CONCEPTUALIZING AND QUANTIFYING THE ENVIRONMENTAL IMPACTS OF BIOLOGICAL PRODUCTION SYSTEMS

by

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“We have to prepare for what life could become in 40 years. We need to outline what is possible and what is impossible with the non-renewable resources of the Earth. What role will technological improvement play? Taking all this into account, what kind of life can we produce in the best way for 10 billion people?”

-Jacques Yves Cousteau

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Abstract

Increasing the amount of food produced while simultaneously reducing the environmental impacts of agriculture is one of the most pressing challenges facing humanity. A promising approach through which this could be achieved is ‘sustainable intensification’. This thesis contributes to the exploration of sustainable intensification using two complementary modes of investigation. First through the development of a conceptual framework that analyzes agricultural systems through the lens of ecosystem services and the trade-offs associated with using external inputs (e.g. fertilizer, pesticides, fossil fuels) as substitutes for them. Then by quantifying the life cycle environmental impacts of a novel aquaculture technology developed as a means for minimizing local ecological impacts. These modes of investigation are linked by using the conceptual framework to analyze trade-offs associated with waste capture in the aquaculture system. This research provides a potentially valuable method for conceptualizing agricultural systems and contributes to the knowledge of the environmental trade-offs associated with aquaculture.

List of Abbreviations Used

AP	acidification potential
APC	actual production cycle
B.C.	British Columbia
BRU	biotic resource use
CEU	cumulative energy use
CO_2eq	carbon dioxide equivalents
ES	ecosystem service
FAO	Food and Agriculture Organization
FEESSA	Framework for the Evaluation of Ecosystem Service Substitution in Agro-ecosystems
GHG	greenhouse gas
GWP	global warming potential
hp	horse power
IPC	intended production cycle
ISO	International Organization for Standardization
kg	kilogram
kWh	kilowatt-hour
LCA	Life cycle assessment
LCI	life cycle impact
MEP	marine eutrophication potential
MJ	megajoule
N	nitrogen
N.S.	Nova Scotia
P	phosphorus
PO_4eq	phoshphate equivalents
RAS	recirculating aquaculture system
SO_2eq	sulphur dioxide equivalents
SWAS	solid-wall aquaculture system
WC	Waste capture

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Chapter 1. Introduction

The environmental impacts associated with biological production systems (BPS: includes any managed ecosystems which are used for productive purposes such as agriculture, aquaculture, silviculture etc…) are larger and more profound than at any other time in human history (Balmford, Green, & Phalan, 2012; Foley et al., 2005; MEA, 2005; Rockström et al., 2009; Steinfeld et al., 2006; Tilman, 1999; Tilman et al., 2001). These systems are important contributors to many environmental problems including: climate change, biodiversity loss, degradation of water and air quality, and the erosion of ecosystem services (ES) (Foley et al., 2005; MEA, 2005). With the number of humans expected to reach approximately 9.6 billion by the year 2050 (UN, 2013) and the increased affluence of a developing world (Myers & Kent, 2003) demand for agricultural products is expected to increase substantially (Godfray et al., 2010; Pelletier & Tyedmers, 2010a). The dual challenge faced is thus to produce enough food, fuel, fibre, and other agricultural products (e.g. farmed seafood, biofuel crops, pharmaceutical crops etc…) while minimizing environmental impacts (Foley et al., 2011; Godfray et al., 2010; Tilman, Balzer, Hill, & Befort, 2011). Failure to do so would result in, at best a deterioration of the average quality of life and at worst catastrophic changes to environmental conditions, potentially precipitating severe hunger and conflict (Barnett & Adger, 2007; Wheeler & von Braun, 2013).

While debate on this issue persists, many agree that intensification – the process by which agricultural yield per unit of area is increased – is the most plausible method by which increased production can be realized without converting much of the planet's remaining natural areas that are considered critical for conservation (Lambin & Meyfroidt, 2011; Phalan, Balmford, Green, & Scharlemann, 2011). Unfortunately, intensification is associated with a large number of environmental impacts, often associated with the increased use of external inputs. Here the term external input is used to describe resources, materials, or energy used in (but not limited to) agro-ecosystems that are not a part of the natural cycling of that system (e.g. fossil fuels, electricity, fertilizers, pesticides) on large monocultures (Chappell & LaValle, 2009). To make agricultural systems both more productive and less impactful there has been a concerted effort by producers, scientists and managers from a variety of fields to improve existing systems and develop alternatives. This includes improving practices so that resources are used more efficiently (e.g. precision agriculture), developing higher yielding varieties (e.g. breeding programs and development of genetically modified organisms), developing new techniques and technologies (e.g. land-based recirculating aquaculture systems, integrated pest management), and reorganizing systems so that nutrient cycles are managed more effectively (e.g. permaculture, integrated multi-trophic aquaculture). What these show is that intensification is a diverse collection of processes and activities which requires critical examination to determine optimal trajectories for future development.

Examining intensification of all systems is an enormous task that is far too broad for one person, let alone one project. Therefore in this thesis, I tackle one small aspect of this larger challenge. This is accomplished in two parallel routes of inquiry. First, in chapter 2, I investigate the environmental impacts of alternative production systems from a conceptual standpoint drawing on theory from the discipline of ecological economics. Here I develop a conceptual framework that draws on the relationship between BPS and ecosystem services (ES) to help in identifying intensification strategies (actions taken to increase yield) that are less likely to induce environmental damage. Then, in chapter 3, I examine a specific production system (a novel salmon aquaculture production setting) using life cycle assessment methodology. These two modes of inquiry provide independent yet complementary information on BPS by exploring how we conceive the environmental impacts that result from BPS in general and then measuring the impacts of a particular system. In chapter 4 I link these two approaches by applying the insights from chapter 2 to the system studied in chapter 3 through an investigation of the environmental costs associated with the application of a waste capture technology to the aquaculture system modelled in chapter 3.

The purpose of this research is to provide novel insight into our understanding of BPS and their environmental impacts by drawing on knowledge of ES. It is hoped that this knowledge can be used to guide future development and intensification of BPS (both in general and specific to salmon aquaculture) so that environmental impacts can be reduced. The objectives of the thesis are threefold and correspond to the three central chapters (2-4):

- to develop and propose a conceptual framework exploring the relationship between ecosystem services and environmental impacts in BPS (chapter 2)

- to quantitatively assess the environmental impacts of a novel aquaculture technology (chapter 3), and
- to explore the utility of the framework by applying it to the aquaculture technology that was investigated in chapter 3 (chapter 4).

1.1 Thesis Structure

The three central chapters of this thesis (2-4) are written such that they are standalone manuscripts to be submitted for potential publication. Together they highlight how decisions about BPS influence the environmental impacts and the ES of pertinent ecosystems. This knowledge will help managers, researchers, and decision-makers in the quest to achieve increased production while minimizing environmental harm. A discussion of these chapters is provided in chapter 5 along with an examination of strengths, weakness and areas for potential research. To provide context for these manuscripts and subsequent discussions, the balance of this chapter explores relevant literature and information pertaining to ES, life cycle assessment methodology, and salmon aquaculture. This is to provide readers of this thesis with an appropriate introduction to topics that are discussed throughout. Additionally, it provides an opportunity to elaborate on concepts which are not fully explored in chapters 2-4 because they have been written for the purpose of publication. I am the primary author of all chapters in this thesis, with substantial contributions from my supervisor Dr. Peter Tyedmers throughout. Dr. Nathan Pelletier, is a co-author of chapter 3 and has made considerable contributions to the methodological development of that chapter.

The conceptual framework I develop in chapter 2 is something I've called the Framework for the Evaluation of Ecosystem Service Substitution in Agro-Ecosystems (FEESSA). I propose that it can be used to explain the environmental impacts of BPS based on their relationship with, and substitution for ES. Of particular interest is the environmental cost associated with substitution of ES by using technology and/or external inputs. I suggest that ES are used to support BPS and in turn are affected by them so that any change in one can be understood in the context of the other. Moreover, the resource requirements and environmental impacts associated with any given BPS is related to the type and extent to which ES substitution takes place. Examining BPS through this lens as they intensify allows the changing relationship with ES to be explicitly described and the consequences of these changes to be understood. I argue that this provides some explanatory power for why environmental impacts differ between BPS that has heretofore been under-utilized and never formally described. The insight provided by this conceptual framework gives new meaning to the understanding of BPS and allows novel trajectories of investigation. For example, using FEESSA one could examine whether there are patterns in BPS whereby systems that utilize ES most effectively result in lower environmental impacts. I believe that this framework is relevant to a variety of disciplines and can be useful as a tool for guiding development of BPS in the future. The purpose of this framework is threefold: 1) it provides novel perspective and insight into the intensification of BPS that can be used to spur innovation, 2) it can be used to assess existing BPS so that differences in environmental impacts can be explained based on the flows of ES, and 3) it can be used to predict the least impactful alternative amoung intensification strategies.

While useful, it is not sufficient to solely explore BPS from a conceptual standpoint. It is also useful to quantitatively assess the environmental impacts of these systems; to this end I employ life cycle assessment (LCA) to quantify the material and energy flows of a floating, solid-wall aquaculture system (SWAS) and to characterize the associated environmental impacts (chapter 3). This provides a baseline measurement of environmental performance against which future iterations of the system and/or other similar systems can be compared. Additionally, the LCA provides useful analysis of environmental hotspots, allowing areas which contribute to poor environmental performance to be targeted for improvement.

To investigate the trade-offs associated with substitution of ES that are suggested by FEESSA in a more explicit manner, a modelling exercise is undertaken in chapter 4. Here the trade-offs associated with using a particular technological package to substitute for an ES from the aquaculture system in chapter 3 is examined. Based on some simple assumptions the life cycle impacts associated with the capture and treatment of solid waste from the aquaculture system in chapter 3 are calculated. FEESSA provides insight into the causes of environmental impacts from a theoretical perspective while using LCA enables the quantification and analysis of a specific substitution in this aquaculture system. This demonstrates in a very straightforward manner the application of the FEESSA framework while simultaneously highlighting the environmental consequences of employing this particular technology. This information is specifically applicable to the aquaculture system that was assessed and broadly relevant as a guide for others to assess the environmental impacts associated with the intensification of other

BPS. It can be used by managers in the conceptual and technical development of future systems so that they have a better understanding of the relationship between BPS and ES. Ideally, this knowledge can be used to develop systems with reduced environmental impacts.

1.2 Ecosystem Services

Primary producing organisms transform radiant and chemical energy into biologically available energy that can be transferred (via consumption) to higher trophic levels. This accumulation of energy has allowed the evolution of myriad complex organisms which co-exist in the varied ecosystems of the earth. Many of the biophysical processes of these ecosystems produce goods and services that benefit humans. It is the provision of these goods and services by ecosystems - breathable air, drinkable water, natural foods and other natural resources - that we call ES (Fisher, Turner, & Morling, 2009). The concept of ES is used throughout this thesis; to support this use a brief overview of ES is provided here.

Humans have recognized the benefits of ecosystems for millennia; however, it has become increasingly clear that modern economic and management systems are not sufficiently capable of incorporating the benefits provided by ecosystems into decision making (MEA, 2005). This is demonstrated by the continued degradation of natural environments, loss of biodiversity, and existence of market failures such as externalities whereby negative environmental impacts of development and production are not reflected in pricing of produced goods and services (Godfray et al., 2010). The development of the FEESSA concept was a reaction to these issues and the identified need for improved decision-making support tools regarding intensification and understanding of ES (Doré et al., 2011; Foley et al., 2011; Garnett et al., 2013). It is hoped by many that incorporating knowledge of ES into modern decision making contexts will promote the recognition of benefits of ecosystems, provide a more holistic accounting for changes in welfare, and facilitate the analysis of trade-offs (De Groot, Alkemade, Braat, Hein, & Willemen, 2010; Lamarque, Quétier, & Lavorel, 2011). Additionally, the use of the ES concept provides an avenue through which it is possible to address market failures such as externalities (Fisher et al., 2009).

ES have been variously defined as:

- the capacity of natural processes and components to provide goods and services that satisfy human needs, directly or indirectly (de Groot, 1992).
- the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life (Daily, 1997).
- the benefits people obtain from ecosystems (MEA, 2005).
- the aspects of ecosystems utilized (actively or passively) to produce human well-being (Fisher et al., 2009).

This is not an exhaustive list of all definitions but rather a sample to demonstrate the variable use of the term ES. Common to all of these definitions of ES however are two points: a) they must benefit humans in some way and b) they are provided by ecosystems. Making explicit that humans are the beneficiary of services is important as it is our existence as 'valuing agents' that allows value to be recognized in components of nature (de Groot, Wilson, & Boumans, 2002). Specifying that ecosystems are the sources from which these services are generated is important for two reasons. Firstly, because it differentiates them from other services which are generated or mediated by humans. More importantly, however, it acknowledges that these services do not arise spontaneously; they are provided by functioning ecosystems which have a physical structure and are maintained by natural processes.

While the majority of scientists agree in principle on the concept of ES and the way in which services arise, there is much less clarity regarding the classification and measurement of ES delivery (Villamagna, Angermeier, & Bennett, 2013). Much of this debate stems from the difficulty in specifying the actual moment at which we recognize the provision of ES. This is especially important in valuation of ES because it helps prevent double counting. For example, in the production of crops, if valuation is done on pollination, soil nutrient cycling and the actual crop produced then the overall production system will be overvalued. To reconcile this issue several authors have suggested frameworks for guiding the classification of ES (Boyd & Banzhaf, 2007; de Groot et al., 2002; Fisher et al., 2009; Villamagna et al., 2013; Wallace, 2007). A significant outcome of these works is to suggest a difference between final services, which directly benefit humans, and intermediate services that lead to the production of final services (Boyd & Banzhaf, 2007; Fisher et al., 2009). Importantly, it is now recognized that a single definition and classification scheme are unlikely to be useful because linked ecological-economic systems are highly complex and dynamic, therefore any classification for ES should be based on the management context and purpose of the research (Fisher et al., 2009).

As valuation is not part of this thesis, double counting is not an issue, I therefore draw on the classification scheme proposed in the Millennium Ecosystem Assessment which considers ES to be any benefit humans obtain from ecosystems and does not distinguish between final and intermediate ES (MEA, 2005). In this classification scheme, ES are classified into four categories: provisioning services which provide materials for sustenance and other uses; regulating services which regulate potentially harmful natural events such as floods and changes in ocean pH; supporting services that underpin the maintenance of other ecosystem services such as carbon cycling; and finally cultural services which provide non-material benefit such as religious and recreational opportunities (MEA, 2005). Some of these are essential to our continued existence, while others merely provide improvement to welfare. This is an appropriate classification scheme because it recognizes the services which underpin agricultural production (i.e. supporting services) which are a primary focus of this thesis.

1.2.1 Ecosystem Structure

An important part of the literature around ES concerns ecosystem structure and the labels/definitions applied to them (Lamarque et al., 2011). Taking a closer look at ecosystems allows the distinction between ecosystem components and processes. Ecosystem components are the biophysical parts and characteristics of ecosystems. The many components of an ecosystem together characterize its composition. These have been described in academic literature under various labels including ecosystem structure, infrastructure, stocks or capital (Fisher et al., 2009; Wallace, 2007). They include: harvestable components such as flora, fauna, and other natural resources; physical structures such as trees, rocks, and soils; and the characteristics of those such as pH and temperature. Ecosystem processes, also called flows or functions, occur as transfers of material or energy (Wallace, 2007). They include all naturally occurring physical, biological, and chemical interactions between the components of an ecosystem (Boyd & Banzhaf, 2007). Weathering of rock, parasitism, and photosynthesis are all examples of ecosystem processes. They include all transformations and interactions of nature which maintain the integrity of ecosystem structure and processes. A key difference between ecosystem components and processes is that components are typically measured in physical amounts (e.g. mass or volume) while processes are typically measured as rates of flow (e.g. kg/s) (Wallace, 2007).

Building on the understanding of ecosystem structure and function, Dominati, Patterson, & Mackay (2010) make the connection that ES flow from natural capital specifically, rather than from ecosystems in general; where natural capital are those components of ecosystems that generate a flow of ecosystem goods or services. This form of capital contrasts with the two other major types: human capital (e.g. knowledge and labour) and manufactured capital (e.g. factories, produced goods), collectively known as human-made capital, that are generated by humans (Costanza & Daly, 1992; Dominati et al., 2010; Ekins, Simon, Deutsch, Folke, & De Groot, 2003). Following de Groot (1992) and Dominati (2010), for the purposes of my thesis, I define ecosystem services as *the flow of direct and indirect benefits to humans from natural capital*. Integrating this definition with our understanding of ecosystem components and processes we are able to develop a working framework of ecosystem dynamics (Figure 1). This is useful because of its simplicity; however, in reality ecosystems are not so simple and are comprised of complex interactions between components and processes. Therefore to avoid confusion, I provide the following additional guidance:

- Ecosystem components that provide benefits are natural capital.
- Ecosystem processes that support natural capital (indirect benefit) and/or deliver direct benefit are considered ES.
- There is not necessarily a one-to-one relationship between natural capital and ecosystem services. Some stocks of natural capital will produce more than one good or service. While the provisioning of some ecosystem services may require the interaction more than a single stock of natural capital (de Groot *et al,* 2002).
- Because of the interconnected nature of ecosystems, many ecosystem processes and components are supported by other processes and/or components (MEA, 2005).
- When supporting ecosystem services are combined with other forms of capital (e.g. manufactured capital) to produce a product or service, they can still be considered an ES (e.g. industrial agricultural production). This contrasts with the view of Boyd & Banzhaf

(2007) who consider these to be human or manufactured capital once they are processed in any way.

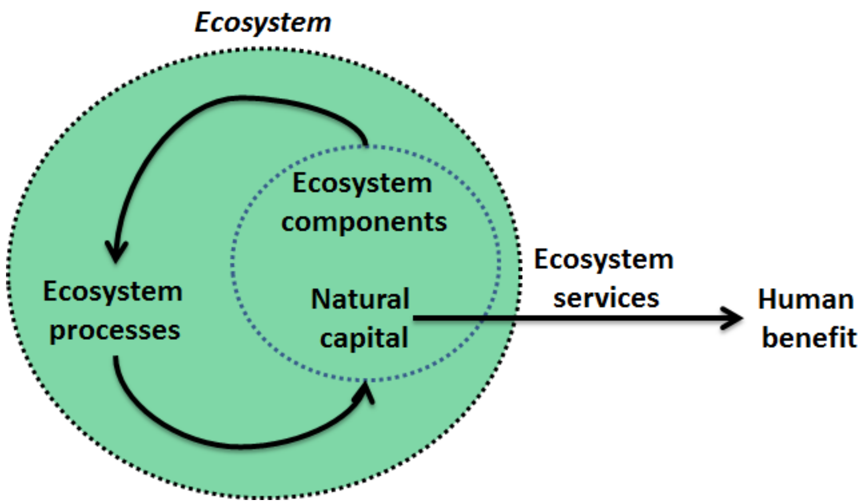

Figure 1. Framework of ecosystem dynamics, illustrating the relationship between ecosystem components, processes, and the provision of ecosystem services

1.2.2 Substitutability of Natural Capital and Ecosystem Services

When ecosystems become degraded for any reason (e.g. through destruction of habitat or changing conditions), natural or otherwise, and the services they provide are negatively impacted, action may be taken in an attempt to conserve them. Likewise, if an ES is recognized as being particularly beneficial, effort may be used to augment it (e.g. habitat restoration/management to increase the abundance of pollinators). An alternative course of action in both of these situations is to use technology and/or external inputs (i.e. human-made capital) to provide a functionally equivalent substitute. While natural capital is generally considered a complement to human-made capital rather than a substitute (Costanza & Daly, 1992), I suggest that it is possible to replace certain ES by using other forms of capital (e.g. producing clean drinking water through distillation of contaminated water or desalination of saltwater) and vice-versa (e.g. reliance on wind-powered transport rather than fossil fuel powered motors). This is not a substitution for the natural capital itself but rather for the goods and services they provide. This is accomplished by using human and manufactured capital, in combination with natural resources, to provide the same benefit as the ES which is being substituted.

Currently, our understanding of where/when these substitutions are possible, the environmental trade-offs, and the role of technology is very poor. In chapter 2 of this thesis I explore this phenomena conceptually and develop a framework upon which trade-offs can be assessed. Additionally I investigate the environmental trade-offs of an aquaculture system in a more focused fashion using LCA. To my knowledge this has never been done before. In order to provide background on the use and utility of LCA the following section explores LCA methodology and practice.

1.3 Life Cycle Assessment

There are many tools and techniques which are used to assess the sustainability and environmental aspects of policies, projects, products and processes (Buytaert et al., 2011). Some notable examples include environmental impact assessment; input/output, energy, emergy, and exergy analyses; and ecological and carbon footprinting. These have been organized in several ways, for example Ness, Urbel-Piirsalu, Anderberg, & Olsson (2007) categorize sustainability tools based primarily on temporal characteristics and the focus of the assessment, while Finnveden & Moberg (2005) do it based on the goal, the types of impacts considered, the focus (similar to Ness et al., 2007), and whether it is descriptive or change-oriented. Each of these tools has strengths and weaknesses and is appropriate for different purposes. In this thesis life cycle assessment has been chosen to assess the environmental impact of a salmon aquaculture production system because it has broad applicability and incorporates many of the capabilities of other tools (e.g. carbon footprinting and energy analysis) in one comprehensive assessment.

LCA is a broadly standardized, though methodologically flexible, quantitative accounting technique that is used to account for flows of material and energy throughout the entire life cycle of a product or process (ISO, 2006a; Pryshlakivsky & Searcy, 2013). It is used to assess environmental impacts associated with resource use and emissions from the provision of products or processes (Baumann & Tillman, 2004). A significant advantage of LCA is that it can be used to characterize the entire process being studied from the acquisition of raw materials through to manufacture, production, use, and disposal (typically described as cradle-to-grave), providing a more holistic approach to characterizing impacts. It is also used to pinpoint the environmental impacts associated with specific production stages that contribute disproportionately to impacts. This allows the key drivers of potential environmental damage to be identified and communicated to interested parties such as production managers, environmental monitoring agencies, and concerned public. This knowledge can then be used to target improvements in environmental performance of production processes (Pryshlakivsky & Searcy, 2013). Additionally, where the fundamental methodological assumptions are consistent, the impacts of competing systems can be compared and the trade-offs evaluated (Henriksson, Guinée, Kleijn, & de Snoo, 2011).

1.3.1 Definition of Goal and Scope

LCA is typically described as a four step process that includes: definition of goal and scope, inventory analysis, impact assessment, and interpretation (Baumann & Tillman, 2004). During the definition of goal and scope it is important to set the study boundaries, define the goal of the study and describe the functional unit, relative to which the results will be communicated. This provides an opportunity to plan the LCA and assess what information is needed and how it can be obtained. Boundaries of the study system include all processes that are relevant and will be included in the proposed study. These are often represented using a flow model that shows the relationship between important inputs and outputs of the study system (Baumann & Tillman, 2004). The functional unit is a measurable amount of product or service that is relevant to the goal of the system being assessed and is typically (although not necessarily) expressed as a multiple of 10 for conceptual ease (e.g. 1kWh, 10kg, 1000 L).

1.3.2 Life Cycle Inventory Analysis

Once the goal and scope of the study has been defined, data on the product or process being studied are gathered during the inventory analysis (Baumann & Tillman, 2004). These include inputs and outputs of material and energy throughout the life cycle. They can be categorized according to foreground data of the system under study and background data from products or processes which feed into the study system. This step is facilitated through the use of databases which contain inventory data on a range of services and products collected by industry, government, and researchers that are used for background data and to approximate foreground data that is unavailable (or the collection of which is constrained due to limited time or resources) (Finnveden et al., 2009). These databases are continually being updated and new ones being created, adding to the resources available for LCA practitioners. Once data are collected they are organized and analyzed relative to the outputs of interest, all of which are organized into relevant subsystems that contribute to the production of the functional unit.

1.3.3 Co-Product Allocation

An important problem associated with the life cycle inventory analysis is the issue of co-product allocation of environmental burdens (Ayer, Tyedmers, Pelletier, Sonesson, & Scholz, 2007). This occurs when there are multiple products being produced that share at least part of a single production process during their life cycles (ISO, 2006a). The question that needs to be addressed in these situations is "how much of the impacts of the shared process are attributable to each co-product?". In the event that co-products occur, allocation can be avoided by system expansion. This is accomplished by attributing environmental burden to the co-products which are not being assessed based on the known environmental burden of a functionally equivalent product (Finnveden et al., 2009). The burden associated with the co-product is then credited to the production process so that all remaining burden is attributable to the product being studied. Often this is not possible because a suitable substitute is not available. In this instance allocation is based on economic valuation or a biophysically relevant characteristic of the co-products (Ardente & Cellura, 2012).

1.3.4. Life Cycle Impact Assessment

After inventory analysis is complete the life cycle impact assessment is conducted. This involves the use of characterization factors to assess the contribution of inputs/outputs (collected in the inventory analysis) to resource use and emissions related impact categories. These impact categories can be custom built to suit particular applications but impact categories with well-developed and peer-reviewed methodologies are the norm. Often individual impact categories are bundled together as a group and published as part of a larger impact assessment method. For example one commonly used impact assessment methodology is the ReCiPe midpoint method which includes impact categories such as Global Warming Potential (GWP), Acidification Potential (AP), Ozone Depletion (OD), Freshwater Eutrophication (FO), Human Toxicity (HT), and Fossil Depletion (FD) (Goedkoop et al., 2009). The impact categories used in LCA can be categorized as mid-point or end-point. Mid-point impact categories assess potential damage to human and environmental issues while end-point assesses actual damage in terms of the real degradation in quality of life or environment (ISO, 2006b). It may seem preferable to use end-point categories however there is often significant uncertainty associated with our understanding of the relationship of emissions or resource depletions with actual damage, making their use problematic (Reap, Roman, Duncan, & Bras, 2008b). For this reason it is often preferable to use mid-point impact categories. The actual calculation of impact categories is based on the characterization of material and energy flows using characterization factors. These relate all relevant emissions and resource use issues to a common unit. For example, there are several types of chemicals that have been identified as contributors to climate change but they do not all contribute equally. Using characterization factors it is possible to translate all of these into their equivalent emissions of CO_2, reported as kilograms of CO_2 equivalents (kg CO_2e). Once all characterization and calculations are complete, the results of the impact assessment are presented relative the functional unit of the study (e.g. X kg of CO_2e per functional unit).

1.3.5 Interpretation

The fourth step in LCA is the interpretation phase. It is an iterative process that actually takes place throughout the completion of an LCA whereby significant results are identified and evaluated (ISO, 2006a). A discussion of these is then presented along with an analysis of the strengths, weaknesses, uncertainties and conclusions. Recently there has been a larger focus on uncertainty in the interpretation of LCA (Finnveden et al., 2009). These issues are described in greater detail in the following section.

The completion of LCA is often supported throughout by the use of specialized software. These software packages can be used to manage and organize data, construct models, calculate and characterize impacts among other things. Moreover they usually contain several databases and impact assessment methods. Here I used SimaPro 7.3 to facilitate the LCA work (PRé Consultants, 2010).

1.3.6 Uncertainty in Life Cycle Assessment

One of the weaknesses of LCA is the difficulty associated with adequately addressing uncertainty (Ross, Evans, & Webber, 2002). Fortunately, there are several techniques which can be used to test results including sensitivity and other uncertainty analyses such as Monte Carlo simulation. These help the LCA analyst to evaluate the significance of the results in the face of uncertainty. Sensitivity analysis is used to test the importance of particular parameters on the results of the life cycle impact assessment. It is done by altering one value to test how that change influences the results. For example, salmon feed conversion ratios (FCRs) represent the ratio between the mass of feed used and salmon produced. These range from above 1.6 to below 1 (e.g. see Pelletier et al., 2009; Wilfart, Prudhomme, Blancheton, & Aubin, 2013). In a typical production cycle FCR may be 1.3 but the analyst may be interested to see how a good (1) or poor (1.6) FCR may impact the results. Monte Carlo simulation is used to test the cumulative uncertainty of multiple parameters. By characterizing the individual uncertainty associated with each parameter and then allowing them to vary simultaneously for a fixed number of runs (e.g. 1,000) it is possible to generate an uncertainty distribution for life cycle impacts. Sensitivity analysis and Monte Carlo simulation are examples of methods used to explore uncertainty in LCA, however, there are many other methods by which uncertainty can be investigated both qualitatively and quantitatively (Lloyd & Ries, 2007). Applying these analyses appropriately helps improve the confidence and reliability of LCA results.

1.3.7 Scenario Analysis

In addition to uncertainty in data used to build LCA models and assess impacts there is variability in the context in which production takes place. Moreover managers exert a degree of control over life cycle impacts based on decisions regarding the production system. Each of these can possibly influence the life cycle impacts of a system under study. It is therefore of interest to the LCA practitioner to use scenario analysis to explore how the outcomes would change under relevant scenarios. Typically these are used to describe competing alternatives for production. For example, feed formulations are a particularly important driver of environmental impact in salmon aquaculture systems (Pelletier & Tyedmers, 2007); by using scenario analysis it is possible to explore changes in life cycle impact associated with using alternative feeds.

1.3.8 LCA in Aquaculture Systems

LCA has been used extensively for assessing agriculture and livestock systems (de Vries & de Boer, 2010; Nijdam, Rood, & Westhoek, 2012; Roy et al., 2009) and it is increasingly being used to assess seafood and aquaculture systems (Ayer & Tyedmers, 2009; Jerbi, Aubin, Garnaoui, Achour, & Kacem, 2012; Mungkung & Gheewala, 2007; Pelletier et al., 2009). This tool is much more comprehensive than single indicator sustainability assessments done on aquaculture in the past such as ecological footprint or indicators related to waste emissions (Samuel-Fitwi, Wuertz, Schroeder, & Schulz, 2012). It provides information on aquaculture systems and their supporting processes such as smolt production, feed manufacturing and transportation. This makes it possible to identify environmental hotspots throughout the life cycle. For example, LCAs on aquaculture have found that for carnivorous species such as salmon, the provision of feed is a very large contributor to the overall impact of production (Pelletier et al., 2009). As a result, there has been a considerable amount of research investigating feeds with alternative ingredients to determine suitable substitutes that have decreased environmental impact (Boissy et al., 2011; Papatryphon, Petit, Kaushik, & Werf, 2004; Pelletier & Tyedmers, 2007).

Several LCA practitioners studying aquaculture have investigated issues related to the methodological development of LCA. Ayer et al. (2007) explored co-product allocation and its implications in seafood systems. Allocation in aquaculture systems was important for feed subsystems, co-products at the farm-gate, and waste from processing. They found economic allocation to be most widely used technique but suggested that biophysically relevant allocation procedures (e.g. allocation based on gross energy content) be used instead. Pelletier et al. (2006) analyzed impact categories used in LCAs of seafood systems. They found that outside of some well-established impact categories, LCA on seafood systems were generally poor at incorporating local ecological and societal impacts. There have even been efforts to remedy this situation both in fisheries (Emanuelsson, 2012) and aquaculture (Ford et al., 2012), however to date this remains a weakness of the LCA methodology.

One of the common goals of LCA of aquaculture is to better understand the impacts associated with production so that optimal methods and technologies can be identified. In this thesis I contribute to this growing body of literature by conducting an LCA of a unique aquaculture system that has been employed on the west coast of Canada. To provide context for this assessment the following section provides a brief overview of salmon aquaculture and the various technologies used to grow them.

1.4 Salmon Aquaculture

Globally production of salmon is increasing rapidly; growing at an annual rate of 9.5% between 1990 and 2010 (FAO, 2012). Total production of salmon in 2011 (the most recent year for which data are available) was 1,721,254 t (FAO Fisheries and Aquaculture Department, 2013). Canada is currently the fourth largest producer of salmon behind Norway, Chile, and the United Kingdom. In 2012, approximately 108,000 tonnes of salmon were produced in Canada (DFO, 2014). Most of this production occurs in British Columbia (66%) and New Brunswick (28%) with the remainder coming from Nova Scotia, Newfoundland and Labrador, and Prince Edward Island (DFO, 2014). The majority of this salmon is Atlantic (*Salmo salar*) but Chinook (*Oncorhynchus tshawytscha*),Coho (*Oncorhynchus kisutch*) and Rainbow trout (*Oncorhynchus mykiss*) are also farmed (CAIA, 2014).

Although there are now some small scale commercial and research projects growing salmon using various grow-out technologies, production in Canada is conducted almost exclusively in net-pens in the near-shore marine environment (CAIA, 2014). These consist of circular or rectangular pens that have a structural collar which floats on the surface (Stead, Selina & Laird, 2002). Attached to the collar are nylon nets that are anchored to the bottom which are used to contain the salmon. Producers will also commonly surround these nets with large, robust, nets to exclude marine predators. To deter avian predators from feeding on fish, bird nets can also be placed above the pens.

Actual production of salmon begins in land-based hatcheries where salmon are grown from eggs and sperm obtained from broodstock. Fertilized eggs are hatched and reared in the hatcheries until they reach approximately 100g (after 12-18 months). They are then transferred to the marine net-pens (or other grow-out technology) and fed a nutrient rich diet until they reach market size (which takes another 12-18 months). The ingredients used to create salmon feeds come from a wide array of agriculture and fisheries systems (Naylor & Burke, 2005; Pelletier & Tyedmers, 2007). These are formulated to optimize growth and nutrient uptake in the most economically efficient way (Pelletier & Tyedmers, 2007) and/or meet sustainability objectives (e.g. certification standards).

Salmon is a high value, intensively cultivated species, which is associated with several environmental concerns. These are summarized in four categories:

- The provision of feed for carnivorous species such as salmon relies heavily on fish meal and oil derived from wild fisheries, thus contributing to over-fishing (Gatlin et al., 2007; Tacon & Metian, 2009). While these species are usually so called ‘forage fish’ (small pelagic fish that have low commercial value), they play a significant role as food in many developing nations (Naylor et al., 2000). In addition, as the feed requirements of aquaculture increase, they are being met at the expense of other more valuable and threatened stocks (Péron, François Mittaine, & Le Gallic, 2010).

- On-site emissions from feed, feces, antibiotics, and other chemicals used in aquaculture can result in localized ecological impacts (Frankic & Hershner, 2003; Tucker, Hargreaves, & Boyd, 2008). Excess nutrients can disrupt local ecological systems by contributing to eutrophication (Gyllenhammar & Håkanson, 2005) and/or benthic impacts (Kalantzi & Karakassis, 2006) while other chemicals can be toxic to marine organisms (Burridge, Weis, Cabello, Pizarro, & Bostick, 2010).
- Negative interactions with cultured fish can impact local wildlife. The high stocking density of aquaculture operations promotes the maintenance and transmission of disease which can then spread to wild populations in the vicinity of the farm (Krkosek, Lewis, Morton, Frazer, & Volpe, 2006). In addition, escapees from aquaculture operations can harm wild fish through inter-breeding, predation, and competition (Fleming et al., 2000; McGinnity et al., 2003). Finally, the presence of fish can also attract predators such as birds and marine mammals which are sometimes disposed of using lethal means.
- The life cycle impacts associated with aquaculture production and supporting processes can contribute to several environmental concerns such as climate change, acidification, and depletion of non-renewable resources (Pelletier et al., 2009; Wilfart et al., 2013). This issue is particularly important because aquaculture systems developed to improve the local ecological impacts have been found to increase the life cycle impacts resulting in environmental trade-offs (Ayer & Tyedmers, 2009).

In an effort mitigate or eliminate some of these concerns, considerable effort has been expended to improve nearly all aspects of production including: feed formulations, husbandry techniques, fish health, and management practices (Belle & Nash, 2008; Gatlin et al., 2007). Chief among these has been the focus on technological solutions such as growing fish in land-based recirculating aquaculture systems (RAS) and other alternative systems (Chadwick, Parsons, & Sayavong, 2010; Martins et al., 2010). These systems typically have two characteristics that differentiate them from net-pens: 1) a physical barrier that separates the culture environment from nearby habitat and 2) some form of waste capture and/or treatment.

The barrier can be solid such as the case in RAS and SWAS or it may be flexible as in marine-bag systems (Chadwick et al., 2010). In either case its purpose is to provide a more secure containment (i.e. reduced likelihood of escapes) and greater control over the culture environment. Moreover, by preventing immediate emission to the environment, the barrier potentially allows waste to be captured and treatment to occur. These treatment systems vary a great deal in design and efficacy; some capture and treat only solids while others also capture soluble wastes through biological or chemical processes. Another important characteristic of these alternative technologies is the degree to which water is recirculated. They range from completely flow-through (e.g. SWAS) to upwards of 99% recirculation (RAS). These are favoured by many environmental non-governmental organizations as a method for reducing local ecological impacts. Unfortunately, there are indications that these systems are more material and energy intensive (due to infrastructure, pumping, heating, and cooling), resulting in higher life cycle impacts than net-pens (for example see Ayer & Tyedmers, 2009; Wilfart, Prudhomme, Blancheton, & Aubin, 2013), potentially creating a situation where local impacts are improved at the expense of global and regional environmental concerns.

In this thesis the environmental impacts of a novel aquaculture production system that was employed in British Columbia, Canada were assessed using LCA (chapter 3). This system represents an alternative to net-pens that has never been assessed using LCA. It is a SWAS that operated near Campbell River, BC, Canada from December 2010 to March 2012. The tank measured 24m in diameter with an internal volume of $3000m^3$. It was cylindrical and constructed primarily of fibre reinforced polymer panels and a reinforced ring made of galvanized steel. The impermeable structure separated the culture environment from the marine ecosystem, affording a certain level of control over influent and effluent. Moreover, the structure of the tank was intended to minimize the risk of escaped fish, transmission of disease, and interactions with predators. The influent was made up of seawater that was continuously pumped in from an intake pipe with additional oxygen injected. Oxygen was generated on-site using oxygen concentrators. Effluent was pumped through a waste capture system that was attached to the tank. This operated intermittently throughout the production cycle due to technical problems. Electricity used to operate the pumps was from the local electricity grid. Many characteristics of this system including the waste capture capabilities and the feed formulation are unique and provide valuable insight into the environmental impacts of salmon aquaculture and its relationship to ES. These are investigated in chapter 4 by drawing on the conceptual framework proposed in chapter 2.

1.5 Guiding Statements

BPS are one of the most important drivers of environmental impact on a planetary scale. The productivity of these systems relies a great deal on the support of ES. In this thesis I explore how the relationship between BPS and the ES they receive influences their environmental impacts. This is accomplished first by developing a conceptual framework that provides the conceptual groundwork for further investigation (chapter 2). Then using LCA, the environmental impacts of an aquaculture system are assessed (chapter 3). The LCA in and of itself does not provide any insight into the relationship of ES and BPS. However, I then relate the substitution of a particular ES in the aquaculture system to the marginal increase in environmental impacts associated with it (chapter 4). This provides, for the first time, a quantification of the environmental trade-offs that result from the substitution of ES in a BPS. To support the work described herein, I have provided here a brief overview of ES, LCA, and salmon aquaculture. A discussion of these topics is provided in chapter 5 along with an examination of limitations and opportunities for future research. Ultimately, this thesis represents one piece of a much larger story but it is hoped that this line of investigation will inspire further work and provide insights into previous work that will help to achieve more productive and less impactful BPS.

Chapter 2. Sustainable Intensification: Conceptualizing Intensification Through the Lens of Ecosystem Services

2.1 Introduction

Environmental impacts of biological production systems (BPS) – including food systems such as agriculture, aquaculture, and livestock and non-food systems such as crops used for fuels and fibre – are more globally significant and widespread than at any other time in human history (Balmford et al., 2012; MEA, 2005; Rockström et al., 2009; Tilman, 1999; Tilman et al., 2001). With population expected to reach 9.6 billion by 2050 (UN, 2013) and the increased affluence of a developing world (Myers & Kent, 2003) demand for agricultural products is expected to rise substantially, contributing to ever more significant environmental degradation (Borlaug, 2002; Fedoroff et al., 2010; Foley et al., 2011; Godfray et al., 2010; Pelletier & Tyedmers, 2010a). The challenge facing humanity is to find ways to increase production while limiting associated environmental impacts, natural resource depletion and the degradation of ever-decreasing amounts of uncultivated land (Rockström et al., 2009; The Royal Society, 2009).

Given the relatively small amount of productive uncultivated land which remains and the desire to protect its ecological value for maintaining biodiversity and ecosystem services (ES, the benefits obtained from ecosystems MEA, 2005), increasing yield of current BPS, or intensification, has been identified as the preeminent source for future increases in agricultural production (Cassman, 1999; Ellis, Klein Goldewijk, Siebert, Lightman, & Ramankutty, 2010; Lobell, Cassman, & Field, 2009). While debate still exists on its ability to achieve environmental (e.g. land-sparing) and other societal goals (e.g. reduction of poverty) in the face of interconnected social, political, and economic forces, intensification is none-the-less an important part of a global solution (Balmford et al., 2012; Godfray & Garnett, 2014; Lambin & Meyfroidt, 2011; Sayer & Cassman, 2013). By reducing yield gaps (i.e. differences in yield between less productive and more productive BPS) of under-performing systems, especially in developing nations, increased production can be achieved while reducing the need for further expansion (Foley et al., 2011; Lobell et al., 2009). Unfortunately, many of the practices by which this has been achieved (e.g. high use of fertilizer, pesticides, water, fossil fuels, landscape simplification, and mechanization) contributes to significant resource use and environmental degradation (MEA, 2005; Robertson & Vitousek, 2009; Rockström et al., 2009; Stoate et al., 2001). As a result, a growing body of literature has developed around the idea of sustainable intensification, promoting practices whereby yield is increased while environmental impacts are reduced (Bommarco, Kleijn, & Potts, 2012; Foley et al., 2011; The Royal Society, 2009; Tilman et al., 2011; Tilman, Cassman, Matson, Naylor, & Polasky, 2002). This research has highlighted methods including: improved biotechnology, precision agriculture, integrated pest management, less intensive tillage, and incorporation of crop residues (Ball, Bingham, Rees, Watson, & Litterick, 2005; Borlaug, 2002; Gebbers & Adamchuk, 2010; Pimentel, 2009; Robertson & Vitousek, 2009). What this research shows is that despite its common application as a simplified descriptor for increased yield, intensification encompasses myriad techniques, technologies, and management strategies which have highly diverse environmental impacts (Tscharntke et al., 2012) (Figure 2). We can therefore consider every decision made in the present regarding increases in productivity as an inflection point which will partly shape the trajectory of future environmental impact (Figure 3). Consequently, intensification cannot be taken as a solution unto itself but rather a potent tool that requires critical examination to help elucidate which intensification pathways are most beneficial and which are least. The relationship between environmental impact and yield of different intensification strategies (represented by curved lines) in Figure 3 is not a simple one, for this reason the following additional guidance is provided: 1) 'environmental impact' encompasses diverse impacts which will be affected differently by various processes of intensification; 2) environmental impacts and yield of specific BPS and specific intensification strategies will vary through space and time as environmental variables change; 3) some intensification strategies will result in negative environmental impacts while others may result in positive ones (e.g. increased biodiversity), though this is typically the exception; 4) the relationship between environmental impacts and yield is not necessarily linear and may reach thresholds where impacts continue to increase while yield remains the same or even decreases; for example application of fertilizer often increases yield but over-fertilization can result in increased runoff of eutrophying emissions (an environmental impact) and plant mortality (decreased yield). Ultimately, the intensification pathways open to any one producer in a given context are limited by available knowledge or resources, and financial implications of alternatives will dominate individual decision-making. At higher organizational levels (e.g. sector, national) where many more options with diverse environmental implications are available, there is pressing need for decision-support tools upon which alternatives can be judged (Foley et al., 2011).

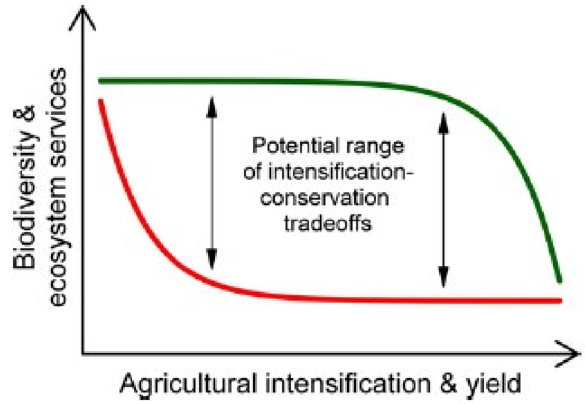

Figure 2. Example from Tscharntke et al. (2012) that demonstrates the variability of intensification on biodiversity and ecosystem services (ES). Here the impacts of intensification on biodiversity and associated ES from tropical agro-forestry (green line) and European agriculture (red line) are demonstrated.

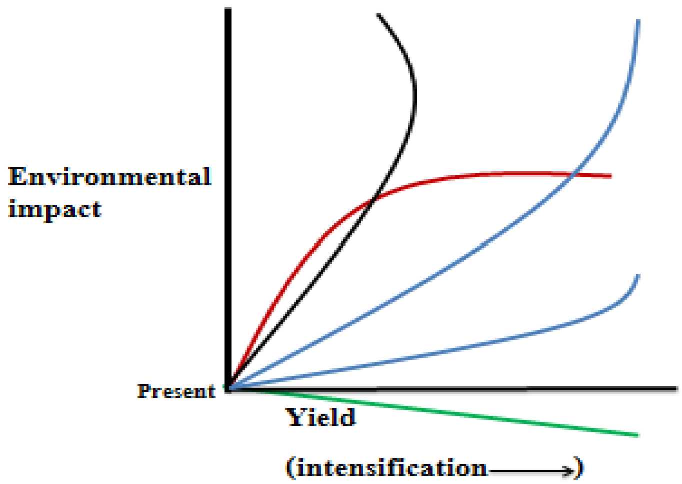

Figure 3. A conceptual illustration of the relationship between yield (x-axis) and environmental impact (y-axis) for diverse intensification strategies. Yield is a measure of agricultural output per unit area. The origin represents the present where decisions regarding intensification are made. Environmental impact is a generic descriptor for a variety of impact categories (e.g. greenhouse gas emissions, biodiversity loss). Each line represents a potential intensification strategy (e.g. increased application of fertilizer).

We suggest that differences in environmental impacts between BPS and intensification strategies are related to the extent to which flows of ES are being altered through the use of technology and external inputs (inputs of material or energy directed by humans that are used to support the productive capacity of BPS e.g. fertilizers, pesticides, and fossil fuels) as substitutes. We contend that each step taken to achieve intensification in BPS can be analyzed through this lens. This can be done by systematically identifying the ES, resource inputs and design of the BPS, then exploring how these change with each intensification step. This will allow alternative strategies of intensification to be compared based on the role of ES and external inputs in that system. We contend that this provides a novel and powerful lens through which intensification (and BPS in general) can be conceptualized. Then, by linking the use of external inputs to their known environmental impacts (for example, see Behrens, Giljum, Kovanda, & Niza, 2007), those intensification pathways that are least impactful can be identified. This knowledge could be used for guiding the design of BPS in such a way that ES and yield are optimized while inefficient resource use and environmental impacts are reduced. In addition to supporting innovation, the knowledge and principles gleaned from the intensification strategies of low impact systems can be operationalized and transferred to other BPS.

To usefully apply the concept of substitution for ES we have developed a classification tool which can be used to assess intensification strategies. We label this concept the Framework for the Evaluation of Ecosystem Service Substitution in Agro-ecosystems (FEESSA). The following is a presentation of FEESSA, its conceptual development (Section 2.2), methodological application (Section 2.3), and an example of its application to the aquaculture sector (Section 2.4). Ultimately this concept is not bound to a specific system of classification but represents a method for conceptualizing the substitution of ES in intensification strategies and relates them to resource use and environmental impact. Thus it is likely that it could be operationalized in many ways other than using FEESSA.

2.2 Ecosystem Services and Substitution in Agro-Ecosystems

Drawing on the framework provided by the Millennium Ecosystem Assessment (2005), ecosystem services are sub-divided here into four types based on the nature of services they provide; provisioning, regulating, cultural and supporting services (MEA, 2005). BPS are cultivated because they provide ES including food, fibre, and fuels but they also provide other less commonly considered benefits including water regulation and purification, wildlife habitat, and promote psychological well-being (Daily et al., 2000; Porter, Costanza, Sandhu, Sigsgaard, & Wratten, 2009; Swinton, Lupi, Robertson, & Hamilton, 2007). In addition to providing services, BPS are part of a larger interconnected agro-ecosystem that is critical to their functioning and provides services including pest control, soil fertility and pollination (Power, 2010). It has been estimated that without pollination, some 75% of major food crop species representing at least 35% of global food production (by volume) would suffer losses of production (Klein et al., 2007). For this reason we consider BPS agro-ecosystems that both provide and receive ES (Swinton et al., 2007). In addition to ES, we also recognize that ecosystems can provide dis-services that negatively affect humans and BPS (Zhang, Ricketts, Kremen, Carney, & Swinton, 2007). These include diseases, pests, natural disasters, and more. In order to increase productivity both ecosystem services and dis-services must be managed. The productive capacities of BPS are supported not just by ES but also by external inputs. Conceptually, we can consider ES and external inputs as flows of materials and energy, the former supplied by ecosystems and the latter by humans. To intensify, producers manipulate these flows to the best of their ability (based on the technology, resources, and knowledge available) to promote supporting and regulating services, limit dis-services, and supplement limiting factors. For example, consider a hypothetical fruit crop that is limited by pollinators. To increase the number of pollinated flowers, the farmer may choose to import honey bee colonies and keep them on-site while the crop is flowering. Alternatively, she may choose to implement habitat management strategies which promote the growth of natural pollinator populations. The first option requires use of external inputs while the second relies primarily on ES. Assuming equal increases in yield then the wild pollinators and the honey bees can be considered substitutes. Traditional methods for intensification, relying predominantly on increasing external inputs are exemplified by the 'green revolution' (Khush, 1999), while promoting the productive capacity of BPS through the use of ES is a newer idea, at least to western research, that is often characterized as 'ecological intensification' (Bommarco et al., 2012).

While both ES and external inputs can be used to intensify, with a certain degree of substitutability between them, there is a critical difference between these two flows. Firstly, the use of external inputs by their very nature represents the appropriation of energy and often materials from outside the agro-ecosystem, frequently derived from non-renewable sources. Thus, using ES for intensification potentially represents a more sustainable pathway as these benefits are generated within the agro-ecosystem and ultimately powered by solar energy. Secondly, the acquisition, production, and use of external inputs often contribute to environmental impacts. This conceptualization of the intensification of BPS is represented graphically in Figure 4.

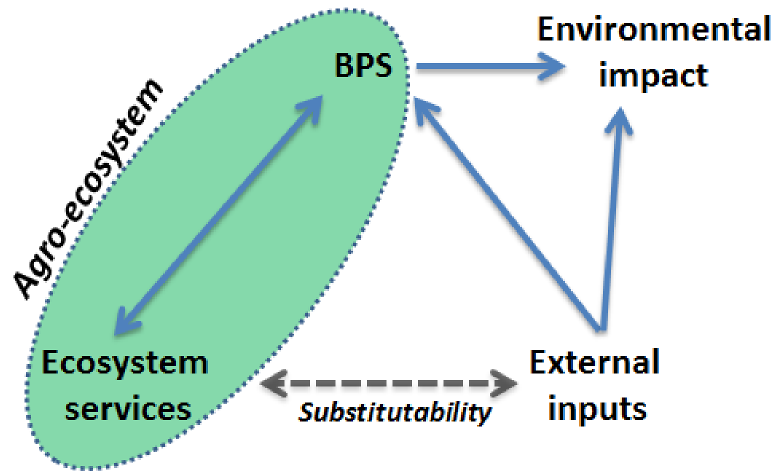

Figure 4. Conceptual model of the relationship between biological production systems (BPS), ecosystem services (ES), external inputs, and environmental impacts. BPS are part of a larger agro-ecosystem which provides supporting ES, while BPS also produce ES. Solid arrows represent causal relationships. Here the BPS is part of a larger agro-ecosystem and as such is both recipient and provider of ES.

The substitutability between ES and other inputs as means of achieving increased yield forms the crux of the FEESSA. It provides a method for conceptualizing BPS that explicitly accounts for the role of both natural and external inputs in production and intensification. Importantly, the interchangeability between ES and external inputs is highly context dependent. The amount, type, and environmental impact of resources required to substitute for ES is not only dependent on the nature of the ES but the technology or technique by which substitution is achieved. This is because the actual delivery of ES is context dependent (e.g. biological control of pests -see Tylianakis & Romo, [2010]). For example, consider the provision of water for an agro-ecosystem through natural cycling (i.e. rainfall) (in this instance we assume that irrigation is not possible). In areas with high rainfall, it may be possible to build rain-capture devices to increase the water supply for intensification. In areas with low rainfall, such as arid coastal regions, this may not be possible therefore desalination can be used, which requires relatively more inputs. In both regions, capital is used to substitute for the provision of water but the provision of water is much more difficult in drier regions. Sticking with this example, we see how technology becomes important because it determines the resources used to substitute for ES. The efficiency of the desalination system will play an important role in determining the energy requirements of this substitution. The consequence of this is that some ES are innately more difficult to modify or substitute than others and will result in more resource use and environmental impact.

Additionally, the cost of substitution is dependent on the context and the technology by which it is achieved.

Taking the concept of ecosystem substitution further we can envision how, the further removed agro-ecosystems are from surrounding ecosystems (i.e. the less it is supported by ES) the more resource intensive it is to support production (Figure 5). This means that it is possible to identify optimal methods for increasing production based on the role of ES. To illustrate this concept it is useful to draw on examples of two different types of BPS: salmon and cattle farming. Life cycle assessments of these systems shows that as salmon aquaculture systems intensify, changing from net-pen systems to more intensive land-based recirculating systems, the per capita greenhouse gas emissions increase dramatically (Ayer & Tyedmers, 2009). On the other hand, in cattle farming, we observe the opposite, whereby per capita emissions of greenhouse gasses tend to decrease in more intensive systems (Nijdam et al., 2012). It is possible to explain why this occurs from a technical standpoint by examining changes in resource use that cause changes in the emissions profile of the systems but conceptually this is not satisfactory. Based on the simple observation that these are both livestock systems that are intensifying we would expect similar outcomes. It is here where the value of evaluating changes in ES becomes apparent. Evaluating the cattle systems we see that through intensification there are only minor changes to the supporting services: a decreased reliance on grazing and increased use of manufactured feeds. In the salmon system, feed use remains the same but many other supporting and regulating services available to net-pens must be managed in the recirculating system. This includes water quality characteristics such as temperature, dissolved gasses (e.g. CO_2, O_2), and waste products (nitrogenous wastes), and physical aspects including water filtration and pumping. Using the conceptual insight provided by analyzing substitution of ES we see that intensifying the salmon system requires the substitution of ES to a much larger degree than the cattle system. Currently, the technological options available for accomplishing these substitutions require large inputs of materials and energy. This includes the pumps, filters, tanks and the electricity used to power them; the production of which results in the larger increases in the emissions of greenhouse gasses (GHG) that are observed in salmon aquaculture compared to cattle farming. In addition to demonstrating how different substitutions for ES result in differences in resource use and environmental impact, this example also shows how important it is to consider the environmental context within which BPS are situated (i.e. the ecosystem).

Differences in which ES are delivered by ecosystems can result in dramatic differences in the resource use intensity of the BPS. Taken to the extreme, we could envision the difficulty and large resource requirements associated with production of cattle on the moon. The technical difficulty of this and associated transport notwithstanding, the resource requirements for production in the total absence of a functioning ecosystem (and therefore any ES) would be immense.

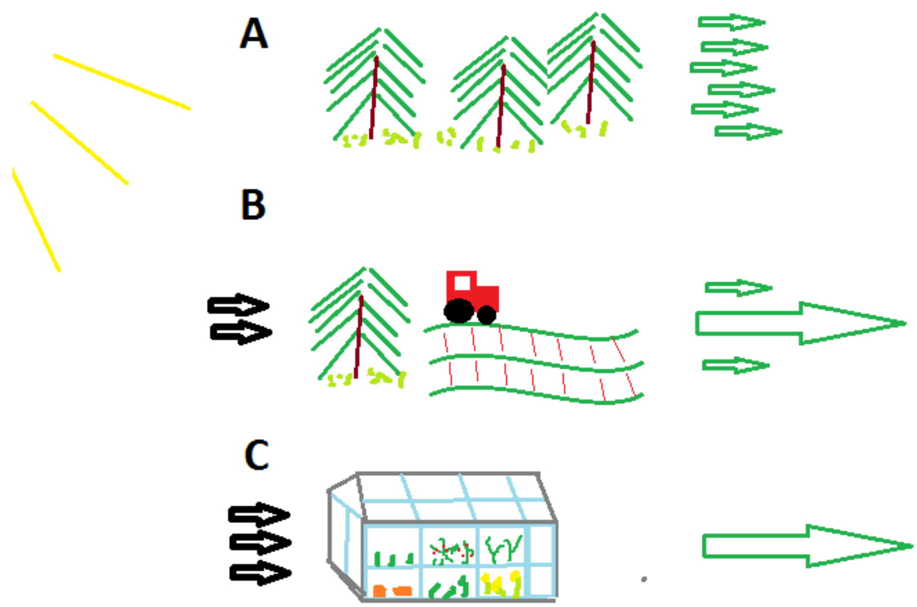

Figure 5. Illustration of the relationship between intensification, external inputs, and the provision of ecosystem services (ES). External inputs are represented by black arrows and ecosystem services are represented by green arrows. The more dramatic the alteration of an agro-ecosystem the larger the external inputs required to substitute for lost ES become. At the same time, the ES generated by these agro-ecosystems become more simplified. Here we present three BPS that vary according to the amount of human alteration. A) A natural unmodified (i.e. wild) ecosystem that provides diverse benefits. B) A managed agro-ecosystem supported by ES and external inputs. This BPS provides multiple ES but its primary function is the production of food. C) A BPS with a high degree of ES substitution. Here the only ES generated is production of food.

As shown in figure 4 above, the productive capacity of BPS and use of external inputs is related to ecosystem services and dis-services that it receives. Due to natural variability in the provision of ES geographically and temporally, achieving the same level of production in similar BPS (e.g. different cattle pasture systems) will require varying amounts of external inputs in different locations or through time. Similarly, different types of BPS (e.g. maize field vs apple orchard) will utilize ES differently, even if they are part of the same ecosystem. This suggests that achieving the same level of production with different types of BPS will require different inputs in order to manage ES. In some instances, systems may be designed or sited poorly so that they under-utilize ES and over-utilize external inputs. These hold true for intensification as well; as BPS are changed, their relationships with ES change too. When analyzing these changes, if the relationship between BPS and ES is not studied then there is the risk that decisions will result in inefficient use of external inputs and unnecessary environmental impacts. Regardless of the specific circumstances, it is clear that examining the relationship between ES and BPS provides a novel lens through which resource use and environmental impact can be examined.

2.3 Framework for the Evaluation of Ecosystem Service Substitution in Agro-Ecosystems (FEESSA)

BPS can be intensified in many ways using external inputs, ES or a combination of the two. Unfortunately, if production is the only parameter that is considered in the development of these alternatives then there is a risk that the development trajectory taken will not optimize ES and result in increased resource use and aggravated environmental impacts (corresponding to the high slope trajectories in Figure 3). FEESSA is a tool for integrating the concept of ES substitution into decision making in the context of intensification of BPS. We suggest that by understanding the consequences of intensification through this lens the likelihood of choosing lower impact alternatives is increased (for example, Porter et al., 2009). FEESSA is used to analyze and score the 1) external inputs (resource use intensity), 2) environmental impacts, and 3) level of ES substitution of intensification alternatives. This makes it possible to identify how ES substitution relates to patterns of resource use and environmental impacts. Using the FEESSA to assess alternative intensification strategies is described as a 7 step process:

1. First, to apply the FEESSA to a particular BPS it is necessary to analyze the agro-ecosystem within which it exists, the ES which support it, and the ES which it provides. This requires specific expertise of the BPS and its ecological relationship to the surrounding environment.
2. Then an inventory of the external inputs supplied to the system must be compiled. This would be completed in partnership with system managers.

3. Once the list of external inputs and ES is completed it is useful to reflect on these two types of inputs and their substitutability. What benefit is provided by the supporting ES? Could this be supplied by an external input? For the external inputs used, what service(s) do they provide? Could this be accomplished by an ES?
4. Next it is necessary to examine the range of possible intensification strategies. What techniques, technologies, or modifications could be used in this specific BPS so to achieve greater production?
5. For each intensification alternative the changing roles of external inputs must be scored and linked to their effect on environmental impacts. What additional external inputs are used? What are the environmental consequences of these changes?
6. With this knowledge, the way which substitutions for ES by external inputs have occurred must be scored for each intensification strategy. How does this intensification strategy make use of (or reduce) external inputs to decrease (or increase) the reliance on ES?
7. Finally, this knowledge can then be used to assess to what extent it is preferable to use each of these inputs (or in what combination) in terms of the environmental impacts assessed. This perspective can be integrated with other knowledge to inform decision-making.

In order to populate the FEESSA classifications for 1) external inputs and 2) environmental impacts we draw on data from Life cycle assessment (LCA). This has several benefits including a standardized methodology (ISO, 2006a, 2006b) that is well suited to accounting for resource and material use of production systems in much the same way as we propose here. LCA practice also has well developed methods for calculating several environmental impacts. Finally, there exists a large body of LCA research already conducted on food production that can be drawn on to facilitate comparisons of intensification strategies (for example: Ayer & Tyedmers, 2009; de Vries & de Boer, 2010; Haas, Wetterich, & Köpke, 2001; Mattsson, Cederberg, & Blix, 2000; Mungkung & Gheewala, 2007; Roy et al., 2009). These attributes make it an ideal tool for exploring the relationship between ES, external inputs, and environmental impact. The following sections (2.3.1-2.3.3) provide guidance on how 1) external inputs, 2) environmental impacts, and 3) ecosystem substitution are scored.

2.3.1 External Inputs - Resource Use Intensity

Understanding the use of external inputs for any intensification strategy is important because many of these inputs are non-renewable resources (e.g. phosphorus and fossil fuels) or their acquisition/production makes use of non-renewables, aggravating concerns of resource depletion (Krautkraemer, 1998). Additionally, the type and amount of resources used are intrinsically linked to intensity of environmental impacts of that strategy. From a life cycle perspective, resource use typically includes extraction, processing, production, distribution, use, and waste. These processes emit environmentally damaging pollutants and degrade natural and human environments. While not all environmental impacts are directly related to resource use, the increased use of resources will nonetheless contribute to increased environmental impacts.

To measure the use of external inputs an inventory of all materials types and amounts should be compiled. These can then be aggregated into resource use categories that provide meaningful information. The actual resource use categories chosen will depend on the purpose of study. Here, four categories of resource use are included: energy use, metal use, chemical use, and biotic resource use.

2.3.2 Environmental Impacts

Ultimately, we are attempting to limit environmental impacts associated with alternative strategies for increased production. For this reason, these environmental impacts must be considered relative to that production. To this end, environmental impacts for each intensification strategy should be quantified. Similar to resource use, the impact indicators that are actually assessed will depend on the socio-environmental context within which the BPS is situated. For our purposes we have chosen to use those categories of environmental impact described as planetary boundaries by Rockström et al. (2009). Following LCA methodology these impacts are measured as the potential contribution of the intensification strategy to each of these planetary boundaries using a relevant indicator (ISO, 2006a).

2.3.3 Ecosystem Services Substitution

As discussed above, the alteration of ecosystem services can occur in several different ways. ES delivery can be enhanced via system design to provide yield increases. Alternatively, they may be augmented or substituted by using external inputs. Here we score the level of substitution for specific ecosystem services from 0-100% based on the amount of the benefit that is derived from ES. In instances where benefits are derived totally from ecosystems they receive a score of 0%. In instances where ES are augmented by using external inputs (e.g. creating artificial habitat for pollinators) then those additional ES are considered as substitutes for the natural provision of services. The percentage of ES substitution given to these types of interventions is then scored out of the new total. For example, if honeybees (an external input) are imported to an agro-ecosystem and increase the total pollination of crops by 200%, then the new total pollination load is 300% of the natural level provided by ES, 66.66% of which is delivered by ES. Alternatively, if habitat management strategies are implemented that improve the populations of natural pollinators (e.g. wild bees) then the substitution would be 0% because the entire service is provided by the ecosystem. Here it is assumed that knowledge of the baseline provision of ES is available (e.g. natural amount of pollination).

2.4 Application of FEESSA to Intensification Alternatives in an Aquaculture Setting

To illustrate the potential applicability of the FEESSA and the concept of ES substitution we draw on the example of salmon aquaculture. As this example is primarily for conceptual demonstration, it is assumed that the background knowledge and inventory of ES/external inputs are available. Moreover, the scores are based on preliminary investigation and are not intended to replace any actual robust assessments. In practice, the type and number of ES being investigated will depend on the characteristics of the system being studied. Here, in place of the knowledge gathering steps 1-3 of the FEESSA process, we provide a brief description of salmon aquaculture in net-pens (the most common method for culture of Atlantic salmon) (Box 3.1). In place of step 4, we describe four hypothetical situations which represent possible alternatives for intensifying production through the control of biofouling organisms (table 1). These alternatives are then scored (steps 5 and 6) using the FEESSA tool described above. To score external inputs and environmental impacts, rather than reporting quantitatively on each (as would be required for a comprehensive examination), we report only whether the intensification alternative is expected to meaningfully contribute to the categories being investigated when compared with the baseline aquaculture system. Here ‘meaningful contribution’ is determined subjectively. Finally, for substitution of ES, we report on only one: the control of the ecosystem disservice biofouling. This is reported as the percentage of biofouling control that is attributable to an ES. A brief rational for these scores is provided (Table 2) followed by the scores themselves (table 3). Here, no final decision regarding the optimal intensification strategy (step 7) is done because this is only a demonstration of the concept and more information is necessary. However, a discussion of potential decisions is provided in Section 2.5.1.

Box 3.1. Salmon aquaculture

Typically salmon are raised in hatcheries. Eggs and milt are collected from broodstock and mixed. The fertilized eggs are then kept until they hatch into fry. Once the fry absorb all the nutrients from the yolk sac they are provided external feeds. After approximately 12 months, once the salmon reach a sufficient size and undergo smoltification (physiological changes that allow survival in salt-water) they are moved to net-pens in the near-shore marine environment. From there they are kept and fed until they reach market size, which takes approximately 12 more months. In net-pens, salmon growth and survival is dependent on both ES and external inputs. Sites are chosen so that water is the proper temperature range, provides sufficient oxygen, and current flow is fast enough that excreted wastes are removed from the growing environment. External inputs required include the infrastructure to hold salmon in the pen, prevent predators from accessing the fish, and allow workers to access the site. Additional inputs include the feed provided and chemicals. The former supply nutrients for metabolic and growth requirements, the latter are used to control pests and disease. On-site energy use includes the fuel used by boats for accessing the site and electricity used by associated facilities for day-to-day operation and processing.

In this generic aquaculture example there are many ways to intensify production but here we only consider one avenue, the control of biofouling organisms, and four alternatives for achieving this goal. Biofouling organisms include mussels, tunicates, macro-algae and many other species that colonize the netting of the net-pen (de Nys & Guenther, 2009). These increase the surface area of the nets, decreasing the openings that allow the free flow of water which limits exchange of oxygen and wastes. These organisms can reach such large proportions and decrease water flow to the point that that they compromise the quality of the growing conditions and cause health issues and even death. If these biofoulers can be removed it results in increased health and growth of the salmon, leading to higher yield. Therefore any method which reduces the incidence of biofouling is a form of intensification.

Table 1. Description of activity, resource use and environmental impact, of four hypothetical intensification alternatives.

Intensification alternative	**Description of activity**	**Resources use***	**Environmental impact***
1) Anti-fouling paints	Copper based anti-fouling paints create a toxic surface on which bio-fouling activity is reduced. These are applied as a coating over the surface of the nets. Paints are typically re-applied for each production cycle.	-Use of anti-fouling paint.	-Release of toxic chemicals to the environment. -Energy and emissions associated with production of paint.
2) Net-pen siting	Biofoulers are typically filter feeding or photosynthetic organisms that require nutrient rich waters to grow. In nutrient poor (oligotrophic) waters the growth rate of these organisms can be dramatically reduced. Unfortunately these areas are not always easily accessible nor do they possess characteristics (e.g. temperature) that make them suitable to the growth of salmon.	-Additional fuel use to access less desirable location. -Potential for reduced efficiency of food conversion due to sub-optimal rearing conditions.	-Potential to harm sensitive species in oligotrophic environments. -Alternatively, the net-pen may provide nutrients that promote growth.
3) Mechanical cleaning	When biofoulers are exposed to sun and wind they become desiccated and perish. By rotating the net-pen such that each area is exposed sequentially, the total load of biofouling organisms are controlled. Once organisms are dead they are removed by manual scrubbing with a high pressure hose.	-Additional infrastructure to allow access to biofoulers. -manual labour, equipment and energy to spray and remove biofoulers.	-Materials and energy use for additional infrastructure. -Emissions associated with additional energy consumption.
4) Biological control	Biofouling organisms that colonize nets are separated from the benthic environment making it difficult for natural predators to feed on them. By introducing predators into net pens, they are theoretically able to control the density of biofoulers.	-Energy and labour required to collect and deploy biocontroling organisms. -Potential for reduced efficiency of food conversion due to negative interactions between salmon and predators (e.g. competition for feed, disease).	-Energy use and emissions associated with additional energy consumption

*Resource use and environmental impact for all alternatives is considered to be identical with the exception of the changes described.

Table 2. Rational for the scores of the framework for the evaluation of ecosystem service substitution in agro-ecosystems (FEESSA) of four intensification alternatives

Intensification alternative	**External inputs**	**Environmental impact**	**ES substitution**
Anti-fouling paint	Energy and chemical use associated with life cycle of antifouling paint.	GHG and acidification associated with energy use. Chemical pollution and biodiversity loss associated with toxic paint	Service provided by paint (100%)
Net pen siting	Energy associated with accessing more remote site.	Loss/gain of biodiversity depending on the receiving environment. May increase biodiversity by providing nutrients and increasing productivity in biologically poor area. May decrease biodiversity by displacing local species adapted to low nutrient environment.	Service provided by environment (100%)
Mechanical cleaning	Increased energy use to operate pressure hose. Increased materials to construct a more complex net-pen	GHG and acidification associated with energy use.	Service provided by wind/sun (50%) and pressure washer/labour (50%)
Biological control	Increased energy to collect or breed predators. Increased labour to manage more organisms.	GHG and acidification associated with energy use.	Service provided by predators (75%) collection of predators by fishing (25%)

Table 3. Results of preliminary investigation of four alternative intensification strategies for using the framework for the evaluation of ecosystem services in agro-ecosystems (FEESSA)

Intensification alternative	**External inputs**	**Environmental impact***	**Ecosystem service substitution**
1) Anti-fouling paints	Energy use (MJ eq) (minor) Chemical use (kg)	Climate change Ocean acidification Biodiversity loss Chemical pollution	100%
2) Net-pen siting	Energy use (MJ eq)	Climate change Ocean acidification Biodiversity loss/gain (depending on the impact on wild species)	0%
3) System design	Energy use (MJ eq) Metal use (kg Fe eq)	Climate change Ocean acidification	50%
4) Biological control	Energy use (MJ)	Climate change Ocean acidification	25%

*Categories of environmental impact are only listed if the intensification pathway is expected to meaningfully contribute to them.

An important consideration for this analysis is that these intensification alternatives are not mutually exclusive, however here we make the assumption that the salmon farmer is resource limited and searching to implement only one solution at the current time. The purpose of this is to demonstrate the relationship between resource use and environmental impact with the substitution of a specific ES. This is done to simplify the analysis and allow for a straightforward examination. Another assumption is that the relative success of these alternatives for reducing the occurrence of biofouling organisms and thus the potential for increased yield is equal. In reality this is likely highly variable. In practice these assumptions are not necessary as real data would be used, however as this example is purely hypothetical they are useful for illustrating the FEESSA concept.

2.5 Discussion

The development of FEESSA began with the realization that not all intensification pathways are equal. In analyzing these pathways it was clear that our ability to explain differences in resource use and environmental impact of different BPS was severely limited. By drawing on technical knowledge and understanding (e.g. by employing LCA methodology) it is possible to explain how these aspects change and what the underlying contributors are (e.g. by analyzing production processes and comparing material and energy use of alternative intensification pathways). However, these do not provide sufficient explanation for the question of why these differences existed. Why should different methods of production require different inputs? Why do different methods for intensification result in different resource use and environmental impacts? It was by incorporating knowledge of 'ecosystem services' that it became possible to explain resource use and environmental impacts, not just from the technical perspective, but as the consequences of changes in the environment that altered the provision of ES. From this vantage point we can perceive how flows of energy embodied in ecosystems underpin BPS and are redirected as intensification occurs.

Conceptualizing BPS in this way and analyzing them using FEESSA provides valuable insight regarding the use of external inputs and occurrence of environmental impacts that is beneficial for several reasons. Firstly, the perspective afforded by FEESSA allows BPS to be examined in a novel way, creating the potential for innovation. Secondly, it allows differences in environmental impact between BPS to be explained in terms of their relationship to ES. This can be used to help identify or even predict those systems and intensification strategies which perform best (i.e. least environmental impact). Lastly, by understanding the role of ES in supporting BPS in a more explicit manner it may be possible to identify optimal management strategies that promote the generation of ES or site the development of BPS based on their ability to integrate into specific agro-ecosystems so that they receive appropriate ES which support production (for example, siting aquaculture, see Borja et al., 2009).

In order to investigate the relationship between BPS, ES and environmental impacts it was necessary to develop a framework within which alternative production systems and intensification strategies could be analyzed. The FEESSA is the result of these. This framework draws heavily on the concept of 'ecological intensification' a form of sustainable intensification which suggests that by establishing and optimizing ES of agro-ecosystems, yield can be improved, while external inputs and environmental impacts are minimized (Bommarco et al., 2012; Cassman, 1999; Doré et al., 2011). This is in contrast to conventional understandings of intensification, whereby yield is continually increased by using external inputs in conjunction with natural productivity (Daly & Farley, 2010).

FEESSA provides a meaningful method through which various forms of intensification can be compared, including ecological intensification. On its face, the FEESSA suggests that the best way to achieve intensification is through the optimization of ES, however it does not necessarily affirm that ecological intensification is the optimal way to achieve intensification. In fact, FEESSA does not require that ecological intensification is even possible. If technological solutions that require external inputs rather than ES provide the least impactful intensification trajectory then these should not be excluded on the basis that they are not ecological intensification. Likewise, redirecting a large portion of the energy that flows through an ecosystem (in the form of ES) solely for the purpose of agricultural production could potentially lead to unintended negative consequences to processes or organisms that previously relied on them. Simply because a service is derived from the ecosystem does not mean that there are no environmental impacts associated with it. For example, implementing habitat management strategies that improve suitability for crop pollinators may increase the pollination of a target BPS but the habitat may be altered at the expense of other organisms which don't provide any direct benefit to production. Alternatively, consumption of water from a river for agriculture is beneficial for the BPS using the water. However, over-consumption of the water can alter downstream ecosystems, causing organisms reliant on that water to suffer. For these reasons, care must be taken when examining BPS not to attribute too much or too little importance to the role of external inputs or ES without careful consideration. Using the FEESSA provides a balanced and objective means for assessing intensification alternatives.

2.5.1 Aquaculture Example

To demonstrate the applicability of the FEESSA, we developed a classification system and applied it to a hypothetical example. Four alternative methods for intensifying salmon production (by managing the ecosystem dis-service of biofouling) were compared. Based on this exercise it appears that the alternative 2) is preferable because it requires little to no additional resource use and may not result in significant environmental impacts. Alternatives 3) and 4) are both similar in that they increase energy use while 3) also requires increased construction materials. Which of these two performs best in terms of environmental impact is highly context dependent. Biodiversity impacts will depend on the local context and the species present. Species in oligotrophic environments may be negatively impacted by 2) if they are sensitive to increased nutrients or this method may induce increased productivity by supplying nutrients. Alternatively, if species used for biological control in 3) are non-native then they have the potential to become invasive. The worst performing system appears to be alternative 1). This strategy relies exclusively on external inputs in the form of energy use and chemical use, while contributing to four environmental impact categories.

As this exercise is purely to demonstrate the potential applicability of the FEESSA concept, it was extremely limited in scope. Compilation and analysis of resource use and environmental impacts were limited to a qualitative assessment by the author rather than a more in depth assessment of quantitative contributions; the change in only one ES was classified rather than a more comprehensive examination of all ES being altered and; alteration of the ES was assessed based on estimations rather than any actual measurements. Fortunately, it is not necessary to conclusively assess the relative performance of alternatives; instead we can use the classification system to explore the link between ES substitution and resource use/environmental impact in a more generalized fashion. For each alternative ES substitution occurs to a different extent. Alternative 2) does not make any substitution (0% substitution) and only requires additional fuel for accessing the site location while 1) completely replaces ES (100% substitution) with the use of anti-fouling paints. Alternative 4) utilizes predators to minimize biofouling, however, the capture and management of those predators requires energy and labour, thus the ES are not provided completely by ecosystems (25% substitution). Finally, alternative 3) uses an even mix of ES and external inputs in the form of wind/sun & pressure washing respectively (50% substitution). In each of these cases the score for substitution is difficult to assess and highly subjective but based on the assumptions made here, there are indications that the larger the substitution the more intensive resource use and environmental impact become.

2.5.2 Problems With Classification Scheme

In order to score the four intensification alternatives investigated here, we have drawn inspiration from LCA research because it is an existing technique with robust methodology and principles that follow international standards (ISO, 2006a, 2006b). It allows the resource use and environmental impact to be scored in a standard way and expressed relative to a functional unit. From there we take another step and link these to ES substitution. Unfortunately, LCA is not without issues (Reap, Roman, Duncan, & Bras, 2008a; Reap et al., 2008b). For example, scoring environmental impacts that are not related to emissions or resource use (e.g. impacts on biodiversity) using LCA methodology is difficult and in some instances impossible.

Issues with LCA aside, we recognize that there are also flaws with the FEESSA classification scheme. Choice of indicators for both resource use and environmental impact are not comprehensive or prescriptive. There is a range of resource use and environmental concerns that may be related to ES substitution and those chosen for any particular situation are highly dependent on the context. As an example of potential impact indicators we have chosen to adopt those planetary boundaries described by Rockström et al. (2009). While not exhaustive, these have been peer-reviewed and cover a broad range of impacts and are thus well suited for demonstrating the applicability of the FEESSA concept.

Another issue related to the FEESSA classification scheme is that the relationship between resource use, environmental impact and ES substitution is not likely as straightforward as we have described here. In some instances the same method of intensification can lead to different trajectories for different environmental impacts. For example, more intensive cattle systems may result in lower GHG emissions (Nijdam et al., 2012) while at the same time increasing the nitrogen cost, contributing to eutrophication (Bleken, Steinshamn, & Hansen, 2005). The relationship between these seemingly opposed trajectories of environmental impact and whether ES substitution provides any insight is unknown. What this example shows is that the complexity and variability between systems makes conclusions difficult to make, and reinforces the importance of context specific knowledge required for evaluating BPS (Bommarco et al., 2012).

2.5.3 Problems With Ecosystem Service Substitution

Perhaps the most challenging problem related to the FEESSA is determining the categories of ES actually delivered to BPS. This is because it is often unclear how ecosystem processes relate to human benefits and the score given to the delivery of ES will change depending on the interpretation of the service. In many instances ecosystem processes related to the provision of benefits are interrelated and there is confusion surrounding the point at which benefits (i.e. ES) are actually realized (Wallace, 2007). The result is that two analysts may come to different conclusions regarding the type, amount and role of ES present in a BPS. For example, consider the service provided by pest predators in a BPS. It is possible to describe this service as the reduced damage to crops and score the percentage of reduced damage which is attributed to predators. However, this same service can be described as an increase to production, a decrease to pest populations, and likely others. It is possible to attribute a different percentage to each one of these services based on the same predator-prey interaction. Another problem occurs when an ES and the regulating or supporting services that underpin it are both counted. Here we see double counting of what should be a single service. To avoid these issues we adopt the classification scheme for ES proposed by Wallace (2007) with one notable exception; in this case the benefits are provided to the BPS rather than to humans, therefore ES occur at the moment which benefits to BPS are realized.

Once categories are determined it is then necessary to quantify the level of substitution. This requires an intimate knowledge of ecosystem function and its relationship to agricultural production. When this knowledge is unavailable we suggest that the most practical method by which this can be accomplished is based on the change in agricultural production, whereby the percentage substituted is the achieved production value divided by the expected production in the absence of the intervention. Take for example the case of fertilizer. The application of fertilizers can be considered substitutes for the natural regeneration and provision of soil nutrients through time. The percentage of ES substitution attributed to fertilizer application can be measured by the difference in production achieved in its presence and absence. If yield of a wheat for a given plot is expected to be 6 t/ha without fertilizer and the actual yield with fertilizer is 8 t/ha then we can say that the fertilizer is responsible for 2 out of 8 tonnes of growth for a substitution rate of 25% ([8-6]/8*100). This method for scoring ES substitution is not likely to work for all situations. Due to the huge variation in production methods, there are likely to be numerous problems with substitution. For this reason it is not practical to list and describe all possible problems and potential solutions. Instead, it is important to be cognizant of these issues moving forward so that a rational and consistent approach is used when they are encountered.

2.6 Conclusion

BPS are a major contributor to environmental impacts on a global scale (Fedoroff et al., 2010; Foley et al., 2011; Tilman et al., 2011; Tilman, 1999). One of the greatest challenges of the coming decades will be limiting these impacts, while continuing to support the economic metabolism of a growing and more affluent human population. ES have been highlighted as a useful framework through which synergies and trade-offs of various human systems could be evaluated to promote better management (Daily et al., 2009; Kareiva, Watts, McDonald, & Boucher, 2007; MEA, 2005; Rodríguez et al., 2006). Here we explore the relationship between provision of ES with the use of external inputs and environmental impacts in BPS. We propose that this relationship can be used to explain observed differences between BPS. This conceptualization of agro-ecosystems provides a novel perspective through which alternative modes of intensification can be viewed, allowing for optimized decision-making and increasing the potential for development of innovative strategies.

Currently, this framework is intended to compare alternatives regarding specific choices for intensification of a particular agro-ecosystem related to the substitution of individual ES, however, as knowledge of these interventions is improved this framework could be used to compare systems that differ to a greater degree. This would first extend to comparing alternative production methods for the same species then to crop or livestock systems of different species. It is important to note this system does not represent the only method for exploring the relationship between these variables and can be modified to reflect the desired application. We also suggests that in addition to differentiating among intensification strategies, this knowledge can be used to understand fundamental aspects of all BPS whether being intensified or not. This could potentially be used to characterize and delineate systems along a spectrum of intensity based on the level of ES substitution.

Recognizing the role that ES play in supporting BPS is an important first step. However, the ability to manage ES in BPS is limited by our knowledge in several important ways. Firstly, there is a need for more research on ecosystem function and the delivery of ES to BPS, exploring under what circumstances these ES are altered. Secondly, there is a need to explore how the substitution of various ES is related to resource use and environmental impact. Is the substitution of all ES equal? Can they all be substituted? Modified? To what extent does the role of technology play in our ability to substitute ES? Thirdly, the role of physical and environmental characteristics on the provision of ES requires investigation. How is the provision (and substitution) of ES related to climate, geography, and biodiversity? At the moment, there are many more questions than answers but by applying the FEESSA concept we gain a valuable perspective on these issues, opening the door for improved understanding of BPS and innovation.

Chapter 3. Life Cycle Assessment of a Novel Closed-Containment Salmon Aquaculture Technology

3.1 Introduction

Aquaculture production worldwide has experienced unprecedented growth over the past several decades and now accounts for over 40% of the seafood consumed globally (FAO Fisheries and Aquaculture Department, 2013). While the pace of this growth has slowed recently, it is predicted that global production will need to increase by at least 23 million tonnes by 2020 in order to meet growing demand in the face of generally stagnant global fisheries landings (FAO, 2012). With the emergence of aquaculture as an ever more important industry, attention has increasingly focussed on understanding and mitigating its resource and environmental impacts (De Silva, 2012; Henriksson et al., 2011). Though representing a small fraction of global aquaculture production, the culture of salmonids in particular has received considerable scrutiny (Ford & Myers, 2008; Hansen & Windsor, 2006).

Environmental concerns associated with salmonid culture vary with the production setting and culture technologies employed but may include: nutrient loading and alteration of local receiving environments through the release of excess feed, fish wastes and other chemicals (Folke, Kautsky, & Troell, 1994; Olsen, Holmer, & Olsen, 2008); the amplification and transmission of pathogens and parasites to wild populations (Krkosek et al., 2006); and reduced fitness of nearby wild conspecifics due to interactions and interbreeding with escapees (Leggatt, O'Reilly, Blanchfield, Mckindsey, & Devlin, 2010; McGinnity et al., 2003). In addition to these potential localized impacts, the scale and density at which carnivorous species such as salmon are reared requires that 100% of growth is predicated on compound diets built from nutrient-dense feedstuffs derived from a wide range of plant and animal sources (Aubin & van der Werf, 2009; Pelletier & Tyedmers, 2007; Pelletier et al., 2009). The provision of feeds along with the range of other operational and capital inputs to salmon culture are material and energy resource intensive practices that inevitably contribute, along with all other animal food systems, to a range of global-scale environmental challenges (e.g. greenhouse gas emissions, eutrophication, etc) (Boissy et al., 2011; Papatryphon et al., 2004; Pelletier et al., 2009; Tyedmers, 2000).

In order to reduce and/or mitigate the environmental impacts of salmon culture considerable effort and research has been invested in many areas, including: salmon genetics (Fjalestad, Moen, & Gomez-Raya, 2003), behaviour (Gjedrem & Baranski, 2010), feed formulations (Boissy et al., 2011; Hua & Bureau, n.d.), and farming practices (Belle & Nash, 2008). One avenue of research that has garnered substantial attention is the potential to move away from the industry dominant net-pen culture system towards some form of 'closed containment' (Chadwick et al., 2010; Partridge, Sarre, Ginbey, Kay, & Jenkins, 2006) or land-based recirculating aquaculture systems (RAS) (Martins et al., 2010; Tal et al., 2009). These alternative culture technologies (whether sited in marine, freshwater, or brackish environments) can be characterized by: 1) the type and degree of water exchange between the culture and non-culture environment; and 2) whether they are land-based or aquatic. They exist on a spectrum from 'flow-through', with no significant barriers to the outgoing flow of water to 'closed' recirculating systems without significant outflow. In land-based flow-through systems (e.g. raceways and ponds) influent water from low-order headwaters or nearby surface water is brought into an enclosed culture environment (Fornshell & Hinshaw, 2008). This water passes through the system and is emitted downstream to receiving environments. Flow-through systems in aquatic environments such as solid-wall and bag technologies are similar except the system itself is situated in the aquatic medium and effluent is released into the same environment from which influent is sourced. On the opposite extreme are RAS, which can have recirculation rates over 99%, where water loss is predominantly due to evaporation (e.g. Gelfand et al., 2003). In these systems, water is pumped into land-based tanks and cycled continuously with the addition of make-up water as necessary.

Alternative aquaculture technologies are favoured by many environmental advocacy groups as they theoretically allow managers to reduce, re-direct or eliminate emissions of nutrients into receiving environments and negative interactions with wild species (e.g. competition, interbreeding and exchange of diseases) (Tal et al., 2009). Moreover, they allow managers a measure of control over several environmental parameters such as temperature, water quality, oxygen content and flow rate, all of which can be optimized to promote fish growth (Wolters, Masters, Vinci, & Summerfelt, 2009). Unfortunately, some of the attributes which improve the environmental performance of closed-containment and RAS may exacerbate other environmental issues, resulting in trade-offs. By isolating cultured animals from supporting ecosystem services (e.g. oxygenated water, waste removal/assimilation and disposal, temperature regulation), these services must now be supplied by managers, which require energy and/or materials (Ayer & Tyedmers, 2009). The result is that these more intensive culture technologies may improve local-scale environmental impacts (e.g. by reducing emissions of eutrophying chemicals) (Aubin, Papatryphon, Van der Werf, Petit, & Morvan, 2006) while exacerbating global-scale environmental impacts (e.g. climate change, terrestrial and ocean acidification) (Ayer & Tyedmers, 2009; Pelletier & Tyedmers, 2010b). Moreover, these alternative technologies can lead to changes in the culture environment that decrease overall efficiency of production (e.g. higher mortalities, lower feed conversion), representing environmental costs that are not necessarily apparent without in-depth assessment. Given the growing importance of aquaculture and the environmental challenges it poses, it is essential that aquaculture develop in ways that intensify production, while simultaneously lowering resource intensity and environmental impacts at all scales. It is thus necessary to comprehensively assess the environmental impacts of alternative aquaculture technologies to provide data on environmental performance necessary for informed decision making.

One method that has been used extensively over the past decade to explore the environmental consequences of finfish aquaculture is life cycle assessment (LCA) (Henriksson et al., 2011; Mungkung & Gheewala, 2007). LCA has been used to research several species and culture technologies (Aubin, Papatryphon, van der Werf, & Chatzifotis, 2009; Aubin et al., 2006; Cao, Diana, Keoleian, & Lai, 2011; D'Orbcastel, Blancheton, & Aubin, 2009; Grönroos, Seppälä, Silvenius, & Mäkinen, 2006; Jerbi et al., 2012; Wilfart et al., 2013) with several studies focusing specifically on salmon (Ayer & Tyedmers, 2009; Ellingsen & Aanondsen, 2006; Pelletier et al., 2009). LCA is a biophysical assessment technique which is used to compile and quantify energy and material inputs/outputs used throughout the production cycle (Heijungs et al., 2002). It is used to assess the resource use and emissions related impacts of production. It can be used to evaluate products or processes from 'cradle-to-grave', illuminating environmental hot spots and potential trade-offs. Moreover, it can be used to assess contributions to a large array of environmental impacts such as global warming, eutrophication, acidification, ozone degradation (Baumann & Tillman, 2004; Heijungs et al., 2002). One of the primary insights gained through application of LCA to aquaculture has been to highlight the importance of feed use as a driver of environmental impacts (Ellingsen & Aanondsen, 2006; Jerbi et al., 2012; Pelletier et al., 2009). This is largely attributable to the impacts associated with the types of ingredients used, fisheries products in particular, and has stimulated interest in assessing alternatives feed formulations, with a focus on finding plant-based substitutes (Boissy et al., 2011; Papatryphon et al., 2004; Pelletier & Tyedmers, 2007). Another insight of this research has been the recognition of the large impact associated with energy carriers (e.g. electricity and fuel) particularly for 'closed' systems such as RAS (Aubin et al., 2009; Ayer & Tyedmers, 2009; D'Orbcastel et al., 2009).

Despite the breadth of LCA research undertaken on production of salmonids grown in various culture environments, it has never been conducted to understand the performance of a floating, solid-walled, containment system. Here we have assessed one such technology that was deployed in British Columbia (BC), Canada using LCA. The technology is a flow-through, solid-walled technology that is referred to here as a solid-wall aquaculture system (SWAS) (see section 1.1). It is a culture technology that has attracted interest as it represents an intermediate between net-pens and RAS, potentially providing some of the local environmental benefits of RAS with reduced global-scale impacts. By isolating salmon from the local environment with a solid barrier there is potential for wastes to be captured and the transmission of some pests and diseases to be mitigated. Additionally, by situating the system in the aquatic environment, the need to pump influent significantly above its static level is reduced, presumably reducing energy demand of pumps.

This research was undertaken to quantify the life cycle impacts (LCI) associated with producing 1 tonne of live-weight salmon using the SWAS. This information was used to identify environmental hotspots and explore strategies for improving environmental performance. Scenario analysis was used to investigate how important system characteristics and decisions made by system managers influenced life cycle impacts of production. Unfortunately, as the SWAS is an emerging technology, operational data were only available from a single tank over most of a single production cycle. The operational horizon from which data were available was truncated by a storm event on March 12, 2012 that compromised the structural integrity of the tank after 13.5 months of operation, resulting in the early harvest of salmon. Consequently, the data do not represent the intended production of salmon but were used to model the actual production cycle (APC). However, a model of the intended production cycle (IPC) was constructed in parallel based on APC data but modified as appropriate to reflect more typical or expected salmonid growth to harvest parameters. Due to the uncertainty associated with using a limited data set and predicting performance, significant effort to quantify uncertainty in the major drivers of life cycle impacts was undertaken. This was achieved by using sensitivity analysis to investigate how performance of key parameters influenced life cycle impacts. Additionally, Monte Carlo simulation was used to characterize the potential distribution of life cycle impacts based on estimated uncertainty of several parameters. This method has been used previously to characterize uncertainty in aquaculture systems (Cao et al., 2011).

3.1.1 Description of Solid-Wall Aquaculture System

The SWAS is a cylindrical tank (24m diameter, 6m depth) composed primarily of steel and thermoplastics, placed directly into the marine environment and anchored to the seafloor by posts and steel chains. The tank was fabricated in sections in Quingdao, China, transported by ship to B.C. then assembled on-site. The culture site was located on the Strait of Georgia in a sheltered environment approximately 150m from shore.

Influent water was pumped from depth, using two 15hp axial flow pumps, from adjacent waters that manifest appropriate water quality (e.g. oxygen levels, temperature, salinity etc). Under operating conditions the maximum head within the tank was 1m above static. Influent water was supplemented with oxygen from a large oxygen concentrator to ensure 100% or greater saturation. All onsite electricity demand for pumps and oxygen supplementation was provided via a connection to the British Columbia electricity grid while use of waste capture technology was powered by on-site diesel generators, albeit intermittently due to technical issues. Feed and smolts were acquired from local producers and transported to the grow-out facility via transport truck and service boat. Approximately 56,000 Chinook salmon smolts were stocked in January, 2011, weighing close to 35g each. Fish were fed on a high-protein feed of fish meal, fish oil, and wheat in proportions of 60:20:20 respectively. Fish were grown in the SWAS for 13.5 months to an average size of 1.73kg when they were harvested. Maximum density in the tank was approximately 26.6kg/m^3.

3.2 Methodology

3.2.1. Life Cycle Assessment

LCA was used to assess the contribution to several global-scale environmental impacts associated with production of 1 tonne of live-weight salmon in the SWAS. LCA is typically described as a four step process that includes: 1) definition of goal and scope, 2) life cycle inventory, 3) life cycle impact assessment, and 4) interpretation (ISO, 2006a).

3.2.2 Definition of Goal and Scope

The goal and scope of this study was to assess the life cycle environmental impacts associated with production of 1 tonne of salmon in the SWAS from cradle to farm-gate. This includes all upstream and background processes of salmon production up until they are produced at the farm site. It does not include processes related to post-harvest processing, packaging, retail, use, or end-of life. Models were constructed based on foreground operational data that encompassed all major subsystems required to produce salmon from cradle to farm-gate (Figure 6). Where necessary, data on background systems (e.g. resource extraction, transportation, energy carriers) was taken from pre-fabricated databases (e.g. EcoInvent version 2.2) and adapted to the current context. Subsystems were delineated based on relevant operational practices, they include: 1) provision of feed, 2) production of juveniles (smolts), 3) production and assemblage of infrastructure, 4) on-site energy use, 5) transportation of materials and 6) on-site emission of nutrients.

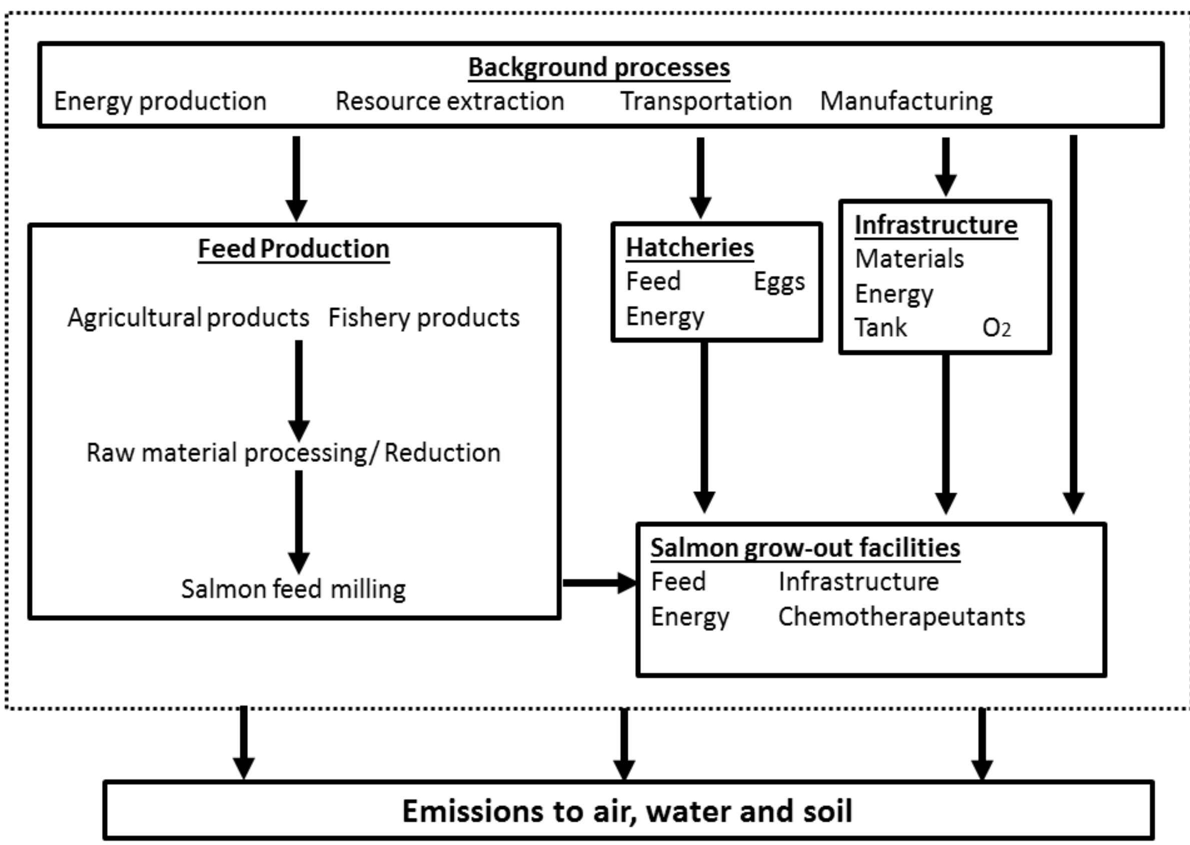

Figure 6. Life cycle flow chart for the life cycle material and energy inputs/outputs associated with the production of salmon in the solid-wall aquaculture system.

3.2.3 Life Cycle Inventory Analysis

To build the model for characterizing the SWAS production cycle, detailed foreground data on inputs and outputs required for total production of all subsystems were acquired. Operational inputs to SWAS operations (e.g. smolts, feed, electricity, fuel use, etc…) for the 13 months of production were elicited from Agrimarine Inc. managers via questionnaires, operational data/reports supplied by Agrimarine Inc., consultation with Agrimarine Inc. staff, and on-site inventory audits. These provided the top-level model which other subsystems fed into for the production of salmon. These were modeled based on total inputs/outputs for the APC. From this and processes associated with all subsystems, a model of the life cycle inventory required for production of 1 tonne of live-weight salmon was calculated by dividing all inputs by total tonnes of salmon produced. Many of the subsystem models were constructed on the basis of a relevant unit and then multiplied by an appropriate factor to give the desired amount. For example, feed models were constructed relative to the production of 1 tonne of feed. To relate these to the APC they were multiplied by total amount of feed used per live-weight tonne of salmon harvested.

Inventory data associated with background processes (e.g. raw mineral extraction, concrete production, steel smelting etc…) were quantified through the use of peer-reviewed datasets such as Ecoinvent database v.2.0 accessible through SimaPro (PRé Consultants, 2010). These were selected based on relevance to the study context. For example inventory data on production of steel used would ideally be chosen from a North American context; unfortunately this was unavailable, therefore data representative of steel production in the European Union were used.

Inputs to construct and maintain infrastructure of the SWAS system were elicited from Agrimarine Inc. personnel (Mr. Rob Walker, personal communication, January, 2013). Data on energy used in the construction of the tank were unavailable due to a break in relations with the organization responsible for its manufacture. These numbers were therefor obtained from Janicki Industries, the organization that was subsequently hired for construction of a second tank with similar specifications (Ms. Sheena Burns, personal communication, May 15, 2013). These data were then attributed to the SWAS production cycle based on its life-expectancy.

Average inputs (e.g. feed & energy use) required for production of one batch of smolts (approximately 50,000 individuals) were obtained from Omega Pacific Hatcheries (Mr. Bruce Kenny, personal communication, January 28, 2013). Data were used to calculate inputs per smolt and then multiplied by the total number of smolts used for the SWAS production cycle.

Total travel required were calculated based on mass of goods transported (e.g. feed, smolts, tank infrastructure etc) and the distances travelled. Modes of transport were determined from communications with Agrimarine Inc. personnel (Mr. Rob Walker, personal communication, January, 2013) and distances were determined using Google Earth. Inventory data for travel for each relevant subsystem were modeled as an independent 'transportation' subsystem. For example, fuel use associated with transportation of feed and smolts were not included in the feed and smolts subsystems but rather in the transportation subsystem. The 'transportation' subsystem therefore refers to the total movement of all materials necessary for the APC.

Life cycle inventory data were collected for two feed types. One feed was specially created for use in the SWAS, and was supplied by Taplow feeds (Dr. Brad Hicks, personal communication, January, 2013). The second is representative of the aggregated average of all feeds produced by EWOS, a large organization that produces salmon feed, in Canada in 2012 (Dr. Jason Mann, personal communication, February 17, 2013). Data for the average feed formulation was collected because the specialty feed is dramatically different from typical feeds that are used in modern salmon aquaculture. Energy/material inputs and outputs related to the production, processing and transportation of all feed ingredients were constructed in a manner consistent with the methods described in Pelletier et al. (2009).

Feed ingredients were derived from agricultural, livestock, and fisheries systems using published literature, reports, and government/ Canadian (or province) specific statistics. For agricultural products this included; an accounting of on-site use of fuel (Pelletier et al., 2009), fertilizer (International Fertilizer Industry Association, IFIA, 2013), pesticides, seed, and electricity (Centre for Energy, CE, 2013); estimations of transport distances; and calculation of field-level emissions of relevant gasses (Intergovernmental Panel on Climate Change, IPCC, 2007). Livestock derived feed ingredients were modeled directly from Pelletier et al. (2009) but updated to reflect more recent production realities (e.g. electricity production mixes).

Inputs/outputs for fisheries based feed ingredients were derived from published reports, personal communications with industry experts and country statistics. This included an estimation of fuel use per tonne of catch and materials used in fishing boats. An estimation of material and energy used in the construction and operation of fishing vessels was adopted from Tyedmers (2000). Fish used in the Taplow feed were derived primarily from Pacific herring (*Clupea pallasii*) and Pacific hake (*Merluccius productus*) fished in waters off the Pacific north-west of North America (Dr. Brad Hicks, personal communication, January, 2013). Fuel use from Canadian hake and herring fisheries was elicited by gear type from Goldseal, a commercial fishing, processing, and distributing organization (Rob Morley, personal communication, January 21, 2013). US hake catch fuel use data were obtained from Aleutian Spray Fisheries, a commercial fishing and processing organization (Mr. Craig Cross, personal communication, March 1, 2013). To account for differences in fuel use between Canadian and American based hake fisheries, a 5-year weighted average from 2005-2010 was constructed using catch data for American and Canadian fisheries (DFO, 2012; Stewart et al., 2011). Fish used in the production of the average Canadian 2012 feed were from herring (BC, Canada), menhaden (United States), and anchovetta (Peru) fisheries. Data on inputs to the herring fishery were the same as that used in the Taplow feed. Data for menhaden and anchovetta fisheries were updated from Pelletier et al. (2009).

Yield of fisheries by-products used for reduction into fish oil/meal was elicited from Delta Pacific (Mr. Don Pollard, personal communication, May 14, 2013) and government fisheries scientists (Dr. Brenda Spence, personal communication, April 30, 2013). Data for yield of meal/oil and energy used in reduction were obtained from West Coast Reductions (Mr. Grant Sarr, personal communication, February 4, 2013).

To estimate the amount of waste emitted by cultured salmon during production a mass-balance model was used (Aubin et al., 2006; Cho & Kaushik, 1990). This model is based on the differences in solids, nitrogen (N), and Phosphorus (P) between the feed provided and the weight gain of harvested fish (Islam, 2005). By accounting for body composition (Tacon & Metian, 2013), feed nutritional content, nutrient digestibility (Papatryphon, Petit, Van Der Werf, Sadasivam, & Claver, 2005), and feed wastage (assumed to be 3% -Mr. Walker, personal communication, April 2013), it was possible to calculate solid and dissolved fractions of these emissions using established mass-balance models (Aubin et al., 2006; Aubin, Tocqueville, & Kaushik, 2011; Jerbi et al., 2012).

3.2.4 Co-Product Allocation

Given the multi-output nature of many salmon feed input subsystems, there was a need to address what is widely referred to in LCA practice as the 'co-product allocation' issue. Co-product allocation arises when there are two or more co-products of a given process that are utilized somewhere in the technosphere (Ayer et al., 2007; Finnveden et al., 2009). For example, capture fisheries produce seafood for consumption and by-products that can be directed to reduction into meal and oil. Allocation must be conducted here based on fisheries co-products then again on the reduction co-products. The ISO-guidelines outline various approaches to ideally first avoid, or where this is impossible, address the need to allocate (ISO, 2006b). Avoidance of allocation is done by dividing the process into individual subsystems or by implementing system expansion, whereby system boundaries are expanded to include the life cycle inventories of affected processes (Finnveden et al., 2009).

The appropriateness of alternative bases of allocation (e.g. allocation based on the financial value or alternatively a physical or energetic property of co-product streams) has always been contentious (Finnveden et al., 2009) and has recently been addressed by various authors (Ayer et al., 2007; Eady, Carre, & Grant, 2012; Pelletier & Tyedmers, 2011; Svanes, Vold, & Hanssen, 2011). In this study we follow the rationale offered by Pelletier & Tyedmers, (2011) and allocate inputs and resulting environmental burdens amongst co-products of the various agricultural and fisheries subsystems providing inputs to salmon feeds modeled in proportion to the nutritional energy embodied in each of utilized co-product. The rationale for this is two-fold: firstly, energetic content represents a relevant biophysical measurement that is common to all co-products, and secondly it is relevant to the purpose of food systems such as salmon aquaculture (i.e. the delivery of food energy).

3.2.5 Modelling Hypothetical 'Intended' Production Cycle

As described above, the SWAS was compromised by a storm in March 2012 after 13.5 months of production, allowing some fish to escape. As a result remaining salmon were harvested prior to completion of the intended harvest time and inventory data collected represents one incomplete production cycle. This is important because between the actual harvest and intended harvest many of the material and energy inputs/outputs of the SWAS change in proportion to the functional unit. For example, inventory data for construction of the tank is independent of total production but the relative amounts of these inputs/outputs will change depending on the total production. This means that life cycle impacts associated with construction of the tank will be higher for the actual production cycle than if it had achieved larger production. To explore the implications of this an inventory of the intended production was calculated by construction of a model of the intended production cycle (IPC). The LCA of the IPC is therefore an approximation of the environmental performance of the SWAS if the storm had not occurred and production continued without interruption to intended harvest.

The inventory data for the IPC were extrapolated from available production data based on growth models. This was accomplished by constructing a model based on data from the APC, and where necessary, peer-reviewed literature and industry experts were consulted (Appendix A).

3.2.6 Life Cycle Impact Assessment

A problem oriented (mid-point) approach was used to evaluate the performance of the SWAS salmon production system. This approach is used to characterize contributions to global-scale phenomena of concern rather than the actual damage levels. Five impact categories were chosen: contributions to global warming potential (GWP), acidification potential (AP), marine eutrophication potential (MEP), cumulative energy use (CEU), and biotic resource use (BRU). Three of which are emissions related impacts (GWP, AP, and MEP) and two are resource use related impacts (CEU and BRU). These impact categories represent issues of particular concern for intensive aquaculture systems that have been highlighted by previous works (Aubin et al., 2009; Henriksson et al., 2011; Pelletier et al., 2006). This is doubly so for GWP, AP, and CEU, which have been found to be a large concern in other alternative aquaculture technologies (Ayer & Tyedmers, 2009). The use of MEP in particular is relevant to an assessment of the impacts of the SWAS because one of the motivations for development of this technology is its potential ability to capture and divert waste nutrients so that eutrophication of receiving environments is reduced. While BRU is a somewhat novel impact category, it has been widely adopted in seafood system LCAs as it is highly sensitive to differences in trophic levels of biological inputs to production systems (Henriksson et al., 2011).

Contributions to GWP (kg CO_2eq), AP (kg SO_2eq), and MEP (kg N eq) were quantified using the Recipe Midpoint version 1.07 impact assessment method (Goedkoop et al., 2009); CEU (MJ eq) was quantified using Cumulative Energy Demand version 1.05 (PRé Consultants, 2010); and BRU (kg C) was quantified following previous practice in recent seafood LCAs (Aubin et al., 2009; Jerbi et al., 2012; Pelletier et al., 2009). All of the calculations to conduct the impact assessment were facilitated by the use of SimaPro 7.1 from PRé consultants (PRé Consultants, 2010).

3.2.7 Sensitivity and Uncertainty Analysis

Given the potential importance of uncertainty in LCA modeling in general and the novel nature of the SWAS and limited real-world operational experience from which data could be derived, effort was made to assess the effect of key input parameters on results. Additionally, since the IPC was modelled using known relationships, operations data, and expert opinion, there is the added uncertainty associated with the model itself and how well it approximates reality.

Broadly speaking, there are two types of uncertainty that can contribute to error: ontological and epistemological (Brugnach, Dewulf, Pahl-wostl, & Taillieu, 2008; Parker & Tyedmers, 2012). Ontological uncertainty occurs where there is variability inherent in a measured value through space and time (natural or otherwise), while epistemological uncertainty represents lack of or imperfect knowledge (Walker et al., 2003). In LCA there is typically a distinction between sources and types of uncertainty, where each source may be subject to different types of uncertainty (Finnveden et al., 2009). Examples of several types of uncertainty are described by Loyd & Ries (2007), these include epistemic uncertainty in the form of random error, statistical variation, and expert uncertainty/disagreement, and ontological uncertainty described as variability (temporal, geographical and technological) and normal fluctuations.

Sources of uncertainty that were relevant to the SWAS included system characteristics (e.g. the rate at which food is converted to biomass and mortality rate) and system inputs (e.g. electricity use), each of which were subject to multiple types of uncertainty. Take for example the body composition of herring and hake which were used to produce fishmeal and oil. The body composition of these fish varies considerably by species, time of year, and environmental conditions etc… (Vollenweider, Heintz, Schaufler, & Bradshaw, 2010) therefore the amount of fish required to produce an equivalent amount of fish meal and oil isn't constant. Consequently, the LCI associated with a given amount of meal or oil is variable and attributing a static value is therefore problematic. This ontological uncertainty is compounded by the fact that the measurement process itself is subject to methodological and human error, a source of epistemic uncertainty. A particularly significant source of epistemological uncertainty in LCA occurs when input processes from databases are used to represent input processes in the system of study. These processes will not be perfectly representative of reality because the two processes – the actual and the one used to characterize the system – are likely to differ geographically, temporally and potentially technologically (von Bahr & Steen, 2004).

Uncertainty was addressed in this study using two methods: sensitivity analysis and Monte Carlo simulation. Both of these methods were applied only to the IPC as this was considered most relevant to the future applications of the SWAS, representing a complete production cycle. Sensitivity analysis was used to evaluate the effect of using alternative values for key inputs that exerted important effects on the LCI. Sensitivity to changes to individual input parameters were chosen based on prior knowledge of important drivers of LCI in aquaculture systems and analysis of preliminary results. Sensitivity analyses included: 1) alternative values for eFCR: minimum 1.1 based on 1.103 from Norwegian industry (Pelletier et al., 2009), maximum 1.5 based on 1.448 for freshwater closed containment of salmonids in eastern Canada (Ayer & Tyedmers, 2009) and 1.493 from Chile (Pelletier et al., 2009); 2) alternative yields of fish meal and oil from reduction, high yield (6% oil, 23% meal) and low yield (4% oil and 17% meal) 3) 10 year life-expectancy for tank infrastructure, and 4) reduced values for trophic level of fish species used in feed production (hake 3.6, herring 2.8). The values used in sensitivity analyses 1) and 2) were chosen based on consultation with experts and published literature to determine realistic distributions (e.g. maximum and minimum values). Reduced values for trophic levels in sensitivity analysis 4) were chosen based on minimum values within estimated error from FishBase (Froese & Pauly, 2013). Values used in sensitivity analysis 3) were based on authors' discretion.

Given the potential compounding effects that uncertainty related to multiple key inputs could have on results, Monte Carlo simulation was used to model the probability of results under a set of estimated input parameter probabilities. Monte Carlo simulation was undertaken using SimaPro software (PRé Consultants, 2010). The cumulative effects of uncertainties in the parameterization of 17 IPC model input parameters were calculated by allowing each input to vary simultaneously (PRé Consultants, 2010). Foreground input parameters were replaced by estimates of minimum, maximum and 'best estimate' frequency distributions for each (Table 4). These values were determined through consultation with industry experts and where possible, distributions were based on the calculation of a standard deviation when data were available. When neither expert consultation nor data analysis was possible, best judgement of the authors was used. A total of 10,000 runs were made to generate a distribution of final model outcomes.

Table 4. Uncertainty parameters used to calculate Monte Carlo simulations for the intended production cycle (IPC). Values are for total production cycle unless indicated. Distribution types: T= Triangular, N= Normal, U= Uniform.

Input parameter	Unit	Distribution type	Minimum - maximum	Mean	Standard deviation
Feed use[1]	kg	T	205,726-249,676	234,608	
Electricity use[2]	kWh	N	777,113-808,029	792,571	15,458
Infrastructure life-expectancy[1]	years	U	10-25	20	
Smolts required[3]	Kg	T	1,683-2,805	2,243	
Hatchery feed use per 1000 kg of smolts[3]	kg	T	1,300-1,700	1,500	
Hatchery diesel use per 1000 kg of smolts[3]	MJ	T	9,025-9,975	9,500	
On-site diesel use [3]	l	T	1,515-1,675	1,595.40	
On-site emissions[4]					
Solids	kg	T	35,211-42,733	40,154	
Phosphorus	kg	T	1,518-2,003	1,837	
Nitrogen	kg	T	7,260-10,090	9,119	
Herring/hake meal yield[1]	%	T	17-23	20	
Herring/hake oil yield[1]	%	T	4-6	5	
Herring by-product yield[1]	%	U	85-90	88	
Hake by-product yield[1]	%	U	60-65	62.5	
Herring trophic level[5]	n/a	N	2.8-3.6	3.2	0.04
Hake trophic level[5]	n/a	N	3.6-4.5[6]	4.4	0.08

[1]Expert consultation
[2]Calculated from recorded electrical use throughout the actual production cycle
[3]Author's judgement
[4]Calculated from mass-balance model
[5]Published values (Froese & Pauly, 2013)
[6]Maximum trophic level for hake is 4.5

3.2.8 Scenario Analysis

Scenario analyses were conducted to evaluate the impact of relevant managerial and analytical decisions on the life cycle impacts of the SWAS IPC. Scenario analyses included: 1) feed formulations representative of Canadian industry average, 2) alternative electrical grid mixes, and 3) allocating impacts amongst agricultural and fisheries feed inputs by mass rather than nutritional energy. Formulations for average Canadian feeds were obtained from EWOS Canada, (pers. comm. Dr. Jason Mann, May 2013). Electricity grid mixes were based on those of BC, Nova Scotia (NS), and Canada in 2012, representing possible production locations and the national average. Allocating impacts by mass was chosen as this represents a readily available alternative biophysical measurement that has been employed in previous aquaculture LCA practice (Ayer et al., 2007).

3.3 Results

3.3.1 Life Cycle Inventory

The SWAS was stocked with an estimated 56,108 Chinook smolts, with an average mass of 35 grams that had been vaccinated against Vibriosis (a bacterial disease). A total of 43,366 fish were harvested with an average mass of 1.73 kilograms. The eFCR for salmon was 1.5 and 1.459 during the hatchery and grow-out phases respectively. Mortality rate throughout the production cycle was 17.8% (excluding mortalities at the hatchery and escapes that occurred after the system was compromised by the storm). Mortality was primarily attributed to the occurrence of bacterial kidney disease, a common disease of Chinook salmon (Elliott, Pascho, & Palmisano, 1995). No antibiotics or anti-fouling paints were used throughout the production cycle. A summary of key life cycle inventory data of the SWAS subsystems for both APC and IPC is provided in Table 5. Formulations for feed used at grow out site and to represent an average Canadian feed in 2012 appear in Table 6. Additional inventory data is provided in Appendix B.

Table 5. Life cycle inventory data for the production of 1 tonne of live-weight salmon in the solid-wall aquaculture system (SWAS) for the actual production cycle (APC) and intended production cycle (IPC). APC parameters reflect conditions as experienced and recorded during system operations between January 2011 & March 2012. IPC parameters modified from APC based on conditions that were projected to apply.

Inventory data		**APC**		**IPC**	
	Unit	**Total**	**Per tonne**	**Total**	**Per tonne**
System characteristics					
Salmon harvested	kg	75,140		171,011	
	#	43,366		42,752	
eFCR			1.459		1.371
Mortalities[1]	kg	10,154	135.1	19,803	115.8
	#	9,989		13,356	
Length of grow-out	days	413		599	
Escapes	#	2753[4]		0[4]	
Estimated waste capture[2]	%	5%		5%	
Inputs					
Juvenile production (smolts)					
Feed	kg	2,946	39.2	2,946	17.2
Diesel	l	7,462	99.3	7,462	43.6
Grow-out					
Feed	kg	109,628	1,459	234,608	1,372
Infrastructure[3]					
Fibre-reinforced plastic (FRP)	kg	38,071	25.3	38,071	11.1
Low-density polyethylene (LDPE)	kg	7,557	5.02	7,557	2.21
High-density polyethylene (HDPE)	kg	600	0.40	600	0.18
High modulus polyethylene (HMPE)	kg	754	0.50	754	0.22
Nylon	kg	406	0.27	406	0.12
Vinyl, fabric reinforced	kg	60	0.04	60	0.018
Steel, galvanized	kg	18,409	12.2	18,409	5.38
Aluminum	kg	2,733	1.82	2,733	0.80
On-site energy use					
Electricity	kWh	546,464	7,272	792,571	4,634
Diesel	L	1,100	14.6	1,595	9.3
Transportation					
Smolts – via truck	tkm	410	5.5	410	2.4
Feed – via truck	tkm	77,836	1,036	166,572	974
Tank[3] – via oceanic freighter	tkm	541,624	360	541,624	158

Inventory data	Unit	APC		IPC	
		Total	Per tonne	Total	Per tonne
Outputs					
On-site nutrient emissions					
Dissolved N	Kg	3,785	50.4	7,755	45.3
Solid N	kg	737	9.8	1,577	9.3
Dissolved P	kg	279	3.7	554	3.2
Solid P	kg	617	8.2	1,320	7.7

[1]Does not include losses related to escapes that occurred due to storm event.
[2]Waste capture rate refers to collection of solids only. Due to technical difficulties, waste capture technology operated only intermittently throughout the production cycle. It is estimated that when fully operational waste capture rate would be 50-75% of solid wastes (Personal communication, Mr. Todd Adamson, April 13, 2013).
[3]For both APC and IPC a life expectancy of 20 years was assumed. In reality, the tank was compromised after a single shortened production cycle, therefore 100% of the life cycle inventory would be attributable to the APC.
[4]All escapes are attributable to the storm event. For the IPC it was assumed that no losses of this kind would occur.

Table 6. Feed formulations for feed used in the production of salmon in the solid-wall aquaculture system (SWAS) and an average Canadian feed.

Ingredient	SWAS Feed	Average 2012 Canadian salmon feed
Fishmeal	60.0%[1]	18.4%[2]
Fish oil	20.0%[1]	9.8%[3]
Wheat	20.0%	13.9%
Canola Seed		5.9%
Canola Meal		3.3%
Canola oil		7.2%
Corn Gluten Meal		9.0%
Peas		6.0%
Soy protein concentrate		1.0%
Feather meal		2.0%
Poultry Blood meal		3.3%
Poultry by-product meal		17.2%
Poultry fat		3.0%

[1]Fishmeal from Pacific hake (90%) and Pacific herring (10%) fisheries.
[2]Fishmeal from Peruvian anchovetta (13.4%) and Pacific herring fisheries (5.0%).
[3]Fish oil from Gulf of Mexico menhaden (5.0%), Peruvian anchovetta (2.4%), and Pacific herring (2.4%) fisheries.

Energy production at the hatchery was generated on-site using diesel generators. Energy use at the grow-out site was sourced from the electrical grid while diesel generators were used to operate the waste capture system. Electricity use of the SWAS was driven primarily by pumping needs of water intake and oxygen provision by a compressor. Mean electricity use throughout the production cycle was 38,044kWh/month.

Transportation related processes required for production of salmon included movement of feed, smolts, and tank materials. Feed and smolts were transported by truck approximately 710 km and 183 km respectively to the grow-out site. There they were transferred to service boats for delivery into the tank. Tank was fabricated in China, and transported by oceanic freighter approximately 7,900 km to BC, where it was transferred to a barge and shipped to the grow-out site.

Direct emissions of wastes (solid and dissolved) were calculated on the basis of a mass-balance model (Aubin et al., 2006). Estimates of waste capture ranged between 5-10% of solid wastes (Personal communication, Todd Adamson, April 13, 2013); therefore a conservative solid capture rate of 5% was applied resulting in avoided emissions of 33 kg P, 39 kg N, and 1,002 kg solids.

Life cycle inventory data for system characteristics of the IPC (Table 5) were modeled based on projected growth and expected conditions. This resulted in decreased eFCR because for the IPC it was assumed that no fish escaped to simulate a production cycle where the storm had not occurred. Inventory data for infrastructure was identical to APC. Electricity use was assumed to be the same as during APC. This is because electricity used for pumping and oxygen supplementation was managed on the basis of influent parameters so that optimal rearing conditions were maintained regardless of density. Thus it is expected that no significant change in the rate of electricity use would occur throughout the production cycle. Transportation related distances and amount of smolts/tank materials transported remained unchanged. Amounts of feeds transported were increased to reflect increased use of feed throughout production. Emissions of wastes were modeled in the same way as for APC using the updated production data. Avoided emissions in the IPC were 70 kg P, 83 kg N, and 2,146 kg of solids.

3.3.2 Life Cycle Impact of Solid Wall Aquaculture System

Contribution analysis was conducted by categorizing production into 6 subsystems including: on-site energy use, infrastructure, transportation, juvenile production, feed production, and grow-out emissions (Table 7). For the APC feed production was the most important driver of all impacts (the sole driver of BRU) with the exception of MEP that was dominated by the on-site emission of nutrients. Other contributors to MEP included feed production (2.0%), smolt production (1.%), and infrastructure (.1%), although these are negligible when compared with on-site emissions (96.8%). The energy use subsystem, associated with on-site use of electricity and fossil fuels, was the second largest contributor to life cycle impacts for GWP (13.5%) and CEU (42.1%). Production of smolts was the second most important contributor to AP (28.3%) and fourth most important for GWP (11.1%). Infrastructure was a minor contributor to GWP (12.5%), AP (8.2%), and CEU (9.9%). Transportation plays only a minor role in all impact categories, the largest which is GWP (3.0%) (Table 7).

Table 7. Contribution analysis of life cycle impacts associated with the production of 1 tonne of live-weight Chinook salmon in a solid-wall aquaculture system (SWAS) for the actual production cycle (APC) and intended production cycle (IPC). Impact categories reported include global warming potential (GWP), acidification potential (AP), marine eutrophication potential (MEP), cumulative energy use (CEU), and biotic resource use (BRU).

	GWP	**AP**	**MEP**	**CEU**	**BRU**
	kg CO2 eq	kg SO2 eq	kg N eq	MJ	kg C
APC					
On-site emissions	0	0	67.1	0	0
(%)	**0**	**0**	**96.8**	**0**	**0**
Feed production	2,318	14.6	1.4	36,324	1,429,362
(%)	**59.8**	**57.2**	**2.0**	**39.7**	**100**
Production of juveniles (smolts)	429	7.22	0.66	5,669	0
(%)	**11.1**	**28.3**	**1.0**	**6.2**	**0**
Transportation	116	0.59	0.027	2,009	0
(%)	**3.0**	**2.3**	**0**	**2.2**	**0**
Infrastructure	4,86	2.09	0.086	9,040	0
(%)	**12.5**	**8.2**	**0.1**	**9.9**	**0**
On-site energy use	5,24	0.99	0.053	38,558	0
(%)	**13.5**	**3.9**	**0.1**	**42.1**	**0**
Total	3,874	25.5	69.3	91,600	1,429,362
	100	**100**	**100**	**100**	**100**
IPC					
On-site emissions	0	0	54.6	0	0
(%)	**0**	**0**	**97.0**	**0**	**0**
Feed production	2180	13.7	1.3	34,156	1,344,038
(%)	**72.1**	**72.3**	**2.3**	**50.9**	**100**
Production of juveniles (smolts)	189	3.17	0.29	2,491	0
(%)	**6.2**	**16.7**	**0.5**	**3.7**	**0**
Transportation	106	0.48	0.023	1,830	0
(%)	**3.5**	**2.5**	**0.0**	**2.7**	**0**
Infrastructure	214	0.92	0.038	3,972	0
(%)	**7.1**	**4.9**	**0.1**	**5.9**	**0**
On-site energy use	337	0.67	0.035	24,616	0
(%)	**11.1**	**3.5**	**0.1**	**36.7**	**0**
Total	3,025	18.9	56.2	67,064	1,344,038
(%)	**100**	**100**	**100**	**100**	**100**

For all impact categories reported, the relative contribution of feed and transportation subsystems increased in the IPC compared to the APC. Additionally, the impacts associated with infrastructure and smolt production decreased. The relative importance of energy use also decreased although not as dramatically. These changes occurred because feed provision and transport increased proportionally with increased production mass. In contrast, total energy inputs increased but at a lower rate than fish production, resulting in a reduction of energy use relative to the functional unit. Total inputs of infrastructure and smolt production remain unchanged between the APC and the IPC, thus their relative contribution decreased in proportion to the increased production, reducing their relative importance. BRU is exclusively caused by the provision of feed in both APC and IPC, thus there is no change in relative contribution between the two.

Comparison of total life cycle impacts from the IPC and APC revealed that for all impact categories reported, IPC resulted in lower impacts per tonne of live-weight salmon, with reductions ranging from 6-26% (Table 7). This result is not surprising given that feed production is the single largest driver of LCI and the improved eFCR used to model the IPC represent approximately a 6% reduction in feed use per tonne of salmon harvested (Table 4). Consequently, the BRU impact category associated with IPC is reduced a comparable amount as it is driven exclusively by feed production.

3.3.3 Sensitivity and Scenario Analyses

Sensitivity and scenario analysis related to provision of feed, provision of electricity, infrastructure and methodological decisions were performed (Table 8). The largest changes associated with sensitivity analyses that allocated environmental burden according to mass were a reduction of 13.9% to BRU and increases in GWP, TA, MEP and CEU from 0.1-6.1%. Altering aspects related to the provision of feed influenced all impact categories. While the production of feed does not directly contribute large amounts to MEP, the uneaten feed and wastes excreted by the salmon do, thus a more efficient eFCR results in lower on-site emissions, reducing the contribution to MEP. Sensitivity analysis using an eFCR of 1.1 resulted in reduction of impacts between 11.6 – 32.1% and increased them by 5.0-15.1% with an eFCR of 1.5. High oil/meal yield sensitivity decreased impacts by 0.2% - 17.1% while low yield increased them by approximately the same amount. Reducing the trophic level of both herring and hake by the standard error provided in FishBase (+/- 0.8 and 0.4 respectively) (Froese & Pauly, 2013) reduced the overall BRU by 84.0% but did not affect the other impact categories.

Table 8. Sensitivity and scenario analysis for total life cycle impacts per tonne of live-weight salmon produced in the solid-wall aquaculture system (SWAS). The relative change (**%**) from the intended production cycle (IPC) is also shown. Impact categories reported include global warming potential (GWP), acidification potential (AP), marine eutrophication potential (MEP), cumulative energy use (CEU), and biotic resource use (BRU).

	GWP (kg CO2 eq)	**AP** (kg SO2 eq)	**MEP** (kg N eq)	**CEU** (MJ)	**BRU** (kg C)
IPC sensitivity analysis					
Mass allocation	3144	20.1	56.3	68,783	1,156,760
(%)	**+3.9**	**+6.1**	**+0.1**	**+2.6**	**-13.9**
FCR 1.1	2,573	16.1	38.2	59,943	1,077,692
(%)	**-14.95**	**-14.8**	**-32.1**	**-10.6**	**-19.8**
FCR 1.5	3,238	20.3	64.7	70,420	1,469,534
(%)	**+7.1**	**+6.9**	**+15.1**	**+5.0**	**+9.3**
High oil/meal yield	2,707	17.4	56.2	62,030	1,114,284
(%)	**-11.5**	**-8.1**	**-0.2**	**-7.5**	**-17.1**
Low oil/meal yield	3,344	20.5	56.3	72,117	1,574,647
(%)	**+10.6**	**+8.1**	**+0.2**	**+7.5**	**+17.2**
Low trophic level	3,025	18.9	56.2	67,065	215,229
(%)	**+0**	**+0**	**+0**	**+0**	**-84.0**
IPC scenario analysis					
Average Canadian Feed	3,167	106.9	68.2	46,636	15,128
(%)	**+4.7**	**+463.9**	**+21.2**	**-30.5**	**-98.9**
Electricity Canada	4,505	23.8	56.4	74,595	1,344,038
(%)	**+48.9**	**+25.6**	**+0.3**	**+11.2**	**+0**
Electricity NS	8,245	43.7	57.1	92,083	1,344,038
(%)	**+172.5**	**+130.5**	**+1.5**	**+37.3**	**+0**
10 year life expectancy	3,119	19.3	56.3	69,095	1,344,038
(%)	**+3.1**	**+1.7**	**+0**	**+3.0**	**+0**

Life cycle impacts of the average commercial feed scenario showed a minor increase in GWP (+4.7%), moderate reduction in CEU (-30.5%), and major reductions in BRU (-98.9%). That same feed would moderately increase MEP (+21.2%) and substantially increase AP (+463.9%). Reductions in GWP, CEU and BRU are reflective of the lower fishery derived ingredients in this feed. The increase in AP is largely caused by the emissions of ammonia associated with the production of poultry required for poultry by-product meals. SWAS powered by electrical mixes representative of average Canadian and NS grids in 2012 both resulted in increased impacts for GWP, AP, and CEU. These are caused by a larger proportion of generation methods that contribute to these impact categories (e.g. fossil fuel combustion) in these electrical mixes. Altering the electricity mix had very little impact on MEP and did not affect BRU.

3.3.4 Monte Carlo Analysis

The effect of multiple sources of uncertainty in LCI of the IPC for the SWAS was evaluated using Monte Carlo simulation using 10,000 runs (Table 9). This showed that in comparison to the LCI of the IPC average GWP, AP, and CEU increased while MEP and BRU are decreased. The distribution of these impact categories showed considerable variation with the coefficient of variation (CV – the ratio of standard deviation to the mean) ranging from 0.43-63.7%.

Table 9. Results of Monte Carlo simulation for the life cycle impacts associated with producing 1 tonne of live-weight salmon in the solid-wall aquaculture system (SWAS) for the intended production cycle (IPC).

Impact category	Mean	Median	Standard deviation	Coefficient of variation
GWP (kg CO2 eq)	3,080	3,030	550	17.9%
AP (kg SO2 eq)	19.2	18.8	1.98	10.3%
MEP (kg N eq)	56.3	56.2	0.242	0.43%
CEU (MJ eq)	67,840	67,532	7,598	11.2%
BRU (kg C)	711,458	577,698	453,406	63.7%

3.4 Discussion

3.4.1 Actual Production Cycle (APC) Versus Intended Production Cycle (IPC)

Ideally, in an LCA characterizing the typical production of a study system, data from multiple growth cycles would be used to obtain average operational inputs/outputs. Unfortunately because the SWAS is novel and only operated for 13.5 months this was not possible; therefore, in addition to the characterization of actual production (the APC) a model was constructed to approximate a complete production cycle (the IPC). Results indicate that for all impacts assessed the IPC performed better than the APC, demonstrating the importance of: 1) increased production as a means of improving life cycle environmental impacts when being assessed according to a production based functional unit and 2) the effect of losses (in this case due to escapes) on life cycle environmental impacts of aquaculture systems. For example, even though electricity use per month was the same for both models, electricity use per tonne of salmon produced was lower in the IPC (4,634 kWh $*t^{-1}$) than the APC (7,272 kWh$*$ t^{-1}).

It is hoped that the IPC provides a reasonable representation of typical production in the SWAS should it continue to operate in the future. However, there are two primary assumptions underlying this model: 1) that the failure of the SWAS caused by the storm was a chance event that is not likely to reoccur (or subsequent designs will be able to survive comparable storms) and 2) the data collected for the APC are representative of future operations. Both of these assumptions are potentially false but the second in particular is problematic. This is because due to the complex nature of aquaculture operations there is typically a learning curve associated with management, especially for a novel system such as the SWAS. It is expected that as managers become more familiar with the system management of issues such as prevalence of disease, feed wastage, and animal stress will be reduced, leading to improved growth performance. This in turn is likely to be reflected in improved system parameters such as eFCR and mortality rate. A second issue with this assumption is that there are many factors which can affect production that are difficult to know and/or predict (e.g. weather, emergence of novel disease, technical malfunctions, etc…). The upshot of these is that they contribute to the already uncertain nature of results (see section 3.3.4).

To characterize the cumulative uncertainty of the IPC in a meaningful way Monte Carlo simulation was performed. This revealed the effect of multiple sources of uncertainty on the LCI of the SWAS (Table 9). The coefficient of variation for GWP, AP, and CEU were all above 10%, while it was below 1% and above 60% for MEP and BRU respectively. Due to this uncertainty and other sources of uncertainty not accounted for, the results obtained here should be interpreted with caution. This is particularly the case with BRU which, based on the parameters modelled here, exhibits very large uncertainty. This is largely attributable to the uncertainty in trophic level of Pacific herring and Pacific hake used in the fishmeal, as it exerts a dramatic role over the BRU (as demonstrated by the sensitivity analysis of trophic level, see Table 8). The reason that the mean BRU is much lower than what is reported in the IPC is because even though the standard deviation for trophic level reported in FishBase (Froese & Pauly, 2013) is representative of a normal distribution, the maximum trophic level for Pacific hake is very close to the trophic level reported, while the minimum falls well below; thus constraining the upper limit of BRU, contributing to a lower average value over 10,000 runs.

The Monte Carlo simulation used here is a useful tool for characterizing the range of uncertainty based on known parameters but it was limited in two important ways; the first which is an issue with the modelling uncertainty and the second is associated approximating real world scenarios. Firstly, it only characterized uncertainty based on known variables, thus parameters which have not been assigned uncertainty were not accounted for. While effort was made to parameterize all those variables which were likely to contribute to uncertainty in important ways, less important variables were ignored. Individually these are not likely to change the results but cumulatively it is possible that they would make a significant impact. Secondly, the parameters modeled here do not account for all possible events that can occur. This is because it is very difficult to predict these stochastic events. Most notably this includes events that would completely compromise the system, such as a disease outbreak that causes complete mortality. Indeed, the storm event that compromised the actual production is an example of these.

3.4.2 Growing Salmon in a Solid Wall Aquaculture System

Employing LCA methodology to analyze the SWAS allows environmental hot-spots to be identified and aids in finding ways to reduce environmental impacts. Results of the life cycle impact assessment reinforce previous findings regarding the importance of feed in contributing to environmental impacts of aquaculture systems (Ayer & Tyedmers, 2009; D'Orbcastel et al., 2009; Jerbi et al., 2012; Pelletier et al., 2009). With the exception of MEP, provision of feed contributed more than 39% for the APC and 50% for the IPC to all impact categories reported (table 7). However the low contribution of feed to MEP is somewhat misleading because on-site emissions (which contribute the majority of MEP impacts) are a direct result of the provision of feed to fish and subsequent emission. These emissions are either uneaten losses or wastes excreted by the fish after digestion. From this perspective, 'on-site emissions' and feed related impacts are intimately linked and the provision of feed can be considered an important factor contributing to MEP (Jerbi et al., 2012). This distinction is important because it shows that reductions in feed use will actually reduce MEP. For example sensitivity analysis of eFCR shows that when it is improved to 1.1:1 the MEP is reduced by approximately 32% (Table 8).

Changing the ingredients used in production of feed can substantially alter impacts. Fish meal and oil in particular are important contributors to the overall impact of fish feeds (Papatryphon et al., 2004; Pelletier & Tyedmers, 2007, 2010b). For this reason, reducing fish meal/oil in feed or replacing it with less impactful substitutes can result in significant reductions in LCI (Pelletier & Tyedmers, 2007). For example, sourcing fish meal/oil from species with high yields reduces the total amount of fish required to produce sufficient meal/oil for feed and therefore reduces impacts associated with their provision (see high/low yield scenarios Table 8). Likewise, sourcing meal/oil from fisheries with lower fuel use reduces the amount of fuel required (and associated environmental impacts) to catch sufficient fish (Tyedmers, 2004). Choosing species from lower trophic levels (e.g. herbivores or primary producers) reduces energy loss through trophic transfer (Pauly & Christensen, 1995). As a result, less primary productivity (as measured by BRU) is required to support the growth of low-trophic fish compared with an equal mass of high-trophic fish. Another option for reducing feed related impacts includes choosing agricultural products as substitutes for fishery-derived ingredients (Pelletier & Tyedmers, 2007; Pelletier et al., 2009). While this holds much promise as a potential avenue for reduction of LCI, research to date has yet to find consistent improvement in any LCI other than BRU (Boissy et al., 2011; Papatryphon et al., 2004). Indeed, many low-impact marine ingredients outperform other high-impact crop and livestock ingredients, indicating that it is perhaps more important to focus on the impacts of specific ingredients rather than whether they are of marine or agricultural origin. Important characteristics of those ingredients include the fuel use required for production and the geographic area where they are produced (Boissy et al., 2011). However, care must be taken when choosing any of these options, as there is potential that they will result in problem shifting where other impacts not accounted for in LCA will be favoured over those that are included. For example, agriculture is often considered one of the most environmentally damaging activities globally (Foley et al., 2011; Tilman et al., 2011), yet LCA methodology is currently unable to adequately address many related concerns such as its impacts on biodiversity. Should decision-making be based solely on the basis of LCA results, many of these impacts that are not considered or do not have robust methods of assessment could become aggravated (Pelletier et al., 2009). A final consideration in choice of ingredients is the performance of fish feed; if lower-impact feed formulations result in decreased growth of the cultured fish, then gains in environmental performance of the feed can be off-set by increased FCR (i.e. increased feed use).

Many of the changes resulting from differences in impacts of ingredients are shown in the average Canadian feed scenario. The average Canadian feed contains a smaller marine derived fraction from species occupying lower trophic levels with a larger fraction of both crop and poultry-derived ingredients (Table 6). In sum, this formulation resulted in increased GWP, AP and MEP while CEU, and BRU were decreased. Noteworthy here is the large increase in AP and the large decrease in BRU. These changes are associated with increased inclusion of poultry by-products and agricultural products, respectively. The poultry products increase AP because of large nitrogen emissions associated with management of poultry excrement while BRU is much reduced because of the low trophic level of species used and the high inclusion rate of primary producers (i.e. crops). This result reinforces the need to address environmental impacts of feeds by ingredients, while simultaneously demonstrating the trade-offs that occur. This is in contrast to the need to design feeds holistically so that they meet the nutritional requirements of the fish.

The role of on-site energy use in contributing to the environmental impacts of SWAS is highly dependent on the primary source from which electricity is generated. Considering the large amount of electricity used, the impacts of electricity use are not especially large in the current application of SWAS because of the high proportion of hydro-electricity in the BC grid. Scenarios using electrical grids representative of average Canadian and NS grids show dramatic increases in the contribution of electricity to GWP, AP, and CEU. In the NS electrical grid scenario (nearly 60% coal) electricity overtakes feed provision as the dominant contributor to GWP and AP. This mirrors similar results of other researchers who have found electricity generation by coal, oil, and natural gas contribute more to many impact categories than sources such as hydro, wind, and solar among others (Ayer & Tyedmers, 2009; Cao et al., 2011). To reduce the impact of the SWAS, electricity should be sourced from the low-impact sources and/or reduction in electricity use should be targeted. While the electrical grid to which aquaculture systems are connected is not typically something producers can control, it can be considered pre-construction during the site selection process. Additionally, for producers in areas with a high reliance on high impact sources of electricity, the reduction of electricity use or installation of renewable energy technologies offers potential for environmental improvement. Reduced electricity use could potentially be achieved through management and design efficiencies. As pumping and oxygen generation account for the majority of electricity use, these are ideal systems which could be examined.

Of tertiary importance, behind the provision of feed and energy are the smolt and infrastructure subsystems. The life cycle environmental impacts attributable to the production of smolts are typically very low in salmon production(Ayer & Tyedmers, 2009; Pelletier et al., 2009). Here smolts contributed as much as 28.3% and 16.7% to reported impact categories for the APC and IPC respectively, an unusually large amount. This occurs because the hatchery from where these fish were sourced was not connected to the grid and all electricity used in their production was produced on-site using a diesel generator. Infrastructure required to construct the SWAS contributed as much as 11.5% and 6.4% of overall impacts in the APC and IPC respectively. Infrastructure typically does not contribute a large amount to life cycle impacts because the inputs/outputs associated with it are distributed over the lifetime production of the system, rather than for a particular production cycle. This phenomenon is demonstrated in the scenario where the life-expectancy of tank infrastructure is reduced by half, from 20 to 10 years, increasing the impacts associated with infrastructure by 50% (which results in an overall increase from 0-3.1%, Table 8).

Transportation contributed very little to overall LCI. Contribution analysis indicates that its largest contribution was 3.0% and 3.5% to GWP for APC and IPC respectively (Table 7). For this reason no scenario analysis or uncertainty analysis related to transportation was undertaken. This result may be contrary to the perception of lay people and environmentalists alike who stress the importance of ‘food miles’ (a concept which emphasizes the importance of the distance food is transported) but it is actually consistent with agricultural production in general, wherein greenhouse gas emissions are dominated by the production phase of the life cycle (Weber & Matthews, 2008). In fact, distance of travel has been found to be a poor indicator of overall environmental impacts because of its narrow focus that ignores most of the supply chain associated with food production (Edwards-Jones et al., 2008).

On-site emissions were of negligible concern with the exception of MEP where they contribute to over 98% of impacts. In order to reduce MEP this is clearly an area of focus for future development. With moderate improvements to current waste capture technology it is estimated that a solid waste capture rate of 50-75% can be achieved (personal communication, Mr. Todd Adamson, April 2013). Due to the importance of on-site emissions in MEP, this would represent a significant improvement over the current system and warrants further investigation. However, should the improved waste capture technology be employed it will require increased use of diesel fuel to power the generators which operate the waste capture system (personal communication, Mr. Todd Adamson, April 2013), increasing impacts of on-site fuel use. A significant challenge in the reduction of MEP is the ability to capture the soluble fraction of nutrients in effluent. While the current waste-capture technology is unable to do this, there are several other techniques in various stages of development which can achieve this through physical or biological means (e.g. nitrification or denitrification reactors, constructed wetlands, algal systems) (Martins et al., 2010). Alternatively, a strategy for reducing MEP without waste-capture involves optimizing nutritional management, whereby nutrient uptake efficiency by salmon is increased and the amount of waste nutrients reduced (Bureau & Hua, 2010). This has the added benefit of improving eFCR, thus likely contributing to improved environmental performance on other impact categories that are influenced by provision of feed.

3.4.3 Comparison With Net-Pens and Other Aquaculture Technologies

Debate concerning the appropriate method for culture of salmonids is widespread (Chadwick et al., 2010; Tal et al., 2009). At issue, in large part, is the environmental sustainability of culture techniques. Commercial scale projects are currently underway to explore the feasibility of alternative culture technologies such as RAS (e.g. 'Namgis First Nation, 2013; Langsand Laks, 2013); the SWAS is a unique example of these. While many of these technologies are designed specifically to avoid local ecological and environmental impacts, it is useful to simultaneously consider the global and regional environmental impacts of these projects.

LCA provides a lens through which contributions to these global and regional impacts (e.g. climate change and acidification) can be examined; however, it is currently very poor at addressing local environmental impacts (e.g. effects on biodiversity, spread of disease, or benthic habitat). The development of local impact categories specific to seafood systems and aquaculture is an active area of research (Ford et al., 2012), however to-date none of these have gained broad acceptance. The impact categories chosen for this paper are broadly representative of those used for other finfish aquaculture systems. Notable exceptions include categories addressing water use, land use, ozone depletion and various ecological/human toxicity potentials (Henriksson et al., 2011). These were omitted because they are not directly relevant to the SWAS (e.g. land use, freshwater use) or there exists considerable uncertainty in their characterization (e.g. categories related to toxicity potential) (Reap et al., 2008a).

A comparison of the SWAS IPC with published LCAs on various finfish aquaculture technologies including seven net-pens and ten alternative systems (i.e. other than net-pens) was done (Aubin et al., 2009; Ayer & Tyedmers, 2009; D'Orbcastel et al., 2009; Jerbi et al., 2012; Pelletier & Tyedmers, 2010b; Pelletier et al., 2009) (Appendix C). The purpose of this comparison was to highlight potential differences in the relative LCI of the SWAS technology with other systems.

Due to methodological, geographic, and temporal differences, and the large amount of uncertainty in LCI of the SWAS, conclusions drawn from this comparison are speculative. Moreover, as the SWAS is an emerging technology it is subject to a whole host of issues that more established aquaculture systems will have dealt with in the past. This means that the current LCI of the SWAS is likely higher than what is expected in the future. For this reason, this comparison is only meant to give a general sense of the relative performance of these various systems, rather than a direct comparison for the purpose of determining the environmentally optimal system. The IPC was used as all of the published LCA reports of other systems were done on complete production cycles.

In terms of GWP the SWAS outperforms two of seven net-pens and four of ten alternative technologies. However, in the NS electricity scenario, SWAS outperforms none of the net-pens and only two of the alternative technologies. Electricity is similarly important for AP where SWAS outperforms all net-pens and six of nine alternative technologies but in the NS electricity scenario AP increases by over 250% and the SWAS outperforms none of the net-pens and only four of ten alternative systems. This increase in AP is directly attributable to the increased use of coal and related emissions. The 56.2 kg N eq MEP of SWAS (61.0 kg PO4 eq when calculated with CML 2) is well within the range of other reported results 17.8-215.0 kg PO4 eq, with an approximately even split between those that have higher and lower emissions. This result is perhaps not unexpected as MEP is largely a function of the FCR of fish (assuming roughly similar nutrient content in feed) and the eFCR of 1.37 for fish grown in SWAS falls in a similar range relative to other published values (min 0.8, max 2.08 mean 1.39).

When comparing alternative aquaculture technologies CEU is an important factor because energy use has been shown to contribute a significant environmental burden for alternative technologies in comparison to conventional net-pens (Ayer & Tyedmers, 2009). Our analysis shows that the 71,820 MJ CEU of the SWAS reported here compares favourably in six of ten alternative systems with reductions in CEU ranging from 6,435 to 281,206 MJ (Table C1). In the four instances where CEU was less than the SWAS, reduced energy use ranged from 16,554 to 45,294 MJ. As expected, CEU for SWAS is higher than that of all net-pens which have a CEU in the range of 18,200 to 55,000 MJ, making the SWAS approximately 1.5 – 4 times as energy intensive. Much of this increased CEU is attributable to the pumping and generation of oxygen which is not necessary in net-pens.

The most striking difference between LCI of SWAS and other aquaculture systems is the BRU. BRU of SWAS is 1,344,038 kg C per tonne of fish produced, nearly 10 times larger than the next highest BRU and over 65 times the smallest BRU reported. While uncertainty in the BRU is very high, the difference between BRU of the SWAS and other systems is still remarkable. The underlying reason for these differences in BRU can be explained by the eFCR of the system and the ingredients used. As demonstrated by the two sensitivity analyses of eFCR, higher eFCR indicates higher feed use and therefor higher biotic resource use. An eFCR of 1.1 and 1.5 resulted in a decrease of 237,593 and an increase of 111,951 kg C, respectively. The impact of ingredients is revealed by the average Canadian feed scenario, where BRU is dramatically reduced, in this case to 15,128 kg C per tonne of live-weight salmon. This occurs because the average Canadian feed includes a higher proportion of crop products and the fish species used to produce meal and oil have lower trophic levels than those used in the SWAS feed. The importance of trophic level in the calculation of BRU is due to the formula used, wherein changes in trophic level result in exponential changes in the result for BRU(Pauly & Christensen, 1995). Pacific Hake which make up the majority of fish ingredients used for the SWAS feed have a trophic level of 4.4, the same level as wild Chinook salmon (Froese & Pauly, 2013). This highlights the problem with “raising tigers of the sea” noted by Naylor & Burke (2005) where high trophic level species require exponentially higher amounts of marine biotic resources. Here where the feed ingredient occupies the same trophic level as the cultured species the problem is compounded by feeding “tigers” to “tigers”. The implications associated with differences in trophic level on BRU are best demonstrated in the “low trophic level” sensitivity analysis where BRU is reduced by nearly 84% by reducing trophic level of ingredients (Table 8). For improvements in BRU it is clear that reductions in feed use and high trophic level ingredients are required.

There was no consistent difference between the MEP of SWAS and other aquaculture systems. It performed better than four of ten alternative technologies and three of seven net-pens. Contribution analyses show that on-site emissions are the primary contributors for all of these systems. Sensitivity analyses indicate that eFCR is the only characteristic tested which resulted in meaningful changes to MEP in the SWAS. Therefore to improve this category efforts should be focused on improving feed utilization. Secondarily, since MEP is primarily a result of on-site nutrient emissions, improving waste capture rate, particularly of dissolved nitrogen, could also reduce these. However, implementing more comprehensive waste capture technology is likely to be technically difficult and require additional material and energy inputs, potentially resulting in environmental trade-offs.

3.4.4 Contained Aquaculture and Ecosystem Services

Previous research has suggested that technological intensification of contained aquaculture systems (e.g. RAS or marine bag systems) isolate those systems from ecosystems and the services they provide (Aubin et al., 2009; Ayer & Tyedmers, 2009; Wilfart et al., 2013). As a result, these services are either lost, leading to negative effects on production, or they must be provided (i.e. substituted) by other means involving the use of materials/energy which can aggravate environmental impact. According to this logic, the further removed aquaculture systems are from functioning ecosystems, the greater their LCI is likely to be. This phenomenon is potentially manifested in the larger value for CEU of the SWAS in comparison to net-pens (Table C1). When compared to alternative systems there appears to be no clear advantage but if only RAS are compared then, in many regards, the LCI of SWAS are less (Table C1). If this observation accurately describes these situations then it would appear that for life cycle impacts the SWAS occupies a middle ground between technologically intensive systems such as RAS and less intensive net-pens.

While potentially informative, there are several complicating factors that would need to be addressed in order to usefully apply this type of observation. Firstly, there is the possibility that the ecosystem service substitution (both type and amount) that occurs when implementing either RAS or SWAS are the same but with different environmental impact. This would suggest that the environmental burden associated with substitution for lost ecosystem services is dependent on the method through which it is achieved. If different aquaculture systems are able to substitute the same ecosystem services at different environmental impact then there is potential for improvement through innovation (technological or otherwise). A second issue with comparing different systems based on the substitution for ecosystem services is that the delivery of these services is highly dependent on the environmental context (i.e. the ecosystem) from which they are derived. This means that the ecosystem services (i.e. benefits) provided to each system is likely different. While this makes it challenging to compare the different systems in a standardized way, it does raise very interesting questions regarding the choice of technology in these applications.

Ultimately this research is not able to address the appropriate or ideal substitution rate or the environmental impacts of any of these aquaculture systems. However, it does provide an ideal starting point for investigations into the environmental trade-offs associated with ecosystem service substitution of alternative systems. Considering the importance of increasing food production in the future while balancing environmental impacts at multiple scales, this area of investigation presents itself as an important area for further research in aquaculture and other agricultural systems. LCA provides an important method for determination of these impacts; however to develop a complete picture of all trade-offs a more holistic approach is suggested, combining LCA with methods for quantifying ecosystem services and environmental impacts not well suited to quantification by LCA.

3.5 Conclusion

Using data of an incomplete production cycle an LCA of a novel salmon aquaculture technology was conducted. In addition to quantifying the life cycle impacts of the APC, the IPC was assessed by building a model based on known operating data, published literature, and expert knowledge. This allowed for an assessment of LCI for a 'typical' tonne of live-weight salmon produced in the SWAS. The predominant contributor to life cycle impacts of this system is the production of feed. For this reason improved feed efficiency should be targeted in order to reduce impacts. Scenario analysis of feed formulations showed the dramatic impact that different feed ingredients exert on the LCI of salmon production. This highlights the need to focus on specific ingredients when developing feeds for reduced environmental impact. Nutritional management strategies can also be targeted to improve feed conversion and reduce the loss of undigested nutrients. This has the added benefit of reducing MEP associated with nutrients lost in fish wastes.

The impacts associated with provision of smolts for the SWAS were relatively higher than previous analysis have found in other salmon aquaculture systems making this life cycle stage a candidate for improvement in environmental performance. In its current application electricity is not a major driver of environmental impact, although if employed in locations with more impactful primary energy inputs, electricity could potentially contribute to larger environmental impact, especially GWP and AP. For this reason, electrical grid mix should be considered when siting future SWAS. On-site emissions of waste nutrients were the primary contributor to MEP. To reduce these, further developments of waste capture systems are recommended, especially the development of methods for removing soluble nutrients from effluent. Other subsystems were of minor importance in determining LCI.

To characterize the influence of uncertainty on LCI of the SWAS, Monte Carlo simulation and sensitivity analyses were conducted. These show dramatic range of environmental impacts and demonstrate the need for continued data collection in the future. Incorporating data of future (and complete) production cycles will help to improve the confidence in found results.

Examination of published LCAs of finfish aquaculture systems revealed that the environmental impact of SWAS is highly variable relative to other systems. Additionally, Monte Carlo analysis revealed that the life cycle impacts of the SWAS are sensitive to uncertainties in key input parameters. Together these make it difficult to determine whether this system is favourable, from a life cycle perspective, to other alternative aquaculture technologies or net-pens at the current time. As more data become available in the future these can be incorporated into the models built here to provide more substantiated conclusions. Moreover, many of the impacts associated with aquaculture and potential benefits of this technology (e.g. reduction in local ecological impacts) were not investigated here. This makes it impossible at this time to draw any definitive conclusions regarding the overall environmental performance of the SWAS relative to other systems. To this end more research investigating specific areas of concern such as the potential for disease transmission and impacts on benthic habitat are suggested.

A possible relationship was identified linking substitution of ecosystem services to environmental impact. Unfortunately, no concrete conclusions were drawn from these because sufficient knowledge of the relationships with ecosystem services is not available. Further research investigating the relationship between environmental impact and ecosystem services are required.

Chapter 4. Investigating Environmental Trade-Offs Associated With the Substitution of Ecosystem Services by Waste Capture in Salmon Aquaculture

4.1 Introduction

Ecosystem services (ES) are the benefits derived by humans from ecosystems (MEA, 2005). They are highly diverse and include services that are critical to maintaining human life as well as services that provide improved happiness and well-being. Conceptually, technology and resources used to provide comparable services can be considered substitutions for ES (Chapter 2). The ability and implications of substituting for ES using technology is a phenomenon about which very little is known but is surprisingly pervasive in modern societies; it includes all benefits conveyed by technological means that could otherwise be delivered by ecosystems. For example, the power required for transporting goods via boat can be supplied by wind-powered sails (an ecosystem service) or fossil fuel-powered motors (a substitute). Likewise, the provision of fresh water is typically thought of as an ES, however using desalination technology it is possible to derive potable water from seawater. Other examples include: the production of synthetic fibres such as polyester to make clothing; building materials such as glass which are made from natural materials, even some fuels (e.g. hydrogen formed by hydrolosis) can be created by humans. An important point regarding the substitutability of ES is that the resource requirements and environmental impacts of different substitutions are highly variable. They are dependent on the technology used to achieve the substitution, the context within which it is being done, and the nature of the ES itself (Chapter 2). For this reason, it is important to assess environmental trade-offs whenever decisions associated with ES substitution are being made in order to avoid inferior environmental outcomes.

An important arena in which substitution for ES takes place is in agricultural systems (Chapter 2). This is because agriculture is a globally important industry that relies to a very large extent on ES to support production (MEA, 2005; Power, 2010). As agricultural systems intensify, it is not uncommon for ES which previously supported production to be compromised and/or replaced (Defries, Foley, & Asner, 2004;Chapter 2). We suggest that the substitution of ES is often environmentally inferior to the baseline condition where benefits are obtained from ecosystems and the trade-offs associated with this substitution requires careful examination. Surprisingly, the cost in terms of energy and material use, as well as the environmental impacts associated with this type of substitution has never been explicitly investigated. Here, these trade-offs are explored in the context of waste management in an aquaculture system.

Technological solutions are often sought to mitigate or manage the environmental concerns associated with production of many types. This is particularly true in salmon aquaculture where considerable effort has been expended in recent years on research and development of alternative rearing technologies and techniques (Ayer & Tyedmers, 2009; Chadwick et al., 2010). Many of these technologies are designed specifically to capture solid wastes composed primarily of feces and uneaten feed. Once captured, these wastes can be disposed of or treated in ways that reduce the alterations that may result when they enter local ecosystems (e.g. composted on land). However, if released into the environment these wastes will be consumed and dispersed through time as part of natural nutrient cycles. The problem is that in many cases, the waste loading associated with aquaculture may overwhelm short term local assimilative capacity, resulting in an altered state of the ecosystem (e.g. organic enrichment and alteration of benthic community, harmful algal blooms) which can persist even once nutrient loading is stopped (e.g. altered benthic communities) (Buschmann et al., 2009; Islam, 2005). Alternatively, reliance on technology to redirect these nutrients may require the use of energy and materials which can contribute to other environmental problems. Since waste capture (WC) (defined here as the neutralization, natural or otherwise, of negative impacts associated with emissions of nutrients in aquaculture effluent) is a service that can be performed by either functioning ecosystems or functionally equivalent technology it is useful to investigate the environmental impacts of these so that trade-offs can be assessed. Such an assessment would help provide clarity regarding which circumstances employing technological solutions for WC might be most appropriate. More generally it will provide insight into the resource and environmental implications associated with ES substitution. To this end, the purpose of this research was to assess the life cycle impacts of implementing a WC system on a commercial aquaculture system that was employed in British Columbia, Canada. This was accomplished using life cycle assessment (LCA) to assess the life cycle environmental impacts associated with production of salmon in a floating solid-wall aquaculture system (SWAS) with, and without, waste capture technology.

This technology is ideal for an investigation of trade-offs associated with the substitution of ES for two reasons. Firstly, because it is used here as an add-on to an existing technology for the sole purpose of WC; this makes it possible to investigate the environmental trade-offs associated with substitution of this ES on a one-to-one basis. This is an advantage over other situations where the technologies used to capture and divert wastes also perform other services (e.g. prevention of escapes), making it difficult to tease apart the environmental burden directly associated with substituting for WC from the other services provided. Secondly, the WC technology described here is implemented solely with the intention of reducing local environmental impacts that may result from waste biomass accumulation within the receiving environment. This means that the trade-offs can be considered only along environmental dimensions, as opposed to ES substitutions which confer benefits that are not related to environmental performance.

LCA was chosen as the analytic framework as it is well suited to assessing the environmental impacts associated with production systems (Ness et al., 2007), it can be used to quantify contributions to a range of impact categories, and has been applied to aquaculture in many settings (Ayer & Tyedmers, 2009; Jerbi et al., 2012; Pelletier et al., 2009; Wilfart et al., 2013). Moreover, it is a broadly standardized and transparent methodology (ISO, 2006a) which facilitates the exploration and communication of results.

4.1.1 System Description

The WC technology being investigated here is an addition to an existing floating solid-wall aquaculture system (SWAS) operated by Agrimarine Industries Inc. near Campbell River, BC, from January 2011 – March 2012 to grow Chinook salmon (for detailed description see Chapter 3). It is designed to collect solid and semi-solid wastes extracted from the effluent stream of the SWAS, from which a portion of the settleable solids are removed, dewatered and then pumped into totes for subsequent transfer to land and composting. Based on the fluid dynamics within the tank, the physical characteristics of fecal pellets, and the nature of the waste capture technology, the projected solids capture rate is between 50-75% (pers. comm. Mr. Todd Adamson, April, 2013). Many of the finer solids would be lost through tank overflow (~90% of tank flow-through). For the analysis conducted here, a capture rate of 75% is assumed to simulate the maximum benefit associated with WC. Methodology used to estimate waste production and type is adopted from Chapter 3. The WC technology is not necessary for the functioning of the SWAS and can be employed or not without affecting production. LCA was used to characterize the material and energy use implications associated with the implementation of WC on the SWAS. The analysis conducted here is based on a hypothetical implementation of the WC technology based on designs that were not yet implemented during the first production cycle of the SWAS. Throughout the modelling, however, efforts were made to characterize the realistic performance parameters under real world conditions. Technical challenges associated with site selection, implementation, power supply and other unforeseen issues were not considered.

4.2 Methodology

4.2.1 Life Cycle Assessment

The life cycle inventory and life cycle impact assessment model of the SWAS used here as the basis of analysis is described in Chapter 3. LCA typically includes four steps: 1) the definition of the goal and scope, 2) inventory analysis, 3) impact assessment, and 4) interpretation (ISO, 2006a). To support this research, a brief overview of LCA methodology and other modelling undertaken is provided here.

4.2.2 Goal and Scope

Here, LCA methodology was used to compare the environmental impacts of producing salmon in a SWAS with and without solid waste capture. The functional unit used was 1 tonne of live-weight salmon at harvest. This functional unit was chosen over one related to waste capture because the WC system was only relevant to the production of salmon and has no purpose if salmon are not being produced. The boundaries of the system were from cradle to farm-gate and did not include a treatment of the waste once it has been captured. This is because the fate of captured waste is highly uncertain and any analytical treatment would require data that is not currently available.

4.2.3 Inventory Analysis

Foreground data for the inventory analysis, including information regarding on-site operations, infrastructure, feed use, and waste capture were solicited from Agrimarine Industries Inc. staff by questionnaire/consultation, reports, and on-site inventory audits. Data on juvenile production was obtained from Omega Pacific Hatcheries (Mr. Bruce Kenny, personal communication, January 28, 2013). Energy and material used in the production and construction of the aquaculture tank was obtained from Janiki industries based off inventory data for a second tank because data for the original tank was unavailable (Ms. Sheena Burns, personal communication, May 15, 2013). Inventory data for travel was calculated from the mass and distance of goods transported. Origin, destination and mode of transport for all materials was determined in consultation with Agrimarine Industries Inc. staff (Mr. Rob Walker, personal communication, January, 2013). Distance of travel was calculated using Google Earth. Feed formulation and source of ingredients was obtained from Taplow feeds (Dr. Brad Hicks, personal communication, January 28, 2013). These included crop and fisheries derived ingredients. Inventory data for these ingredients was obtained from peer-reviewed journals, online databases, personal communications with industry experts, and databases compiled by government and/or industry (see Chapter 3 for details). Inventory data for background processes (e.g. raw material extraction, processing, primary electricity generation) were obtained from peer-reviewed databases (e.g. Ecoinvent v2.0) that were included with Simapro software used as a support for modelling throughout the LCA (PRé Consultants, 2010).

Mass-balance models for nutrients provided in feeds and retained in fish were used to calculate on-site emissions of nutrients (Aubin et al., 2006, 2011; Cho & Kaushik, 1990). By analysing digestibility of nutrients it was possible to calculate the fraction of wastes which were solids and could potentially be captured.

To construct the inventory associated with the capture and diversion of wastes for the SWAS it was necessary to estimate the number, size, and type of all components (e.g. pumps, filters, conveyors) used to capture waste material and the generator used to power the waste capture system (section 2.1.3). Once this was complete, hourly fuel use was estimated based on manufacturer specifications for an appropriate generator. The waste capture technology was assumed to run continuously throughout the production cycle. Materials associated with the components (pumps, filters, generators) of the waste capture system were not included in the inventory as these were not expected to meaningfully contribute to overall life cycle impacts.

Data on an incomplete production cycle that spanned 13.5 months were compiled. These data were used to model a hypothetical complete production cycle (Chapter 3). The base case SWAS production cycle modeled here mirrors this model with one exception: the WC system that was used during actual production operated intermittently, achieving a waste capture rate of 5% and consumed 1,595l of diesel. Here it was assumed that this WC was not used, therefore a waste capture rate of 0% and diesel use of 0l is attributed to the base case. This base case is compared with the same production cycle except with a waste capture rate of 75% and diesel use calculated based on estimated generator size.

4.2.4 Estimating Generator Size for the Waste Capture System

To estimate the size of the generator required for the WC system it was necessary to know the kilowatts (kW) and kilovolts-amperes (kVA) required for starting the system. This was calculated by adding the startup kW and kVA of the largest motor and the running kW and kVA for the other motors. The calculations for these are shown below (equations 1-6). All motors modelled here were assumed to be 3-phase with a 92% running efficiency, running power factor (PF) of 0.85, and a starting PF of 0.2 (Engineeringtoolbox.com). Locked rotor kVA/hp is different for each motor and was retrieved from (Engineeringtoolbox.com).

(1) Running kW (RkW) = [(nameplate hp) x (0.746 kW/hp)]/ efficiency

(2) Running kVA (RkVA) = RkW/ (running PF)

(3) Starting kVA (SkVA) = (nameplate hp) x (locked rotor kVA/hp)

(4) Starting kW (SkW) = (SkVA) x (starting PF)

(5) Total SkW = Largest SkW + other RkW

(6) Total SkVA = Largest SkVA + other RkVA

Once the total starting kVA (SkVA) and total starting kW (SkW) were calculated the required generator was estimated by choosing the minimum sized generator that could safely supply those totals.

4.2.5 Life Cycle Impact Assessment

In accordance with the methodology used in Chapter 3, a mid-point (problem-oriented) approach was used to assess the potential life cycle impacts of the SWAS with and without waste capture. Life cycle impact categories used here included global warming potential (GWP), acidification potential (AP), marine eutrophication potential (MEP), and cumulative energy use (CEU). The impact category biotic resource use (BRU) was not included as part of this analysis because no changes regarding the use of feed or feed ingredients was investigated. As a result, BRU would remain unchanged, providing no valuable information. The impact categories chosen here are representative of the most commonly employed and widely accepted impact categories used for LCA of aquaculture systems (Henriksson et al., 2011; Pelletier et al., 2006).

The impact assessment method ReCiPe version 1.07 was used to assess contributions to GWP ($kgCO_2eq$), AP ($kgSO_2eq$), and MEP (kgNeq) (Goedkoop et al., 2009), while CEU (MJeq) was quantified using Cumulative Energy Demand version 1.05 (PRé Consultants, 2010). All of the calculations done to characterize these impact categories were facilitated through the use of SimaPro v 7.1, a specialized LCA software package (PRé Consultants, 2010).

4.2.6 Sensitivity and Scenario Analysis

To investigate the impact that important variables and management decisions have on the life cycle impacts sensitivity and scenario analysis were conducted. Sensitivity was conducted on the fraction of solid waste that was captured (50%, 75%, and 100%). Scenarios included, using a 40kW generator (assuming a less efficient design) and a 60kW generator (e.g. one that would be required for configuration requiring parallel startup). Moreover, the implications of connecting the WC technology to the electrical grid were investigated. Three potential electricity mixes were used to represent the electricity mixes of Canada, British Columbia (BC), and Nova Scotia (NS).

4.3 Results

4.3.1 Inventory Analysis

Data for production of salmon in the SWAS with and without WC were compiled (Table 10). Inventory results for the SWAS without WC are the same as in Chapter 3 with the exception of estimated waste capture rate and associated fuel use which are both zero. For the SWAS production with WC, a waste capture rate of 75% is assumed. Based on the estimated generator size (a 30 kW generator running at ¾ load) a total of 130,584l of diesel fuel are required to operate the WC system until salmon are harvested (599 days). This equates to 764 l of diesel per 1000kg of salmon produced. Implementation of WC results in avoided emissions of nitrogen and phosphorus equal to 1245 kg and 1042 kg respectively, to the local environment. This is equal to reductions of 7.3 kg of nitrogen and 6.1 kg of phosphorus entering the local environment per 1000kg of salmon produced.

Table 10. Life cycle inventory data for the production of 1 tonne of live-weight salmon in the solid-wall aquaculture system (SWAS) with and without waste capture (WC).

Inventory data		Without WC		With WC	
	Unit	Total	Per tonne	Total	Per tonne
System characteristics					
Salmon harvested	kg	171,011		171,011	
	#	42,752		42,752	
eFCR			1.371		1.371
Mortalities[1]	kg	19,803	115.8	19,803	115.8
	#	13,356		13,356	
Length of grow-out	days	599		599	
Escapes	#	0		0	
Estimated waste capture[2]		0%		75%	
Inputs					
Juvenile production (smolts)					
Feed	kg	2,946	17.2	2,946	17.2
Diesel	l	7,462	43.6	7,462	43.6
Feed					
feed	kg	234,608	1371.9	234,608	1371.9
Infrastructure					
Fibre-reinforced plastic (FRP)	kg	38,071	11.1	38,071	11.1
Low-density polyethylene (LDPE)	kg	7,557	2.21	7,557	2.21
High-density polyethylene (HDPE)	kg	600	0.18	600	0.18
High modulus polyethylene (HMPE)	kg	754	0.22	754	0.22
Nylon	kg	406	0.27	406	0.12
Vinyl, fabric reinforced	kg	60	0.018	60	0.018
Steel, galvanized	kg	18,409	5.38	18,409	5.38
Aluminum	kg	2,733	0.80	2,733	0.80
On-site energy use					
Electricity	kWh	792,571	4,634	792,571	4,634
Diesel	L	0	0	130,584	763.6
Transportation					
Smolts	tkm	410	2.4	410	2.4
Feed	tkm	166,572	974	166,572	974
Tank	tkm	541,624	158	541,624	158

Inventory data		Without WC		With WC	
	Unit	Total	Per tonne	Total	Per tonne
Outputs					
On-site nutrient emissions					
Dissolved N	kg	7,755	45.3	7,755	45.3
Solid N	kg	1,660	9.7	415	2.4
Dissolved P	kg	554	3.2	554	3.2
Solid P	kg	1,389	8.1	347	2.0

[1]Does not include losses related to escapes that occurred due to storm event.
[2]Waste capture rate refers to collection of solids only. It is estimated that when fully operational waste capture rate would be 50-75% of solid wastes (Personal communication, Mr. Todd Adamson, April 13, 2013).

The WC system requires a floating drum filter (2hp), an axial flow pump (15hp), two sludge pumps (1hp each), a fan press (5hp), and a conveyor (2hp) (Figure 7). At full load the combined total running power required is slightly less than 26hp (approximately 19 kW) (the drum filter and one of the sludge pumps do not operate continuously). However, due to additional startup power demands required by all components, it is estimated that the WTC system modelled would require a generator capable of producing 37.5 hp (approximately 27.8 kW) (see Section 2.1). For this reason a 30,000 W Perkins Diesel home generator was chosen. According to manufacturer specifications for this model, the fuel use at ½ load is 9.84 l/h.

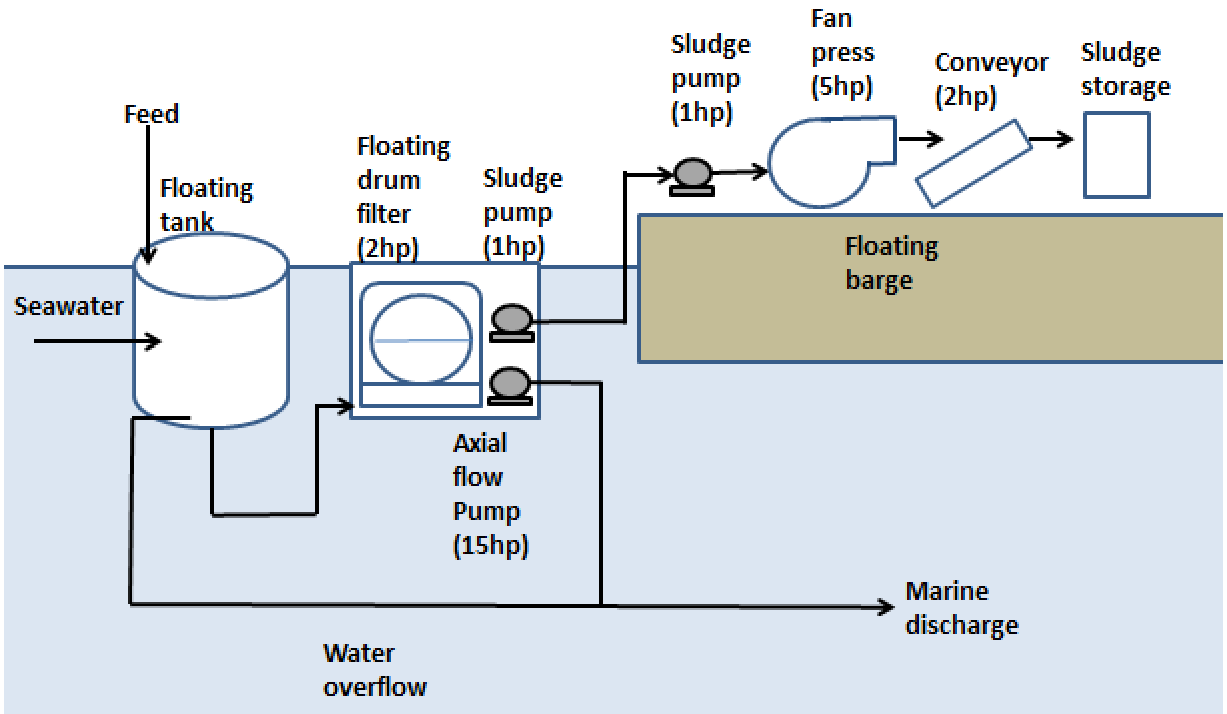

Figure 7. Process flow chart for collection of settleable solids using waste capture technology in the solid-wall aquaculture system. *Courtesy of Agrimarine Industries Inc.

Table 11. Technical specifications for components of proposed waste capture system.

Component	Operating power (hp)	RkW (kW)	SkW (kW)	RkVA (kVA)	SkVA (kVA)
drum filter (2hp)	2	1.62	3.60	1.91	18.00
axial pump (15hp)	15	12.16	18.87	14.31	94.35
sludge pump (1hp)	1	0.81	2.00	0.95	9.99
sludge pump (1hp)	1	0.81	2.00	0.95	9.99
fan press (5hp)	5	4.05	7.55	4.77	37.75
conveyor (2hp)	2	1.62	3.60	1.91	18.00

4.3.2 Life Cycle Impact Assessment

Life cycle assessment results indicate that when WC technology is implemented the environmental impacts increase for GWP, AP, and CED relative to the base case model of the SWAS (Table 12). These increases range from 52.4 – 139% and are a result of increased use of diesel fuel required to run the generator which is used to power the WC system. The capture rate of 75% of solids emitted from the tank reduces the MEP by 9.4%.

Table 12. Life cycle environmental impacts associated with the production of 1 tonne of live weight salmon in the solid-wall aquaculture system (SWAS) with and without waste capture.

Impact Category	SWAS without WC	SWAS with WC	LCI attributable to WC	% change
GWP (kg CO2 eq)	3,025	5,410	2,385	78.9
AP (kg SO2 eq)	18.9	45.2	26.3	139
MEP (kg N eq)	56.2	50.9	-5.3	-9.4
CED (MJ)	67,064	102,209	35,145	52.4

4.3.3 Sensitivity and Scenario Analysis

Sensitivity analysis conducted on the percentage of solid waste capture indicates that it only slightly changes MEP (Figure 8). In the scenario where 100% of solid waste is captured the MEP decreases by 13.9% over no waste capture while a 50% capture rate results in a modelled reduction of 5.1% relative to the base case without WC. Scenario analyses indicate that the power source used for WC exerts considerable influence on the impact categories GWP, AP, and CEU. Increasing the size of the generator from 30kW to 40kW increases these categories as would be expected due to increased diesel use. Contrary to expectations, however, is the relatively better performance of a 60kW generator compared to a 40kW generator. The improved environmental performance is because the 60kW generator only needs to run at ½ load to generate sufficient power to run the WC technology. Based on manufacturer estimates, the 60kW generator uses less fuel at ½ load than the 40kW generator at ¾ load.

Scenario analyses investigating the consequences of connecting the SWAS and waste capture system to an electrical grid show variable results (Figure 8). Using an electrical grid representative of the average Canadian electrical mix results in increased GWP and CEU over a 30kW generator but decreased AP. Connecting to a grid representative of BC electricity production decreases GWP, AP, and MEP but results in slightly increased CEU. Alternatively, using an electrical mix representative of the NS electrical grid increases all impact categories, with dramatic increases to GWP, AP, and CEU.

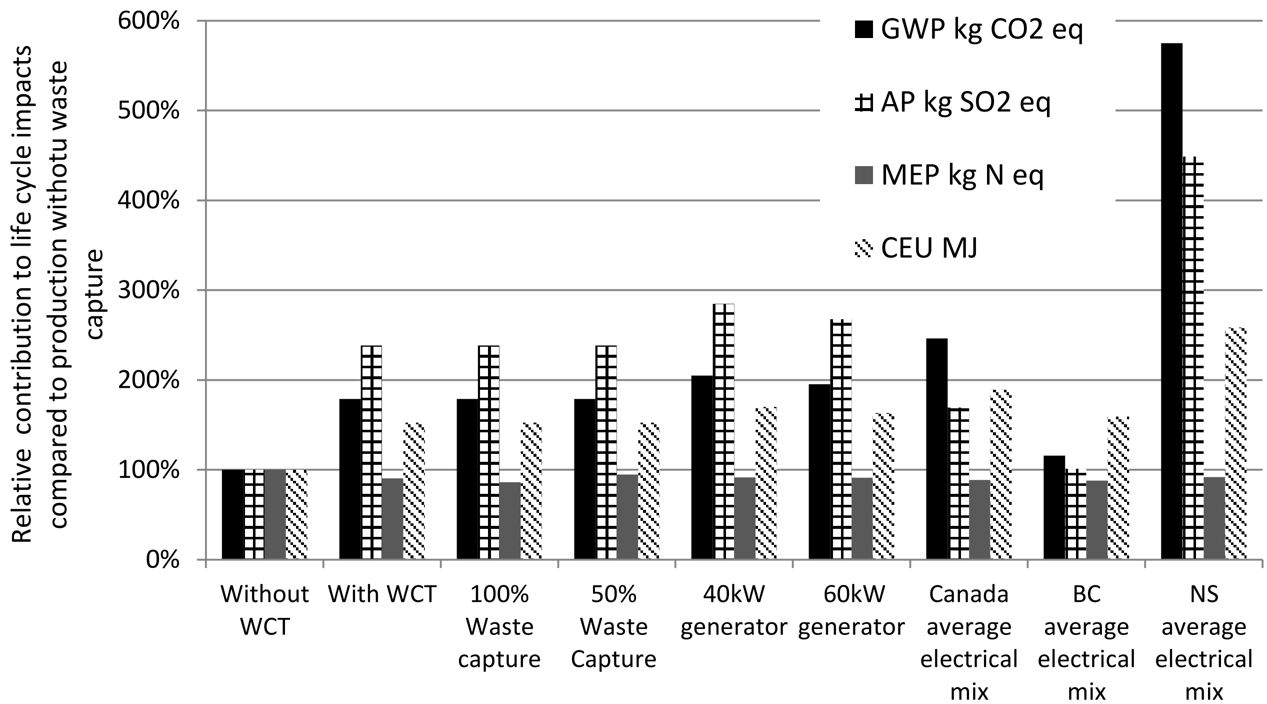

Figure 8. Sensitivity and scenario analysis associated with the production of 1 tonne of live-weight salmon in the solid-wall aquaculture system (SWAS) when waste-capture technology is employed. Results are shown relative to the base case production cycle without waste capture.

4.4 Discussion

4.4.1 Waste Capture and Treatment Technology

Overall the results of the LCA of the SWAS with and without waste capture indicate that, for the impact categories investigated here (with the exception of MEP), it is environmentally preferable from a life cycle perspective not to implement WC technology. Implementing waste capture greatly increases GWP, AP, and CED, while resulting in only minor decrease to MEP. If a WC system is to be used, it is advised that a connection to the local grid be made. If not possible, then the 30kW generator would be the next least impactful alternative. This is in agreement with previous research that has found increased life cycle impacts associated with implementation of technologically intensive aquaculture (Ayer & Tyedmers, 2009).

However, the methodology employed here is limited in two important ways, making it impossible to reach a conclusion regarding the environmentally optimal solution without further study. Firstly, we have not addressed the consequences of what occurs with waste once it is captured. Presumably it could be transported to land and composted. From there it could be used in many ways, one of which is as a fertilizer. From this perspective, the wastes could be treated as a co-product of the SWAS and would need to be modelled appropriately, either through co-product allocation or system expansion (Ayer et al., 2007). For example, estimates of impacts associated with the production of fertilizers with an equivalent amount of nitrogen and phosphorus would be subtracted from the impacts modelled here. This approach was avoided because of the considerable uncertainty concerning the fate of waste once captured.

The second, and perhaps more important, limitation is that it is not possible to definitively state the optimal alternative because LCA is not able to adequately assess the range of the environmental impacts associated with aquaculture systems, particularly those related to local ecological effects (Ford et al., 2012). In this case, environmental concerns related to the emissions of waste that are not investigated include impacts on: biodiversity, water quality, local eutrophication (as opposed to the impact category used here 'marine eutrophication potential') and benthic organic enrichment. These are critical for a complete assessment of this WC technology as it is claimed to be able to "eliminate undesirable pollution to local marine eco-cultures thus preventing eutrophication" (Agrimarine Industries Inc., 2014). While it is difficult to know the benefits of this technology without data collected on the status of the local environment and its response to nutrient loading, research shows that organic solids associated with aquaculture waste can accumulate below the farm-site leading to anoxic environments with reduced biodiversity (Brooks, Mahnken, & Nash, 2002; Kalantzi & Karakassis, 2006). Therefore it is not unreasonable to conclude that the WC technology employed here could potentially ameliorate these impacts by reducing the amount of emitted solids. Importantly however, these benthic impacts are known to be correlated to other variables, including depth, current speed, and sediment type (Kalantzi & Karakassis, 2006) therefore the environmental improvement might only be realized in those areas or situations which possess unfavourable environmental conditions and are susceptible to emissions of solid waste or where the assimilative capacity of the ecosystem is pushed beyond its limits and environmental conditions are altered. The ability of this WC system to reduce eutrophication are even less clear as this is overwhelmingly caused by the emission of soluble wastes, most importantly nitrogen (Islam, 2005; Robertson & Vitousek, 2009), which are not captured. Without further research of the farm-site and local environment, analyzing a range of environmental indices it is not possible to determine the benefits of the WC employed here (Borja et al., 2009), which would be necessary for making an informed decision on whether the trade-offs are advisable.

Even with large waste capture there is not a significant reduction in MEP. This is because the primary contributor to MEP is the soluble nitrogen dissolved in seawater that is emitted as ammonia through the gills of the salmon (Bureau, Gunther, & Cho, 2003). This soluble fraction cannot be captured by the current WC technology and is lost in the marine discharge. However if the WC system was designed such that the soluble fraction of nitrogen was removed then it would make significant improvements to the eutrophication potential of the SWAS (e.g. denitrification in anaerobic reactors). These are used in land based RAS systems (Martins et al., 2010) but to date it has not been employed in any floating systems such as SWAS, marine bag, or net pens, and is therefore not considered in this analysis. As is the case here, employing additional technologies to capture wastes is likely to require energy and material, resulting in life cycle trade-offs. Moreover, the extraction of soluble nitrogen would almost certainly entail a further increase in energy requirements for WC, further increasing the associated impacts.

4.4.2 Environmental Trade-Offs of Ecosystem Service Substitution

The results of this LCA provide understanding of the environmental impacts associated with utilizing technology to substitute for the WC in the context of aquaculture system. This will support efforts to design systems with optimal environmental performance in the future. More generally, these results are an example of how it is possible to quantify the environmental impacts associated with substitution for an ES on a one-to-one basis. This provides insight into decisions regarding the implementation of substitutes for ES and their associated trade-offs.

It is anticipated that substitutions for ES are likely to have different impacts. Indeed, as demonstrated by the sensitivity and scenario analysis, even substitution for WC by using a specific technology can result in highly variable environmental impacts, depending on the nature of underlying energy flows (Figure 8). As such this research is not meant to provide definitive information on the impacts of all ES substitution but rather to offer an example upon which future studies can draw. These could support managers and decision makers in the quest to achieve environmentally optimal situations when faced with development alternatives regarding ES substitution.

In many situations, it may turn out that it is not environmentally optimal to substitute for ES. This is because ES are largely powered by solar energy and naturally regenerative, so long as the natural capital from which they are generated remains intact (MEA, 2005). The benefits they provide are thus realized without the use of external inputs of matter or energy and the accompanying environmental impacts. Additionally, ecosystems are multifunctional and provide a variety of services which can potentially benefit agricultural production (Tscharntke, Klein, Kruess, Steffan-Dewenter, & Thies, 2005). Therefore in situations where ecosystems become increasingly degraded and substitution is required for multiple services then the resources required also increase. This is demonstrated by Moberg & Rönnbäck (2003) in their investigation of the services provided by seascapes (mangroves, coral reefs, and sea grass beds). They found that the multitude of services provided by these ecosystems often requires a large continual investment of resources. Moreover, they demonstrated that there is a knowledge deficit regarding the complexity and dynamics of these systems so that complete replacement of the ES provided is likely not possible (Moberg & Rönnbäck, 2003). Ultimately, it may be possible to substitute for these ES in efficient ways but this is likely to require knowledge and technology beyond what is currently available.

Analysis of trade-offs associated with the substitution for ES is likely to have utility in many contexts but especially in regard to the development of modern agricultural systems (Foley et al., 2011; Sachs et al., 2010). This is because agriculture is a dominant force in driving global environmental change (MEA, 2005; Power, 2010; Rockström et al., 2009). Moreover, as both recipients and providers of ES (Sandhu, Wratten, & Cullen, 2010) agricultural systems are particularly sensitive to changes in their relationship to ES. In extensive, low-input forms of agriculture the majority of processes supporting production are provided by ES (Bommarco et al., 2012). However, modern intensification of agriculture has been achieved primarily through increased reliance on external inputs of energy and agrochemicals with reduced importance of ES. As agriculture continues to intensify in the future it will be of great importance to understand the consequences of these substitutions and their role in contributing to environmental impacts so that sustainable solutions can be pursued (Foley et al., 2011; Tilman, 1999). This includes building on existing approaches that improve yield (e.g. pollination and pest regulation by biodiversity) or reduce environmental impacts (e.g. WC by wetlands) and the development of novel approaches. To this end, analyses such as conducted here and conceptual frameworks that provide guidance will be valuable resources (Chapter 2).

4.5 Conclusion

The life cycle environmental impacts associated with the application of a WC technology in the context of a SWAS were quantified using LCA. The results indicated an overall increase in all impact categories measured, with the exception of MEP where a slight decrease was estimated. Scenario analysis demonstrates that the best option for implementing WC involves connecting it to an electrical grid rather than a generator, although the sources which underpin the grid are important. Neither local environmental impacts of implementing the WC technology nor the downstream impacts of collecting organic wastes on land (i.e. composting) have been quantified here. These are likely to be highly context dependant but are necessary for a holistic assessment of environmental trade-offs. For this reason, more research on the impacts of this technology is needed.

The use of WC technology is considered here as a substitute for a service that might otherwise be provided by ecosystems. This represents the first time that this phenomena has been explicitly quantified. This type of analysis is applicable to many situations but is considered especially germane for agricultural systems because of their large global environmental impact and intimate relationship with ES (Tilman et al., 2011). As this approach represents a novel conceptualization of ES there is a great deal of research required to understand its implications. Questions that remain include: Are all ES substitutable? Is it possible to classify substitutes in meaningful ways related to resource use and/or environmental impact? What role does technological innovation play in determining the impacts of substitution for ES? Are there situations where it is always preferable to substitute, or vice versa? Moving the human enterprise in a more environmentally sustainable direction is critical for the continued well-being of individuals and societies. Being able to assess environmental trade-offs of developmental decisions, particularly in agriculture, is necessary. Here we have demonstrated one way in among many in which this can be accomplished.

Chapter 5. Discussion

Sustained population growth and shifting patterns of consumption continue to increase global demand for agricultural products (Godfray et al., 2010; Tilman et al., 2011). Unfortunately the methods and scale at which these are produced contributes to severe environmental impacts (Fedoroff et al., 2010; Foley et al., 2011; Rockström et al., 2009; Tilman, 1999). One method through which further environmental harm from biological production systems (BPS) can be mitigated that has received considerable attention is by intensifying those systems while simultaneously reducing impacts; so called "sustainable intensification" (Balmford et al., 2012; Tilman et al., 2011). This emerging concept does not describe specific techniques but instead denotes an underlying concept which can manifest itself in myriad conventional and unconventional ways (Garnett et al., 2013). To shed light on this issue, I have taken a conceptual route to explore BPS and their environmental impacts. In particular, I investigated how the relationship between ecosystem services (ES) and intensification of BPS can offer insight into their associated environmental impacts. I then quantified the environmental impacts of a specific BPS (an aquaculture system that was developed in an effort to reduce local environmental impacts) using life cycle assessment (LCA). Finally, I applied the insight from the conceptual exploration of BPS to the aquaculture system that was studied to demonstrate the feasibility of the concept. The objectives of this thesis were threefold, each of which corresponds with one of the three central chapters:

- To develop a conceptual framework exploring the relationship between ecosystem services and environmental impacts in BPS.
- To quantitatively assess the environmental impacts of a novel aquaculture technology using LCA
- To explore the utility of the framework by applying it to the same novel aquaculture technology

The purpose of this research was to investigate the environmental impacts of intensification of BPS (both generally and to an aquaculture system) in order to provide a novel perspective which is informed by ES. It is hoped that this knowledge will help identify pathways for intensification that have the least environmental impacts.

In chapter 2 I developed the Framework for the Evaluation of Ecosystem Service Substitution in Agro-ecosystems (FEESSA). This provides a basis for the evaluation of environmental trade-offs associated with substitutions for ES in BPS. The logic underpinning FEESSA is that in managed agro-ecosystems such as BPS the requirements for production are supplied by ES and external inputs. To intensify production BPS can be managed so that ES are replaced through the use of external inputs and other technology. Alternatively, it may also be possible to increase the benefits from ES and/or utilize them more efficiently as production systems intensify. FEESSA provides a method to explore the environmental impacts of these various alternatives so that optimal decisions can be identified.

In chapter 3 I employed LCA to quantify the life cycle environmental impacts associated with a novel aquaculture technology employed on the west coast of Canada. This technology represents an alternative to net-pen systems (commonly employed for salmon production) that has potential to reduce local environmental impacts. By using LCA to identify environmental hotspots it was possible to explore avenues for improving the environmental performance of the system. In agreement with previous assessments of aquaculture systems, the results of this LCA indicated that the provision of feed was a major driver of environmental impacts (Ayer & Tyedmers, 2009; Jerbi et al., 2012; Pelletier et al., 2009). This is because feed use was relatively high and relies on ingredients from impactful activities such as capture fisheries, agriculture, and poultry farming. Two strategies for reducing impacts from feed use are to: 1) formulate feeds using less impactful ingredients and 2) improve the efficiency at which feed is transformed into fish biomass (feed conversion ratio: FCR). While research has suggested that reducing impactful types of feed ingredients (e.g. fisheries ingredients) in favour of less impactful ones (e.g. crop ingredients) can improve environmental performance (Papatryphon et al., 2004), results of this study suggest that it is actually more important to focus on specific ingredients (e.g. hake meal, wheat, menhaden oil). This is because the individual context within which ingredients are produced is often more important than the type (Boissy et al., 2011). For example, in other feeds that have used less marine-derived ingredients, the environmental impacts of several impact categories (e.g. global warming potential, acidification potential) are actually more impactful then the feed used here, which was 80% marine-derived (see feed used in Scottish aquaculture from Pelletier et al., 2009). To improve feeding conversion efficiency both improved nutritional management and improved husbandry techniques in general should be targeted.

Nutritional management (diet formulation, feed digestibility) has been found in the past to significantly reduce the use of feeds and emission of wastes (Bureau & Hua, 2010). Improved husbandry, on the other hand, can be pursued as managers and operators become more familiar with the idiosyncrasies of the nascent SWAS technology.

Other subsystems identified as areas of importance included the production of juveniles (smolts) and on-site energy use. The smolt production system was unusual in that electricity production was done on-site using diesel generators. As a result, the production of juveniles was more impactful than in other aquaculture systems modelled previously. Regarding energy use, while not an overwhelmingly large contributor to life cycle impacts of the SWAS in its current application; scenario analysis indicated that if the SWAS was sited in a jurisdiction with higher proportion of fossil fuel powered electricity (e.g. Nova Scotia) then it would result in substantially higher impacts, particularly in terms of global warming and terrestrial acidification potential. As a result, siting future SWAS technologies in locations with access to an electrical grid with a low proportion of impactful primary energy sources is an important consideration. Additionally, further development of the SWAS technology that reduces energy use, particularly related to pumping and oxygen generation would help in reduction of life cycle impacts.

In addition to providing the opportunity to assess environmental hot-spots, the LCA of the SWAS can be used as a baseline upon which future iterations of the project can be compared. Unfortunately, at the current time it was not possible to conclude whether the life cycle environmental impacts of this technology are favourable when compared with other aquaculture systems (e.g. RAS or net-pens). This is because of the limited data available (only one incomplete production cycle) and the considerable uncertainty associated with its performance, as demonstrated by the uncertainty analysis. For this reason, comparison of the SWAS to other systems was limited to a superficial analysis (Appendix C). To conduct a more direct comparison of these systems it is necessary to gather more data through further research.

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In chapter 4, the insight gained from development of FEESSA was applied to the SWAS. This was an interesting opportunity because using FEESSA as a lens to explore the SWAS provided an opportunity to link the two main contributions of this thesis. Moreover, much of the infrastructure and energy use of the SWAS was associated with performing functions that might otherwise be delivered by ecosystem services (e.g. provision of fresh oxygenated water and removal of wastes); a demonstration of the fundamental premise of FEESSA. These technological interventions were intended to improve the environmental performance of the SWAS but it was apparent that there were environmental trade-offs associated with them. To quantify some of these trade-offs, LCA was used to assess the life cycle environmental impacts associated with the substitution of waste assimilation by the local environment with waste capture and treatment (WC) technology. The results of the LCA of the SWAS indicate that using the WC system described in chapter 4 to capture solid wastes increases the life cycle environmental impacts for three out of four impact categories considered, with increases to global warming potential and acidification potential being particularly large. Sensitivity and scenario analysis demonstrate that these results are variable and the source of energy, size of generator and waste capture efficiency are highly important in determining environmental impacts. While only a limited assessment of a particular type of environmental impacts, this research demonstrates that there are significant downsides to using this type of waste capture technology to mitigate the ecological impacts associated with nutrient emissions from aquaculture.

5.1.1 Limitations of Research

The completion of this thesis involved two independent yet complementary modes of investigation into the environmental impacts of BPS: the development of a conceptual framework linking BPS with ES and an LCA of an aquaculture system. These were linked together by using the aquaculture system as a case study for exploring the insights gained from the conceptual framework. In many ways this was a unique opportunity and resulted in a potentially important contribution to our academic conceptualization of BPS. On the other hand, many of the insights contained herein are reactions to observations made throughout rather than the conclusions of pre-meditated research design. The upshot is that although the system of study (the SWAS) and the methodology employed (LCA) played a significant role in the development of the direction of research, they were not specifically chosen for their suitability to exploring the conceptual framework. It is therefore possible that other examples of BPS and methods may provide more elegant demonstrations (or rebuttals) of the conclusions drawn herein. For this reason, it is of critical importance that these ideas be tested in multiple contexts and the validity of FEESSA explored.

5.1.2 Framework for the Evaluation of Ecosystem Service Substitution in Agro-Ecosystems

The purpose of the conceptual framework developed in chapter 2 is to describe the relationship between ES, BPS and environmental impacts and how it might be quantified/ integrated into decisions regarding the intensification of BPS. This is beneficial because it allows the role of ES in supporting BPS to be described and interpreted in relation to use of external inputs. This can be used to assess individual systems (e.g. what is driving the use of external inputs in system X?), compare different systems (e.g. why is system X more impactful than system Y?), and provide insight which can be used to improve design (e.g. how can we incorporate the role of ES in BPS X so that impacts are reduced?).

FEESSA does not provide a set of rules or causal relationships between substitution for ES and environmental impacts. This is because agro-ecosystems are complex and these relationships are dependent on the context within which ES are delivered and the technologies used to substitute them. Instead, FEESSA represents a conceptual platform from which BPS can be analyzed and upon which future research can be built. Indeed, the concept of ES substitution can be applied more broadly than just to intensification of BPS. It can be used to compare different BPS that are not intensifying or even to any situation where it is possible to substitute an ecosystem service using technology and vice-versa. Here, I have limited this discussion to intensification of BPS because it is here that its application appears most relevant. Ideally, FEESSA will be applied to other situations where it provides meaning and insight in the future. Additionally, it would be very interesting to explore under what conditions the substitution for ES is possible, the environmental impacts of that substitution, and how utilizing different technology to achieve the substitution influences those impacts.

The strength of FEESSA lies in its ability to include some of the benefits provided to BPS from ES in a way that is not possible otherwise. However, in as much as it may provide important information to decision-makers, there are many other important aspects of a holistic decision process that are not accounted for in FEESSA (e.g. economic and social). As such, it can be thought of as one of many tools (e.g. Finnveden & Moberg, 2005; Ness, Urbel-Piirsalu, Anderberg, & Olsson, 2007) which can be used to assess development and should not be viewed as a replacement for those.

5.1.3 Life Cycle Assessment of the Solid Wall Aquaculture System

LCA methodology provides a systematic approach to assessing environmental impacts that is particularly relevant for BPS such as aquaculture that are reliant on material and energy inputs from a global supply chain (Henriksson et al., 2011; Pelletier & Tyedmers, 2008). It is an excellent tool for informing the decision making process of system managers by identifying environmental hotspots and providing a basis for comparison. However, it provides only certain pieces of a much larger puzzle. This includes a limited suite of environmental impacts associated with resource use and emissions. For this reason, it should be used in combination with other information (e.g. proximate ecological impacts, economic, social) for decision making (Baumann & Tillman, 2004). While development of novel impact categories for improving the scope of impacts covered by LCA (e.g. social impacts, local ecological impacts) is ongoing (see Finnveden et al., 2009; Ford et al., 2012), they have not yet been sufficiently developed or gained widespread acceptance and are therefore not included in this thesis (Finnveden et al., 2009; Pelletier et al., 2006).

A significant issue in LCA practice and particularly for the LCA of the SWAS is uncertainty. Uncertainty is generally caused by poor data quality (e.g. lack of foreground data, errors in measurement, inappropriate use of data), variability (e.g. attempting to quantify a variable phenomenon such as weather), and mis-characterization of relationships (e.g. incorrect models) (Finnveden et al., 2009). These typical sources of uncertainty are exacerbated in the LCA of the SWAS because data on only one incomplete production cycle were available. Thus results are based on a production cycle that is not representative of intended production. Moreover, the SWAS is a nascent technology and is subject to a number of problems that might be managed more effectively in the future, leading to more efficient production. This means that the results obtained here are likely biased towards larger impact than what might be expected in the future. For these reasons, substantial effort was undertaken to characterize a more realistic production cycle by constructing a model of intended production and the impact of uncertainties in several key parameters using sensitivity analysis and Monte Carlo simulation. These showed a considerable amount of uncertainty in the life cycle impacts of SWAS.

A common use of LCA is to make comparisons between alternative systems so that environmentally preferable ones can be identified (e.g. Ayer & Tyedmers, 2009; Pelletier &

Tyedmers, 2007; Pelletier et al., 2009). While a presentation of results of other aquaculture LCAs was done, a direct comparison was purposefully avoided. This is in large part because of the uncertainty associated with current results. However, it was also avoided because of the difficulty associated with direct comparisons of aquaculture LCA results in general. These systems all exist within unique environmental, economic and legislative contexts (Samuel-Fitwi et al., 2012). This means that it is very difficult to attribute differences in life cycle impact to a particular technology or production type. Unless these uncertainties and variables can be addressed the results of the LCA of the SWAS are better utilized as a baseline for comparison of future iterations of the system.

5.1.4 Study System

Much of the insight for the FEESSA came out of the LCA work with the SWAS and previous work on other aquaculture systems (e.g. Ayer & Tyedmers, 2009) where it appeared that there was a causal relationship between the degree of linkage with a functioning ecosystem and life cycle environmental impacts. It was fortunate that the SWAS waste capture and treatment system provided an opportunity to study environmental tradeoffs associated with ES substitution but it was not chosen with this research in mind. As a result, the SWAS is not necessarily the optimal system for studying this phenomenon. In the future a more purposeful research design could be used. This would allow a more experimental approach that could test an appropriate hypothesis, rather than describing a potential relationship. Due to practical constraints, this approach was not taken here.

In addition to demonstrating the applicability of the FEESSA, one of the key motivations for investigating the tradeoffs associated with employing a WC system for the SWAS was to actually assess the WC technology. In other words, I wanted to determine whether it was environmentally preferable to use WC or to simply pump the waste into the receiving environment and allow the natural assimilative capacity of the ecosystem to recycle the nutrients where the conditions are such that this is possible. Results showed that for three out of four impact categories investigated it is preferable not to employ WC technology; however, as described above (section 5.1.3) LCA is a limited assessment. To make an informed decision, the benefits associated with the removal of waste on the local environment would need to be quantified. This analysis should therefore be used to inform a broader assessment process before any decision regarding the utility of WC technology in the SWAS is made.

5.2 Conclusion

Continued intensification of BPS throughout the 21st century is expected to exacerbate already significant environmental issues of global, regional and local concern. It is therefore very important to find ways of mitigating these impacts so that human activity and prosperity can be sustained. This effort will require innovation and creativity so that new and better techniques of 'sustainable intensification' can be developed. Novel conceptualizations, such as is provided by FEESSA, can be used to spur this innovation by offering new perspectives. In order to judge progress and compare these techniques, sustainability tools, such as LCA can and should be used. While the challenge is enormous, it must be tackled one piece at a time. This thesis represents one small piece of this effort where quantification of impacts was limited to a specific aquaculture production technology. It is hoped that it provides utility but it must be refined through critical examination and complemented by further research. To this end, further exploration of the underlying concepts of FEESSA for variety of BPS and continued use of LCA to quantify environmental impacts is recommended. Moreover, additional research on BPS using LCA and other sustainability tools are required to provide the data necessary for making informed and holistic decisions.

Appendix A: Construction of the Model of the Intended Production Cycle (IPC)

To build the LCI for the IPC, the unit processes were categorized in two ways: foreground and background. Background processes included descriptive system characteristics which remained unchanged regardless of the amount of production. Examples of background processes are: feed formulations, types of agricultural and fishery related inputs, transportation modes, transportation routes, material production/processing inputs, electricity generation, electric grid mixes etc... Foreground processes, or those important for on-site production were categorized in three ways: static, biomass dependant, and temporally dependant. Those inputs that were static remained the same in both the APC and the IPC. This included tank and anchoring materials, inputs for production of smolts, equipment used, and other one-time inputs that were sufficient for production regardless of timeline or biomass. In both the APC and IPC, tank infrastructure was assigned a 20 year life-expectancy. This represents the intended life-expectancy. In reality, the tank infrastructure was compromised after 13.5 months and all environmental costs are thus attributable to the APC. However, to facilitate comparison without the distortion of the impacts of infrastructure, a 20 year life-expectancy was used. Life-expectancy was modeled in consultation with experts, based on estimates for a second generation tank that is currently under construction.

Biomass dependant inputs were those that increased as the fish biomass increased, for example the amount of feed, and mass of goods for transport. Temporally dependant inputs included those inputs which were used at a constant rate (or roughly constant) throughout the production cycle and therefore depended on the total time that fish were held in the tanks. This included total electricity use (for pumping water and oxygen production), total fuel use, mortality rate, and escape rate.

Background processes and static inputs were the same for the LCI of both the SWAS and IPC. To determine the biomass and temporally dependant inputs it was necessary to determine the total biomass at the intended moment of harvest and the total time to reach that biomass. In order to accomplish this it was necessary to calculate the Thermal unit Growth Coefficient (TGC) of the system using equation (1) based on known characteristics of the SWAS up until the fish were harvested (Iwama & Arthur, 1981; Jobling, 2003).

$$TGC_1 = \frac{\left(Wh^{\frac{1}{3}}\right)-\left(Wi^{\frac{1}{3}}\right)}{[(T_1)(days_1)]} * 1000 \qquad (1)$$

TGC_1 = Thermal unit growth coefficient
W*h* = Average harvest weight of individual fish
W*i* = Average initial weight of individual fish
T_1 = Average temperature
$days_1$ = number of days between initial weight and harvested weight

Once this is known, the equation can be rearranged to predict future growth rates and total time to reach harvest size (2).

$$days_2 = \frac{\left(Wf^{\frac{1}{3}}\right) - \left(Wh^{\frac{1}{3}}\right)}{\sum[(T_2)(TGC_1)]} \tag{2}$$

$days_2$ = number of days between harvest weight and final weight
Wf = Final weight at intended harvest
T_2 = Average temperature between actual harvest and predicted harvest

Using the value for total time to reach harvest from stocking to intended harvest (3), the total number of mortalities at the time of predicted harvest was calculated (4).

$$days_h = days_1 + days_2 \tag{3}$$

$days_h$ = Total number of days between stocking and intended harvest

$$m_t = m_1 + [mr * n_1 * days_2] \tag{4}$$

m_t = predicted total number of mortalities between stocking and intended harvest
m_1 = number of mortalities between stocking and actual harvest
mr = average individual daily mortality rate (see equation 8 below)
n_1 = estimated number of remaining individuals at actual harvest

Next, biomass at time of predicted harvest was calculated (5).

$$Bf = Wf_2 * (N - m_t) \tag{5}$$

Bf = Predicted total biomass at intended harvest
N = total number of stocked individuals

Finally, using the values for total biomass (Bf) and total time to intended harvest ($days_h$), the biomass dependant and temporally dependant inputs of the IPC were calculated. For example predicted feed use (6) and predicted electricity use (7). Economic FCR of the IPC was calculated based on the estimated eFCR of the APC assuming that no fish had escaped and that escaped fish were the same average weight as those harvested.

$$Feed = eFCR * Bf \tag{6}$$

Feed = predicted feed use at predicted harvest
eFCR = economic feed conversion ratio of SWAS

$$E = E_{avg} * days_h \quad (7)$$

E = predicted electricity use at predicted harvest
E_{avg} = predicted feed use at predicted harvest

Other important assumptions used to build the IPC include:

- Average mortality rate (mr) of 0.0392% was used (calculated as daily mortality rate without transportation losses)
- Monthly electricity use = 1323.2 kWh/day (mean electricity use during production)

Average individual mortality rate (mr) was calculated as the sum of mortality, less the losses that occurred during the first week of production, divided by the total number of days during the APC less one week, all which is divided by the total number of fish less transportation losses (8). These were not included in calculation of average mortality because losses during the first week were significantly higher than during other periods. The high level of initial mortality is attributed to the stress associated with transportation and stocking.

$$mr = \frac{(m_1 - m_2)/(days_h - 7)}{(N - m_2)} \quad (8)$$

m_2 = Total number of mortalities during first week of production (transportation related losses)

Appendix B: Supplemental Inventory Data

Table B1. Life cycle inventory data for production of 1000 kg of smolts (approximately 20,000 smolts)*.

Inventory	Unit	Total
Smolts	kg	1,000
	#	20,000
Feed	kg	1,500
Diesel	l	3,800

*Mr. Bruce Kenny, personal communication, January 28, 2013

Table B2. Life cycle inventory data for construction of the solid-wall aquaculture system (SWAS).

Inventory	Unit	Total
Fibre-reinforced plastic (FRP)	kg	38,071
Low-density polyethylene (LDPE)	kg	7,557
High-density polyethylene (HDPE)	kg	600
High modulus polyethylene (HMPE)	kg	754
Nylon	kg	406
Vinyl, fabric reinforced	kg	60
Steel, galvanized	kg	18,409
Aluminum	kg	2,733
Electricity	kWh	289,570

* Ms. Sheena Burns, personal communication, May 15, 2013

Table B3. Fuel use required for the capture of 1 tonne of fisheries used as inputs for salmon feed.

Inventory	Unit	Total
Diesel per tonne of Hake (trawler, US)[1]	l	53.4
Diesel per tonne of Hake (trawler, Canada)[2]	l	46.0
Diesel per tonne of Herring (seine net, Canada)[2]	l	70.6
Diesel per tonne of Herring (gillnet, Canada)[2]	l	73.4

[1] US hake catch fuel use data was obtained from Aleutian Spray Fisheries, a commercial fishing and processing organization (Mr. Craig Cross, personal communication, March 1, 2013).
[2] Fuel use from Canadian hake and herring fisheries was elicited by gear type from Goldseal, a commercial fishing, processing, and distributing organization (Rob Morley, personal communication, January 21, 2013).

Table B4. Life cycle inventory data for processing and reduction of fisheries derived feed ingredients.

Inventory	Unit	Total
Processing		
Inputs		
Herring	kg	1,000
Hake	kg	1,000
Outputs		
Herring roe	kg	125
Herring by-product	kg	875
Hake fillet	kg	625
Hake by-product	kg	375
Reduction		
Inputs		
Herring by-product	kg	100
Hake by-product	kg	900
Natural gas[1]	MJ	2325
Biodiesel[1]	MJ	179.9
Outputs		
Herring/hake meal[2]	kg	190
Herring/hake oil[3]	kg	50

[1]National Renderers Association (nationalrenderers.org)
[2]Yield of 19% (personal communication, Grant Sarr, February 4, 2013)
[3]Yield of 5% (personal communication, Grant Sarr, February 4, 2013)

Table B5. Life cycle inventory data for one acre of wheat production from a model wheat farm in Manitoba[1].

Inventory	Unit	Total
System characteristics		
Area	acre	1
Wheat yield	kg	1,224.7
Inputs		
Water	l	46.4
Ammonium anhydride	kg	31.7
Urea	kg	20.7
Monoammonium phosphate	kg	16.5
Potash	kg	6.8
Dividend see treatment	kg	0.19
Infinity herbicide	kg	0.3
Tilt fungicide	kg	0.2
Glyphosate	kg	1.0
Diesel	MJ	831.8
Electricity	MJ	13.23
Outputs		
Nitrogen dioxide	kg	0.468
Ammonia	kg	0.378
Nitrogen	kg	3.37

[1]Life cycle inventory data from LCA of Wheat in Manitoba (http://www.gov.mb.ca/agriculture/soilwater/climate/fcc03s01.html#wheat)

Table B6. Total electrical production by source for Nova Scotia, British Columbia and Canada (MWh)*.

Electricity source	NS 2012	BC 2012	Canada 2010
Petroleum	1,205,711	0	19,325,261
Natural gas	1,743,393	5,300,000	58,109,457
Coal	6,750,896	0	47,393,283
Nuclear	0	0	85,300,000
Hydro	1,100,000	61,200,000	363,500,000
Wind	300,000	38,970	1,600,000
Tidal	55,000	0	55,000
Biomass	602,856	3,025,000	972,000
Total	11,757,856	69,563,970	576,255,000
Estimated distribution losses (%)	2	15	7.93

*Centre for Energy http://www.centreforenergy.com/

Table B7. Nutrient mass-balance modelling for salmon grown in the actual production cycle (APC) and intended production cycle (IPC) in the solid-wall aquaculture system (SWAS).

	Feed distributed	Unconsumed food[1]	Feed ingested[2]	Fish weight gain[3]	Faecal loss[4]	Total solid emission[4]	Total dissolved emissions[4]	Total emissions[4]
APC								
Total amount (kg)	109,628	3,289	106,339	83,051				
Solids (kg dry matter)	98,665[a]	2,960	95,705	24,168[c]	17,100	20,059		20,059
P (kg)	1,261	38[b]	1,223	332	611	649	279	929
N (kg)	7,209	216[b]	6,993	2,648	559	776	3,785	4,561
IPC								
Total amount (kg)	234,608	7,038	227,570	188,571				
Solids (kg dry matter)[a]	211,148[a]	6,334	204,813	54,874[c]	36,594	42,928		42,928
P (kg)	2,698	81[b]	2,617	754	1,309	1,389	554	1,944
N (kg)	15,428	463[b]	14,965	6,013	1,197	1,660	7,755	9,415

[1]Assumed a 3% wastage of feed (personal communication, Mr. Robert Walker, January, 2013)

[2]Feed ingested = Feed distributed-unconsumed food

[3]Fish weight gain = (Mass of harvest + mass of mortalities)-mass of stocked fish

[4]Based on digestibility coefficients and mass-balance model parameters from Aubin et al. (2011)

[a]10% moisture content of feed

[b]Calculated from feed ingredients and nutritional tables (Sauvant, Perez, & Tran, 2004)

[c]Moisture content of salmonid flesh 70.9% (Dumas, de Lange, France, & Bureau, 2007)

Appendix C: Comparison of the Life Cycle Impacts for the Intended Production Cycle of the Solid-Wall Aquaculture System With Other Aquaculture Systems

Table C1. Published life cycle impacts for selected species and aquaculture production systems. SWAS – Solid-wall aquaculture system; RAS – Recirculating aquaculture system

Source	Species	System	Method	FCR	GWP kg CO2 eq	AP kg SO2 eq	MEP kg N eq	CEU MJ eq	BRU kg C
Chapter 3 McGrath & Tyedmers, 2014	Chinook	SWAS	Recipe	1.37	3,025	18.9	56.2	67,064	1,344,038
Ayer & Tyedmers, 2009	Atlantic	Net Pen	CML	1.3	2,073	18.0	35.3	26,900	n/a
	Atlantic	Marine bag		1.17	1,900	15.8	31.8	32,800	n/a
	Atlantic	Raceway		1.165	2,770	16.6	29.9	97,900	n/a
	Arctic charr	RAS		1.448	28,200	255.0	20.1	353,000	n/a
Pelletier & Tyedmers 2009	Atlantic	Net-pen - Norway	CML	1.103	1,790	17.1	41.0	26,200	111,100
	Atlantic	Net-pen - UK		1.331	3,270	29.7	62.7	47,900	137,200
	Atlantic	Net-pen - Canada		1.313	2,370	28.1	74.9	31,200	18,400
	Atlantic	Net-pen - Chile		1.493	2,300	20.4	51.3	33,200	56,600
Aubin et al 2009	Trout	Raceway	CML	1.21	2,753	19.2	65.9	78,229	62,200
	Sea bass	Net Pen		1.77	3,601	25.3	108.9	54,656	71,400
	Turbot	RAS		1.23	6,017	48.3	77.0	290,986	60,900
Jerbi et al 2012	Sea bass	Traditional raceway	CML	1.81	11,087	54.0	180.0	175,000	76,000
	Sea bass	Cascade raceway		2.06	17,449	70.0	215.0	280,000	64,000
Roque d'Orbcastel et al 2009	Trout	Raceway	CML	1.1	2,015	13.4	28.5	34,869	27,968
	Trout	RAS		0.8	1,602	10.5	17.8	57,659	21,432
Pelletier & Tyedmers 2010	Tilapia	Lake	CML	1.7	1,520	20.2	47.8	18,200	2,760
	Tilapia	Pond		1.65	2,100	23.8	45.7	26,500	2,700

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