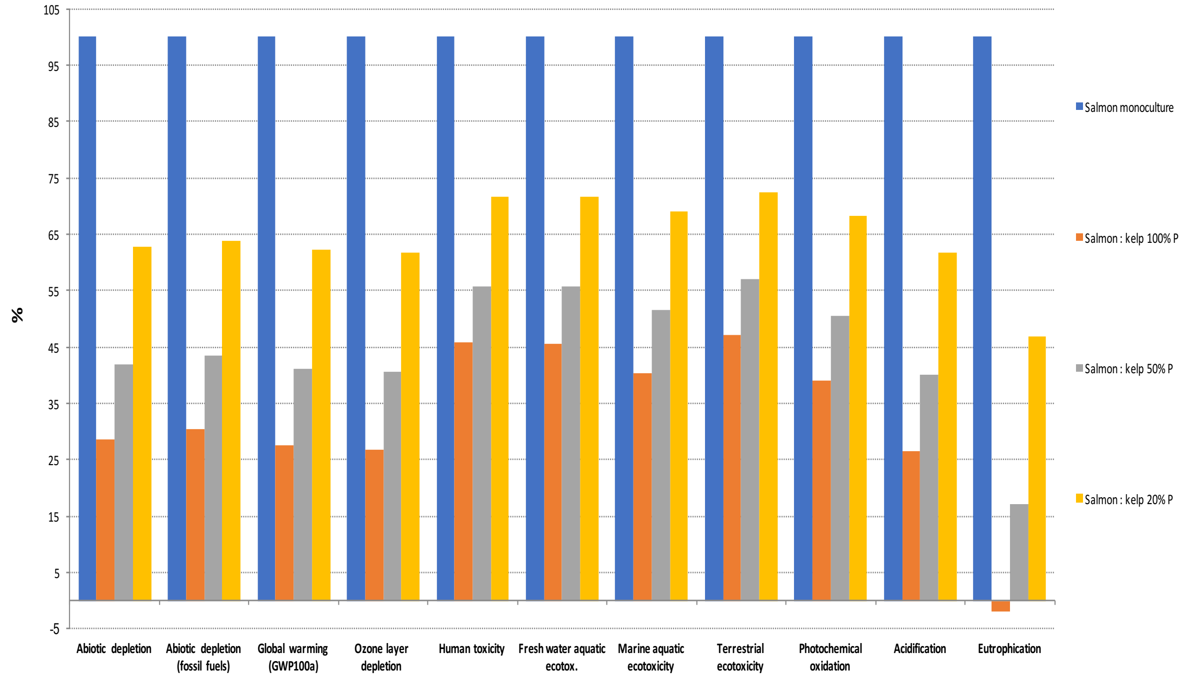


Figure 9.6. Comparison of the results from the characterised impact assessment of salmon monoculture, with those from salmon : kelp IMTA combined at ratios for 100 %, 50 % and 20 % of total phosphorous emissions from fish metabolism. The contributions are expressed as a percentage of the highest contribution towards each impact category. Functional unit = 100 kg Protein (mass-adjusted). Calculated using the CML-IA-baseline method V3.03.

Table 9.8. The contributions of salmon : mussel IMTA with different bioremediation efficiencies: (100 %, 50 %, and 20 % of nitrogen and phosphorous), expressed as a percentage relative to the total impacts of salmon monoculture. For example, a contribution of -50.76 % mean s that the contribution quantity of IMTA is 50.76 % lower than the respective contribution quantity from salmon monoculture. Functional unit: 100 kg protein (mass-adjusted). Calculated using the CML-IA-baseline method V3.03.

	Contributions of salmon : mussel IMTA, expressed as a percentage relative to contributions from salmon monoculture					
	100 % balancing of..		***50 % balancing of..***		***20 % balancing of..***	
Impact category	***..N***	***..P***	***..N***	***..P***	***..N***	***..P***
Abiotic depletion	-50.76	-57.84	-36.51	-44.32	-19.82	-26.05
Abiotic depletion (fossil fuels)	-15.91	-18.13	-11.45	-13.89	-6.21	-8.17
Global warming (GWP100a)	-35.92	-40.93	-25.84	-31.36	-14.03	-18.43
Ozone layer depletion	+326.92	+372.52	+235.16	+285.42	+127.66	+167.75
Human toxicity	-34.32	-39.10	-24.68	-29.96	-13.40	-17.61
Fresh water aquatic ecotox.	+69.36	+79.04	+49.89	+60.56	+27.09	+35.59
Marine aquatic ecotoxicity	-19.85	-22.61	-14.28	-17.33	-7.75	-10.18
Terrestrial ecotoxicity	+736.00	+838.66	+529.41	+642.56	+287.40	+377.66
Photochemical oxidation	-9.56	-10.90	-6.88	-8.35	-3.73	-4.91
Acidification	-45.81	-52.20	-32.95	-40.00	-17.89	-23.51
Eutrophication	-82.11	-93.56	-59.06	-71.69	-32.06	-42.13

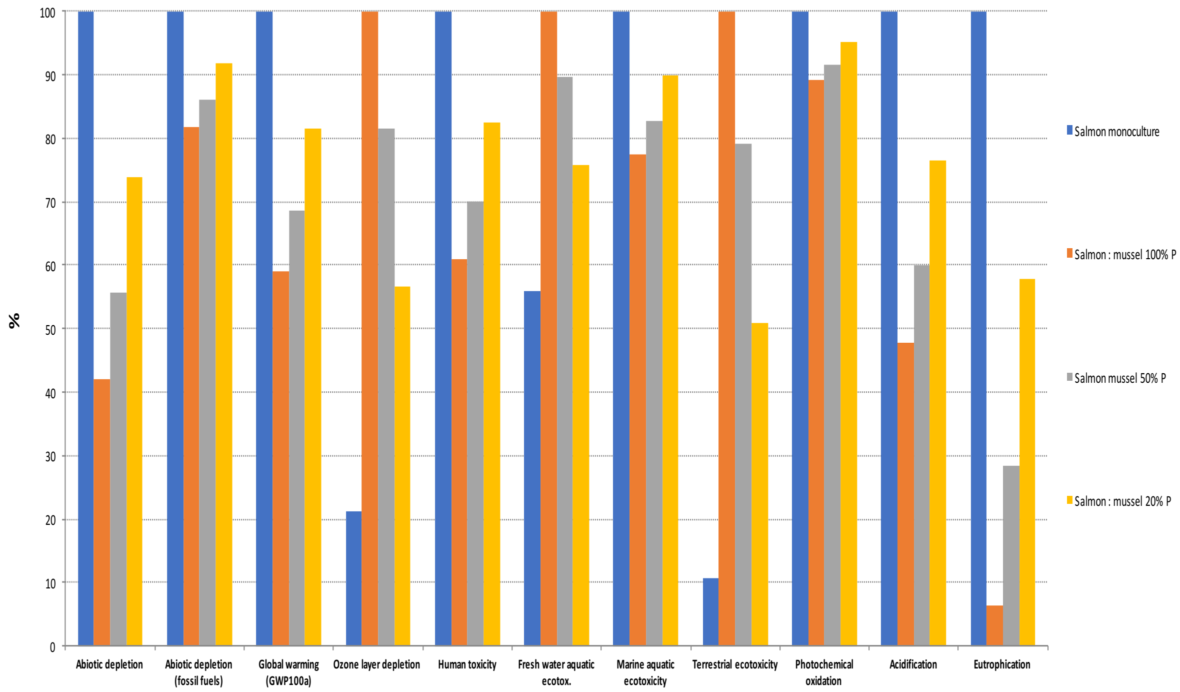


Figure 9.7. Comparison of the results from the characterised impact assessment of salmon monoculture, with those from salmon : mussel IMTA combined at ratios for 100 %, 50 % and 20 % of total phosphorous emissions from fish metabolism. The contributions are expressed as a percentage of the highest contribution towards each impact category. Functional unit = 100 kg Protein (mass-adjusted). Calculated using the CML-IA-baseline method V3.03.

9.4.3. Functional unit: Economic value, 100 US$

Neither mass, nor protein content, can provide an equivalent basis for comparison in terms of the nutritional value that food products present to human society. A mass of salmon is clearly not equivalent to an equal mass of kelp when nutritional content is taken into consideration. Protein, although an important indicator in future projections of food security, does not take into account the provision of other nutritional aspects necessary for human health, such as the supply of essential fatty acids that are prevalent in salmon, and are also found in the edible meat portion of mussels. For this reason, mass and protein do not allow for complete standardisation of nutritional function necessary for a like for like comparison. Indeed, it will be difficult to develop a fully, multi-nutrient based standardised unit for use in life cycle assessments of alternative food products. This later consideration is discussed in section 9.6. As a way to ameliorate this scenario, it could be plausible to use economic value as a proxy for the nutritional value that food products offer to society. Of course, the use of economic value for such a purpose is an imperfect solution, because the economic worth of food products also represents hedonistic values not directly essential to the provision of food security (although they can influence consumer choice in such a way that may alter the nutritional profile of our diet). Economic value of food products can also be influenced by many other economic variables unrelated to their nutritional content. Despite these opportunities for non-linearity between economic value and nutritional value, economic value is a reasonable choice when contrasted against the inadequateness of mass, or protein, for the purposes of standardised units of comparison.

Table 9.9. The contributions of salmon : kelp IMTA with different bioremediation efficiencies: (100 %, 50 %, and 20 % of nitrogen and Phosphorous) expressed as a percentage relative to the total impacts of salmon monoculture. Functional unit: 100 USD (mass-adjusted).

	Contributions of salmon : kelp IMTA, expressed as a percentage relative to contributions from salmon monoculture					
	100 % balancing of..		***50 % balancing of..***		***20 % balancing of..***	
Impact category	***..N***	***..P***	***..N***	***..P***	***..N***	***..P***
Abiotic depletion	+0.68	+1.46	+0.36	+0.81	+0.15	+0.34
Abiotic depletion (fossil fuels)	+3.77	+8.04	+1.97	+4.44	+0.82	+1.89
Global warming (GWP100a)	-1.00	-2.13	-0.52	-1.18	-0.21	-0.50
Ozone layer depletion	-2.20	-4.68	-1.15	-2.58	-0.47	-1.10
Human toxicity	+29.09	+62.00	+15.22	+34.22	+6.26	+14.60
Fresh water aquatic ecotox.	+28.71	+61.17	+15.02	+33.77	+6.18	+14.41
Marine aquatic ecotoxicity	+20.36	+43.38	+10.65	+23.95	+4.38	+10.22
Terrestrial ecotoxicity	+31.33	+66.76	+16.39	+36.85	+6.75	+15.72
Photochemical oxidation	+17.98	+38.32	+9.41	+21.16	+3.87	+9.02
Acidification	-2.95	-6.28	-1.54	-3.47	-0.63	-1.48
Eutrophication	-50.01	-106.56	-26.16	-58.82	-10.76	-25.09

Table 9.9. shows the contributions from salmon : kelp IMTA, as they relate as a percentage of contributions from salmon monoculture, and Figure 9.8. shows the comparison of total contributions when phosphorous is the main target of bioremediation. The equivalent comparisons for salmon : mussel IMTA are shown in Table 9.10. and Figure 9.9. Whereas, when the functional unit is described by product mass or protein content, salmon : kelp IMTA compares favourably to salmon monoculture across all impact categories, when a functional unit of economic value is used as the basis of comparison, the performance of kelp : salmon IMTA becomes noticeably poorer. For each of the three ratios, for both the balancing of nitrogen and phosphorous, the salmon : kelp IMTA systems present higher contributions than salmon monoculture, towards 7 out of the 11 impact categories. Whilst the relative increases for some impact categories are only marginal, they still do not present a favourable position. For the impact category 'abiotic depletion', the contributions across all IMTA ratios for the differential removal efficiencies of both N and P, are increased between 0.15 % and 1.46 %, relative to salmon monoculture. However, most of the increases towards other impact categories are significantly higher, especially for the IMTA ratios calibrated to remove 100 % of the target nutrient.

The reason for these relative increases is easy to explain. Kelp commands a low price in Chile, and is usually harvested from natural populations by members of very low-income, rural communities. Even the price used in the calculations is optimistic and depends upon a largely hypothetical (although biologically possible) end-use scenario, capable of creating demand (i.e. Correa et al. 2016). In obvious contrast to kelp, an equal weight of salmon is sold at a much higher market price.

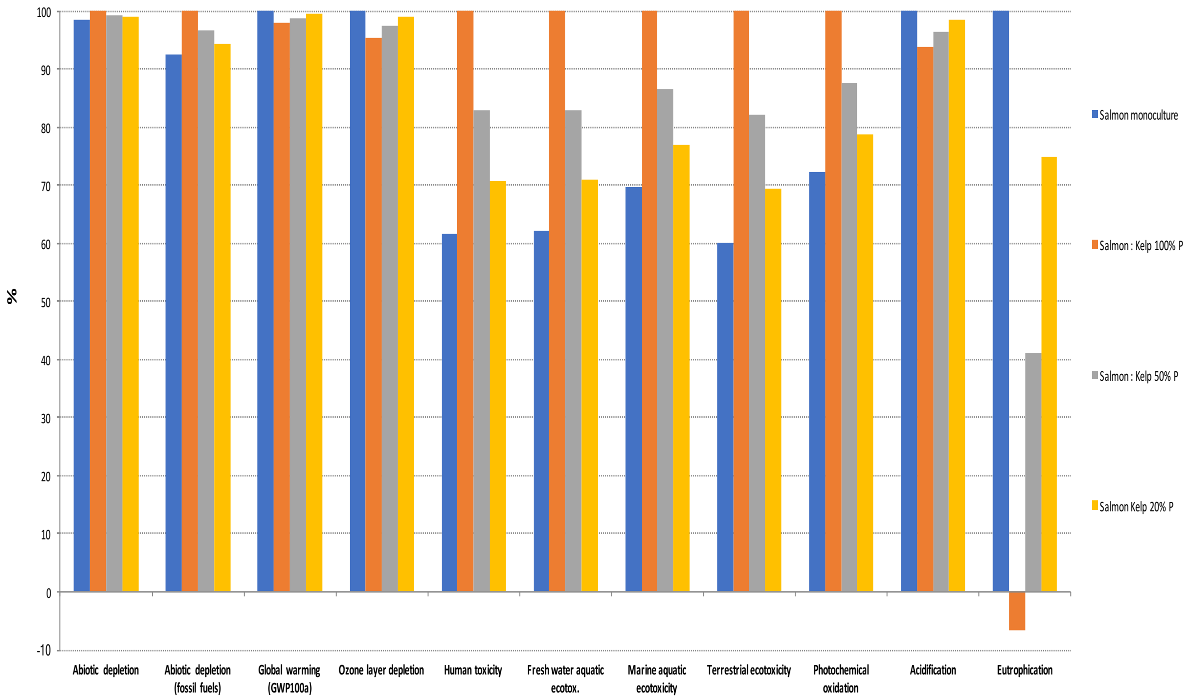


Figure 9.8. Comparison of the results from the characterised impact assessment of salmon monoculture, with those from salmon : kelp IMTA combined at ratios for 100 %, 50 % and 20 % of total phosphorous emissions from fish metabolism. The contributions are expressed as a percentage of the highest contribution towards each impact category. Functional unit: 100 USD (mass-adjusted). Calculated using the CML-IA-baseline method V3.03.

Table 9.10. The contributions of salmon : mussel IMTA with different bioremediation efficiencies: (100 %, 50 %, and 20 % of nitrogen and Phosphorous) expressed as a percentage relative to the total impacts of salmon monoculture. Functional unit: 100 USD (mass adjusted).

	Contributions of salmon : mussel IMTA, expressed as a percentage relative to contributions from salmon monoculture					
	100 % balancing of..		***50 % balancing of..***		***20 % balancing of..***	
Impact category	***..N***	***..P***	***..N***	***..P***	***..N***	***..P***
Abiotic depletion	-70.29	-75.89	-56.89	-64.62	-36.20	-44.70
Abiotic depletion (fossil fuels)	-49.26	-53.19	-39.87	-45.29	-25.37	-31.33
Global warming (GWP100a)	-61.33	-66.22	-49.64	-56.38	-31.58	-39.00
Ozone layer depletion	+157.62	+170.18	+127.57	+144.90	+81.17	+100.22
Human toxicity	-60.36	-65.18	-48.86	-55.49	-31.09	-38.39
Fresh water aquatic ecotox.	+2.20	+2.37	+1.78	+2.02	+1.13	+1.40
Marine aquatic ecotoxicity	-51.63	-55.75	-41.79	-47.47	-26.59	-32.83
Terrestrial ecotoxicity	+404.47	+436.72	+327.37	+371.84	+208.29	+257.19
Photochemical oxidation	-45.43	-49.05	-36.77	-41.76	-23.39	-28.89
Acidification	-67.30	-72.67	-54.47	-61.87	-34.66	-42.80
Eutrophication	-89.21	-96.32	-72.20	-82.01	-45.94	-56.73

In comparison to salmon : kelp IMTA, the outcome for salmon : mussel IMTA is quite different. Each of the assessed configurations compare favourably against salmon monoculture, in 8 out of the 11 impact categories. Additionally, its contributions relative to those of salmon monoculture, are lower than when the functional unit is defined by product protein content. Unlike Chilean kelp, cultivated *M.chilensis* usually achieves a profitable price among an established market for species of edible blue mussel (*Mytilus spp*.). As already discussed, the co-cultivation of salmon and mussels performs poorly when protein content is used as a functional unit, owing the an approximate shell to meat weight ratio of 70 : 30. Using economic value somewhat compensates for this factor, whilst reflecting, to an extent, the energy and protein profile of the edible meat which contains various fatty acids important to human health. However, salmon : mussel IMTA still has a considerably higher contribution potential than salmon monoculture, in the categories 'ozone layer depletion' (between 81.17 % and 170.18 % greater) and 'terrestrial ecotoxicology' (between 208.29 % and 436.72 % higher). Despite this, it is important to remember that even though these contributions are much greater than those of monoculture, they are not proportionately significant when the CML EU 27+3 normalisation procedure is applied (Figure 9.14).

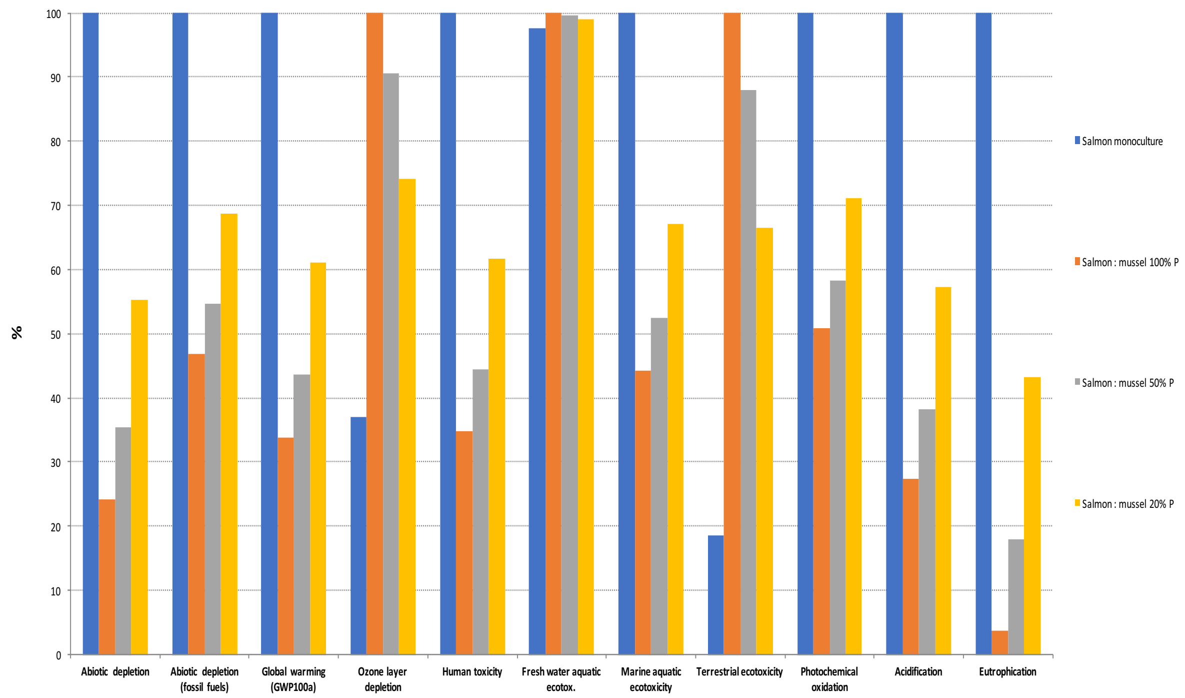


Figure 9.9. Comparison of the results from the characterised impact assessment of salmon monoculture, with those from salmon : mussel IMTA combined at ratios for 100 %, 50 % and 20 % of total phosphorous emissions from fish metabolism. The contributions are expressed as a percentage of the highest contribution towards each impact category. Functional unit: 100 USD (mass-adjusted). Calculated using the CML-IA-baseline method V3.03.

Table 9.11. The change in the contributions from IMTA relative to salmon monoculture, when the functional unit is changed from 100 kg protein to 100 USD. For example, if, when the protein is the unction unit, the contribution of IMTA towards a category is +3.72.52 % higher than those from monoculture, but it is +170.18 % higher when economic value is the function unit, the contribution of IMTA relative to the contribution of monoculture is 54.32 % lower when economic value is the functional unit.

	Change in contribitions from IMTA relative to contribultion from monoculture, when FU is switched from Protein to USD					
	Salmon : kelp IMTA			*Salmon : mussel IMTA*		
Impact category	***100 % P***	***50 % P***	***20 % P***	***100 % P***	***50 % P***	***20 % P***
Abiotic depletion	+102.04	+101.39	+100.92	-31.21	-45.80	-71.59
Abiotic depletion (fossil fuels)	+111.57	+107.85	+105.22	-193.38	-226.06	-283.48
Global warming (GWP100a)	+97.05	+98.00	+98.67	-61.79	-79.78	-111.61
Ozone layer depletion	+93.60	+95.66	+97.11	-54.32	-49.23	-40.26
Human toxicity	+214.18	+177.45	+151.51	-66.70	-85.21	-118.00
Fresh water aquatic ecotox.	+212.16	+176.09	+150.60	-97.00	-96.66	-96.07
Marine aquatic ecotoxicity	+172.85	+149.42	+132.86	-146.57	-173.92	-222.50
Terrestrial ecotoxicity	+226.05	+185.52	+156.87	-47.93	-42.13	-31.90
Photochemical oxidation	+162.85	+142.64	+128.35	-350.00	-400.12	-488.39
Acidification	+91.46	+94.21	+96.15	-39.21	-54.68	-82.05
Eutrophication	-4.63	+29.03	+52.80	-2.95	-14.40	-34.65

Finally, working under the assumption that economic value presents a more reasonable reflection of nutritional content than the alternatives, it will be interesting to see how an IMTA system composed of both kelp and mussels as nutrient removing species, compares against IMTA in which either kelp, or mussels are the only extractive crop. For this purpose, a roughly 50 : 50 weight ratio between kelp and mussel has been assessed. This weight ratio between the two-extractive species cannot be perfectly maintained due to the differing nutrient content of each crop and, and due to their different economic values.

Figure 9.10. shows a comparison between salmon monoculture and IMTA systems consisting of either salmon and kelp, salmon and mussel, or salmon, kelp and mussels, with 100 % removal of phosphorus being the target bioremediation efficiency. The result is somewhat predictable, in that salmon : kelp : mussel IMTA has contributions that are lower than salmon : kelp IMTA, and higher than salmon : mussel IMTA for 7 out the 11 categories. For the other four categories, IMTA consisting of the three species has contributions that are lower than salmon : mussel IMTA, and higher than salmon : kelp IMTA. The desirability of these outcomes in regard to nutritional value and environmental impact potentials, is discussed in the discussion and conclusions section of this chapter.

Table 9.12. The contributions of salmon : kelp : mussel IMTA with different bioremediation efficiencies: (100 %, 50 %, and 20 % of nitrogen and Phosphorous) expressed as a percentage relative to the total impacts of salmon monoculture. For example, a contribution of -55.72 % means that contribution quantity is 55.72 % lower that the contribution quantity of salmon monoculture. Functional unit: 100 USD (mass-adjusted).

	Contributions of salmon : kelp : mussel IMTA, expressed as a percentage relative to contributions from salmon monoculture					
	100 % balancing of..		***50 % balancing of..***		***20 % balancing of..***	
Impact category	***..N***	***..P***	***..N***	***..P***	***..N***	***..P***
Abiotic depletion	-55.72	-62.21	-40.54	-48.33	-22.31	-28.95
Abiotic depletion (fossil fuels)	-38.37	-42.36	-27.92	-32.90	-15.37	-19.71
Global warming (GWP100a)	-48.95	-54.88	-35.62	-42.64	-19.60	-25.54
Ozone layer depletion	+124.81	+139.24	+90.82	+108.17	+49.98	+64.80
Human toxicity	-42.00	-42.68	-30.56	-33.15	-16.82	-19.86
Fresh water aquatic ecotox.	+7.64	+12.78	+5.56	+9.93	+3.06	+5.95
Marine aquatic ecotoxicity	-36.86	-38.21	-26.82	-29.68	-14.76	-17.78
Terrestrial ecotoxicity	+327.87	+371.26	+238.58	+288.41	+131.30	+172.76
Photochemical oxidation	-32.41	-33.59	-23.58	-26.10	-12.98	-15.63
Acidification	-54.09	-60.92	-39.36	-47.33	-21.66	-28.35
Eutrophication	-81.16	-98.13	-59.06	-76.23	-32.50	-45.66

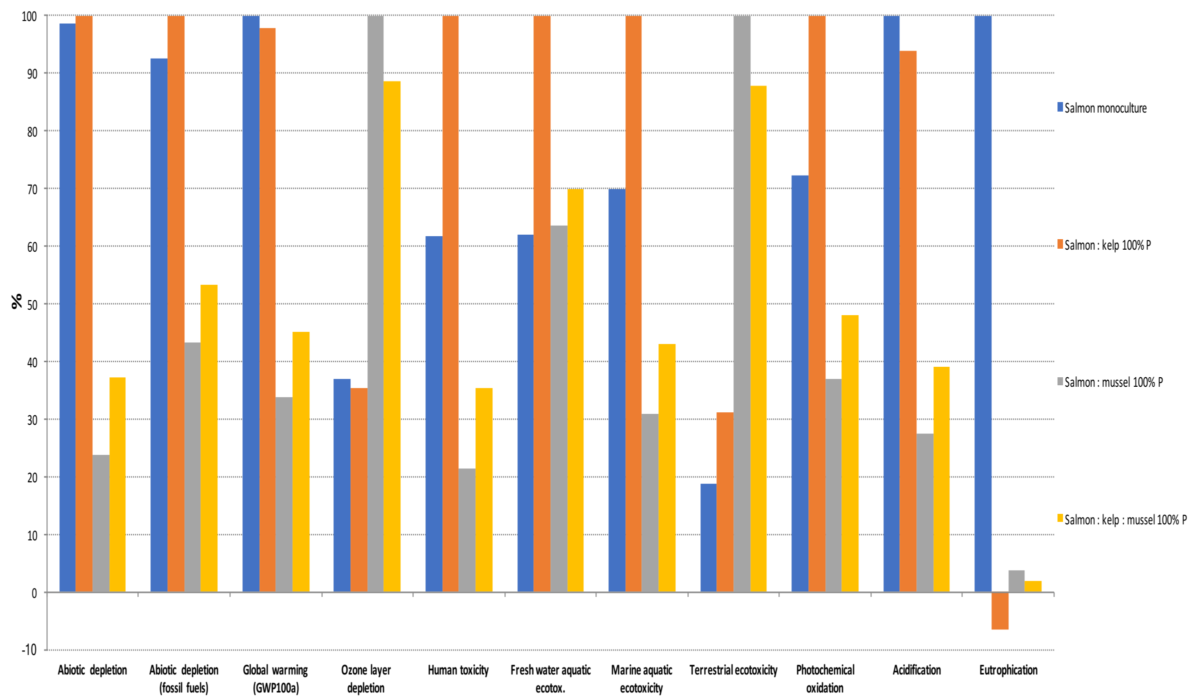


Figure 9.10. Comparison of the results from the characterised impact assessment of salmon monoculture, salmon : kelp IMTA, salmon : mussel IMTA, and salmon : kelp : mussel IMTA. Each IMTA system has a bioremediation efficiency of 100 % of fish phosphorous emissions. The contributions are expressed as a percentage of the highest contribution towards each impact category. Functional unit: 100 USD (mass-adjusted). Calculated using the CML-IA-baseline method V3.03.

9.5. Uncertainty analysis

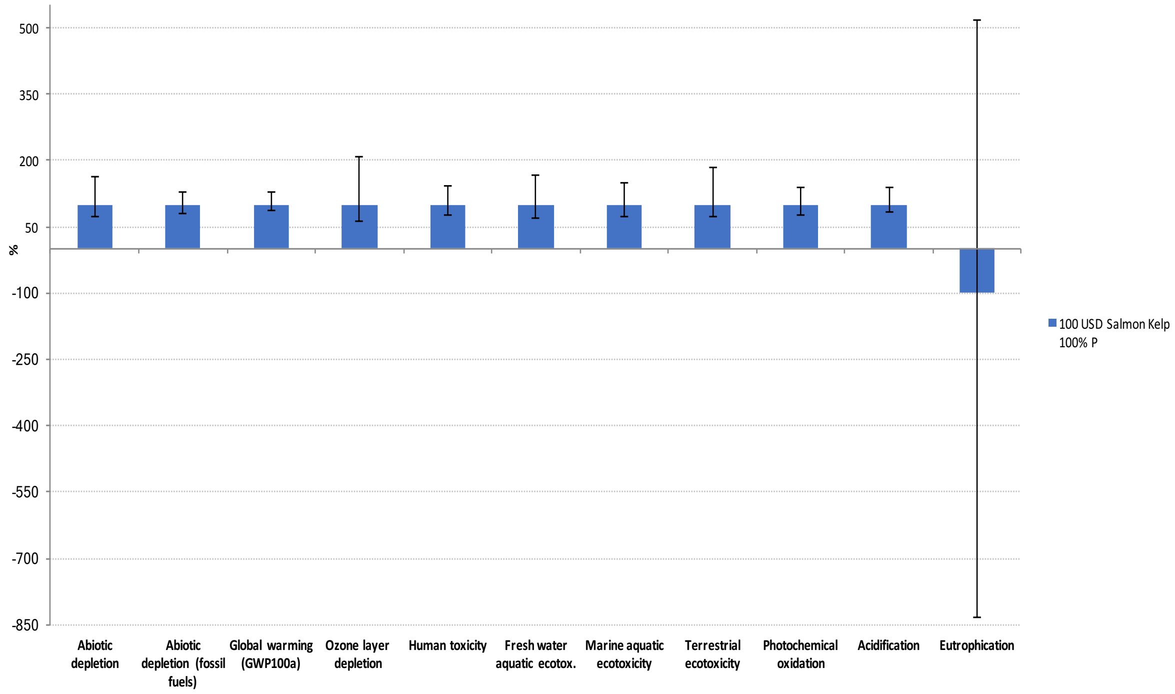


Figure 9.11. Uncertainty ranges (95 % confidence interval) for the impact assessment of biomass with an economic value of 100 USD, produced in salmon : kelp IMTA 100 % P. Calculated using the Monte Carlo assessment method, with 1000 test runs.

The uncertainty ranges (95 % confidence intervals) generated by the Monte Carlo runs are shown for the characterised impact assessments for salmon : kelp IMTA 100 % P (Figure 9.11.), salmon : mussel IMTA 100 % P (Figure 9.12.), and salmon : kelp : mussel IMTA 100 % P (Figure 9.13). For salmon : kelp IMTA, all of the ranges are within what is commonly acceptable, except for eutrophication potential. However, this result need not be alarming, because is an artefact of the procedure used, which gives a false impression. This occurs because the Monte Carlo method randomly selects values from the inventory that contribute towards eutrophication potential (based upon their associated uncertainty values). As it sometimes selects values from the nutrient emissions to water from fish production, and other times negative values that represent uptake of nutrients by kelp, it generates large uncertainty ranges. Thus, in the case of eutrophication, the scenarios tested by the Monte Carlo runs are scenarios that cannot exist, because the removal of nutrients always occurs as a factor of ratio between salmon and kelp. The true uncertainties surrounding the nutrient uptake values are grossly overestimated. The results of the uncertainty analysis for salmon : mussel IMTA and salmon : kelp : mussel IMTA, produce a similar situation, but not as extreme. Disregarding the artefact in the uncertainty range for eutrophication potential, the ranges are well within what is acceptable for LCA studies, and in this respect they are quite low. The lack of high uncertainty ranges (relative those typical of LCA), are due to the quality and quantity of primary data that has been used to model the foreground systems. It could be said, that these uncertainty ranges are a result of the effort that has been made to collect appropriate data. Without the reporting of uncertainty data, LCAs can be produced using poor quality data and there will be no gauge of their potential value, relative to competing studies.

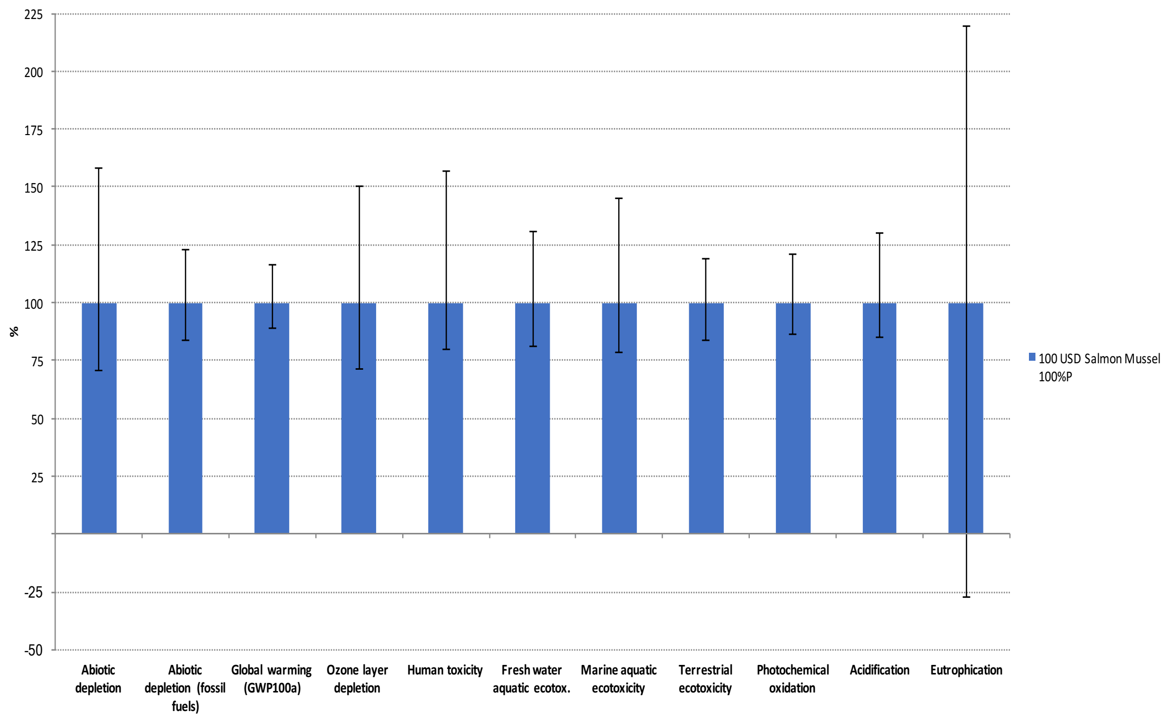


Figure 9.12. Uncertainty ranges (95 % confidence interval) for the impact assessment of biomass with an economic value of 100 USD, produced in salmon : mussel IMTA 100 % P. Calculated using the Monte Carlo assessment method, with 1000 test runs.

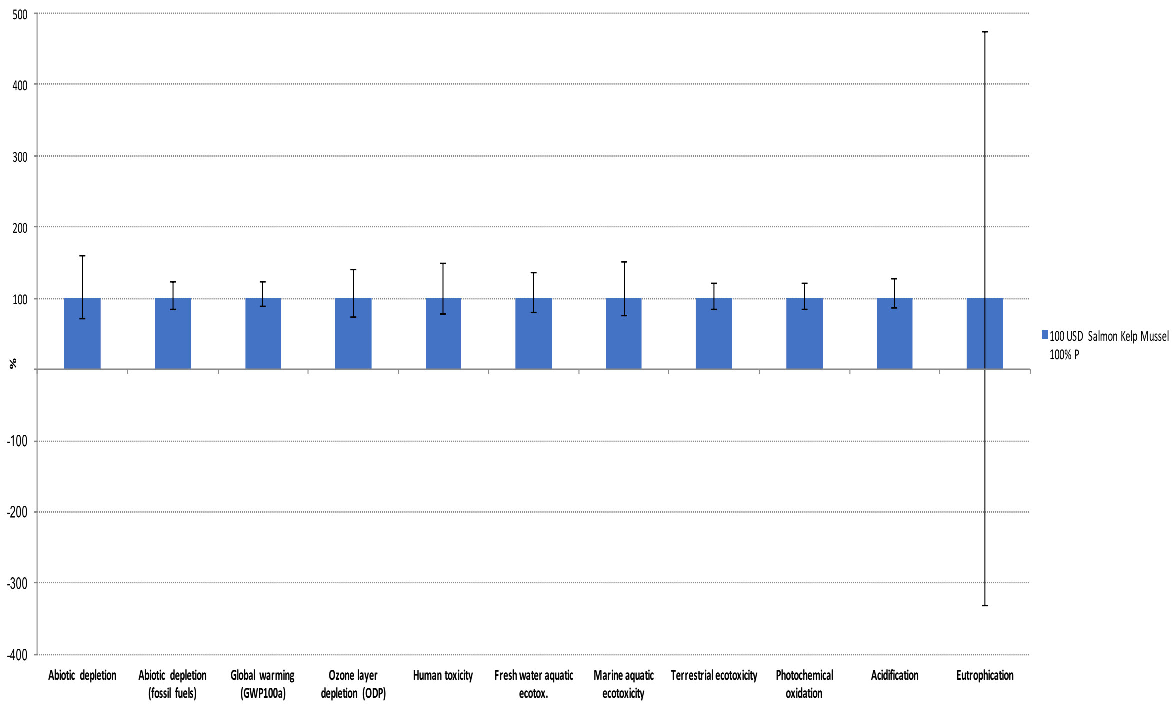


Figure 9.13. Uncertainty ranges (95 % confidence interval) for the impact assessment of biomass with an economic value of 100 USD, produced in salmon : mussel : kelp IMTA 100 % P. Calculated using the Monte Carlo assessment method, with 1000 test runs.

9.6. Discussion and conclusions

Three functional units have been used to assess the potential contributions of IMTA systems towards global scale environmental impacts. The first of these, 1 kg of product mass, suggests that salmon : kelp IMTA has a significantly better environmental impact profile than salmon monoculture. This is the case regardless of whether the equivalent of 100 %, 50 %, or 20 % of either nitrogen or phosphorous emissions from fish, is removed upon the harvesting of kelp. Salmon : mussel IMTA also compares favourably to monoculture in most categories. However, its contributions almost match those of salmon monoculture in the category 'ozone layer depletion,' and greatly exceed those of monoculture in the category 'terrestrial ecotoxicology.' This is a potentially alarming result, especially considering that contributions towards these categories increase significantly when the functional unit is described by either protein content or economic value. The further increases upon the change of functional unit are such that the contributions of salmon : mussel IMTA towards these impacts become greatly in excess of those of monoculture. To help understand the severity of these impacts, a normalisation procedure was carried out, which normalised the contributions of salmon : mussel IMTA to those of the year contributions from the citizen population 'EU 27+3.' This is a commonly employed method of normalisation. Interestingly, normalisation of results showed that even though contributions towards these two categories are particularly high, they are insignificant in comparison to the normalised quantity of potential contributions towards marine ecotoxicology. What this tells us, is that even though some potential impacts of an IMTA system may compare poorly in comparison to alternative monoculture production, it may be that contributions towards another category are of more concern, even though IMTA compares favourably to monoculture in this area. The downside to this method of normalisation is that, quite obviously, the contributions from IMTA production of 1 kg of biomass is not going to be remotely close to the yearly contributions of an average European citizen that consumes many kilograms of food within this time period, as well as numerous other consumer products. However, despite this limitation, it still holds true that contributions towards other impact categories appear to be more concerning than either those to ozone layer depletion, or terrestrial ecotoxicology. This point is illustrated in Figure 9.13, for which the impacts of producing ten billion kilograms of salmon : mussel IMTA, are normalised using the CML EU 27+3 method. The results of the normalisation procedure, as with those of the characterised impact assessment, are by no means conclusive, and may change as a factor of the normalisation procedure being employed. However, they do help to put the relative importance of the impact assessment results into a more tangible context.

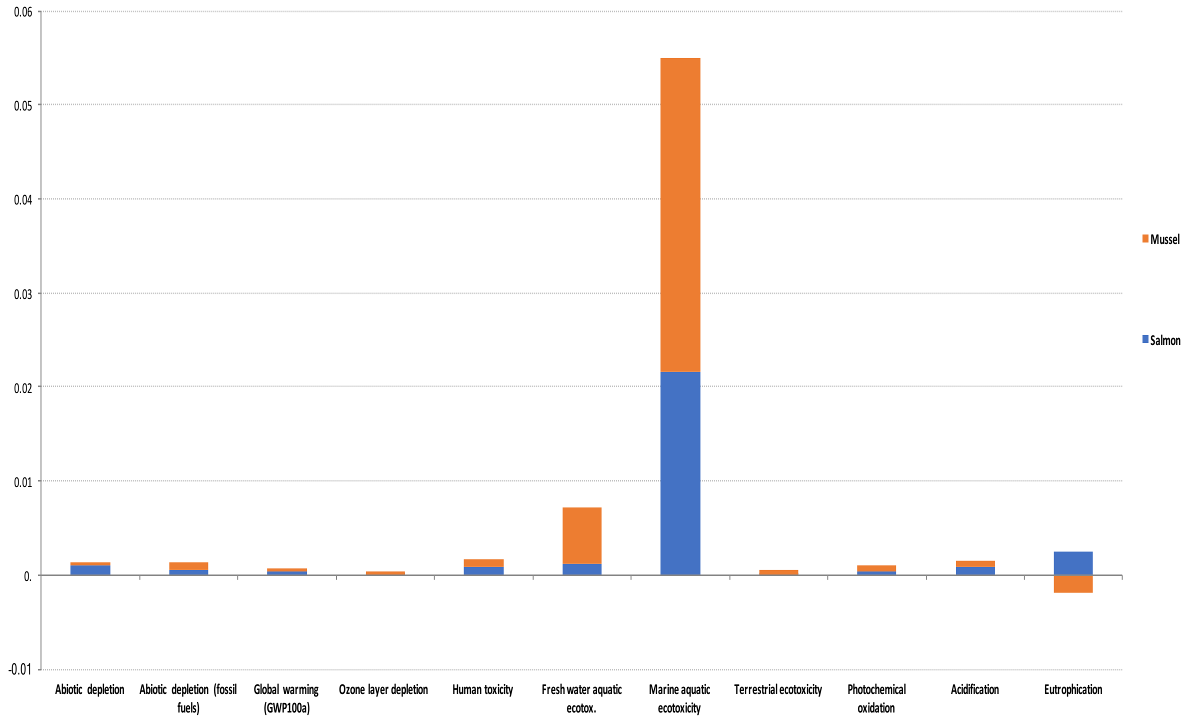


Figure 9.14. Characterised impact assessment results of 10 billion kg of biomass produced in a salmon : mussel IMTA system (100 % P), normalised to the respective yearly contributions from the population of the 'EU 27+3.' Calculated using the CML EU 27+3 method.

Defining functional units by product mass is acceptable when a single food product is being assessed. However, when comparing alterative products, or systems consisting of two or more species, mass does not account for multi-nutritional function. From this perspective, using product protein content to define the functional unit, more accurately defines the true function of the system. Predictably, due to comparatively lower protein content of kelp, and an approximate shell to meat ratio of 70 : 30 of cultivated mussels, salmon : kelp IMTA and salmon : mussel IMTA compares less favourably against salmon cultivation when protein content defines the functional unit, than when using product mass (Table 9.5 and 9.6 respectively). Despite global protein production being an important predictor of future food security (hence its use within this study), protein alone does not allow for a fair comparison between alternative food products that differ substantially in their content of other nutritional components essential for adequate human health. Farmed salmon offers nutritional benefits, in the form of high quantities of various fatty acids, such as the polyunsaturated omega-3 acids (e.g. Shepherd et al. 2017), that kelp simply cannot provide. Salmon : mussel IMTA compares much more poorly against salmon monoculture than does kelp, despite mussel meat being clearly superior than kelp in terms of its nutritional profile. However, it is true that the production of calcium carbonate within mussel cultivations currently has little contribution towards human dietary needs. Therefore, from the perspective of nutritional functional, calcium carbonate shells are a co-product with no contribution toward the systems function. Downstream of grow-out production and harvesting, the inedible shells are removed, and either enter landfill sites, or fulfil another function in the non-food economy. In Chile, shells removed at mussel processing facilities are crushed, and then used to provide surfaces for minor roads, avoiding the financial cost of road surfacing materials such as concrete. Additionally, there is some interest in the use of mussel shells for extracting industrial grade calcium carbonate (e.g. Iribarren el al. 2010a). In this study, product multifunctionality is not only limited to mussels. The success of kelp as a financial profitable component of IMTA systems may depend upon a distinctly non-nutritional purpose, as a substrate for conversion to biofuel (Buschmann et al. 2014). Kelp has also been investigated for use as an animal feed (e.g. Correa et al. 2016). Regarding salmon its entire carcass is not always fully utilised in the provision of nutrition. When salmon are not sold as whole fish, some portions of the salmon, such as 'bone-scrapings,' may be used to provide material for value added products such as pâté (personal observation). Other components, such as blood and viscera may be discarded via authorised methods of disposal, although there is also interest in the potential utilisation of salmonid co-products, such as extracting fish-oil from viscera, for use as a dietary supplement (Newton et al. 2014). In some isolated cases, the skin of farmed salmon is used a fabric for the production of clothing and fashion items, such as belts, wallets and purses, handbags, and even slippers, which can be occasionally found in craft markets in Puerto Montt (personal observation). It is also worth bearing in mind, that the average fillet yield of an Atlantic salmon is approximately 61 % (Newton et al. 2014), and for the majority of western consumers, this is the only portion of the fish that is eaten. However, missed opportunities for a more complete utilisation of available nutrition are not immediately relevant to this study's focus, and exist outside of the system boundary.

The above examples of nutritional inequality, and non-nutritional function, are unpresented by functional units defined by product mass, or product protein content. The solution to these problems is not necessarily to divide the system into separate, fully comprehensive systems, each with a representative function unit. In fact, complete division is not possible if nutrient and / or other flows are considered to be shared between the different products. However, the problem could be avoided by separating IMTA into subsystems with separate co-products, between which flows and emissions can be allocated. This was the approach taken by Mendoza Beltran and Guinée (2016), and it is one that might be especially appropriate if IMTA systems progress to become the suppliers of biomass to a variety of nutritional and non-nutritional applications. However, to remain consistent with the approach taken within this current study, mass-adjusted economic value has been used as a standardised unit of comparison, working on the assumption that economic value can be used as a proxy for the nutritional function that food products deliver to society. When a functional unit of 100 USD is used as the basis of comparison, salmon : kelp IMTA compares badly to salmon monoculture across all impact categories, apart from eutrophication. The economic value of kelp is low, and it will be difficult, although potentially possible (Correa et al. 2016), to develop a commercially profitable cultivation of *M.pyrifera*. This low economic value means that salmon : kelp IMTA must produce much more biomass than salmon monoculture, in order to produce the equivalent economic value. This is less of a problem for salmon : mussel IMTA, because the economic value of mussel is higher. Salmon : mussel IMTA compares favourably to salmon monoculture across all but three impact categories when economic value is used.

Different weight ratios of salmon : extractive species have been tested. These weight ratios have been calculated to result in three levels of bioremediation efficiency, 100 %, 50 % and 20 % of either N or P. Changing the ratios changes the environmental impact profile of IMTA. It also changes the impacts of IMTA relative to those of salmon monoculture. When the ratio is calculated for 100 % bioremediation, the proportion of kelp or mussels is higher than what is required for a bioremediation of 20 %. Especially in the case of salmon : kelp IMTA, the greater the ratio of salmon : kelp, the more total biomass the IMTA system must produce to, be equivalent to salmon monoculture in terms of economic value. As the ratios increase in the direction of increased bioremediation efficiency, the differences increase between the impacts of salmon monoculture, and those of IMTA. In categories where IMTA compares favourably, its comparative performance improves as bioremediation efficiency increases. In categories where IMTA has a higher impact potential than salmon monoculture, IMTA become comparatively worse as bioremediation efficiency increase.

Whilst still working with a functional unit of economic value, salmon : kelp : mussel IMTA has been assessed. Salmon : kelp : mussel IMTA has contributions that are lower than salmon : kelp IMTA, and higher than salmon : mussel IMTA, for 7 out the 11 categories. For the other four categories, IMTA consisting of the three species has contributions that are lower than salmon : mussel IMTA, and higher than salmon : kelp IMTA. It will be possible to alter the ratio between kelp and mussel, so to achieve the most optimal balance between environmental impact potentials, whilst maintaining the desired bioremediation efficiency. Whether this should be done, depends upon a variety of considerations. One of these is the relative importance of the impact potentials generated for each impact category. The higher the proportion of mussel cultivation, the higher the contributions towards ozone layer depletion and terrestrial ecotoxicology impacts. But if the magnitude of these contributions are to be considered as relatively small when compared to others (as the normalisation procedure suggested), then this poses a question. Why should any kelp be cultivated, when it increases impacts overall, and has a low nutritional (and economic) value? The answer might lie in the comparative localised impacts of salmon and kelp. As discussed in chapter 2, mussel cultivation can lead to localised benthic enrichment, through the loading of organic material produced as mussel faeces and pseudofaeces. In contrast, the only potential source of organic loading from kelp, comes from the exudation of tissue, which tends to be in the region of about 20 % of total biomass (Lobban and Harrison 1997). However, when the benthic environment was studied below the kelp cultivation assessed as part of this study, there was little observable difference to that of the surrounding benthic environment. Other considerations include those, such as the possibility of improved yield or quantity of the extractive crops as a result of direct nutrient uptake (as discussed in Chapter 2).

In general, the results of this LCA show that the environmental profile of IMTA is an example of trade-offs. IMTA can lead to improvements in some areas, but a of worsening in others. These trade-offs change as a factor of the species chosen and the ratios at which they are combined. Trying to establish the relative importance of these trade-offs is no easy task. Impact categories cannot be directly compared. It is also difficult to determine which impact categories are more important than others. Is it ok to implement IMTA if it reduces the eutrophication impacts of aquaculture, but leads to an increase in emissions of CO_2 equivalents? The value of IMTA is a matter of context, and it cannot be assumed that it is a more sustainable method of producing crops than monoculture.

Chapter 10: Final Discussion and Conclusions

Throughout this thesis, each separate chapter (excluding Chapter 1 and 4) has its own discussion and conclusions section. This final chapter is a synthesis of these sections, but more detail is added when appropriate.

When embarking upon this project, a choice had to made regarding the best way to allocate resources between the projects activities. Overall, there was a choice between two strategies. The first was to create an LCA based upon comprehensive, high quality production data. The second, was to develop novel approaches towards modelling IMTA within the context of life-cycle assessment, but making use of limited data sets. I decided upon the former strategy, largely because an opportunity existed to access various actors within the Chilean aquaculture industry.

The objective of collecting comprehensive, representative production data was an ambitious task, which at times was made difficult by a reluctance among industry to participate. In Chapter 3, I have explained how the Chilean salmon industry in particular, operates within a unique social and political setting. The industry faces numerous challenges, arguably, many of which have roots in the combination of a prolonged period of cultural suppression, and the state promotion of intensive salmon farming that has resulted in an almost untethered expansion of the industry. In general, the challenges faced so far can be defined as being part of a conflict between traditional and neoliberal culture, a conflict between industrial expansion and the environmental capacity to withstand it, and a systemic inadequacy to effectively manage issues of biosecurity. The industry has moved increasingly towards self-regulation, and it has made much progress in dealing with these problems. I have argued that the industry association, Salmonchile, and its technical and scientific research department, INTESAL, have been, and will likely continue to be, essential to the sustainable development of the Chilean industry. Reluctance to provide data among industry, finds its origin not only in the nature and quantity of data required, but also, in my opinion, from within the social, political, and administrative background, in which the industry has to operate. INTESAL (whose activities are consciously and intrinsically focused upon a global market perspective), AVS Chile. S.A. and the I~mar research institute, enabled the collection of data to be a success, through their recognition of the importance of life-cycle thinking, to the sustainability of Chilean aquaculture.

The incentive to conduct this project originated through consideration of the current state of research regarding the sustainability of open-water, marine IMTA. In chapter 2, I have discussed research that provides evidence for the bioremediation of nutrient emissions from intensive, marine fish-farming. I argue that, despite claims to contrary, the state of research is far from being sufficient to confirm open-water marine IMTA, as being a more sustainable alternative to, or modification of, modern intensive fish farming. The lack of focus upon the contributions of IMTA towards global-scale environmental impacts, significantly reduces confidence in the wisdom of the IMTA approach. I also propose, that it is not definitely necessary to demonstrate the direct removal uptake of fish-farm nutrient emissions, by the co-cultivated, extractive species. If the objective of IMTA is to prevent the adverse environmental effects of nutrient loading, then indirect uptake, through the harvest of species that have a net-uptake of nutrients, should be sufficient. Thus, within the appropriate context, IMTA does offer benefits from this limited, although important perspective.

In Chapter 4, I have described the standard methodology used in attributional LCA. I was my intention whilst writing this, to make the subject more accessible to researchers interested in the sustainability of aquaculture, but who might not have a background in LCA. I believe this to be important, because the available resources describing the methodology typically provide explanations of complicated concepts, through a reliance upon the use of technical terminology. Importantly, within this chapter, I describe a novel approach towards estimating the uncertainty surrounding inventory data. This approach, originally proposed by Henriksson et al. (2014), is based upon the horizontal averaging of a collection of data points which describe a specific input value of a given process. This average is weighted, by assigning a data quality score to each value of the numerical population being averaged. This data quality score is calculated using a chosen variant of the NUSAP method for quantifying qualitative aspects of data. To apply these methods, I constructed a spreadsheet using the appropriate formulas provided in Henriksson et al (2014), and combined them with the NUSAP scoring system proposed by Weidema (1998). Advancing methods of calculating uncertainty is an important agenda within LCA. As is typical of post-normal sciences, data uncertainty is something to be managed, rather than eliminated entirely. Within LCA, the uncertainty ranges are calculated and then reported, so that the confidence level of each result can be considered as part of its interpretation. The method proposed by Henriksson et al. (2014), is an interesting development in the methods for calculating uncertainty. However, I do have some reservations about the approach. To explain, I will use the specific example of the data quality indicators proposed by Weidema (1998), for application to the average values of lognormal distributions. One of the data quality indicators is ‘completeness.’ In order to describe the data quality in terms of ‘completeness,’ one of five data quality descriptions must be selected. The chosen description determines the numerical value that is used to quantify that specific level of quality. These descriptions define data quality based upon the representativeness of the data to the market considered. When there is only one value available to describe a specific input value of a process, selecting the appropriate quality description is relatively easy, as the market in which the process is part of, is usually known. However, when several input values from different sources are being averaged to provide an input value for a process, this procedure becomes more complicated. Typically, different data sources will provide a value from a different location, and from a different market. It is then unclear, as to which market is being considered. Does the 'market considered' refer to the market from which the individual value has been sourced? Or does it refer to the market that the average of each individually sourced value is intended to represent? There are other such instances of ambiguity, such as when describing the geographical representativeness of the data values. As a consequence, there is uncertainty originating from within the method of describing uncertainty itself. The route of this problem is that the NUSAP method is intended to describe an individual data point, and not to be used as part of a method for generating a quality weighted average of separate values. The horizontal averaging technique developed by Henriksson et al. (2014) promises to be a very useful tool. In order to further improve the method, it may be possible to design a set of data quality indicators intended for this specific purpose.

The horizontal averaging method was the main method employed for collecting data describing the agricultural production of aquaculture feed ingredients. Unfortunately, it was impossible to collect primary, directly sourced data for this purpose. Doing so would allocate all of the project resources away from the collecting of data for aquaculture production. This is an almost universal problem faced by those conducting life-cycle assessments of intensive aquaculture systems. As has been shown in Chapter 6, agricultural production of feed ingredients accounts for the majority of impacts resulting from the farming of Atlantic salmon. As a best-effort resort, secondary data was sourced from a variety of literature sources for each individual input to a process, rather than from only one source, to produce quality-weighted average input values. The horizontal averaging method enabled the calculation, and subsequent reporting, of the uncertainty ranges among the data collected. In this respect, this is an advancement in the current state of research, because it is the first time this has been done as part of an LCA of farmed salmon. However, by far the most superior way of collecting accurate life cycle inventory data, is to obtain it directly from the source. It is my opinion that future efforts to improve the state of research, should focus not upon the salmon growing phases, but upon the agricultural production of feed ingredients. The attainment of fish for reduction to meal and oil, is also an area requiring improvement. Performing LCAs of these various production systems will be an ambitious undertaking, but doing so is crucial for improving stakeholder confidence in the ability of LCA to offer a significant contribution to the environmental assessment of farmed salmon.

The life cycle assessments of giant kelp (*Macrocystis pyrifera*) and Chilean blue mussels (*Mytilus chilensis*), are necessary components for the final LCA describing production in integrated multi trophic aquaculture. The capacity of these LCAs to remain competitive, lies within the quality of data which has been used in their construction. The data were obtained directly from the producers. In the case of kelp, detailed, comprehensive data were obtained directly from the production facilities, and so they are of the highest possible level of representativeness. This is, I believe, the only LCA so far, that describes the production the *M.pyrifera* using such quality, primary data. As it is the largest cultivation of kelp within the western hemisphere, it certainly makes an interesting case study. The LCA of this kelp has been produced with the intention that it can be used as part of further LCA studies, in addition to its present use within the analysis of IMTA. As part of a continuing project being coordinated by Alejandro Buschmann of the i~mar research institute in Puerto Montt, it is intended to form the basis of future life cycle assessments incorporating the use phase of kelp, one such example being as a feed for abalone (Correa et al. 2016), another being as a substrate for conversion to ethanol (Buschmann et al. 2014). The LCA of mussel production was produced in collaboration with AVS Chile S.A. It is the only LCA of Chilean mussel cultivation, that has been produced using comprehensive, primary datasets. To be truthful, some results of both these LCAs are generally predictable, in that infrastructure presents the main contributor to impacts, along with those from fuel consumption. However, both LCAs provided some results that are, perhaps, surprising. In the LCA of kelp, the contribution of the seed production processes was higher than might be expected. This is most likely a result of economies of scale, and if the production facility is expanded, the intensity of input quantity per unit product, might decrease. In the LCA of mussel production, the significant contribution of cotton mesh bags was not expected, and its significance is supported by the relative low uncertainty range generated for this process. If the producers of mussels and kelp are serious about improving the sustainability of their products, there are two main things they can do regarding life-cycle impacts. Projects can be initiated to improve the quality of inventory data describing processes such as rope production. They can also make efforts to source infrastructure materials with an improved environmental profile. The sourcing of more 'environmentally friendly' infrastructure materials is a realistic option for the producers of either kelp or mussels, to make genuine improvements to the sustainability of their products.

Throughout the LCAs produced as part of this project, economic value has been used as the basis for the allocation of impacts between co-products. I have also proposed that economic value is a useful proxy for nutritional value, which is the major function of food products. The function of a food product is a key driver of consumer demand within market systems, and economic price is intended to denote the value of those products to human society. However, it has been argued that economic value is an unsuitable descriptor of the functionality of food products, and in particular, as the basis of co-product allocation. This position is bolstered by the assertion within ISO 14044 (ISO 14044 2006), that all life cycle assessments should be conducted in such a way, that they represent the true biophysical flows within a system. Indeed, whilst detailing the correct approach toward selecting the basis of co-product allocation, ISO 14044 (ISO 14044: 2006) does state that physical attributes of any given product should always be given precedence over economic value. Of particular relevance to life cycle assessments of food products, is a critique presented by Pelletier and Tyedmers (2011). The authors of this article propose that the use of market price is inherently incapable of delivering an undistorted representation of the relationships between economic production and environmental consequences. Their conclusions are particularly relevant to this present discussion, not only because the article focuses upon the LCA of food products, but because they maintain a particularly firm stance towards the modelling of multifunctional systems. Their relevance is further enhanced by the prominence of these authors as the producers of key life cycle-assessments of aquaculture systems (e.g. Pelletier et al. 2009). The concluding assertion of Pelletier and Tyedmers. (2011), is that market information should be excluded from life-cycle assessment to the fullest extent possible, and that it should be entirely rejected as a basis of co-production allocation within multifunctional systems. They further argue in favour of energy content as the most suitable descriptor of food product function, and as the basis upon which co-product allocation should be performed. Indeed, the article highlights some important problems concerning the use of economic value, and it represents a genuine and important contribution the advancement of food-focused LCA. It is true that market prices are subject to numerous distorting factors. Also true, is that economic value cannot be expected to perfectly describe the biophysical relationships between the extraction of materials and their subsequent conversion to products, or the biophysical connection between industrial activity and its environmental impacts. Thus, it can seem logical, that any attempt to model the impacts of economic activity upon the natural world, should, to the fullest extent possible, be based upon the biophysical reality of its material. In response to the article being discussed, a counter argument was offered by Wienzettel (2012). It argues that economic production takes place within an economic system, that the drivers of production are a consequence of demand and supply, and that using biophysical factors as a basis of allocation between co-products can sometimes result in seemingly absurd outcomes.

Regarding the latter statement, Wienzettel (2012) provides an explanation based upon the hypothetical, mutually dependant extraction of gold and copper, with gold being the target of extraction. The physical quantity of copper obtained is many times greater than the quantity of gold, but the economic value of the gold extracted is many times more than that of copper. If allocation is performed upon the basis of physical mass, the majority of environmental burdens are allocated to the copper, even though the purpose of the activity in the first place, is the extraction of gold. Using economic value as the basis of allocation assigns most of the burdens to gold. This hypothetic example is taken further, and Pelletier and Tyedmers (2012), offer an effective argument as to why physical based allocation (although not that of product mass), would still be a preferable choice, because it indicates the environmental consequences of economic activities, from a biophysically realistic perspective. In general, the debate can be seen as part of an interdisciplinary field of research that attempts to understand highly complex, and uncertain interactions between human economy and natural spheres (e.g. Liu et al. 2007; Rodriques et al. 2017). It need not be surprising that this argument introduces the relevance of such a technical area of research. The concept of sustainability embodies the very definition of post-normal science, as offered by Funtowicz and Ravetz (1993):

"facts uncertain, values in dispute, stakes high, and decisions urgent."

Comprehensive discussions of how issues of sustainability can be managed throughout society, involve a dynamic and evolving debate, drawing from the contributions of interdisciplinary sciences, economics, and political theory. It also includes a significant dimension of ethical and moral interpretation. The post-normal condition of '*facts uncertain'* is inextricably encountered in issues of co-product allocation, and the function of food. Although their ideas do find some support within the relevant research (e.g. Lui et al. 2007), adopting an absolute position, such as that proposed by Pelletier and Tyedmers (2011), might seem like a premature manoeuvre to take, when working within a conceptual framework so complex, that there are few definite answers. Most frequently, life cycle assessments are based, at least an extent, upon value-based judgements, which arise from individual, or group interpretations, of how the product being studied relates to the question they wish to answer. Thus, the appropriate methodology can change as a function of the study goal. It is my own understanding, that within this context of uncertainty, no single approach should be exclusively prescribed for every situation. That is not to say that the debate should end here, and there is much more that can be offered by many other experienced participants.

The final LCA, is that of the different open-water, marine, IMTA scenarios. This LCA is one of two that covers this subject. The other, is that produced by Angela Mendoza Beltran, as part of the multi-partner project 'IDREEM' (Mendoza Beltran and Guinée 2016), coordinated by the Scottish Association for Marine Science (SAMS). The approaches used in these two studies are distinct, but in some ways, complementary. One key difference is that, whereas the LCA described in this thesis is based upon a comprehensive, primary dataset, the study by Mendoza Beltran, is based upon limited production data, and uses estimated inputs. It is focused more directly upon methodological approaches. Another key difference, is in the approach to the modelling of the IMTA system itself. In this study, the product outputs of the IMTA system have been incorporated within the umbrella of a single functional unit, with the aim of describing the nutritional function that IMTA production delivers. This creates difficulties due to the relative inequalities between nutritional function that each harvested species delivers. The study by Mendoza Beltran avoids this problem by allocating flows (and thus, impacts) between the different species. This is a valid and worthwhile approach, but one I avoided, due to at least one reason in particular. As discussed in Chapter 2, there has been difficulty in demonstrating consistent uptake of nutrient emissions by co-cultivated species. If the main goal is to reduce the impacts of nutrient loading upon the environment, I have argued that direct uptake need not be demonstrated, and that a black box approach can be applied. If flows are allocated between the co-products of IMTA, then this includes allocation of nutrient emissions taken up by mussels and seaweed. Due to lack of conclusive evidence that significant uptake occurs over a prolonged period, the allocation of these nutrient flows between co-products, will be based upon significant uncertainty. If there is no direct uptake of nutrient emissions, it is difficult to describe the system as being integrated. Rather, it is the cultivation of separate monocultures within close proximity to one another. In this case, demonstrating integration depends upon the presence of indirect assimilation. Such issues bring into question some of the proposed benefits of IMTA. Without direct uptake, there is no improved biomass productivity per unit of feed introduced into the IMTA system. Lack of direct uptake also means that integration confers little benefit to the extractive species thorough enhanced nutrient provision, unless a benefit is acquired through indirect assimilation (such as that which may occur in cases of enhanced phytoplankton production being available to mussels). One of the findings of Mendoza Beltran, is that the factor of co-product allocation alters the environmental profile of the system. This is not surprising, as the system was modelled by allocating flows between the IMTA co-products. In response, a pseudo-statistical method was developed to deal with the uncertainty surrounding the choice of allocation method and its influence upon the results (Mendoza Beltran 2015).

In this study, three factors have been used to describe the function of the system. These are product mass, product mass-adjusted protein content, and product mass-adjusted economic value. These functional units are the unit of comparison between alternative systems, and so must represent a standardisation of product function. As the LCA has been performed within the context of food security, mass is clearly inadequate for this purpose. A more suitable descriptor is protein. However, product protein content does not adequately define the true function of the products, because, similarly to mass, it does not account for the inequality of nutritional function between the two products. For this reason, economic value has been used as a proxy for nutritional function. The suitability of a factor for describing function, lies in its ability to proportionally reflect the nutritional value of various food products. From this perspective, economic value might be suitable as proxy for nutritional value, from the perspective that it reflects, although imperfectly, the nutritional value that a product offers to society. The results of the comparison between IMTA and salmon monoculture change significantly depending upon whether protein content, or economic value, is used to define the functional unit. When protein content is used, salmon : kelp IMTA compares favourably to salmon monoculture across all impacts. Salmon : mussel IMTA performs poorly in two impact categories, but compares favourably across the rest. When economic value is used, salmon : kelp IMTA compares badly to salmon monoculture across all impact categories, apart from eutrophication. Contrastingly, salmon : mussel IMTA compares favourably across all but three impact categories. What this shows, is that there may be clear trade-offs between an improved performance in some areas of environmental concern, and a worsening in others.

A variety of weight ratios have been assessed, with different ratios resulting in different efficiencies of bioremediation. As the ratio increases in the direction of increased bioremediation efficiency, the differences increase between the impacts of salmon monoculture, and those of IMTA. In categories where IMTA compares favourably, its comparative performance improves as bioremediation efficiency increases. In categories where IMTA has a higher impact potential than salmon monoculture, IMTA becomes comparatively worse. Again, this provides evidence of environmental trade-offs.

These results are important, because they demonstrate, for the first time, that IMTA may potentially have a worse environmental profile than salmon-monoculture. Of course, the results depend upon the methodology used in their calculation, as well as upon the combination of species used within the system. The occurrence of trade-offs between environmental benefits and environmental costs, ought to be interpreted within an appropriate context. The balance between these costs and benefits is case dependant. In a semi-enclosed bay with high nutrient inputs originating not only from aquaculture, but from other anthropogenic sources, the value of the benefit of bioremediation may be very high. In this case, bioremediation itself could be incorporated into the product function. Bioremediation might be assigned a monetary value (e.g. Ferreira and Ferreira 2012), enabling it to be embedded within a functional unit defined by economic price. However, when cultivation is taking place within a marine environment with good dispersive capacities, there may be no measurable, or expectable negative consequences from production. In this case, the value of bioremediation is comparatively reduced.

Chapter 11. References

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