LIFE CYCLE MEASURES OF BIOPHYSICAL SUSTAINABILITY IN FEED PRODUCTION FOR CONVENTIONAL AND ORGANIC SALMON AQUACULTURE IN THE NORTHEAST PACIFIC

by

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We have at our disposal the human and material resources to achieve sustainable development, not as an abstract concept but as a concrete reality. Our efforts must be linked to the development of cleaner and more resource efficient technologies for a life cycle economy.

- Malmö Declaration, 1st Global Ministerial Environment Forum, 2000

Conservation is a state of harmony between men and land. By land is meant all things on, over, or in the earth. Harmony with the land is like harmony with a friend; you cannot cherish his right hand and chop off his left. That is to say, you cannot love game and hate predators; you cannot conserve the waters and waste the ranges; you cannot build the forest and mine the farm. The land is one organism.

- Aldo Leopold

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Abstract

The cumulative impacts of industrial society's high material/energy throughput compromise the stability of planetary biogeochemical cycles. Life Cycle Assessment (LCA) provides a framework for quantifying how industrial products/processes contribute to these macroscale human/environment interactions. This thesis explores the strengths and weaknesses of using LCA to understand and improve sustainability in seafood production, and undertakes a Life Cycle Assessment of alternative feed production scenarios for conventional and organic salmon aquaculture. It was found that the environmental impacts of feed production could be reduced by decreasing the fraction of animal-derived ingredients employed, and that current standards for organic salmon aquaculture actually compromise biophysical sustainability in this industry. Moreover, it is proposed that achieving sustainability in aquaculture (and industrial society, generally) requires recognition of the finite nature of resources and the limited capacity of ecosystems to absorb waste and respond to change. Such recognition highlights the desirability of developing maximally eco-efficient production technologies.

List of Abbreviations and Symbols

AP	Acidification Potential
BC	British Columbia
BRU	Biotic Resource Use
c	carbon
C	conventional salmon feed
CDN	Canadian
CO_2	carbon dioxide
EP	Eutrophication Potential
EROI	Energy Return on Investment
EU	Energy Use
FAO	(United Nations) Food and Agricultural Organization
FCR	feed conversion ratio
g	gram
GWP	Global Warming Potential
h	hour
ha	hectare
hp	horsepower
HT	Human Toxicity
IFOAM	International Federation of Organic Agriculture Movements
ISO	International Organization for Standardization
kg	kilogram
kJ	kilojoules
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Lice Cycle Impact Assessment
MAEP	Marine Aquatic Ecotoxicity Potential
MJ	megajoules
NAS	National Academy of Science
NO_x	nitrous oxide
NPP	Net Primary Production

OA	salmon feed containing organic crop ingredients, reduction fisheries meals and oils, and poultry by-product meal
OBP	salmon feed containing organic crop ingredients and by-product meals and oils from the British Columbia herring roe fishery
OD	Ozone Depletion
ORF	salmon feed containing organic crop ingredients/no poultry by-product meal/25% of reduction fisheries fishmeal replaced with soy meal/100% of the reduction fisheries fish oil replaced with canola oil
PO	Photochemical Oxidation
PO_4	phosphate
SEI	Socio-economic Impact
SELCA	Social and Environmental Life Cycle Assessment
SETAC	Society for Environmental Toxicology and Chemistry
SIK	Swedish Institute for Food and Biotechnology
SO_2	sulphur dioxide
SRES	School for Resource and Environmental Studies
TE	Terrestrial Ecotoxicity
US	United States
USD	United States dollar
1,4-DCB	1,4-dichlorobenzene

Acknowledgements

As with any academic endeavor, this work builds on the contributions of countless others. I was fortunate to carry out this research amidst a community of practitioners whose labours and generous wisdom greatly facilitated and expedited my work. In particular, I owe thanks to my supervisor, Dr. Peter Tyedmers, for providing such excellent opportunities for academic development, useful contacts and data sources, and for the many thoughtful and provocative intellectual exchanges we have shared. I also wish to acknowledge the work of Nicole Arsenault, whose data, literature collection, and insights into LCA theory and practice made invaluable contributions to my research. Finally, I thank Professor Raymond Côté and Dr. Anne Kapuscinski for their clear and constructive input to this document.

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Chapter 1: INTRODUCTION

We live at an interesting time in human evolution. Only recently have we become a truly globalized society whose cumulative industrial activities are of sufficient magnitude to overshoot the homeostatic capacity of planetary biogeochemical cycles (Bowman 1998, Rabouille *et al.* 2001, Klee and Graedel 2004). Yet the norms upon which we organize ourselves are woefully inadequate to respond to this overshoot. Our economies are founded on premises that were historically workable so long as our consumptive demands and waste flows were within the assimilatory capacity of host ecosystems. This is no longer the case. We cannot continue to pretend that, on a planet of finite size, natural resources are inexhaustible and waste does not matter. The emergence of global-scale environmental issues including the biomagnification of persistent organic pollutants across trophic levels (Colburn and Thayer 2000; Fisk *et al.* 2003), depletion of stratospheric ozone (Crutzen 1992; Madronich *et al.* 1995), acid precipitation (Likens *et al.* 1996; Bouwman *et al.* 2002), climate change (Hughes 2000; Robertson *et al.* 2000; Levitus *et al.* 2001; Walther *et al.* 2002) and biodiversity loss (Nee and May 1997; Duffy 2003; Olden *et al.* 2004) speaks to the unavoidable necessity of restructuring human activities with respect to biophysical limits. A central challenge of sustainable development, then, is to advance the theoretical foundations and practical tools necessary for a biophysically informed political economy.

The purpose of this thesis was to tackle one tiny piece of this complex puzzle. My approach was two-fold. First, I aimed to become sufficiently conversant with Life Cycle Assessment, a biophysical accounting tool used to quantify the environmental impacts of the inputs and emissions associated with products or processes, to advance a methodological contribution. Second, I used this tool to compare the environmental performance *(i.e.* the relative magnitude of contributions made by different production systems to a specified suite of environmental problems) of competing industrial food production scenarios in order to both identify specific environmental performance improvement recommendations and to speak, more broadly, to the relative sustainability of alternative food production technologies.

1.1 Overview of Life Cycle Assessment

1.1.1 Origins and Evolution of Life Cycle Assessment

Life Cycle Assessment methodology had its genesis in the 1960s as novel approaches were sought to study packaging and waste management issues. A packaging study carried out by Hunt and Franklin in 1969 on behalf of the Coca Cola Company, which compared resource demands associated with use of glass and aluminum containers, is often credited as the first Life Cycle Assessment (Baumann and Tillman 2004). Fuel cycle studies in both Britain and the US, spurred on by the oil price shocks of the early 1970s, created further impetus for methodological development (Baumann and Tillman 2004).

The LCA concept remained somewhat obscure outside of packaging industry and waste-management circles throughout the 1970s and 80s. During the late 80s, renewed debate around environmental issues related to waste management raised broader interest in Life Cycle Assessment as a management tool. At this time, the divergent results and methodologies of several high profile LCA studies of packaging (Christiansen *et al.* 1990; Tillman *et al.* 1992) generated considerable controversy and gave rise to a flurry of methodological discussion and development (Baumann and Tillman 2004).

The 1990s saw considerable effort invested in harmonizing and expanding LCA methodology. Following a series of international workshops, the Society for Environmental Toxicology and Chemistry (Consoli *et al.* 1993) published the first Code of Practice for LCA studies. Between 1997 and 2002, the International Organization for Standardization (ISO) also published several methodological standards (ISO 2003). At present, Life Cycle Assessment is gaining increasing currency as a decision-support tool – both within industry for cleaner production and product stewardship (Environmental Protection Agency 1995) and among policy makers as a basis for sustainable development (Wrisberg *et al.* 1997).

Although the vast majority of early life cycle studies involved the products/processes of manufacturing industries, Life Cycle Assessment methodology is now used frequently for other purposes (Baumann and Tillman 2004), such as criteria development for green marketing (Llorenc *et al.* 2002; Baldo *et al.* 2002), transportation studies (Fet *et al.* 2000;

MacLean and Lave 2003), waste treatment (Tukker 1999; Lundin 2003) and product design (Baumann and Tillman 2004). Much effort has also been invested in adapting the LCA framework for evaluating food products and food production systems (Andersson *et al.* 1994; Andersson 2000). These efforts have been particularly productive in northern Europe, where numerous food LCAs have been conducted (for example, see Mattsson 1999; Hogass Eide 2002; Jones 2002; Berlin 2002; or Stern *et al.* 2005).

Although most LCAs of food production have focused on agricultural systems, comparable work for fisheries and aquaculture production systems has increased substantially in recent years (Mattsson and Ziegler 2004). A review of the published literature indicates that a number of methodological challenges remain, including the development of additional appropriate impact categories for evaluating fisheries and aquaculture production systems, and the further standardization of allocation decision-making and reporting procedures.

1.1.2 Life Cycle Assessment Methodology

Life Cycle Assessment may be conducted with varying degrees of detail. *Detailed* LCAs are generally favored, although *conceptual* and *simplified* LCAs are also common. The distinction between the three is best represented as a continuum of increasing detail (Jensen *et al.* 1999). The simplest is the conceptual LCA, which uses a limited, qualitative inventory as the basis for an impact assessment. A conceptual LCA might be used to answer questions such as "Is the product markedly different from competing products" or "Does the product have obvious benefits or shortcoming with regards to specific environmental issues?" The results can be presented using simple scoring systems or qualitative statements, but would not support decisions requiring quantitative analysis (Jensen *et al.* 1999).

A simplified LCA, although similar to a detailed LCA, focuses on specific areas of interest in the product/process life cycle, and may use generic data. This requires less time, but may compromise the accuracy and reliability of the results (Jensen *et al.* 1999). Simplified LCA's are commonly used to identify environmental "hotspots" for ecolabeling criteria, or to identify which life cycle stages contribute to specific areas of environmental concern.

A methodology standardized by the International Organization for Standardization (ISO 2006) is now available for conducting detailed Life Cycle Assessments and has been widely adopted by the LCA community. According to ISO standards (Guinee *et al.* 2001), a full LCA must have four components: goal definition and scoping, life cycle inventory (LCI), impact assessment, and improvement assessment. The first stage of a Life Cycle Assessment is Goal Definition and Scoping. Here, the purpose of the study is clearly defined, research expectations are articulated, and the study boundaries, including any assumptions, are delineated. The nature of the system boundaries chosen will, in large part, determine the final outcome. These boundaries must therefore be chosen according to a clearly defined rationale, and must include not only all system operations that make significant contributions to the life cycle, but spatial and temporal limits as well. Goal Definition and Scoping also requires definition of the functional unit, which is the quantity of the product under study to which all product/process flows, and their associated impacts, will be related (Guinee *et al.* 2001).

The second LCA stage is the Life Cycle Inventory (LCI), which involves identifying all process steps and collecting data relating to the material and energetic inputs and emissions associated with each life cycle stage. Although a variety of LCI databases of material inventories are currently available for common construction materials and energy production, government statistical reports, industry association reports, private companies and utilities are also typical data sources (Baumann and Tillman 2004).

The environmental impacts associated with specific life cycle activities are derived during the Impact Assessment Phase (Pennington *et al.* 2004). This stage typically involves the use of dedicated LCA software programs to translate the Life Cycle Inventory data into actual, quantitative contributions to a suite of specified environmental impact categories based on peer-reviewed impact assessment models. According to ISO standards, the LCIA framework includes both mandatory and optional elements.

The first mandatory element is the selection of impact categories, which are classes representing specific environmental issues of concern such as global warming, acid precipitation, resource depletion, etc. (see Table 2.1). The second mandatory element is choosing category indicators and characterization models. Category indicators are reference species used to represent all contributions made by various substances to a specific impact category. For example, carbon dioxide is commonly used as the category indicator for the "global warming" impact category, and all other substances that may contribute to global warming are expressed in terms of carbon dioxide equivalents. Characterization models are the peer-reviewed impact assessment methodologies used to translate LCI data into actual quantitative contributions to specific impact categories. For example, the Intergovernmental Panel on Climate Change model for calculating the global warming potential of greenhouse gas emissions is typically used as the characterization model for the global warming impact category. Classification is the next mandatory element of the Impact Assessment phase. It involves assigning the inputs and emissions tabulated in the Life Cycle Inventory to the appropriate impact categories. Characterization is the final mandatory element, and involves calculating the magnitude of potential impacts associated with each input/emission in the product/process life cycle for each relevant impact category. For example, in the case of a specific nitrous oxide emission, this would include calculating the magnitude of potential climate change and acid precipitation impacts (Consoli *et al.* 1993).

Non-mandatory elements of the Impact Assessment phase are normalization, grouping, weighting, and data quality analysis (Guinee *et al.* 2001). Normalization is a process to enhance the comparability of data between different impact categories by dividing indicator values by a reference quantity in order to convert differing units into a common, unitless format. A common approach to normalization is to divide the indicator values by estimates of total global contributions to each impact category, which allows an assessment of the relative "importance" of specific life cycle inputs and emissions. Grouping is a value-based process that involves sorting or ranking the impact categories on a nominal basis (for example, by relevance at global, regional or local scales) or according to a given hierarchy *(e.g.* high, medium or low-priority). Weighting is similarly subjective, and involves using selected weighting factors to convert the indicator results from different impact categories based on value choices regarding their relative importance. For example, if global warming impacts were deemed to be of greater importance than eutrophication impacts, the indicator results for global warming would be transformed using a higher weighting factor. Data quality analysis involves the use of additional techniques to better understand the significance, uncertainty and sensitivity of LCIA results in order to remove negligible LCI results, help distinguish whether or not significant differences are present, and guide the iterative LCIA process (Guinee *et al.* 2001). Sensitivity analysis, which measures the extent to which changes in the LCI results, characterization models, allocation decisions, etc. may influence the indicator results, is a frequently used data quality analysis technique (Guinee *et al.* 2001).

The final stage of a LCA study is the Improvement Assessment phase, which involves a systematic evaluation of the results of the Impact Assessment in order to determine which life cycle stages contribute disproportionately to specific environmental impact categories (Baumann and Tillman 2004). This information can be used to formulate environmental performance improvement recommendations and, more generally, to compare environmental performance between competing products and production technologies.

1.1.3 Allocation in Life Cycle Assessment

Many production systems are multi-functional in that they yield more than one product. For example, fish processing produces two co-product streams – filets for human consumption and processing trimmings. In order to evaluate the life cycle impacts of the filets, it would be necessary to allocate appropriate shares of the environmental impacts of the fishery and processing stages of the life cycle to the two co-products. In other words, what share of the environmental burdens should be apportioned to the filets and what share to the processing trimmings? Such allocation decisions are frequently necessary in Life Cycle Assessment research when several products are derived from the same process, but the environmental burdens are to be expressed in relation to only one of the products (Guinee *et al.* 2001). Despite the existence of international methodological standards, there is still considerable debate and inconsistency surrounding allocation decisions in LCA research (Azapagic and Clift 1999). According to ISO (2006), the preferred method is to avoid allocation by sub-dividing the process in order to isolate the component of interest, or to employ system expansion. The latter option involves modeling alternative production systems for each co-product, then crediting the associated impacts against the total impacts generated in the multi-product system in order to isolate the share of burdens attributable to the product under study. Since many processes cannot be subdivided and system expansion is rarely feasible either because alternative production systems for each product do not exist (for example, it would not be possible to model a production system that produces only fish processing trimmings) or because of time/data limitations, the next best option in the ISO hierarchy is to partition the burdens according to some underlying physical relationship or by mass, energy content, or relative economic value (Ekvall and Finnveden 2001).

For a comprehensive treatment of allocation problems and strategies in LCA research of seafood production systems see Ayer *et al.* (2006). A discussion of the allocation criterion employed in this thesis research can be found in Chapter Four.

1.1.4 Applications of Life Cycle Assessment

Traditionally, the most common use of Life Cycle Assessment has been product design and development (Baumann and Tillman 2004). Life cycle-based approaches to product design facilitate the kind of holistic, environmental perspectives necessary to developing environmentally superior goods and services for industrial society. The challenge is to identify where and how to best incorporate environmental considerations in the product development process.

A less common but important application of LCA is process design. This involves modeling a series of alternative production processes to determine which scenarios generate the most or least significant environmental impacts (Baumann and Tillman 2004). This is particularly effective for primary processing sectors, which extract raw materials and generate single-material products. It is somewhat less practical for manufacturers of complex products, who have little control over the upstream systems that provide the diverse material and energy inputs to their production systems.

Waste management and recycling concerns have been central to the evolution of LCA. Although packaging waste has been a recurring theme, LCA has also been applied to sewage treatment and the handling of sewage sludge, hazardous wastes, energy recovery, landfilling, treatment of wastewater, and the recycling and reuse of a wide variety of products. Generally, the central concern is to ascertain which waste management option minimizes environmental harms (Baumann and Tillman 2004).

Much effort has also been invested in applying the LCA framework to food products and food production systems (Andersson *et al.* 1994; Andersson 2000). The abundance of published LCA studies in the food sector suggests these efforts have proven successful, although the degree of methodological innovation achieved may be relatively insubstantial. Life Cycle Assessments in the food sector have included both product and process studies (Andersson *et al.* 1994; Andersson 2000). Sandars *et al.* (2003) used LCA to evaluate the environmental benefits of livestock manure management practices and technology. Brentrup *et al.* (2001, 2004) compared the impacts of several application regimes for nitrogen fertilizer use in sugar beet and winter wheat production systems. Nicol (2004) discussed life cycle thinking in the dairy industry. Product-specific studies have included Australian grains (Narayanaswamy *et al.* 1992) dessert apples (Jones 2002), swine (Stern *et al.* 2005), sugar cane (Ramjeawon 2004), bread (Andersson and Ohlsson 1999), milk (Hogass-Eide 2002) and cheese (Berlin 2002). Several researchers have also used LCA to compare the relative impacts of conventional and organic products and production processes (Mattsson 1999; Cederberg and Mattsson 2000). Although most LCAs of food production systems have focused on agricultural systems, comparable work for fisheries and aquaculture production systems has increased substantially in recent years (Mattsson and Ziegler 2004).

1.1.5 Life Cycle Assessment Research of Fisheries and Aquaculture

The application of Life Cycle Assessment to fisheries and aquaculture is a relatively recent phenomenon. To date, LCA researchers in fisheries and aquaculture have variously examined Finnish trout production (Seppälä *et al.* 2001), Danish cod products (Zeigler *et al.* 2003), French trout farms and feeds (Papatryphon 2003, 2004), Danish fish products (Thrane 2003, 2004), Spanish tuna fisheries (Hospido and Tyedmers 2005), Thai shrimp products (Mungkung 2005; Mungkung *et al.* 2006), and Norwegian cod and farmed salmon production (Ellingensen and Aanondsen 2006).

1.2 Sustainability in Salmon Aquaculture

Global seafood consumption has doubled since the early 1970's, and continues to grow. This demand is fueled by growing populations, income, and urban growth in the developing world, as well as the rising popularity of shrimp and carnivorous finfish in developed countries (Naylor and Burke 2005). With increasing demand for seafood products and concurrent declines in wild-capture fisheries (Watson and Pauly 2001; Myers and Worm 2003; Worm and Myers 2004), aquaculture has become the fastest growing food industry in the world (FAO 2004). Almost 50% of marine products are farmed (Figure 1.1) and the United Nations Food and Agricultural Organization (FAO) predicts a 70% increase in aquaculture production by the year 2030 (FAO 2004, 2006). Although aquaculture has contributed to economic growth in many regions, rapid and poorly regulated development has also generated adverse socio-economic and ecological consequences (Folke and Kautsky 1992; Primavera 1997; Stonich and Bailey 2000; Paez-Osuna 2001; Naylor *et al.* 1998, 2000; Muir 2005; Naylor and Burke 2005).

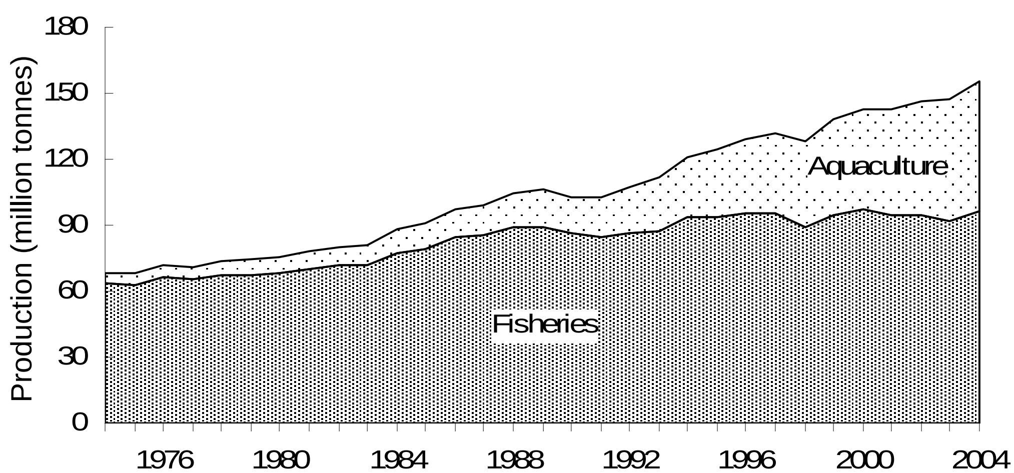

Figure 1.1 Total global fisheries and aquaculture production trends 1974-2004 (source FishStat Plus).

According to the United Nations Food and Agriculture Organization (FAO) glossary, aquaculture is the farming of aquatic organisms in inland and coastal areas, involving intervention in the rearing process to enhance production and the individual or corporate ownership of the stock being cultivated (FAO Glossary 2006). Aquaculture is a highly diverse activity. World-wide, over 240 species of fish, shellfish and seaweeds are farmed. Culture environments encompass fresh, brackish and marine waters and production technologies range from low-intensity culture in earthen ponds to highly intensive industrial production facilities (Iwama 1991; Folke and Kautsky 1992; Troell *et al.* 2004). The majority of fish aquaculture production is extensive or semi-intensive, relying on the use of fertilizers to stimulate natural food production in culture ponds and stocking several species to take advantage of available food sources. In contrast, the intensive culture of high-value salmonids, which represents a small fraction of global production, employs concentrate feeds and produces yields an order of magnitude or more greater than extensive production technologies allow (Hardy 1996).

The sustainability of salmon aquaculture has been much debated in recent years (Folke and Kautsky 1992; Folke *et al.* 1994; Buschmann *et al.* 1996; Naylor *et al.* 1998, 2000; Muir 2005; Naylor and Burke 2005; Kristofersson and Anderson 2006). With a quadrupling of farmed salmon production since 1990 (Figure 1.2), and considerable industry interest in expanding the commercial cultivation of other carnivorous fish species according to the industrial salmon aquaculture model (Naylor and Burke 2005), this debate has garnered international attention.

Although industry proponents have heralded aquaculture as the solution to declining fish stocks and food security in developing countries, critics are quick to point out that not all production systems are created equal. Net-cage salmon farming, which is characterized by intensive production systems generating a high-value product for sale in the developed world, has been implicated in numerous environmental problems, such as habitat destruction (Findlay *et al.* 1995) nutrient pollution (Folke *et al.* 1994), the introduction of genetic material into compromised conspecific populations (Einum and Flemming 1997; Youngson and Verspoor 1998) and the spread of pathogens (Krkosek *et al.* 2006). Another aspect of particular concern is the suite of environmental impacts associated with the production and use of salmon aquafeeds (Naylor *et al.* 2000). These include both proximate biological effects and the diverse range of impacts stemming from resource extraction and processing technologies (Folke 1988; Tyedmers 2000).

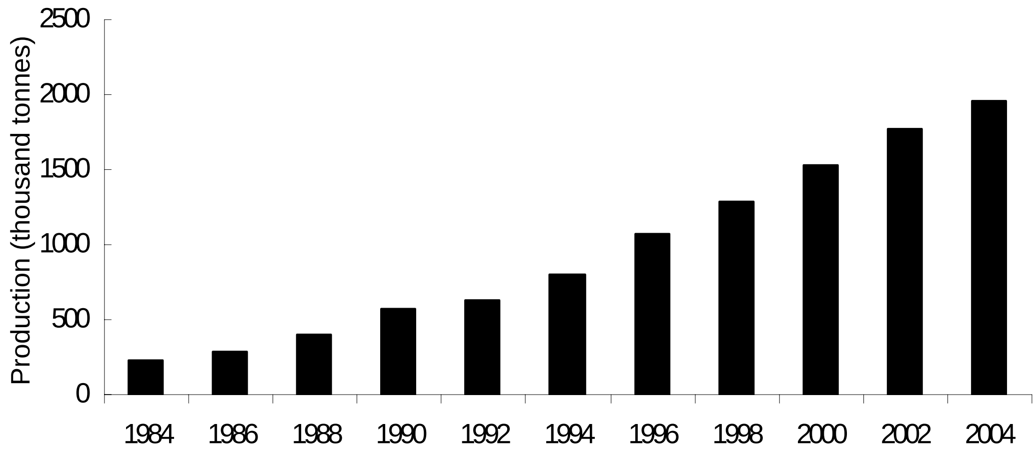

Figure 1.2 Total salmonid aquaculture production 1984 – 2004 (source FishStat Plus).

As many fisheries for high-value, carnivorous species have reached and overshot sustainable exploitation limits, catch volumes are increasingly comprised of smaller, low-value species (Pauly *et al.* 1998), many of which are not currently used directly for human consumption. Four of the five top capture species between 1986 and 1997 were small pelagic fishes used in animal feed production – a substantial and increasing fraction of which is used in aquaculture production of both carnivorous species and herbivorous/detrivorous species that are being raised as functional carnivores (FAO 1998). In fact, fully one-third of global fisheries landings are dedicated to the animal feed production sector (Naylor *et al.* 2000).

Although reduction fisheries landings have remained stable over several decades, the allocation of fishmeal and oil between competing sectors has undergone rapid change, with increasing volumes diverted to the aquaculture sector (Naylor *et al.* 2000; New and Wijkstrom 2002; Tacon 2005). Between 1988 and 1997, the amount of fishmeal used by the aquaculture industry rose from 10% to 33% of total supplies (Naylor *et al.* 2000). Tacon (2005) reported that aquaculture currently consumes 46% of fishmeal and 81% of fish oil produced globally (Figure 1.3). This represents a tripling of fishmeal and oil use in aquafeeds between 1994 and 2003 (Tacon 2005).

Although farmed salmon contributes less than 3% to total aquaculture production (excluding seaweeds), compound aquafeed production for salmon farming accounts for 8.4% of global production (Tacon 2005). Moreover, due to the exacting dietary requirements of carnivorous fish and an industry focus on high-energy feeds, salmon aquaculture currently appropriates 19.5% of global fishmeal supplies and fully 50% of fish oil supplies (Tacon 2005) (Figure 1.4).

Critics of salmon farming frequently refer to the fact that it takes over 2 kg of wild fish to produce 1 kg of farmed salmon (Naylor *et al.* 1998, 2000; Naylor and Burke 2005), which, they argue, represents an inefficient use of resources due to the net loss of available fish protein. Alternatively, it has also been argued that the feed-to-flesh conversion efficiency in salmon production relative to wild fish production and terrestrial animal husbandry actually makes salmon aquaculture a highly efficient means of upgrading marine resources currently considered unpalatable in western markets into products acceptable for human consumption (Asgard and Austreng 1995; Tidwell and Allen 2001).

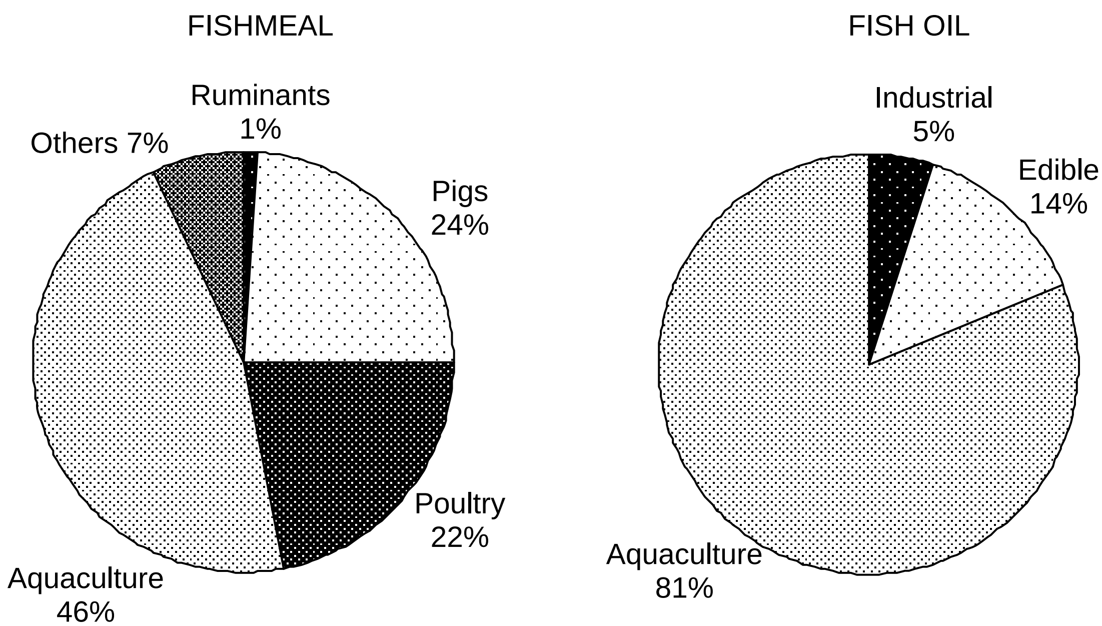

Figure 1.3 Global appropriation of fish meal and oil supplies by sector (source Tacon 2005).

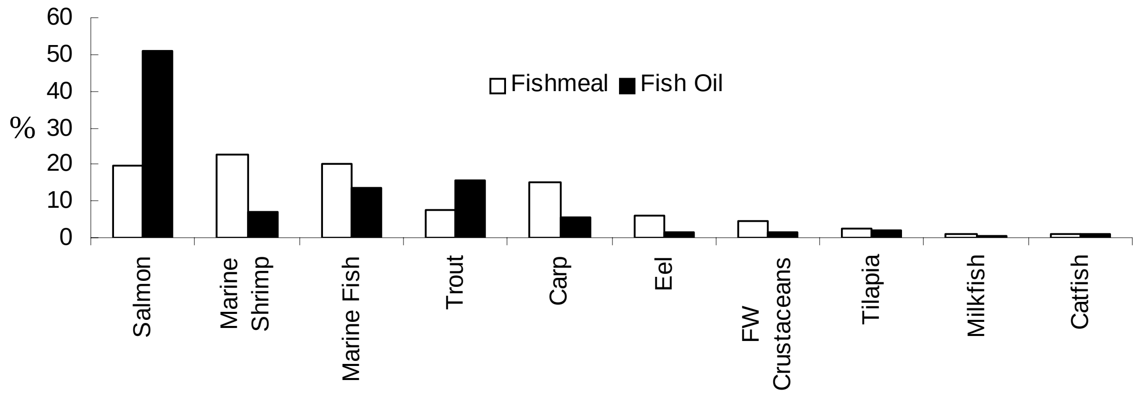

Figure 1.4 Appropriation of global fishmeal and oil supplies by aquaculture sector (based on data from Tacon 2005).

Regardless of where the truth lies between these polarized perspectives, it has long been recognized that the expansion of certain sectors of the aquaculture industry may ultimately be limited by the finite nature of marine feed resources (New and Wijkstrom 2002). At current rates of expansion, it is predicted that the global aquafeed industry will require 70% of the average historical fishmeal supply and 145% of the fish oil supply by 2015 (New and Wijkstrom 2002). This situation is clearly untenable. Finding appropriate dietary replacements for fishmeal and oil in aquafeeds has thus become a research priority (Storebakken *et al.* 1998; Carter and Hauler 2000; Refstie *et al.* 2001; Watanabe 2002; Opstvedt *et al.* 2003), and both crop-derived and terrestrial animal by-product ingredients are accounting for an ever-increasing fraction of feed formulations (Tacon 2005). Soybean meal has been identified as a promising candidate for protein substitution, and several vegetable oils have been tested as lipid replacements (Olli *et al.* 1995; Carter and Hauler 2000; Watanabe 2002; Glencross 2003). However, substitution of marine-derived ingredients with inputs from terrestrial food production systems does not necessarily imply improved ecological efficiency. Such a comparison requires comprehensive examination of the environmental costs and benefits of producing alternative crop and animal by-product ingredients. This includes certified organic crop ingredients and fisheries by-product meals and oils, which are currently touted as sustainable alternatives to conventional feed ingredients (Pelletier 2003; Soil Association 2005; Marine Harvest 2006).

1.3 Organic Aquaculture: A Sustainable Alternative?

Organic agriculture arose largely in answer to the social and environmental impacts associated with the industrial agriculture model. According to the International Federation of Organic Agriculture Movements (IFOAM 2005), the primary goal of organic agriculture is to optimize the health and productivity of interdependent human and non-human communities. Increased consumer awareness has resulted in widespread interest in organic foods. Current growth in global demand for organic foods is estimated at 20-25% per annum (El Hage-Scialabba and Hattam 2002).

Organic standards are developed to regulate the materials and practices used in food production systems. A central goal of certified organic production is to verify and communicate to consumers that production systems are in place that will promote ecological efficiency and social justice by managing the production system as an integrated whole (IFOAM 2005). IFOAM has drafted an overarching statement of principles that organic operators should abide by as far as possible in their interactions with human populations, domestic and wild animals, and the environment (IFOAM 2005).

Numerous researchers have evaluated the comparative environmental performance of conventional and organic agriculture production systems according to criteria such as impacts to soil fertility (Gerhardt 1997; Siegrista *et al.* 1998; Stockdale *et al.* 2002; Oehl *et al.* 2004; Green *et al.* 2005), biodiversity (Clark 1999; Chamberlain *et al.* 1999; Beecher *et al.* 2002; Bengtsson *et al.* 2005) and nutrient dynamics (Halberg *et al.* 1995; Pacini *et al.* 2003; Crews and People 2004). Others have compared energy efficiency (Sarapatka 2002; Sartori *et al.* 2003; Mendoza 2005; Pimentel *et al.* 2005) and contributions to climate change (Foereid and Hogh-Jensen 2004). With few exceptions, organic agriculture has been found to generate less severe environmental impacts compared to conventional agriculture.

In recent years, there has been increasing interest in the concept of organic aquaculture, and various certified fish products are now available in high end markets in both Europe and North America. Moreover, based on current estimates of organic aquaculture production and anticipated compound growth rates the FAO predicts a 240-fold increase in certified organic aquaculture production by 2030 (El Hage-Scialabba and Hattam 2002). Although the relative contribution to total aquaculture production will remain small, the majority of this increase will occur in the carnivorous finfish aquaculture sector, which has attracted the majority of criticisms regarding the environmental impacts of intensive aquaculture. Since market and consumer awareness can influence unsustainable behaviour within industries, the application of organic principles to aquaculture production systems could provide a market-driven approach to reducing environment impacts in this sector – providing that the certification standards actually result in environmentally preferable production practices (Pelletier 2003).

Developing standards for organic aquaculture poses a number of significant challenges. Given the basic differences between terrestrial and aquatic animals and environments, most terrestrial organic standards cannot be directly applied to aquaculture systems. Rather, it is necessary to adapt the overarching general principles of organic production to the specific conditions of aquaculture systems in an appropriate manner (Pelletier 2003).

Standards for organic aquaculture have been developed in several jurisdictions throughout the world, although many are still in draft form (Pelletier 2003). These standards regulate production parameters such as stock density, chemical inputs, production materials, benthic impacts, and siting specifications. They also include specifications for aquafeed production, including allowable origins and amounts of marine resources, agricultural ingredients, and feed additives (for example see Naturland 2005; Soil Association 2005). However, as with many terrestrial organic standards, a rigorous biophysical foundation has not been established for organic aquaculture standards, and comparative data for the environmental impacts of conventional and organic production systems is lacking.

1.4 Literature Review

1.4.1 Energy Analyses and Ecological Footprint Analyses of Aquaculture

Various biophysical accounting techniques have previously been employed to provide insight into the sustainability of aquafeed production and of aquaculture production systems generally (Folke *et al.* 1998; Troell *et al.* 2004). Folke (1988) investigated the energy economy of salmon aquaculture in the Baltic Sea. Tyedmers (2000) similarly used energy analysis to assess the biophysical costs of producing salmon through both the commercial salmon fishery and intensive salmon aquaculture industry in British Columbia, Canada. Both studies indicated that the provision of feeds accounts for the majority of material and energy inputs to salmon aquaculture.

Ecological Footprint Analysis has also been used by several other researchers to quantify the ecosystem services appropriated by various forms of aquaculture (Deutsch *et al.* 2000; Tyedmers 2000). Kautsky *et al.* (1997) reported that the spatial ecosystem support area necessary to produce feed inputs and assimilate wastes for a typical semi-intensive shrimp farm in Caribbean Colombia was 35-190 times the surface area of the farm itself. Berg *et al.* (1996) compared the ecological demand of semi-intensive pond farming and intensive cage farming of tilapia in Zimbabwe according to appropriation of Gross Primary Production (GPP) and the ecosystem areas required to produce feeds, supply oxygen, and assimilate nutrients. Intensive cage farming was found to require substantially more energy and off-farm support area. Tyedmers (2000) compared intensive net-cage salmon farming and salmon fisheries in British Columbia, and found that salmon farming had a substantially larger ecological footprint than salmon fishing, regardless of gear type and species.

Although this technique successfully quantifies the ecosystem support area required for aquaculture activities, it has been argued that interpretations based on area alone fail to account for the comparative efficiency of resource use between systems of varying intensity (Bunting 2001). Furthermore, it does not provide an adequate foundation for informing specific product/process improvements.

1.4.2 Life Cycle Assessment Research of Aquaculture

Life Cycle Assessment has also been used by several researchers to evaluate the environmental performance of various aquaculture production technologies. Seppälä *et al.* (2001) conducted an LCA of trout production in Finnish freshwater lakes and found that eutrophication impacts were a priority area for environmental performance improvements. It was recommended that substituting soy meal as a protein source in feeds would greatly decrease phosphorous emissions and the associated eutrophication potential.

Papatryphon *et al.* (2003) conducted an LCA of rainbow trout production in France. The analysis focused on the production process, to the farm gate, of one tonne of trout at each of eight farms. In an effort to capture the variability of on-farm production practices, product size and production intensity were used as the basis of farm categorization. System boundaries were defined as on-the-farm production processes, as well as upstream processes including: the production and use of primary inputs (oxygen, medicine, and feed); the production and transformation of feed ingredients; production of farm equipment; infrastructure production and construction; and transportation at all stages. Significant variability was found between farms in respect to all assessed environmental impacts. Feed production dominated contributions to Biotic Resource Use, Climate Change, and Acidification, while fish production contributed most to Eutrophication impacts.

Using LCA to evaluate shrimp aquaculture production in Thailand, Mungkung (2005) found that the most significant life cycle impacts were associated with energy use, feed production, and the use of burnt lime, which is employed to control pH in shrimp ponds. Eutrophication effects related to wastewater discharge from the ponds were also identified as significant.

More recently, Ellingsen and Aanondsen (2006) reported the comparative life cycle impacts of salmon farming, cod fisheries, and poultry production in Norway. Feed production was found to dominate contributions to all impact categories considered for both salmon and poultry production.

To date, only one published study has reported an in-depth analysis of the life cycle impacts of aquafeed production (Papatryphon *et al.* 2004). This research examined the life cycle of feeds produced for rainbow trout aquaculture in France, and compared the environmental performance of a series of hypothetical feeds containing varying proportions of fisheries-derived products. However, because the researchers chose to allocate environmental burdens according to the relative economic value of the feed components, the actual environmental costs of the background production systems for these ingredients are not accurately reflected in the reported results.

1.5 Purpose

Understanding the ecological burdens associated with the provision of aquafeed is critical to assessing and improving the sustainability of salmon aquaculture as a whole. This requires a clear understanding of the environmental tradeoffs of using alternative feedstuffs, including crop, fisheries and animal by-product materials. Moreover, given that "organic" salmon aquaculture is being touted as a sustainable alternative to conventional production, it is important to establish a clear biophysical foundation for this claim. The insights derived from this work are intended to provide guidance for aquafeed producers and aquaculturists wishing to improve the biophysical sustainability of their products, and also to inform the continued evolution of organic standards for salmon aquaculture, as well as relevant ecolabeling/consumer education initiatives.

The remainder of this thesis is divided into three parts. Chapter 2 presents a review of impact categories used in LCA research, how these might be expanded to account for the broader range of environmental concerns associated with seafood production, and challenges to such methodological development. Chapter 3 presents the results of a comparative Life Cycle Assessment of aquafeeds for conventional and organic salmon aquaculture in British Columbia, Canada. The information derived is intended to illuminate opportunities for improving the environmental performance of salmon aquaculture and to establish the improvements (if any) that might be gained by a transition to organic aquafeed production. Both of these chapters were written as stand-alone manuscripts for submission to peer-reviewed journals. The first has been accepted for publication in the *International Journal of Life Cycle Assessment*. Submission of the second manuscript is pending. Chapter 4 summarizes the key insights derived from this research, and discusses the findings in the context of sustainability in global food production. Research priorities are also identified.

CHAPTER 2: IMPACT CATEGORIES FOR LIFE CYCLE ASSESSMENT RESEARCH OF SEAFOOD PRODUCTION SYSTEMS: REVIEW AND PROSPECTUS

2.1 Publication Information

This paper has been accepted for publication in the *International Journal of Life Cycle Assessment*. It was co-authored by Nathan L. Pelletier[1], Nathan W. Ayer[1], Peter H. Tyedmers[1], Sarah A. Kruse[2], Anna Flysjö[3], Greg Robillard[2], Friederike Ziegler[3], Astrid J. Scholz[2], and Ulf Sonesson.[3] Nathan Pelletier was the lead author and was primarily responsible for researching and writing the paper. However, the paper was much improved by contributions from all co-authors.

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2.2 Abstract

Goal, Scope and Background. In face of continued declines in global fisheries landings and concurrent rapid aquaculture development, the sustainability of seafood production is of increasing concern. Life Cycle Assessment (LCA) offers a convenient means of quantifying the impacts associated with many of the energetic and material inputs and outputs in these industries. However, the relevant but limited suite of impact categories currently used in most LCA research fails to capture a number of important environmental and social burdens unique to fisheries and aquaculture. This article reviews the impact categories used in published LCA research of seafood production to date, reports on a number of methodological innovations, and discusses the challenges to and opportunities for further impact category developments.

Main Features. The range of environmental and socio-economic impacts associated with fisheries and aquaculture production are introduced, and both the commonly used and innovative impact categories employed in published LCA research of seafood production are discussed. Methodological innovations reported in agricultural LCAs are also reviewed for possible applications to seafood LCA research. Challenges and options for including additional environmental and socioeconomic impact categories are explored.

Results. A review of published LCA research in fisheries and aquaculture indicates the frequent use of traditional environmental impact categories as well as a number of interesting departures from the standard suite of categories employed in LCA studies in other sectors. Notable examples include the modeling of benthic impacts, by-catch, emissions from anti-fouling paints, and the use of Net Primary Productivity appropriation to characterize biotic resource use. Socio-economic impacts have not been quantified, nor does a generally accepted methodology for their consideration exist. However, a number of potential frameworks for the integration of such impacts into LCA have been proposed.

Discussion. LCA analyses of fisheries and aquaculture call attention to an important range of environmental interactions that are usually not considered in discussions of sustainability in the seafood sector. These include energy use, biotic resource use, and the toxicity of anti-fouling paints. However, certain important impacts are also currently overlooked in such research. Although prospects clearly exist for improving and expanding on recent additions to environmental impact categories, the nature of the LCA framework may preclude treatment of some of these impacts. Socio-economic impact categories have only been described in a qualitative manner. Despite a number of challenges, significant opportunities exist to quantify several important socio-economic impacts.

Conclusion. The limited but increasing volume of LCA research of industrial fisheries and aquaculture indicates a growing interest in the use of LCA methodology to understand and improve the sustainability performance of seafood production systems. Recent impact category innovations, and the potential for further impact category developments that account for several of the unique interactions characteristic of fisheries and aquaculture will significantly improve the usefulness of LCA in this context,

although quantitative analysis of certain types of impacts may remain beyond the scope of the LCA framework. The desirability of incorporating socio-economic impacts is clear, but such integration will require considerable methodological development

Recommendations and Perspectives. Although the quantity of published LCA research for seafood production systems is clearly increasing, the influence this research will have on the ground remains to be seen. In part, this will depend on the ability of LCA researchers to advance methodological innovations that enable consideration of a broader range of impacts specific to seafood production. It will also depend on the ability of researchers to communicate with a broader audience than the currently narrow LCA community.

2.3 Introduction

According to the United Nations Food and Agriculture Organization (FAO) the global production of seafood from fisheries and aquaculture reached 133 million tonnes, and provided direct employment to an estimated 38 million people, in 2002 (FAO 2004). With many traditional fisheries depleted, over-exploited, or fully exploited, it appears that global carrying capacity for seafood production has been reached or even exceeded (Pauly *et al.* 2002; Myers and Worm 2003; Worm and Myers 2004), and that restrictive management regimes are imperative (Pauly *et al.* 2003). In response to this decline and increasing demand for seafood products, aquaculture has become the fastest growing animal food-producing sector (FAO 2004). At present, approximately 40% of all consumed fish and shellfish are farmed, and a 70% increase in aquaculture production by the year 2030 is predicted (FAO 2004).

Both fisheries and aquaculture have been implicated in a variety of environmental and socio-economic impacts. Besides direct impacts to targeted stocks (Pauly *et al.* 2002; Christensen *et al.* 2003; Myers and Worm 2003), some fisheries have been criticized for generating substantial by-catch (Alverson *et al.* 1994; Glass 2000), the disturbance and displacement of benthic communities (Johnson 2002; Chuenpagdee *et al.* 2003) and the alteration of trophic dynamics (Jackson *et al.* 2001). Criticisms of aquaculture, which are largely associated with the intensive cultivation of highly-valued shrimp and salmon species, include eutrophication of local water bodies (Folke *et al.* 1994), deterioration of the benthos (Findlay *et al.* 1995; Paez-Azuna 2001), introduction of genetic material into compromised conspecific populations (Einum and Fleming 1997; Youngson and Verspoor 1998; Fleming *et al.* 2000), the amplification and retransmission of diseases and parasites to the wild (Krkosek *et al.* 2006), the discharge of pharmaceuticals and other chemicals into the marine environment (Hastein 1995), the depletion of wild stocks through broodstock or seed harvesting (Mungkung *et al.* 2006) , and a dependence on fish-based feeds (Naylor *et al.* 2000; Naylor and Burke 2005).

Although discourse regarding the environmental repercussions of seafood production is often dominated by such proximate biological interactions, other research indicates that the material and energetic demands of both industrial fisheries and aquaculture can also precipitate considerable impacts. In the fisheries sector, these include the environmental and socio-economic costs of fishing vessel construction and maintenance (Watanabe and Okubo 1989; Hayman *et al.* 2000), the provision of fishing gear (Ziegler *et al.* 2003), the discharge of wastes and loss of fishing gear at sea (Derraik 2002), the use of fossil fuels in vessels (Ziegler and Hansson 2003; Thrane 2004[b]; Tyedmers 2004, Hospido and Tyedmers 2005; Tyedmers *et al.* 2005), and the transportation and processing of landings (Karlsen and Angelfoss 2000; Andersen 2002). In aquaculture, these secondary impacts are largely related to the high material and energetic demands associated with the provision of concentrated feed for intensive production systems and the maintenance of water quality in closed containment systems (Tyedmers 2000; Troell *et al.* 2004; Papatryphon *et al.* 2004). Furthermore, many of these impacts may also threaten the health, social, economic and cultural well-being of adjacent human communities as well as consumers of seafood products (Anderson and Fong 1997; Hites *et al.* 2004).

The rapid expansion of the aquaculture sector and the vulnerability of global fisheries to further degradation underscore the urgent need to understand and manage the environmental and social interactions of seafood production systems. Life Cycle Assessment (LCA) provides a convenient, standardized means of quantifying and describing many of these interactions and targeting specific process and product improvements. However, the efficacy of this tool in providing comprehensive insights as to the scope and scale of such impacts will ultimately be limited by the availability of appropriate impact categories.

2.4 Discussion

2.4.1. Impact Assessment in LCA

The magnitude and significance of environmental or social costs associated with specific life cycle activities are identified during the Life Cycle Impact Assessment (LCIA) phase (Pennington *et al.* 2004). This is achieved by quantitatively expressing the results of the Life Cycle Inventory (LCI) using impact categories (classes representing environmental issues of concern) and their associated category indicators (quantifiable resources/emissions/substances representing each impact category) (Guinee *et al.* 2001). ISO defines both mandatory and optional elements of the LCIA framework. Mandatory elements are: the selection of impact categories, category indicators and characterization models; the assignment of LCI results (classification); and the calculation of category indicator results (characterization). Optional elements are: calculation of the magnitude of category indicator results relative to reference information (normalization); grouping; weighting; and data quality analysis (Guinee *et al.* 2001).

SETAC standards (Consoli *et al.* 1993) define four key impact categories for LCA: ecological health, human health, resource depletion, and social welfare. ISO (2003) does not include social welfare as a mandatory impact category, but does state that where existing impact categories, category indicators, and characterization models are not sufficient to satisfy the defined goal and scope of the research, new ones must be defined. The ISO standards further require that the selection of impact categories reflects a comprehensive set of environmental issues relevant to the system under study.

These impact categories are very broad and, depending on the nature of the study, are generally sub-divided to represent more specific impacts. The European Environment Agency (Jensen *et al.* 1999) identifies abiotic resources, biotic resources, land use issues, global warming, stratospheric ozone depletion, ecotoxicological impacts, human toxicological impacts, photochemical oxidant formation, acidification, eutrophication, and the human work environment as priority impact categories. With the exception of human work environment, most of these categories are commonly employed in published LCA studies (Table 2.1).

Table 2.1 Impact categories commonly employed in published LCA research.

Impact Category	Description of Impacts
Global Warming	Contributes to atmospheric absorption of infrared radiation
Acidification	Contributes to acid deposition
Eutrophication	Contributes to Biological Oxygen Demand
Photochemical Oxidant Formation	Contributes to photochemical smog
Aquatic/Terrestrial Ecotoxicity	Contributes to conditions toxic to flora and fauna
Human Toxicty	Contributes to conditions toxic to humans
Energy Use	Contributes to depletion of non-renewable energy resources
Abiotic Resource Use	Contributes to depletion of non-renewable resources
Biotic Resource Use	Contributes to depletion of renewable resources
Ozone Depletion	Contributes to depletion of stratospheric ozone

2.4.2 Commonly Employed Impact Categories in Seafood LCAs

Although Life Cycle Assessment in the agriculture sector is relatively well established (Andersson *et al.* 1994; Andersson 2000), the application of this tool for evaluating seafood production systems is a more recent phenomenon. To date, LCA researchers in fisheries and aquaculture have variously examined Norwegian cod fisheries (Ellingsen and Aanondsen 2006), Spanish tuna fisheries (Hospido and Tyedmers 2005), Danish fish products (Thrane 2004[a], 2006), Swedish cod products (Ziegler *et al.* 2003), Finnish trout production (Seppälä *et al.* 2001), farmed salmon (Ellingsen and Aanondsen 2006), farmed Thai shrimp products (Mungkung 2005), and French trout farms and feeds (Papatryphon 2003, 2004), The conclusions generated by these studies suggest that the LCA framework is well-suited to informing eco-efficiency measures in fisheries and aquaculture, and that the life cycle perspective holds considerable promise for informing seafood product-oriented environmental policy. However, it is apparent that the development of appropriate impact categories will be essential to arriving at more comprehensive evaluations of the environmental and social interactions of seafood production.

Of the studies reviewed (Table 2.2), all employed global warming, acidification, and eutrophication as impact categories. In some cases, eutrophication has been sub-divided to reflect the relative contributions of nitrogen and phosphorus emissions (Ziegler *et al.* 2003), or emissions to land or water (Seppälä *et al.* 2001). As with photochemical oxidant formation, abiotic resource use (alternately referred to as energy use, abiotic depletion, or depletion of fossil fuels) has also been commonly, if not consistently, employed (Seppälä *et al.* 2001; Papatryphon *et al.* 2003,2004; Ziegler *et al.* 2003; Mungkung 2005). Toxicity-related emissions have been variously quantified in terms of marine, aquatic, freshwater, terrestrial, and human toxicological impacts. Papatryphon *et al.* (2003, 2004) were alone in accounting for biotic resource consumption. The use of these categories is consistent with the impact category choices typical of LCA research in other sectors. In so far as accounting for broad-scale environmental impacts characteristic of human industrial activities, these choices are certainly defensible. However, if the goal is to arrive at more comprehensive assessments and opportunities for improvement in seafood production systems, then further impact category development is desirable.

Table 2.2 Impact categories and category indicators employed in selected LCA research of seafood production systems.

Author	Research	GW	Ac	Eu	PO	AE	TE	HT	EU	BRU	OD
Hospido and Tyedmers (2005)	Spanish tuna fisheries	CO_2	SO_2	PO_4	C_2H_4	1,4DCB		1,4DCB			CFC
Mungkung (2004)	Thai shrimp aquaculture	CO_2	SO_2	PO_4	C_2H_4	1,4DCB	1,4DCB	1,4DCB	Sb		CFC
Papatryphon *et al.* (2003, 2004)	French trout aquaculture and feeds	CO_2	SO_2	PO_4					MJ	NPP	
Thrane 2004[a], 2006	Danish fisheries	CO_2	SO_2	PO_4	C_2H_4	H_2O			MJ		CFC
Ziegler *et al.* (2003)	Swedish cod fisheries	CO_2	SO_2	NO_3 PO_4	C_2H_4	H_2O			MJ		

Impact categories are: GW = Global Warming, Ac = Acidification, Eu = Eutrophication, PO = Photochemical Oxidant, AE = Aquatic Ecotoxicity, TE = Terrestrial Ecotoxicity, HT = Human Toxicity, EU = Energy Use, BR = Biotic Resource Use, OD = Ozone Depletion. Category indicators are: CO2 = Carbon Dioxide, SO2 = Sulphur Dioxide, PO4 = Phosphate, NO3 = Nitrate, O2 = Oxygen, 1,4 DCB = 1,4 Dichlorobenzene, H2O = Water, MJ = Mega Joules, Sb = Antimony, NPP = Net Primary Productivity, CFC = Chlorofluorocarbon.

2.4.3 Challenges to Seafood Production-Specific Impact Category Development

Perhaps due to its origin and evolution in the context of traditional industrial production systems, the standard suite of impact categories used in most LCAs accounts for an important but limited range of environmentally significant chemical parameters. In contrast, the life cycle impacts of seafood production often encompass a much greater range of environmental concerns. A major challenge facing LCA researchers of seafood production systems is to develop novel impact categories appropriate to quantifying the environmental and social interactions characteristic of specific seafood production technologies and, where this is not possible, to clearly articulate the limitations of the information generated.

LCA is most often used to describe product life cycle impacts that contribute to broad-scale, global environmental concerns such as climate change, ozone depletion, and abiotic resource use. The physical/chemical pathways that contribute to these impacts are reasonably well-known and relatively easy to quantify. Standardized analyses and comparisons between production systems and technologies are therefore largely straight-forward, whatever the geographical context. For example, a tonne of greenhouse gas emissions will contribute to climate change in a specific manner, regardless of where the emissions occur. This is similarly true of the release of ozone depleting substances and the consumption of non-renewable resources. With certain other traditional impact categories, including eutrophication, acidification, and marine ecotoxicity, the relationships between local emissions and global impacts are less clear. In the case of eutrophication, for example, the actual eutrophication potential of nutrient emissions will be very much context specific, depending on a wide variety of variables which together define the sensitivity and assimilatory capacity of the ecosystem in question. Nonetheless, the pathways leading to eutrophication impacts are well-known and easy to quantify.

There are numerous other environmental concerns that manifest only at local levels but also contribute to globally-recognized issues. These include biodiversity loss, biotic resource depletion, and the erosion of ecosystem structure and function through habitat alteration. However, such impacts are often much more difficult to quantify, and accepted methodologies for use in ISO-compliant Life Cycle Assessment have yet to be developed. A logical starting point, then, is to determine which impacts might be meaningfully quantified using the LCA framework, and which cannot. This will largely depend on two factors: the quality of available information regarding the impact in question; and the ability to link the impact to a functional unit in a realistic manner. In many cases, such as the quantification of benthic impacts, the development and application of an appropriate impact category appears promising but may be constrained by a lack of scientific consensus, data, and accepted models. In other cases, as when the impacts in question arise from a complex interplay of several variables and cannot be reduced to direct causal relationships, it may simply not be possible to defensibly link the impacts to a functional unit. Examples include changes to biodiversity, such as when seabird colonies exhibit declines attributable only in part to competition with fisheries for food resources, or declines in wild salmon stocks impacted by a variety of factors including sea lice emissions from net-cage aquaculture operations. Moreover, although a variety of local impacts may, indeed, prove relatively easy to quantify in relation to a functional unit, the value of incorporating these aspects into an LCA must be assessed with respect to the goals of the research. In all situations where quantitative analysis of specific impacts proves unfeasible, the value of qualitative analyses should be considered.

The following discussion is intended to review various approaches that seafood and other LCA researchers have used to attempt to quantify a range of non-traditional life cycle impacts, and to highlight potential areas for future impact category development efforts for seafood LCAs. Rigorous evaluations of the technical merits of specific approaches have not been undertaken.

2.4.4. Methodological Innovations

A review of published LCA research in fisheries and aquaculture indicates that modest efforts have been made to expand the suite of impact categories beyond those commonly employed in other sectors. In addition, several authors have discussed the desirability of new impact categories or described possible additional categories.

Ziegler *et al.* (2003) pioneered the use of seafloor effects due to bottom trawling as an impact category in a Life Cycle Assessment of frozen cod fillets. Seafloor effects were calculated by quantifying the area of seafloor swept by trawls per functional unit landed. Although this initial attempt to quantify benthic impacts was somewhat limited in that it only distinguished two seafloor types (oxic and anoxic) and did not differentiate between the associated benthic communities and their resilience to disturbance, considerable effort has since been expended to incorporate these complexities into the impact assessment method (Nilsson and Ziegler 2006). In a more recent study, a Geographical Information System (GIS) was used to analyze the spatial distribution and biological impact of fishing effort in relation to seafloor habitat types. An index of gear impact per hour trawled was calculated from vessel speed as well as otter board and trawl width for the demersal trawl types in use. This index was multiplied by the fishing effort for each trawl haul to arrive at the total seafloor area swept during each fishing event. Fishing effort intensity was then modeled by dividing the area into squares in which the number of times swept was calculated, based on the gear set position and seafloor impact calculated as above. Next, this dataset was overlaid with the habitat map showing the fishing intensity in the individual habitats. It was found that large areas were not used for fishing during the study period (three years) and that the habitats were targeted to highly different degrees. The biological impact of the fishing intensities was assessed using a British database detailing marine habitat sensitivity and recoverability from fishing disturbance. Results which could be used as a seafloor impact indicator include the percentage of each habitat considered not to recover between fishing events *(i.e.* being in a permanently altered condition) (Nilsson and Ziegler 2006).

Ziegler *et al.* (2003) also quantified by-catch of non-target and juvenile target species based on landing and discard data for the cod fishery in the Baltic Sea. A qualitative assessment of the sustainability of the resource use in relation to cod stock status was included. Given widespread concern regarding the ecosystem effects of by-catch in a diversity of fishing systems (Harrington *et al.* 2005; Hall and Mainprize 2005; Catchpole *et al.* 2005; Read *et al.* 2006), this category certainly merits further elaboration. It also provides a means of partially accounting for biotic resource use.

Although many studies employ 1,4-dichlorobenzene as the category indicator for aquatic ecotoxicity, Ziegler *et al.* (2003) reported ecotoxicity in terms of volume of water polluted by the copper-based anti-fouling paints used on the hulls of fishing vessels. Thrane (2004[a]) and Hospido and Tyedmers (2005) similarly quantified anti-fouling paint emissions in LCA studies of Danish flatfish and Spanish tuna fisheries. These studies indicated non-trivial ecotoxicity impacts, suggesting the importance of modeling this emission in future LCA research of seafood production.

Papatryphon *et al.* (2004) used appropriation of Net Primary Production (NPP), which is the net flux of carbon from the atmosphere into green plants per unit time, as a proxy for biotic resource use impacts in an investigation of the life cycle impacts of salmonid feed production. For simplicity, analysis was restricted to direct NPP use, and did not include long-term effects on ecosystem NPP levels. In order to quantify global life cycle NPP use, a single category combining terrestrial and marine NPP was employed. For crop-based feed ingredients, NPP use was calculated according to the carbon content of the harvested and utilized crop components. NPP use for fisheries-derived ingredients was calculated based on the carbon content and trophic level of the species harvested.

The use of NPP to characterize biotic resource use represents one of the most interesting impact category developments to date. Appropriation of Net Primary Productivity provides a convenient and logical measure of biotic resource use. According to basic thermodynamic principles, energy can neither be created nor destroyed; all forms of energy are inter-convertible; but every transformation of energy results in increased entropy. These principles are evident in biological systems, almost all of which are driven by solar energy. Photosynthetic plants and algae convert solar energy into heat (dissipated) and chemical (stored) energy in the form of carbon-based organic molecules. In turn, as these carbon complexes flow through trophic webs, the amount available to successive consumers decreases. Generally, only 10% of chemical energy is transferred between trophic levels in aquatic ecosystems (Pauly and Christensen 1995). This implies that food webs are effectively carbon-based energy pyramids, with high biomass of (numerous) primary producers at the bottom, and few, high-level consumers at the top. In this light, trophic dynamics can be interpreted as a competition for carbon resources, which represent the transferable products of primary production.

Human appropriation of global terrestrial NPP has been estimated at close to 40% (Vitousek *et al.* 1986), and an estimate of 8% has been advanced for the appropriation of marine NPP by fisheries (Pauly and Christensen 1995). The finite nature of primary production implies that optimal resource use in human food production systems must be informed by considerations of how best to allocate these resources between competing interests. This must include not only human needs, but the implications of such appropriation for biodiversity and ecosystem stability.

Measuring the appropriation of Net Primary Productivity provides opportunities to address allocative efficiency in fisheries and aquaculture on several counts. This metric can be used to incorporate the biophysical costs of by-catch in fisheries. It can also be used to elucidate the ecological demands of alternative aquafeed formulations, the culture of species occupying different trophic levels and, more generally, as a yardstick for comparing the relative ecological efficiency of different seafood production technologies. Such a measure is analogous to Ecological Footprint Analysis, which has been used to quantify the area of productive ecosystem support appropriated by various forms of aquaculture (Folke 1988; Larsson *et al.* 1994; Berg *et al.* 1996; Folke *et al.* 1998; Tyedmers 2000). One challenge in the use of this indicator is the conflict inherent in treating primary production as a limited resource while simultaneously pegging eutrophication as an environmental impact. In both cases, considerations should be made for local carrying capacity, and the status of the organisms in question in relation to baseline conditions.

Since traditional LCA research focuses on industrial production systems that consume primarily abiotic resources, the underdevelopment of appropriate measures for biotic resource use is somewhat understandable. This certainly does not hold for food production systems, where biotic resources are of central concern. Despite this, the majority of LCA studies of food products have failed to include any measure of biotic resource use – in effect, externalizing biotic resources by treating them as free system inputs.

In a LCA of Danish fish products, Thrane (2004[b]) reported an in-depth investigation of abiotic energy consumption in Danish fisheries. His analysis considered fuel consumption at the fisheries stage both as a function of fish species and fishing gear type.

Fisheries for demersal fish and shellfish were found to be much more energy intensive than fisheries for pelagic fish for human consumption, mussels, and those fish destined for reduction to meal and oil. It was also concluded that significant reductions in fuel use could be achieved by using seine or gillnet gear instead of trawls in the Danish fishery. This is consistent with Tyedmers (2004), who reported that fuel use in fisheries can range from 20-3400 liters/tonne of fish landed. Hospido and Tyedmers (2005) similarly found that fuel inputs dominated environmental impacts in Spanish tuna fisheries, and concluded that efforts to rebuild stocks could result in improvements in the environmental performance of the fishery by decreasing distances traveled to catch and transport fish. In general, research of seafood production systems indicates that fuel use in the fisheries stage often contributes a disproportionate share of impacts, although the actual proportion can vary widely depending on the fuel-intensity of the fishery (Thrane 2006; Hospido and Tyedmers 2005; Tyedmers 2004; Ziegler *et al.* 2003). LCA research in aquaculture also suggests the importance of energy consumption to overall environmental impacts (Mungkung and Clift 2006; Mungkung 2005; Papatryphon *et al.* 2003, 2004). It would appear, then, that energy use should perhaps be treated as a stand-alone indicator of environmental performance in LCA research in the seafood sector - as, for example, was reported in Papatryphon *et al.* (2004).

Thrane (2004[a]) made qualitative evaluations of seabed impacts, land use, waste, use of non-renewable abiotic resources, use of groundwater, exploitation of fish, discards and by-catch, occupational health and safety, noise and accidents, and animal welfare. In a LCA of salmonid feeds, Papatryphon *et al.* (2004) expressed concern regarding the current lack of relevant impact categories, characterization factors and emission factors specific to the aquatic environment. Eutrophication effects, aquatic toxicity, Net Primary Productivity, damage to the benthos and biodiversity, the effects of solid waste, the use of chemicals, genetic impacts, and disease transmission were identified as areas requiring further research for impact category development. Mattsson and Ziegler (2004) also discussed the state of knowledge and research needs in regards to using seafloor effects, mortality of target and non-target species, working environment, anti-fouling agents, and fish welfare as impact categories for seafood LCAs. Although these impacts are, indeed, highly relevant, convincing methods for relating them to a functional unit must be developed before they can be incorporated in standard LCA research.

2.4.5. Lessons from Agriculture

Although the creation of new impact categories for LCA has, in general, been limited, innovations and theory generated within the context of agri-food LCAs might usefully inform impact category developments for seafood. For example, Cowell and Clift (2000) discuss a methodology for assessing soil quantity and quality in Life Cycle Assessments of agricultural production systems. Relevant quantifiable factors describing changes to soil resulting from agriculture are identified. These include soil mass, nutrients, weeds and weed seeds, pathogens, nutrients, salts, pH, organic matter, and soil texture and structure. Given that water provides a comparable role as growth medium in fisheries and aquaculture, a similar suite of parameters could be developed to evaluate impacts to water from seafood production systems. Owens (2002) proposes a suite of detailed category indicators for assessing water quantity and quality in Life Cycle Assessments that could provide a foundation for further refinement of a Water Resources impact category in seafood LCAs. Eutrophication, Dissolved Oxygen Demand, Thermal Effects, Pathogenic Micro-organisms, Colour and Turbidity, Suspended Solids, Toxic Hazard, and Effluent Toxicity are identified as potential indicators.

The Land Use category sometimes employed in agricultural LCAs might also have a potential parallel in fisheries and aquaculture. Just as the environmental costs of land use include the loss of habitat and potential associated loss of biodiversity, use of the marine environment for harvesting or producing seafood generates similar impacts. Brentrup *et al.* (2002) discusses the Hemeroby concept, which is a measure of human influence on ecosystems, as a possible quantifiable indicator for land use in agriculture. Hemeroby evaluates different types of land use according to intensity. Characterization factors can then be used to calculate landscape degradation, with reference to the natural state of the biogeographic region, due to specific land uses (Brentrup *et al.* 2002). In the context of seafood production, such characterization factors might include benthic macro-faunal diversity, zooplankton abundance and diversity, or sediment and water chemistry. However, it would first be necessary to define clear causal links between the described impact and the product system – a challenge compounded by the necessity of both extensive on-site sampling and considerable baseline information.

In a LCA study comparing organic, extensive and intensive grassland farming in southern Germany, Haas *et al.* 2001 employed several unconventional impact categories, including soil function, water quality, biodiversity, landscape image and animal husbandry. All of these are either directly applicable or have a potential parallel as impact categories for the marine environment should appropriate methodologies be developed.

2.4.6. Potential Socio-economic Impact Category Development

In addition to recent biophysical and ecological impact category developments, there is considerable potential for the creation of socio-economic impact categories (Dreyer *et al.* 2006) - the addition of which would much enhance the value of LCA in assessing and improving the sustainability of seafood production. However, despite the fact that SETAC standards recommend inclusion of 'social welfare' as an impact category in all detailed LCAs (Consoli *et al.* 1993), only a limited number of published studies have attempted to quantify the socioeconomic impacts (SEIs) associated with specific products/processes.

The integration of SEIs into the LCA framework is considerably less advanced for several reasons. The first is related to limitations inherent to the LCA framework, which was originally developed by industrial engineers to measure and mitigate impacts linked to specific flows of raw materials and energy. Although these relationships are generally easy to establish, causal links describing relationships between a process and its socioeconomic impacts are more difficult to define. To date, a set of metrics suitable to describing such links for SEIs has not been established, nor is there a shared understanding as to how this is best achieved. O'Brien *et al.* (1996) advance a general framework for integrating Social and Environmental Life Cycle Assessment (SELCA) that might usefully inform the incorporation of social criteria in future impact category developments. In addition, a SETAC-Europe Working Group review of best practices for defining impact categories and category indicators provides further guidance (Udo de Haes *et al.* 1999). More recently, Dreyer *et al.* (2006) proposed a framework for Social

Life Cycle Assessment (SELCA) based on a combined top-down and bottom-up approach that addresses the conduct of companies in relation to universal consensus documents such as the *Universal Declaration of Human Rights*, the *International Labour Organization's Conventions and Recommendations*, and the *Tripartite Declaration of Principles Concerning Multinational Enterprises and Social Policy*.

The successful integration of socioeconomic indicators into the LCA framework requires that impacts be measured using a metric that is additive along the value chain, or that they can be described in a way that facilitates comparisons between process steps. In others words, socio-economic impacts might be described quantitatively, in relation to the functional unit, or qualitatively – both of which can meaningfully inform inter and intra-framework comparisons.

Examples of SEI categories or indicators identified in several existing studies include: occupational health & safety, noise and accidents, and illegal workers and age of workers. However, these indicators and/or categories vary significantly in size, scope and metrics. A more useful and relevant way of classifying indicators might be to place them in categories of a similar size and scope. These impact categories could include: (1) Working conditions (including occupational health & safety, accidents), (2) Workforce (including illegal workers and age of workers), and (3) Business/company practice/influence (including corporate social responsibility).

Sabatella and Franquesa (2004) consider the use of SEIs in fisheries, but outside the context of an LCA framework. Similarly, Ellingsen (2004) discusses the potential use of LCA to inform improvements to working environment in fisheries and aquaculture by reducing risk of fatalities. A recent agricultural LCA (Heller, 2002) featured the development of indicators to describe the sustainability of the U.S. food system (excluding seafood products), with different indicators developed for the various stages in the value chain. Some key socioeconomic indicators used in this study were "percentage of illegal farm workers", "increasing age of farm operators" and "declining entry of young farmers."

There are a variety of other socioeconomic sub-categories that can be used to describe impacts, but most topical LCA literature has focused on only one particular sub-category (*e.g.* Weidema, 2002), as opposed to developing a comprehensive set of SEIs. Only one existing LCA study of fisheries considers some integration of socioeconomic aspects (Thrane 2004[a]) by qualitatively describing Occupational Health & Safety and Noise and Accidents (including odor and visual aspects) as potential impact categories.

An exhaustive review of potential indicators of social welfare that might be used within the LCA framework is beyond the scope of this paper. However, attention to certain broad social objectives relevant to food production can illuminate potential foci for impact category development. For example, if one accepts the premise that social welfare is positively influenced by a net gain in food security, then relative contributions to food security could serve as a quantifiable means of comparing the social costs and benefits of various food production systems. Using edible Energy Return On Investment (EROI) (Mitchell and Cleveland 1993) as a characterization factor, the LCA researcher could report net caloric gains or losses in relation to both fossil energy inputs and the appropriation of Net Primary Productivity. Refining this indicator to reflect net gains or deficits in food security within and between geo-political regions would also be important to evaluating how fisheries and aquaculture production systems contribute to the equitable distribution of resources. Moreover, such a measure would effectively bridge biophysical and socio-economic considerations, making visible the dynamic interplay of social preferences, human needs, and the supporting context provided by ecosystem goods and services.

Contributions to job security could similarly be used as a social welfare impact category. Where several production systems exist for the same product (for example, different fisheries targeting the same species), costs and benefits could be characterized according to the number of jobs created, human work-hours, reported accidents, and salary levels. This would also provide a quantitative frame of reference for comparing fisheries and aquaculture production systems generating similar products.

2.5 Conclusions

An assessment of the state-of-knowledge of Life Cycle Assessment, as it is applied to fisheries and aquaculture production systems, indicates significant potential for the use of this tool in promoting environmental and social improvements in these industries. As an analytic tool, LCA generates quantitative, biophysical data that can be used as a basis for making specific product/process improvements. More generally, it focuses attention on core sustainability issues related to eco-efficiency and broad-scale environmental impacts that are often overlooked in public discourse concerning the environmental interactions of fisheries and aquaculture production. In addition, as each reported Life Cycle Assessment draws from and contributes to a rapidly expanding information base and methodology, the potential for meaningfully informing biophysically-based management decisions increases.

However, the limited scope of existing impact categories significantly impairs the value of Life Cycle Assessment as a management tool. Further impact category development is therefore desirable to facilitate more comprehensive evaluations of the myriad environmental and social costs of seafood production. Methodological innovations from LCAs in the agri-food sector and the pioneering efforts of seafood LCA researchers provide sound guidance for future efforts, although the nature of this tool (which requires the establishment of direct, causal relationships between impacts and the product system that are measurable across process stages) limits the ability to quantify certain impacts within the LCA framework.

2.6 Recommendations and Perspectives

Although the quantity of published LCA research for seafood production systems is clearly increasing, the influence this research will have on the ground remains to be seen. In part, this will depend on the ability of LCA researchers to advance methodological innovations that enable consideration of a broader range of impacts specific to seafood production. It will also depend on the ability of researchers to communicate with a larger audience than the current LCA community. Reporting mechanisms that transform Life Cycle Impact Assessment data into palatable and easily understood formats, and present these in relation to definable goals are required if the insights derived from LCA research in this sector are to be successfully applied to improve policy/management prescriptions. Researchers should make efforts to contextualize their results in terms of regional, national, or international standards or targets relevant to specific impact categories, and should carefully select impact categories with respect to the goals of the research project.

CHAPTER 3: LIFE CYCLE MEASURES OF SUSTAINABILITLY IN FEED PRODUCTION FOR COVENTIONAL AND ORGANIC SALMON AQUACULTURE IN THE NORTHEAST PACIFIC

3.1 Publication Information

The publication venue for this paper has not yet been determined.

3.2 Abstract

Feed provision for intensive net-cage salmon aquaculture accounts for the majority of material and energetic inputs to farmed salmon production. Assessing and ameliorating the environmental performance *(i.e.* the relative magnitude of contributions made by different production systems to a specified suite of environmental problems) of feeds is therefore central to improving the sustainability of salmon farming as a whole. Life Cycle Assessment was used to compare the environmental impacts of producing ingredients for four hypothetical feeds for conventional and organic salmon aquaculture. Fish and poultry-derived ingredients generated substantially greater impacts than crop-derived ingredients. Despite the fact that organic crop ingredients had markedly lower life cycle impacts compared to equivalent conventional ingredients, substituting organic for conventional crop ingredients therefore resulted in only minor reductions to the total impacts of feed production. Replacing fishmeals/oils from dedicated reduction fisheries with fisheries by-product meals/oils markedly increased the environmental impacts of feed production, largely due to the higher energy intensity of fisheries for human consumption, and low meal/oil yield rates of fisheries by-products. Environmental impacts were considerably lower when feeds contained reduced proportions of fish and poultry-derived ingredients. These results indicate that current standards for organic salmon aquaculture, which stipulate the use of organic crop ingredients and fisheries by-product meals and oils, fail to reduce the environmental impacts of feed production according to the impact categories considered in this study.

3.3 Introduction

Salmon farming makes a small, but economically and environmentally significant, contribution to global aquaculture production volumes. From a fledgling industry pioneered in the early 1970's, net-cage salmon aquaculture operations have proliferated in numerous regions including Europe, North America, Australasia and, most notably, South America (Naylor *et al.* 2000; Naylor and Burke 2005). Once considered a luxury item, farmed salmon is now the most widely consumed seafood product in the industrialized world, competing in price with chicken, pork and beef (Naylor *et al.* 2005). The meteoric rise of salmon aquaculture has been accompanied by a growing and vociferous host of criticisms, largely related to the potential impacts of intensive net-cage operations on local ecosystems (Folke *et al.* 1994; Findlay *et al.* 1995; Einum and Flemming 1997; Youngson and Verspoor 1998; Fleming *et al.* 2000; Krkosek *et al.* 2006) as well as a reliance on wild-caught fisheries products for feedstuff (Naylor *et al.* 2000; Naylor and Burke 2005). An emerging body of work is also calling attention to the broader range of biophysical impacts related to the material and energetic intensity of intensive aquaculture production systems (Folke 1998; Tyedmers 2000; Troell *et al.* 2004). Previous research has shown that the provision of concentrate feeds accounts for the majority of material and energy inputs to intensive salmonid aquaculture (Folke 1988; Seppälä *et al.* 2000; Tyedmers 2000; Papatryphon *et al.* 2003; Ellingsen and Aanondsen 2006). Understanding and improving the environmental performance *(i.e.* the relative magnitude of contributions made by different production systems to a specified suite of environmental problems) of feed ingredients and alternative feed formulations is therefore central to improving the sustainability of salmon farming as a whole.

Salmon aquafeeds derive ingredients from a broad array of fisheries and agricultural production systems, as feed mills strive to deliver formulations that are both cost-effective and nutritionally optimal. Traditionally, salmon feeds have relied heavily on fishmeals and oils from dedicated reduction fisheries (Hardy 1996; Naylor *et al.* 2000; Naylor and Burke 2005). Given the finite nature of available marine resources and predictions that demand by aquaculture may outstrip global supplies within the next decade (Tacon 2005), substituting these core components is a challenge that the industry must necessarily embrace.

Organic certification of salmon aquaculture is seen by some industry players as a potential means of gaining consumer confidence regarding the environmental performance of farmed salmon products (Pelletier 2003; Graig Farm 2006; Marine Harvest 2006). Although currently accounting for a small fraction of total production volumes, a 240-fold increase in organic aquaculture production by 2030 is predicted (Hattam and El-Hage Scialabba 2002). Numerous organic certification schemes for salmon aquaculture currently exist, and all include specifications for organic feeds (for example, see Soil Association 2005; Naturland 2005). These standards typically require that fishmeals and oils be derived from reduction fisheries independently certified as “sustainable” (none currently exist) or the by-products of fisheries for human consumption, and stipulate that plant-derived material must be of certified organic origin. However, such substitution does not necessarily imply reduced environmental burdens and, to date, the efficacy of organic salmon aquaculture standards in enhancing environmental performance in aquafeed production has not been evaluated in a rigorous manner.

Life Cycle Assessment is an ISO-standardized biophysical accounting tool for quantifying the environmental costs of products and services. Based on a “cradle-to-grave” approach, the LCA framework is used to inventory the energy and material inputs and emissions associated with each stage of a product life cycle, from resource extraction through use and eventual disposal, and to express these in terms of their quantitative contributions to a specified suite of environmental impact categories (Guinee *et al.* 2001). This information can then be used by industry to target process-specific improvements and by policy makers and managers to create an enabling environment for sustainable development.

The purpose of this study was to use Life Cycle Assessment methodology to evaluate the comparative environmental performance of conventional and organic salmon feed ingredients and feed formulations and to identify the feed-related environmental improvements, if any, that might be gained by a transition to organic salmon aquaculture. The insights derived are intended to guide aquafeed producers and aquaculturists wishing to improve the biophysical sustainability of their products and to inform the continued evolution of organic standards for salmon aquaculture, as well as relevant ecolabeling/consumer education initiatives.

3.4 Methods

ISO-compliant Life Cycle Assessment methodology (Guinee *et al.* 2001) was used to determine the life cycle environmental impacts associated with the provision of concentrate feed for intensive net-cage salmon aquaculture in British Columbia, Canada. Feed components were evaluated on an individual basis, and four hypothetical feed formulations were compared. The first formulation (C) represented the average inputs to all conventional salmon feeds produced over the course of one year by a feed mill in British Columbia (Anon). The second formulation (OA) was identical to the first, except that all agricultural ingredients were certified organic. The third formulation (OBP) included the same organic crop ingredients, but all animal-based ingredients were derived from the by-products of fisheries for human consumption. In the fourth formulation (ORF), which similarly included certified organic crop ingredients, poultry by-product meal was replaced with fishmeal, 25% of the fishmeal was replaced with organic soy meal and all fish oil was substituted with organic canola oil (Table 3.1). These latter scenarios represent the range of hypothetical organic feed formulations that might be employed based on existing or proposed standards for organic salmon aquaculture, but the third (OBP) most closely conforms to the specifications of the major existing organic standards (Pelletier 2003, Soil Association 2005, Naturland 2005).

Geographically and temporally, this study was limited to contemporary feed production for salmon aquaculture in British Columbia. However, the system boundaries encompassed all direct material and energetic inputs and emissions related to the fisheries and agricultural production and processing systems from which feed ingredients were derived, as well as transportation of these ingredients to British Columbia, and the processing of ingredients into concentrate salmon feed in a BC feed mill. This included inputs to vessels and equipment used to produce/harvest the ingredients, but excluded processing, housing, and transportation infrastructure (Figure 3.1). Vitamin/mineral premixes, pigments and anti-oxidants were also not considered. As far as possible, current data reflecting modern production technologies were used.

Table 3.1 Composition of one tonne of each of four hypothetical feeds for salmon aquaculture in British Columbia, Canada.

Ingredient (kg)	C	OA	OBP	ORF
Fishmeal (Peru)	260	260		296.25
By-product Fish Meal (BC)	25	25	395	
Fish Oil (Peru)	60	60		
Menhaden Oil (US)	30	30		
By-product Fish Oil (BC)			90	
By-Product Poultry Meal (BC)	110	110		
Wheat (Alberta)	160	160	160	160
Corn Gluten Meal (Ontario)	110	110	110	110
Canola Seed (Alberta)	70	70	70	70
Canola Meal (Alberta)	40	40	40	40
Canola Oil (Alberta)	70	70	70	160
Soy Meal (Ontario)	45	45	45	143.75
Premixes (not included in analysis)	20	20	20	20
Total	1000	1000	1000	1000

note: C = conventional, OA = organic crop ingredients/conventional fish and poultry ingredients, OBP = organic crop ingredients/fisheries by-product ingredients, ORF = organic crop ingredients/reduced fisheries ingredients.

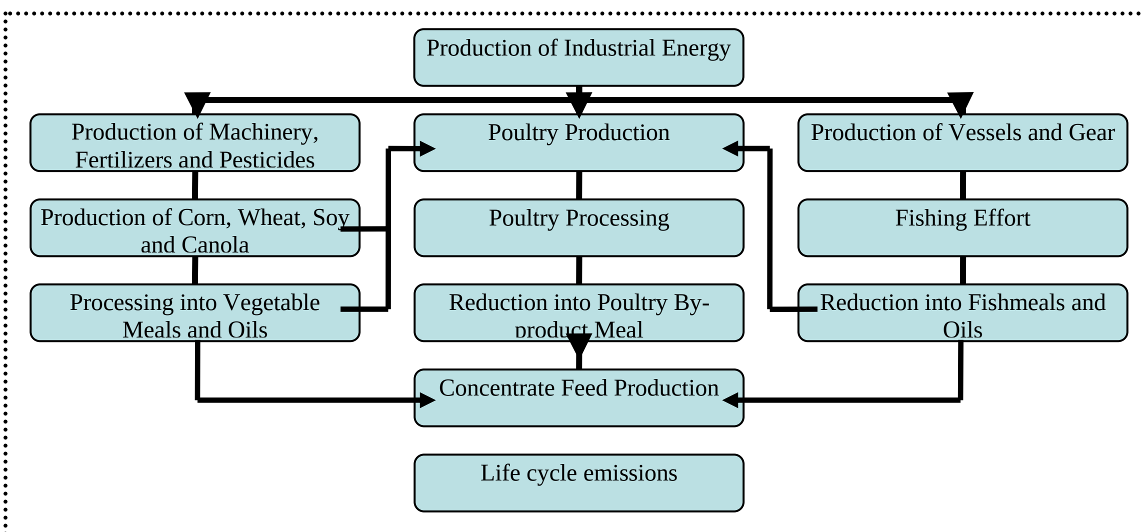

Figure 3.1 System boundaries for a Life Cycle Assessment of salmon feed production.

3.4.1 Life Cycle Inventory

The Life Cycle Inventory (LCI) stage of LCA involves collecting relevant data regarding the material and energetic inputs and emissions associated with each stage of the product life cycle. Although a variety of LCI databases of material inventories are currently available for many products and processes, to the extent possible efforts were made to rely on primary data from peer-reviewed literature, government statistical reports, industry association reports, and private companies and utilities. For the complete life cycle inventory for this study, including key assumption made, see Appendices A-I.

3.4.2 Allocation

Allocation in LCA is necessary when several products are derived from the same industrial process, but the environmental burdens are to be expressed in relation to only one of the products (Guinee *et al.* 2001). In such cases, the researcher must arrive at a defensible means of allocating burdens between co-products. For the purpose of this study, gross nutritional energy content was used as the allocation criterion to partition environmental impacts in all production systems yielding co-product ingredients. It was felt that this would more accurately represent the actual material/energy flows and associated emissions attributable to the co-product streams compared to alternative allocation criteria such as the relative mass or economic value of co-products (see Ayer *et al.* 2006).

3.4.3 Impact Assessment

Impact Assessment, which is the third stage of a LCA, involves calculating the potential environmental burdens associated with specific life cycle activities by quantitatively expressing all inputs and emissions tabulated in the Life Cycle Inventory according to their contributions to a suite of specified environmental impact categories (Pennington *et al.* 2004). The impact categories considered in this study were Energy Use, Global Warming Potential, Acidification Potential, Eutrophication Potential, Marine Aquatic Ecotoxicity Potential and Biotic Resource Use (Table 3.2). With the exception of Biotic Resource Use, all impacts were calculated using the SimaPro 7.0 LCA software package from PRé Consultants (PRé 2006). Biotic Resource Use for agricultural ingredients was calculated based on the carbon content of the fraction of crop used following Papatryphon *et al.* (2004). These were 460, 528, 607, and 465 g C/kg crop (dry matter) for wheat, soybean, canola and corn respectively. Biotic Resource Use for fishery-derived ingredients was calculated following Pauly and Christensen (1995) according to the formula $P = (M/9) \times 10^{(T-1)}$, where P is the kg of C (representing Net Primary Productivity) appropriated, M is the mass of fish required (wet weight), and T is the trophic level of the organism used.

Table 3.2. Impact categories and characterization factors employed.

Impact Category	Description	Characterization Factor
Global Warming	contributes to atmospheric radiative forcing	CO_2 equivalents
Acidification	contributes to acid deposition	SO_2 equivalents
Eutrophication	contributes to Biological Oxygen Demand	PO_4 equivalents
Marine Eco-toxicity	contributes to conditions toxic to marine flora/fauna	1,4-DCB equivalents
Energy Use	use of industrial energy	MJ equivalents
Biotic Resource Use	appropriation of net primary productivity	C

The life cycle impacts of delivering one tonne of each of the conventional and organic crop ingredients and animal-derived ingredients to the mill gate were calculated and comparisons were made between conventional and organic crop ingredients, and between vegetable meals/oils and fishmeals/oils. The life cycle impacts of feed ingredients were also compared by sector of origin *(i.e.* the average life cycle impacts associated with delivery to the mill gate of conventional crop, fisheries, and poultry-derived ingredients) and the cumulative life cycle impacts of producing each of the four hypothetical feeds considered were evaluated. In addition, the relative contributions of ingredient production, processing and transportation to the overall industrial energy used to produce one tonne of conventional salmon feed were assessed.

The total life cycle impacts associated with the provision of sufficient feed to grow one tonne of conventionally or organically produced Atlantic salmon were calculated, assuming a gross feed-to-flesh conversion ratio (FCR) of 1.3:1 (as per Ellingsen and Aanondsen 2006). In combination with data for the non-feed related inputs to net-cage Atlantic salmon aquaculture reported by Tyedmers (2000), this information was used to compare the edible protein energy return on industrial energy investment (EROI) associated with both conventional and organic salmon aquaculture production as well as published edible protein EROI values for a range of other food production systems. Biotic resource use efficiency *(i.e.* the amount of primary production, as measured in tonnes of carbon, appropriated to produce sufficient feed to grow one tonne of salmon), and the greenhouse gas intensity per tonne of salmon produced were also calculated and compared with reported values for other food products. In addition, values were calculated for the FCRs at which the feed-related environmental burdens of organic salmon production using the three organic feeds would be equivalent to those of salmon production employing conventional feeds at an FCR of 1.3:1. Finally, a sensitivity analysis was conducted to evaluate the impact of using alternative allocation criteria (mass and economic value).

3.5 Results

3.5.1 Feed Ingredients

Of the conventional agricultural ingredients used in BC salmon feeds, corn gluten meal and canola oil were the poorest performers (*i.e.* generated the highest impacts) overall. For corn gluten meal this was due to the high processing energy inputs and, in the case of canola, the considerable nitrogen fertilizer used in crop production. These were followed by canola seed, canola meal, soy meal and wheat. Specifically, wheat contributed on average 8% across impact categories relative to the combined impacts for each category of all of the crop-derived ingredients, while soy meal contributed 12%, canola seed 15%, canola meal 12%, canola oil 26%, and corn gluten meal 27% (Table 3.3).

Among fisheries-derived ingredients, Canadian by-product fishmeals and oils from processing wastes of the BC herring fishery performed poorest due to the higher fuel intensity of the fishery and the low meal/oil yield rates achieved from the fishery by-products. US fish oil from a dedicated menhaden reduction fishery, which had the lowest fuel inputs, outperformed Peruvian reduction fishery oil (anchoveta) in all impact categories besides Marine Aquatic Ecotoxicity and Acidification (Table 3.3). Although Canadian poultry by-product meal contributed least to Biotic Resource Use, it ranked highest in contributions to Acidification and Marine Aquatic Ecotoxicity, and was second to Canadian by-product fish oil in contributions to Energy Use and Global Warming Potential. These impacts were largely generated during poultry production. Overall, Peruvian fish meal contributed on average 6% across impact categories relative to the combined impacts of all animal-derived ingredients, while Peruvian fish oil contributed 11%, US fish oil 9%, Canadian by-product fishmeal 18%, Canadian by-product fish oil 34% and Canadian poultry by-product meal 22%. Excluding Biotic Resource Use, fisheries-derived ingredients generated only 60% of the impacts associated with the production of poultry by-product meal (Table 3.3).

Soy meal was found to generate consistently lower impacts compared to US and Peruvian fishmeals. On average, soy meal generated 32% and 49% of the impacts associated with Peruvian and US fishmeals respectively (Table 3.3). Similarly, canola oil performed much better than fish oils, generating on average 51% and 58% of the impacts associated with Peruvian and US fish oils. However, global warming impacts for canola oil were 13% higher than for US fish oil (Table 3.3). This was heavily influenced by the substantial nitrogen fertilizer requirements of canola cultivation, because the synthesis of N fertilizer requires considerable energy and its use results in the emission of nitrous oxide, a powerful greenhouse gas.

All organic crop ingredients except wheat generated slightly higher Marine Aquatic Ecotoxicity impacts compared to equivalent conventional ingredients, largely as a consequence of using phosphate rock (the beneficiation of which is energy-intensive) in place of conventional phosphorous fertilizer. However, organic ingredients outperformed conventional ingredients in every other impact category (Table 3.3, Figure 3.2). Specifically, Marine Aquatic Ecotoxicity impacts of organic crop ingredients averaged 111% of those generated by equivalent conventional ingredients while aggregate Energy Use averaged only 65% relative to conventional ingredients, Global Warming 56%, Acidification 73% and Eutrophication 66%. For individual ingredients (excluding Biotic Resource Use, which is necessarily equivalent), average differences across impact categories were greatest for organic wheat (63% of the impacts associated with conventional wheat), intermediate for canola (71%), canola oil (73%), canola meal (74%), and soy meal (78%), and least for corn gluten meal (87%) (Figure 3.2). These differences were largely due to the use of green manure production to supply nitrogen to organic crops as opposed to the synthetic nitrogen fertilizers used in conventional production.

Comparing the magnitude of average life cycle impacts generated by delivering one tonne of ingredients to the mill gate by sector of origin *(i.e.* average impacts of all conventional crop, fisheries, and poultry-derived ingredients), crop-derived ingredients contributed .9% to Biotic Resource Use relative to fish-based (90%) and poultry-derived (9.1%) ingredients (Figure 3.3). Excluding Biotic Resource Use, the crop-derived agricultural ingredients used in conventional salmon feeds contributed, on average, 8.7% across the impact categories considered relative to fishery and poultry-derived ingredients while fishery-derived ingredients contributed 34.1% and poultry by-product meal 57.2%. The relative magnitude of contributions made by crop, fishery and poultry-derived ingredients were largely consistent across impact categories (Figure 3.3).

Table 3.3 Life Cycle Impact Assessment results for ingredients used in BC salmon feeds (brackets indicate % relative to conventional ingredients, bars indicate relative proportions of impacts contributed by raw material production, processing and transportation stages).

CROP-DERIVED INGREDIENTS	EU MJ equiv	GWP kg CO_2 equiv.	AP kg SO_2 equiv.	EP kg PO_4 equiv.	MAEP kg 1,4-DCB equiv.	BRU kg C
Conventional Production						
Wheat	3 020	429	2.95	0.91	9 880	460
Canola Seed	5 510	802	5.07	3.08	16 700	607
Canola Meal	4 630	584	5.84	2.12	19 000	404
Canola Oil	9 860	1 260	9.16	4.64	41 600	887
Soy Meal	3 990	333	3.24	2.89	24 400	429
Corn Gluten Meal	12 800	960	11.1	2.4	98 000	618
Organic Production						
Wheat	1 290 (43)	184 (43)	1.90 (64)	0.65 (71)	9 090 (92)	460 (100)
Canola Seed	2 860 (52)	881 (48)	3.25 (64)	1.94 (63)	21 700 (130)	607 (100)
Canola Meal	2 860 (62)	303 (52)	3.04 (52)	1.37 (65)	22 400 (118)	404 (100)
Canola Oil	5 980 (61)	643 (51)	6.50 (71)	2.97 (64)	49 000 (118)	887 (100)
Soy Meal	3 440 (86)	221 (66)	2.45 (76)	1.63 (56)	25 800 (106)	429 (100)
Corn Gluten Meal	11 400 (89)	725 (76)	10.10 (91)	1.78 (74)	100 000 (102)	618 (100)

FISH-DERIVED INGREDIENTS	EU	GWP	AP	EP	MAEP	BRU
Peruvian Fish Meal	15 500	1 050	6.79	3.59	43 000	19 880
Peruvian Fish Oil	27 000	1 830	12.10	6.65	79 700	37 340
CDN By-product Fish Meal	40 000	2 690	36.50	10.50	76 400	79 420
CDN By-product Fish Oil	75 100	5 040	68.50	19.60	143 000	149 100
US Fish Oil	18 200	1 110	15.40	5.71	142 000	8 230

POULTRY-DERIVED INGREDIENTS	EU	GWP	AP	EP	MAEP	BRU
Poultry By-product Meal	50 500	4 260	35.7	21.3	177 000	5 960

Production Processing Transportation

note: EU = Energy Use, GWP = Global Warming Potential, MAEP = Marine Aquatic Ecotoxicity Potential, AP = Acidification Potential, EP = Eutrophication Potential, BRU = Biotic Resource Use.

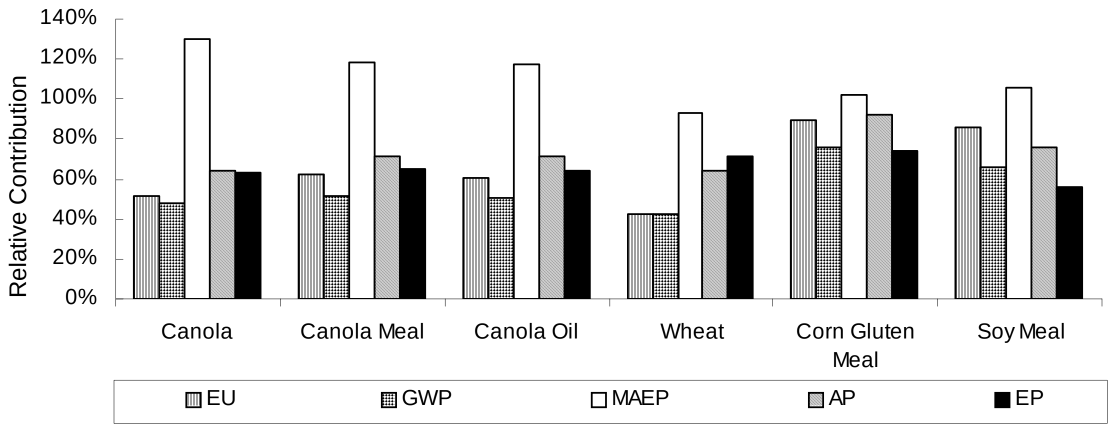

Figure 3.2 Comparative cradle-to-gate life cycle impacts of organic agricultural feed ingredients delivered to a BC feed mill as a percentage of impacts generated by producing equivalent conventional agricultural feed ingredients. EU = Energy Use (MJ equiv.), GWP = Global Warming Potential (CO_2 equiv.), MAEP = Marine Aquatic Ecotoxicity Potential (1,4-DCB equiv.), AP = Acidification Potential (SO_2 equiv.), EP = Eutrophication Potential (PO_4 equiv.).

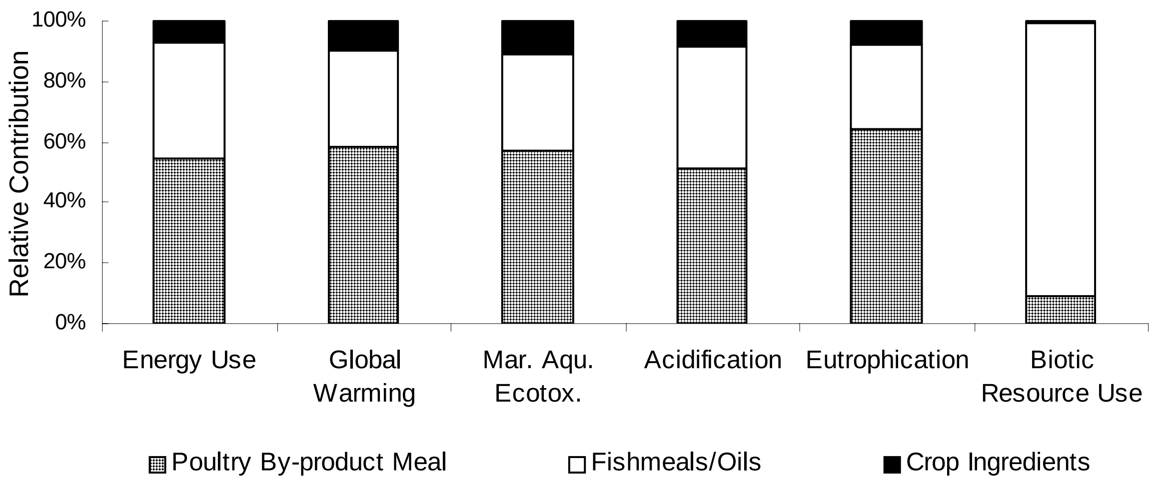

Figure 3.3 Average cradle-to-gate life cycle impacts of salmon feed ingredients by sector of origin *(i.e.* one tonne each of poultry by-product meal, fish meals and oil, and conventional agricultural crops). EU = Energy Use (MJ equiv.), GWP = Global Warming Potential (CO_2 equiv.), MAEP = Marine Aquatic Ecotoxicity Potential (1,4-DCB equiv.), AP = Acidification Potential (SO_2 equiv.), EP = Eutrophication Potential (PO_4 equiv.), BRU = Biotic Resource Use (Carbon).

3.5.2 Feed Formulations

With the exception of Marine Aquatic Ecotoxicity (higher) and Biotic Resource Use (identical), substituting organic for conventional crop ingredients (OA) resulted in only minor improvements across impact categories, with an average of 96.6% of the impacts of the conventional formulation (C) overall (Table 3.4, Figure 3.4). Substituting by-product fish meals and oils for both meals and oils derived from reduction fisheries as well as poultry by-product meal (OBP) resulted in much poorer performance (*i.e.* higher impacts) in every impact category measured. This feed generated 190% of the impacts associated with the conventional (C) feed (Table 3.4, Figure 3.4). The reduced fish meal/oil formulation (ORF), in which vegetable oil replaced fish oil, soy meal substituted for 25% of the fish meal, and no poultry by-product meal was used, demonstrated considerably superior performance, generating on average 54% of the impacts of the conventional (C) formulation (Table 3.4, Figure 3.4).

Table 3.4 Total cradle-to-gate life cycle impacts associated with producing one tonne of each of four alternative feed formulations (brackets indicate % relative to conventional feed, bars indicate relative proportions of impacts contributed by crop ingredients, fish/poultry ingredients and feed production at the feed mill).

Formulations	EU MJ equiv	GWP kg CO_2 equiv.	AP kg SO_2 equiv.	EP kg PO_4 equiv.	MAEP kg 1,4-DCB equiv.	BRU kg C
C	18 100	1 400	12.6	5.3	60 700	10 600
OA	17 100 (95)	1 250 (89)	11.8 (94)	4.9 (93)	61 100 (102)	10 600 (100)
OBP	26 900 (149)	1 810 (129)	24.6 (196)	6.7 (127)	63 300 (104)	45 100 (426)
ORF	9 860 (54)	690 (49)	6.9 (55)	2.3 (43)	47 600 (78)	6 300 (60)

Crop Ingredients   Fish/Poultry Ingredients   Feed Mill

note: C = conventional (average BC salmon feed), OA = organic crop ingredients/conventional animal meals and oils, OBP = organic crop ingredients/fisheries by-product meals and oils, and ORF = organic crop ingredients/no poultry by-product meal, 25% of fishmeal substituted with organic soy meal, and 100% of fish oil substituted with organic canola oil. % of impacts relative to C are given for OA, OBP, and ORF.

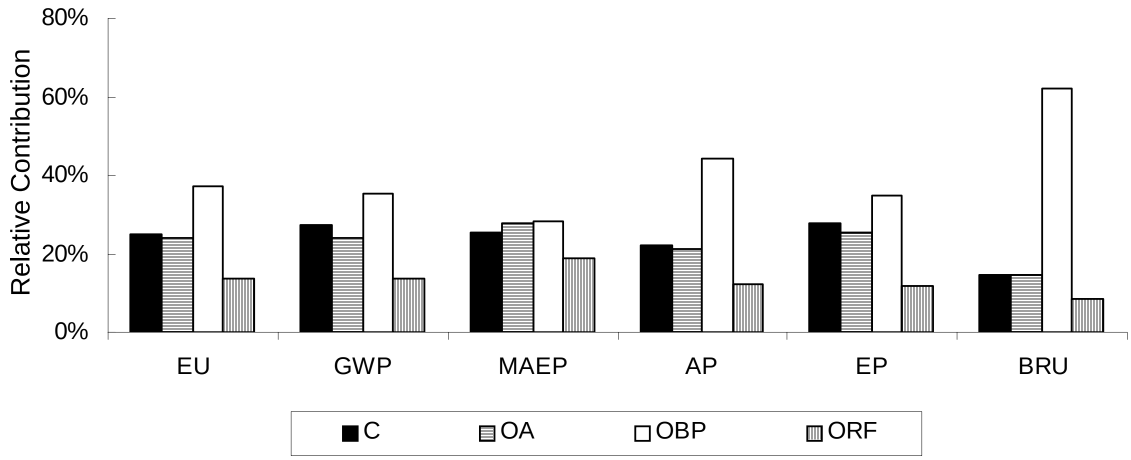

Figure 3.4. Comparative cradle-to-gate life cycle impacts of producing one tonne of each of four alternative salmon feed formulations in a BC feed mill. EU = Energy Use (MJ equiv.), GWP = Global Warming Potential (CO_2 equiv.), MAEP = Marine Aquatic Ecotoxicity Potential (1,4-DCB equiv.), AP = Acidification Potential (SO_2 equiv.), EP = Eutrophication Potential (PO_4 equiv.), and BRU = Biotic Resource Use (Carbon). C = conventional (average inputs to BC salmon feed), OA = organic crop ingredients/conventional animal meals and oils, OBP = organic crop ingredients/fisheries by-product meals and oils, and ORF = organic crop ingredients/no poultry by-product meal/25% of fishmeal substituted with organic soy meal/100% of fish oil substituted with organic canola oil.

Production of primary materials from agricultural, fisheries and poultry husbandry accounted for 54% of total energy use associated with producing one tonne of conventional salmon feed, with fisheries and poultry ingredients requiring considerably more energy than crop ingredients, despite the fact that the feed contained 50% plant-derived materials by mass. Processing accounted for 29% of total energy use. The energy required to process fish into meals and oils was twice that of poultry and crop ingredients. All transportation steps combined contributed only 5% to total energy demand, while the milling of ingredients into a concentrate salmon feed accounted for 11% of energy use (Figure 3.5).

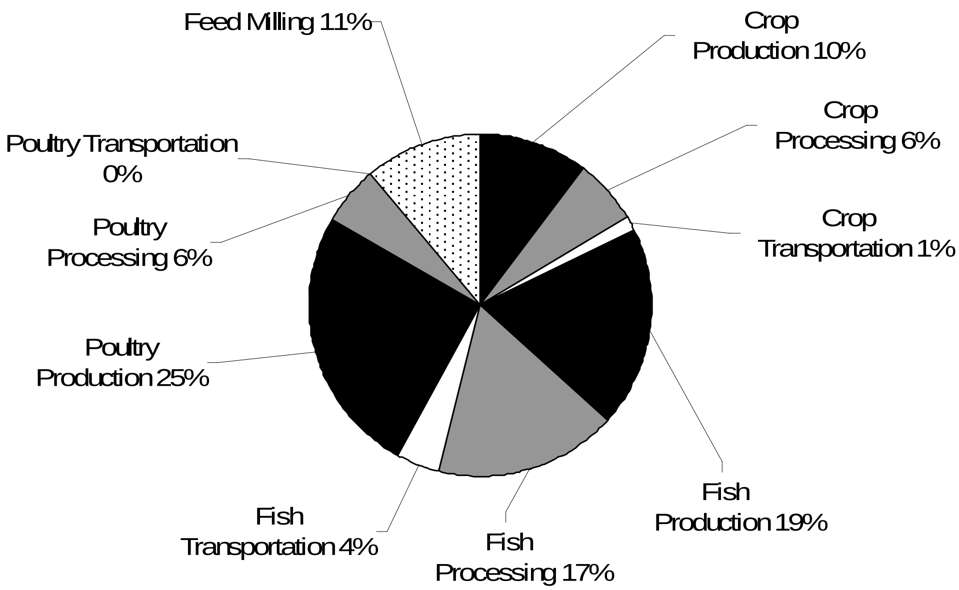

Figure 3.5. Breakdown of industrial energy inputs to the production, processing and transportation of feed ingredients used in one tonne of conventional salmon feed as well as energy for feed milling (total energy inputs = 18,100 MJ/tonne).

3.5.3 Energy Consumption, Biotic Resource Use Efficiency and Carbon Intensity of Salmon Production

The total amount of industrial energy associated with the production of sufficient feed to grow one tonne of farmed salmon (assuming a gross FCR of 1.3:1) using conventional feeds was 23,500 MJ tonne, but varied from as low as 12,900 MJ/tonne using ORF feeds to as high as 35,100 MJ/tonne using OBP feeds. Including 3,400 MJ/tonne of salmon as non feed-related inputs to grow-out (Tyedmers 2000), predicted edible protein EROI in salmon aquaculture ranged from as high as 17.8% when ORF feeds were used, to as low as 7.8% using OBP feeds (Figure 3.6).

In general, biotic resource use in conventional salmon aquaculture was similar to that calculated for lower trophic level salmon fisheries and less than that of higher trophic level salmon fisheries, although it was much higher when OBP feeds were used and substantially lower using ORF feeds (Figure 3.7).

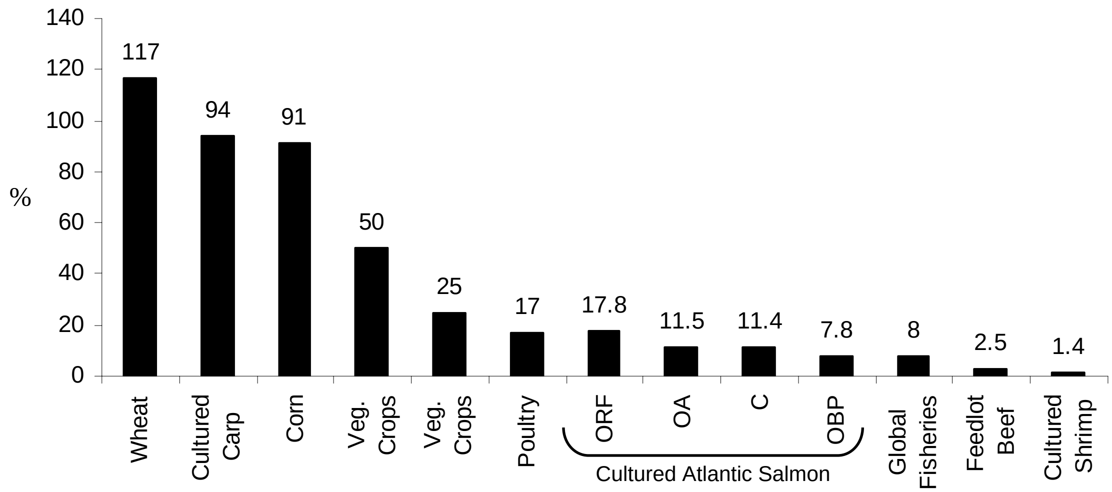

Figure 3.6 % Edible protein energy return on industrial energy investment (EROI) in the production of Atlantic salmon using four alternative feed formulations (FCR 1.3 assumed) compared with EROI in other food production systems. Data for wheat, corn, poultry and cultured salmon from this study; all others as summarized in Troell *et al.* (2004) and Tyedmers *et al.* (2005). EROI calculated assuming 65% of carcass is edible, 20% of edible fraction is protein, and energy density of protein is 23.6 MJ/kg.

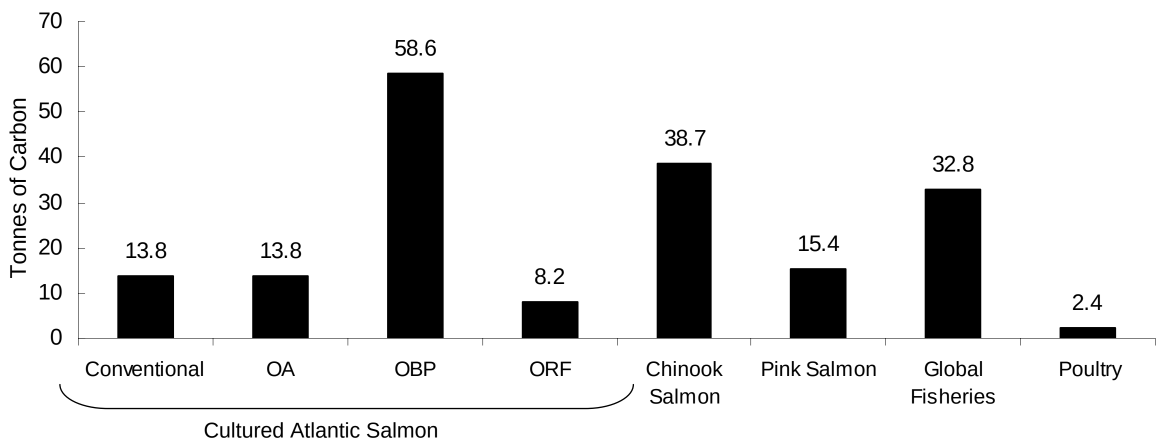

Figure 3.7. Biotic Resource Use (as measured in tonnes of carbon/tonne of product) in alternative meat production systems. Note: assumes a feed conversion ratio of 1.3:1 for Atlantic salmon (this study); data for Chinook and pink salmon in the BC salmon fishery from Tyedmers (2000); data for global fisheries for human consumption assumes a mean trophic level of 3.47; data for poultry from this study. C = conventional (average inputs to BC salmon feed), OA = organic crop ingredients/conventional animal meals and oils, OBP = organic crop ingredients/fisheries by-product meals and oils, and ORF = organic crop ingredients/no poultry by-product meal/25% of fishmeal substituted with organic soy meal/100% of fish oil substituted with organic canola oil.

The greenhouse gas intensity of farmed salmon production ranged from 1.2 – 2.7 tonnes of carbon dioxide equivalents per tonne of salmon produced (Figure 3.8). These values compared favourably with reported greenhouse gas emissions associated with sheep, beef and pork production, and were similar to reported values for poultry production.

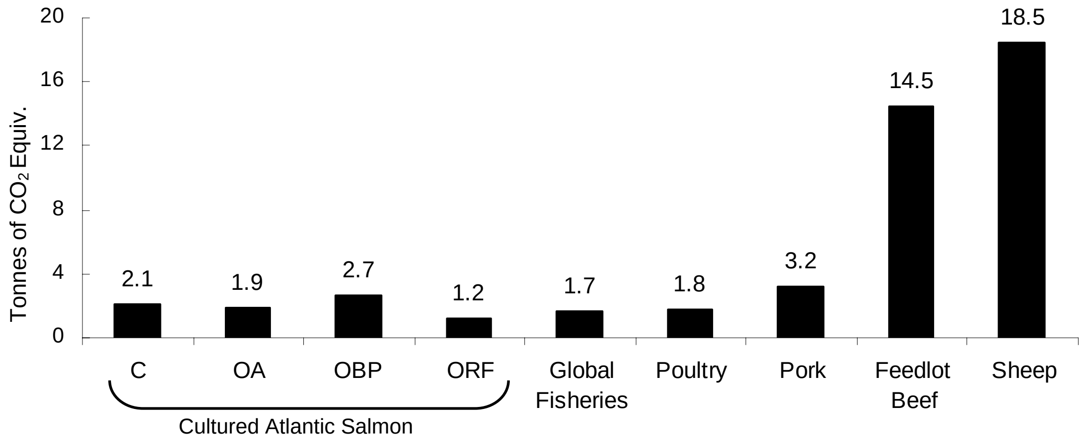

Figure 3.8 Life cycle greenhouse gas emission intensities (in tonnes of CO_2 equivalents) per live weight tonne in alternative meat production systems (values for sheep, beef, pork and poultry as reported by Nemry *et al.* 2001).

To result in equivalent impacts to conventional salmon production, FCR in organic salmon production would have to average 1.35:1 using OA feeds, 0.90:1 using OBP feeds and 2.38:1 using ORF feeds (Table 3.5).

Table 3.5 Feed conversion ratios (FCR) at which the use of alternative feed formulations for salmon aquaculture would result in equivalent feed-related impacts to using conventional feeds at an FCR of 1.3:1.

Impact Category	**FCR**		
	OA	**OBP**	**ORF**
Energy Use (MJ equiv.)	1.37	.87	2.37
Global Warming (CO_2 equiv.)	1.46	1	2.62
Marine Aquatic Ecotoxicity (1,4-DCB equiv.)	1.18	1	1.76
Acidification (SO_2 equiv.)	1.37	1.16	2.36
Eutrophicatin (PO_4 equiv.)	1.4	1.03	3.0
Biotic Resource Use (carbon appropriation)	1.3	.31	2.18
Average	**1.35**	**.90**	**2.38**

A sensitivity analysis was conducted to evaluate the potential impacts of using alternative allocation criteria on the results of this study. This was accomplished by running three separate iterations of the impact assessment phase (in SimaPro 7.0) using gross nutritional energy, mass, or economic value of co-products as the allocation criterion to partition environmental burdens (Table 3.6). Not surprisingly, it was found that economic allocation resulted in a lower attribution of impacts to feeds containing animal by-products since these products have little economic value compared to the primary meat products for human consumption. Conventional feeds containing reduction fisheries meals and oils as well as poultry by-product meal averaged 70% of the impacts measured when economic values were used to allocate burdens. In contrast, the feed in which all animal-derived ingredients were by-product fishmeals and oils generated only 39% of the average life cycle impacts, while the feed containing reduced proportions of fishmeals/oils and no animal by-products was attributed a greater share of environmental burdens than when gross energy content was used as the allocation criteria. Using mass allocation generally resulted in slightly higher attribution of impacts, suggesting that the co-product streams destined for human consumption were of higher nutritional energy content than those used in salmon feeds.

Table 3.6 Sensitivity analyses for the comparative average life cycle impacts of alternative salmon feeds when calculated using different allocation criteria.

Allocation Criteria	Comparative Average Life Cycle Impacts			
	C	OA	OBP	ORF
Gross Energy	1	1	1	1
Economic	.70	.69	.39	1.09
Mass	1.05	1.06	.94	1.04

note: C = conventional (average inputs to BC salmon feed), OA = organic crop ingredients/conventional animal meals and oils, OBP = organic crop ingredients/fisheries by-product meals and oils, and ORF = organic crop ingredients/no poultry by-product meal/25% of fishmeal substituted with organic soy meal/100% of fish oil substituted with organic canola oil.

3.6 Discussion

3.6.1 Comparative Performance of Feed Ingredients and Formulations

Feed ingredients derived from fisheries and poultry production generate considerably higher environmental burdens then do ingredients from agricultural production systems. Given that poultry production is underpinned by feed inputs from both fisheries and agriculture, the higher costs of poultry by-product meal compared to crop ingredients are hardly surprising. After all, regardless of the levels of efficiency that might be achieved in poultry production, even the briefest of nods to thermodynamic principles forces the recognition that any animal husbandry system will necessarily consume more energy (in crops) than it generates (in animal products) due to the inefficiencies inherent in biological feed conversion. Since poultry by-product meal is attributed a share of the impacts associated with poultry feed production as well as direct energy inputs to poultry production, processing and rendering, its life cycle impacts are substantial.

The relationship between the relative environmental burdens of fisheries and agriculture is less intuitive. Fisheries require material and energy inputs for vessels, gear and fuel in order to harvest resources that are often widely dispersed but whose productivity is otherwise underpinned by largely unmanaged ecosystem processes. These resources are, hence, essentially “free” in terms of additional required human intervention. Depending on the species fished, the status of the stock, and the technology used, inputs can vary widely. In a comprehensive review of fuel inputs to fishing fleets, Tyedmers (2004) reported a range of 20-3,400 l of fuel/tonne of fish or shellfish landed.

In contrast, industrial agricultural production involves the intensive management of comparatively small areas of land in order to produce high yields from crop species selected for this purpose. Here, material and energy inputs for farm machinery, cultivation, fertilizers, and plant protection are used to increase yields to levels higher than what would otherwise be achieved. In other words, industrial inputs partially replace and augment ecosystem services. However, although the degree of intervention in facilitating the growth of the harvested crop is much higher, it is not necessarily as energy intensive as many fisheries. Reported edible protein energy returns on industrial energy investment (EROI) for vegetable crops range from 25-50%, while global fisheries average an 8% return (Troell *et al.* 2004; Tyedmers *et al.* 2005). Amongst the agricultural systems referred to in this study, total energy inputs ranged from 1030 MJ/tonne for organic wheat cultivation to 5250 MJ/tonne for conventionally grown canola whilst for fisheries, energy inputs ranged from 1970 MJ – 5980 MJ/tonne landed for menhaden and herring respectively (Table 3.3).

Of course, the life cycle impacts of feed ingredients are not derived exclusively from the production stage. Processing can also figure prominently. For example, although corn production is less energy intensive than canola production, the wet-milling of corn into corn gluten meal and its associated co-products requires considerably more energy than does the pressing and extraction of canola meal and oil from canola seed. For this reason, the life cycle energy use associated with the production of corn gluten meal is more than twice that of canola meal, which in turn drives higher acidification and ecotoxicity impacts (Table 3.3). Similarly, the processing of fish into meals and oils may require more energy and generate greater impacts per tonne than does fish harvest, as in the case of the US menhaden reduction fishery (Table 3.3).

In general, the transportation stages contributed minimally to the overall life cycle impacts of salmon feed production because rail and ocean freight transport were assumed over long distances (Table 3.3). Nonetheless, transportation-related impacts for feed production could potentially be substantial, depending on the distances traveled and the mode of transportation. Long-haul oceanic shipping is much less energy-intensive than rail freight, which in turn performs much better than truck transport (Schipper *et al.* 1997). An ingredient that travels thousands of kilometers by ship might therefore have similar transport-related impacts to one transported a fraction of that distance by rail. For example, shipping one tonne of fishmeal via ocean freighter from Peru to BC requires a specific amount of energy and generates a characteristic set of impacts. Shipping an equivalent mass of corn gluten meal half that distance from Ontario by train performs similarly. Shipping via tractor trailer rather than rail would be twice as impactful. Overall, the addition of processing and transport-related impacts can mean that ingredients at the feed mill gate may have much higher life cycle impacts than those associated strictly with the production of the primary materials.

This insight is key to understanding why, despite the fact that organic crop production outperforms conventional crop production by a large margin in all impact categories except Marine Aquatic Ecotoxicity (Table 3.3), the substitution of conventional with organic ingredients in salmon feeds results in only small improvements in environmental performance (average 4.3% across impact categories) (Table 3.4). In essence, where transport and processing-related inputs are high, they can effectively dilute the magnitude of gains made in the field by using organic methods. The apparently minor differences between the life cycle impacts of conventional and organic corn gluten meal up to delivery to the feed plant provide a case in point. The transport and processing costs for these ingredients are identical, and contribute 64% to the life cycle impacts of the conventional ingredient and 73% for the organic ingredient (Table 3.3). Conversely, if transport and processing costs are minimal relative to crop production costs, the gains can be much greater – as in the case of organic versus conventional wheat, where the life cycle impacts of the former up to the point of inclusion in salmon feed average 57% of the impacts of the latter (Table 3.3). Unfortunately, feed ingredients like corn gluten meal are valued for the high protein density achieved through processing whereas ingredients like wheat, included partly for use as a binding agent (New 1987), have lower protein density and thus reduced potential for substitution. Organic soy meal and canola meal, which are protein-rich but have medium-intensity processing costs and generate 74% and 68% of the impacts of their conventional equivalents (Table 3.3), may present best-case substitution opportunities for crop ingredients.

The life cycle impacts of fisheries-derived ingredients may exhibit even greater variability. Not only do fuel inputs differ substantially between fisheries in general and even between reduction fisheries (Tyedmers 2004), the environmental costs of reduction can be radically different depending on the energy-efficiency of the reduction plant, the kinds of fuel used to power reduction operations and, most importantly, the fishmeal and oil yields achieved. In addition, transport modes and distances are highly distinct for each. Of the fish-derived products considered in this study the fisheries stage contributed, on average, 54% of the life cycle impacts (excluding Biotic Resource Use), while reduction into meals and oils contributed 42% and transportation 4% (Table 3.3). However, the relative performance of different meals and oils differed tremendously. BC herring meal and oil is derived from the least fuel efficient fishery and has the highest energy costs of reduction due to low yield rates. Thus, although these products have little distance to travel to reach a BC feed mill compared to meals and oils from Peru and the southern US, their life cycle impacts are substantially higher. Biotic Resource Use for these products is also much greater because herring occupy a higher trophic level than do menhaden and anchoveta (FishBase 2006). In contrast, because the US menhaden fishery is less energy intensive and yield rates for meals and oils are much greater, the life cycle impacts of these products are minimal in most impact categories compared to the herring products, despite the greater distances traveled.

Like herring meal and oil, BC poultry by-product meal has minimal transport costs but the impacts of producing the feedstock (poultry processing trimmings) are substantial. Although poultry production is generally considered to be a relatively efficient means of producing meat compared to many other forms of animal husbandry (Flachowsky 2002), the energy intensity is similar to the herring fishery (Table 3.3). Substituting poultry by-product meal for fish meal will therefore not result in improved environmental performance in salmon feeds.

Evidently, producing animal-derived feed ingredients for salmon aquaculture has substantially higher environmental costs compared to producing crop ingredients. Due to the growing demand for and limited availability of fishmeals and oils, there is widespread interest in finding acceptable alternatives. Tacon (2005) reports that the aquaculture industry currently consumes 46% of fishmeal and 81% of fish oil produced globally. Moreover, at current rates of expansion, the global aquafeed industry will require 70% of the average historical fishmeal supply and 145% of the fish oil supply by 2015 (New and Wijkstrom 2002). It is therefore worthwhile to carefully consider the magnitude of improvements in environmental performance that might be achieved through using vegetable meals and oils in their place.

Soy meal is generally considered to be the most obvious replacement for animal-derived meals and, for this reason, the prices of fish meal and soy meal have been closely aligned historically. A recent report by Kristofersson and Anderson (2006), however, suggests a break in this relationship due to constraints on fishmeal availability and projects rising prices of fishmeal relative to soy meal into the future. This suggests that, from an economic perspective, substituting soy meal for fish meal should prove increasingly attractive in the long-term. Moreover, soy meal demonstrates consistently better environmental performance than fish meal in all impact categories considered, averaging 30% of the impacts generated by Peruvian fishmeal and 44% of those generated by US menhaden fishmeal (Table 3.3). The use of organic soy meal, which averages 75% of the life cycle impacts of conventional soy meal, would result in even greater improvements (Table 3.3).

The percentage of fishmeal used in salmon feeds has decreased substantially in recent decades, from 60% in 1985 to a present average level of 35% (Tacon 2005). Further substitution of fishmeal with soy meal could result in considerable additional improvement in environmental performance. However, price and environmental performance are not the only factors determining the desirability and degree of substitution that can be achieved. Previous research with Atlantic salmon has suggested that soy protein sources contain less digestible energy than fishmeal (Storebakken *et al.* 1998; Carter and Hauler 2000; Refstie *et al.* 2001; Opstvedt *et al.* 2003), but growth performance was unaffected in Atlantic salmon fed diets containing 20-33% soy bean meal (Olli *et al.* 1995; Carter and Hauler 2000). Advances in methods to overcome anti-nutritional factors may substantially increase potential substitution rates in the future (Olli *et al.* 1994; Refstie *et al.* 1999, 2001, 2005).

The decreasing level of fishmeal inclusion in salmon feeds has been accompanied by equivalent increases in dietary lipid levels, with inclusion rates rising from an average of 10% in 1985 to 35-40% in 2005 (Tacon 2005). Much of this has been achieved through greater use of fish oils - to the extent that demand by the aquaculture industry for fish oils will soon exceed supplies (New and Wijkstrom 2002). Considerable research efforts have therefore also been invested in finding suitable vegetable oil replacements for fish oils

(Glencross 2003). Similar to substitution of fishmeals, replacing fish oil with vegetable oil can lead to marked improvements in the environmental performance of salmon feeds. For example, canola oil outperforms Peruvian and US menhaden fish oils in almost all impact categories considered, generating on average 37% and 42% of the impacts of Peruvian and US oils respectively (Table 3.3). Using organic canola oil would further increase the environmental benefits of such substitution (Table 3.3). However, although inclusion of significant amounts of canola oil and other vegetable oils may not compromise growth (Cho et al. 1974; Dosanjh et al. 1984, 1988,1996; Thomassen and Røsjø 1989; Polvi and Ackman 1992; Koshio et al. 1994; Guillou et al. 1995), tissue composition is affected (Thomassen and Rosjo, 1989; Green and Selivonchek, 1990; Dosanjh *et al.* 1998). Vegetable oils such as canola oil and soy oil do not contain an ideal profile of long-chain polyunsaturated omega fatty acids, which is desirable for the health of both the cultured organism and consumers of fish products (Glencross 2003). One possible strategy to reduce fish oil use in aquafeeds and still ensure high levels of omega fatty acids is to include fish oil in feeds during the final stage of the production cycle only (Glencross 2003).

Due to the varied environmental performance of crop and animal-derived ingredients, alternative salmon feeds will have very distinct life cycle impacts depending on the formulation employed. It was found that substituting organic for conventional crop ingredients resulted in only minor improvement to the environmental performance of feed production. This was due to the much larger impacts associated with the production of animal-derived ingredients, which overwhelmed the gains associated with organic/conventional crop substitution. Improving the environmental performance of salmon feeds will therefore largely be a factor of reducing the fraction of animal-derived ingredients employed. Using fisheries by-product meals and oils over conventional meals and oils – a practice required by most current organic aquaculture standards – will only further compromise environmental performance in terms of the impact categories considered in this study.

3.6.2 Comparisons with Other Studies

Several authors have previously investigated the environmental performance of aquaculture feeds. Using energy analysis, Tyedmers (2000) reported that the production of one tonne of feed for conventional salmon aquaculture in British Columbia appropriated 48,082 MJ of fossil fuel equivalents. In contrast, the results of this study indicate that total industrial energy use for conventional feed production in BC is currently 18,100 MJ per tonne. Much of this difference can be attributed to changes in feed composition *(i.e.* origins and amounts of animal-derived ingredients used) as well as substantially lower energy costs of production for poultry by-product meal due to the use of updated feeds, production energy, and FCR data.

Using LCA to evaluate alternative feeds for trout aquaculture, Papatryphon *et al.* (2004) reported that energy use for feed production ranged from 17,800 to 21,000 MJ per tonne of trout produced. Energy use was highest for feeds containing conventional reduction fisheries meals and oils, and lowest for feeds containing by-product meals and oils. However, as the authors indicate, these results reflect the use of economic allocation to partition environmental burdens between fish products for human consumption and by-products used in meals and oils – which resulted in a very low share of burdens being assigned to the by-product meals. In contrast, using gross energy content to allocate burdens among co-products, the results of this study show that feeds containing fisheries by-products consume considerably more energy and generate higher impacts than all other feeds considered. It is the opinion of the authors of the present study that allocation by energy content provides a much more realistic representation of actual biophysical flows and associated environmental impacts for alternative feed production strategies (see Ayer *et al.* 2006). Moreover, given that the results reflect stable biophysical relationships between co-products, they should be of use to others interested in modeling similar systems elsewhere, regardless of time and geographical context.

Overall, the provision of sufficient feed to produce one tonne of conventionally farmed salmon (live weight) requires more energy and generates higher global warming and acidification impacts than those associated with feeds for the production of pig (Carlsson-

Kanyama 1998) and poultry (Spies *et al.* 2002). This relationship is reversed if salmon are produced using feeds containing less animal-derived ingredients.

3.6.3 Environmental Performance in Salmon Aquaculture as a Function of Feed Use

Since feeds are major drivers of environmental performance in salmon aquaculture generally, it is worthwhile comparing the magnitude of impacts and potential efficiency gains associated with growing salmon using alternative feeds, and how these compare with other industrial food production systems. Edible protein energy return on industrial energy investment (EROI) provides a simple measure of efficiency for such comparisons (Mitchell and Cleveland 1993; Troell *et al.* 2004; Tyedmers *et al.* 2005). Reported EROI values, as well as those measured in this study, vary widely depending on the production system. Among published reports, extensive carp culture shows the highest returns at 94% edible protein EROI, while intensive feedlot beef and shrimp aquaculture production achieve as little as 1.4-2.5% returns. Vegetable crop production ranges from 25-50% EROI (summarized in Troell *et al.* 2004). The results of this study indicate that edible protein EROI in salmon aquaculture can range from 7.8-17.8%, depending on the feed formulation employed. This is markedly higher than has been reported in previous studies (Folke 1988; Tyedmers 2000). When feeds containing reduced proportions of animal-derived ingredients are used, EROI is similar to that achieved in broiler poultry production (this study), while the use of fisheries by-product based feeds results in an edible protein EROI similar to the average reported value for global fisheries (Tyedmers *et al.* 2005).

Given the finite nature of energy resources and the environmental impacts associated with producing and consuming industrial energy, EROI should be a central consideration in food policy. However, it is important to note that this measure is not sensitive to energy quality, which can also strongly influence environmental performance. For example, a high-intensity production system that consumes only hydro-electric energy may generate lower impacts than a less intensive system consuming electricity derived from fossil fuel combustion (Ayer *et al. in prep).* This discrepancy can be overcome by simultaneously considering life cycle contributions to environmental issues closely linked to energy use such as acidification, ecotoxicity, and global warming. Given the magnitude of concerns associated with climate change, the latter consideration is particularly pertinent. Conventional salmon aquaculture generates approximately 2.1 tonnes of CO_2 equivalents for every tonne of salmon produced. Using alternative feeds can increase greenhouse gas emissions to 2.7 tonnes of CO_2 equivalents when fisheries by-product meals and oils are employed, or result in decreased production when less animal-derived ingredients are included (1.2 tonnes) (Figure 3.8). These values are similar to reported values for poultry, and much lower than reported values for sheep and beef (Subak 1999; Nemry *et al.* 2001).

Biotic resource use efficiency can also provide a worthwhile measure of comparative environmental performance. Biotic resources are derived almost entirely from photosynthesis by plants and algae, which convert solar energy into chemical energy in the form of carbon-based organic molecules. Due to the inefficiency of biological food conversion, the amount of food energy available to successive consumers decreases as these carbon complexes flow through trophic webs. This implies that food webs are effectively carbon-based energy pyramids, with high biomass of (numerous) primary producers at the bottom, and few, high-level consumers at the top. In this light, trophic dynamics can be interpreted as a competition for carbon resources, which represent the transferable products of primary production.

Human appropriation of global terrestrial net primary productivity (NPP) has been estimated at close to 40% (Vitousek *et al.* 1986), and an estimate of 8% has been advanced for the appropriation of marine NPP by fisheries (Pauly and Christensen 1995). As human populations continue to grow and levels of consumption increase, this appropriation will place increasing strain on ecosystems and biodiversity. Maintaining the integrity of ecosystem goods and services and of biodiversity generally will therefore require concerted efforts to maximize the efficiency of biotic resource use and limit the overall scale of human appropriation. It is thus useful to carefully evaluate biotic resource use within industrial production systems in order to identify ways to improve efficiency and, more generally, to compare efficiencies between competing production technologies.

The efficiency of biotic resource use in aquaculture will largely be determined by the proportion of animal-derived ingredients used in aquafeed. For this reason, herbivorous fish can be expected to yield much higher returns on biotic resource investments relative to carnivorous fish. For example, it was found that the production of one tonne of Atlantic salmon using a conventional aquafeed appropriated the net products of primary production at a rate of 13.75 tonnes of carbon per tonne of salmon produced (Figure 3.7), while using feeds containing by-product fishmeals and oils that are derived from higher trophic level fish and have low fishmeal/oil yield rate increased biotic resource use dramatically. Reducing the proportion of animal-derived ingredients in feeds had the opposite effect.

It is important to note that the actual proximate environmental impacts of carbon appropriation will depend on the relative importance of the extracted resources in the context of local trophic dynamics – in particular, the abundance of the targeted species and their importance to indirectly impacted species. However, the development of a weighting scale to reflect the ecosystem-specific implications of biotic resource use is challenging, and beyond the scope of this paper.

A major assumption that should be considered is that the use of alternative feeds may actually result in higher or lower feed conversion ratios, which would greatly affect the environmental performance of the farming system. It should be noted that efforts were not made to ensure that the feed formulations considered were nutritionally equivalent or biochemically optimal for salmon production. Rather, the intention was to illuminate the environmental implications of using different ingredients in salmon feeds. Feed formulators may therefore be faced with tradeoffs in maximizing fish performance and minimizing the environmental impacts associated with feed production. However, of the four feeds considered, only the organic feed containing reduction proportions of fishmeal/oil differed substantially from the conventional feed (which represented the average inputs to all salmon feeds produced over the course of one year by a major B.C. feed mill). Moreover, although reductions in the gross FCR for Atlantic salmon grown with feeds containing varying proportions of soy proteins have been reported by several authors (Olli *et al.* 1994, 1995; Storebakken *et al.* 1998; Opstvedt *et al.* 2003), no reductions have been observed when purified soybean concentrate was used (Olli *et al.* 1994; Refstie *et al.* 2001). Furthermore, Carter and Hauler (2002) reported that substitution of up to 33% soy bean meal for fishmeal did not reduce growth rates. It has also been suggested that substantial replacement of fish oil with vegetable oil in salmonid diets is feasible and will not affect performance (Thomassen and Rosjo, 1989; Green and Selivonchek, 1990; Dosanjh *et al.* 1998; Glencross 2003). These results indicate that significant reductions in FCR should not be expected using the reduced fishmeal/oil feed considered in this study. Moreover, FCR would have to increase to 2.38:1 before the average life cycle impacts of producing this feed would be equivalent to using conventional feeds at an FCR of 1.3 (Table 3.5).

3.7 Conclusions

The environmental performance of salmon aquaculture is profoundly influenced by the nature of the concentrated feeds employed. In general, reducing potential contributions to the range of impact categories considered in this study will largely be a factor of decreasing the fraction of animal-derived ingredients used - particularly poultry by-product meals and fisheries by-product meals and oils, which generate the greatest environmental burdens. These findings indicate that current organic aquaculture standards (Pelletier 2003; Naturland 2005; Soil Association 2005) do not result in improved environmental performance with respect to energy use, global warming, marine aquatic ecotoxicity, acidification, eutrophication and biotic resource use. Rather, adherence to these standards actually results in markedly higher burdens in each of these areas of environmental concern. However, potential substitution of vegetable meals and oils in place of animal-derived ingredients does offer substantial opportunities to decrease the environmental costs of salmon aquaculture, and to vastly improve resource use efficiency.

However, it must be recognized that this study accounts for only a narrow range of the numerous environmental impacts potentially associated with feed production for salmon aquaculture. Among the many omissions, impacts associated with intensive agriculture such as pesticide use and soil erosion are not included in this analysis. More generally, biodiversity impacts at all scales are unaddressed. In addition, this research does not consider other potential environmental improvements that might result from organic aquaculture production practices such as lower stocking density, and the prohibition of both copper-based anti-fouling compounds and most pharmaceutical agents. For this reason, the information derived from this work should not be used in isolation to inform decision-making, but should rather complement a more comprehensive suite of considerations.

CHAPTER 4: DISCUSSION

4.1 Life Cycle Assessment: Methodological Strengths and Limitations

During the last century, the cumulative impacts of industrial activities have become of sufficient magnitude to overwhelm the homeostatic capacity of biogeochemical cycles at multiple scales. Ozone depletion resulting from the release of chlorofluorocarbons and other ozone-depleting substances has compromised the capacity of the atmosphere to filter damaging UV radiation (Crutzen 1992; Madronich *et al.* 1995). Anthropogenic emissions of greenhouse gases are altering climatic conditions by contributing to radiative forcing of the atmosphere (Hughes 2000; Robertson *et al.* 2000; Levitus *et al.* 2001; Walther *et al.* 2002). Flows of biologically available reactive nitrogen have doubled since 1960, resulting in local eutrophication impacts and raising concerns regarding potential broad-scale ecosystem effects (Smil 1999; Galloway *et al.* 2004). Acid precipitation linked to nitrogen and sulfur-based atmospheric emissions is similarly generating both local and regional impacts (Likens *et al.* 1992; Bouwman *et al.* 2002). Inarguably, addressing these issues is of paramount importance and will necessarily involve implementing a range of tools to assess and improve the environmental performance of industrial activities (van Berkel *et al.* 1997, 1999). Life Cycle Assessment (LCA) is a highly useful tool in this context.

Life Cycle Assessment was designed by process engineers to quantify the environmental impacts associated with material and energy flows in industrial production systems (Baumann and Tillman 2004). The measures traditionally employed in LCA therefore deal exclusively with a narrow suite of resource extraction and chemical emissions-related impacts. This focus provides powerful resolution into the relative importance of, and linkages between, the inputs and emissions related to individual life cycle stages of specific production systems and the cumulative environmental impacts of industrial activities. Such insights are key to improving environmental performance on two fronts: first, they facilitate the identification of environmental “hot spots” within production chains, thus allowing managers to prioritize process improvements; and, second, they provide comparative measures of efficiency between competing production technologies. Although complex, this systematic approach to understanding human/environment interactions is necessary if we are to structure our activities such that they do not overshoot the stability domains of planetary biogeochemical cycles. LCA cannot signal how well a system performs with respect to specific biophysical limits (the upper thresholds of dynamic systems are, by their nature, unpredictable). However, the information derived can inform means of minimizing the overall scale of human-induced perturbations of these systems.

In particular, the life cycle perspective is invaluable to informing a mature understanding of the environmental implications of energy production and use, both in terms of quality and quantity. Given that energy is the fundamental currency of the economy of nature and, by default, human economies, this understanding will be central to developing alternative pathways to sustainable development. However, the high degree of resolution that the life cycle perspective affords with respect to the energy-related impacts of specific life cycle stages can be a strength or a weakness, depending on the scale and objectives of the analysis. LCAs of single production systems situated in explicitly defined geographical contexts can provide detailed information relevant to process improvements for the system in question. However, much of the information derived may not apply in other contexts due to the sensitivity of life cycle impacts to energy quality. Two identical systems powered by different industrial energy sources will generate profoundly different life cycle impacts. This confounds the potential for making meaningful comparisons between production systems generally, since their life cycle environmental performance will be so heavily weighted by the locally available energy source. LCA was used in this study to evaluate comparative environmental performance between production systems using a combination of models, some of which were context specific but many of which were intentionally constructed to be broadly representative of a class of activities. For example, the electricity used for salmon feed processing in a BC feed mill is largely derived from hydro-electric power (90%), whereas the electricity fueling corn processing is assumed to be derived from an average Canadian electricity mix (18% coal, 4% oil, 4% natural gas, 13% nuclear, 61% hydro). Since a large fraction of this mix comes from oil and coal-powered generating plants, the apparent energy related impacts of corn processing (particularly Marine Aquatic Ecotoxicity and

Acidification) are much higher than those of feed processing. Were the corn processing operations situated in BC, this outcome would have differed substantially.

4.1.1 Impact Categories for Life Cycle Assessment Research of Food Production Systems

Life Cycle Assessment is gaining increasing recognition as a valuable systems analysis tool, and is now frequently used in contexts other than industrial production systems. In particular, it has been widely applied to evaluating environmental performance within and between numerous food production technologies (Narayanaswamy *et al.* 1992; Andersson and Ohlsson 1999; Mattsson 1999; Cederberg and Mattson 2000; Hogass-Eide 2002; Berlin 2002; Jones 2002; Nicol 2004; Arsenault 2006). It has also been used to evaluate various seafood production systems (Seppälä *et al.* 2001; Ziegler *et al.* 2003; Thrane 2003, 2004; Papatryphon *et al.* 2003, 2004; Mungkung 2005; Hospido and Tyedmers 2005; Ellingsen and Aanondsen 2006). The conclusions generated by these studies suggest that the LCA framework is well-suited to informing eco-efficiency measures in food production, and that the life cycle perspective provides important insights for food product-oriented environmental policy.

However, the efficacy of LCA in evaluating environmental performance in food production systems is limited by the scope of existing environmental impact categories. In part, this limitation reflects the origins of the tool. With many industrial processes, the system boundaries with nature are clearer and the traditional impact categories for abiotic resource consumption and chemical emissions are adequate to account for most of the direct environmental repercussions of industrial activities. This is certainly not true of food production systems, where both industrial inputs and ecosystem goods and services underpin productivity and the range of potential impacts is much broader.

In response to this shortcoming, numerous LCA practitioners have attempted, with varying degrees of success, to develop impact categories to account for environmental impacts characteristic of specific food production technologies. For example, Nilsson and Ziegler (2006) developed a method for quantifying benthic impacts associated with trawl fishing. However, developing impact categories for quantitative LCA is constrained in two important ways: first, by the quality of available information regarding specific environmental impacts; and second, by the necessity of linking a product life cycle to the impact in question in a defensible and quantifiable manner. In some cases, such relationships are relatively straight-forward and easily quantified if adequate information exists. For example, a tonne of greenhouse gas emissions will contribute to climate change in a specific manner, regardless of where the emissions occur. Many environmental problems, however, are the manifestation of a complex interplay of intersecting variables and are difficult to reduce to direct relationships. This is particularly true of impacts to biodiversity, where the decline of specific populations or species may be the result of numerous synergistic influences (Perrings *et al.* 1992; McNeely 1992; Kerr and Cihlar 2004).

For these reasons, it is important to recognize the strengths and limitations of LCA in evaluating environmental performance, and to communicate this clearly in the presentation of LCA results. Moreover, these limitations imply that LCA should be considered one tool among many in assessing environmental performance.

4.1.2 Allocation in Life Cycle Assessment of Seafood Production Systems

A review of LCA research of seafood production systems indicates that economic allocation *(i.e.* allocation according to the relative economic values of the co-products) is the most frequently chosen option (Ayer *et al.* 2006). This choice is often justified according to the rationale that it best reflects the motivation for industrial activities and, hence, co-products should be assigned environmental burdens based on their value to society. Besides the obvious point that food products have value to society other than generating revenues, there are several weaknesses to this approach. Most notable among these is that economic values do not reflect the actual material and energetic flows and emissions associated with co-product streams. Since the very purpose of using LCA is to understand and manage these flows it seems at best counterproductive to fall back on economic values, which often reflect market failures and other distorting influences that have resulted in the problems we seek to address. Rather, in an insidious sense, economic allocation produces results that reinforce dysfunctional societal values.

Contrasting the results of this study with similar work carried out using economic allocation (Papatryphon *et al.* 2004) provides an excellent example. Concentrate salmon feeds contain diverse ingredients, many of which are by-products of food production for human consumption *(i.e.*. poultry by-product meal from poultry production, soybean and canola meals from oil production, corn gluten meal from starch and oil production, by-product fishmeals and oils from capture fisheries, etc.). In some cases, such as soy meal (where large amounts of meal are produced compared to oil), the "by-product" actually generates more economic value for the producer than does the primary product despite its lower unit price (Nonhebel 2004). In other cases, including poultry production and capture fisheries, the processing trimmings (which often constitute the bulk of the product stream by mass) are considered worthless relative to the primary products. Papatryphon *et al.* (2004) used economic allocation when comparing the life cycle impacts of four salmonid feeds containing varied proportions of fishmeals and oils derived from either dedicated reduction fisheries or the by-products of fisheries for human consumption. It was found that feeds containing reduction fisheries meals and oils generated higher impacts relative to feeds containing by-product meals and oils because, as the authors explicitly stated, the processing trimmings from which the by-product meals and oils were derived were inexpensive. In other words, despite the fact that fisheries for human consumption are typically more energy intensive than reduction fisheries (Tyedmers 2004) and yield rates for meals and oils are lower from processing trimmings (therefore requiring larger volumes of trimmings to produce a given amount of meal or oil), the quantitative environmental impacts of producing by-product meals/oils were deemed to be less than those of producing reduction fisheries meals/oils simply because the feedstock was cheap. Although these findings may be of interest to an economist, they are less useful for informing environmental management.

In contrast, the gross nutritional energy content of co-products was used as the allocation criterion for the comparative Life Cycle Assessment of alternative salmon feeds reported in this study, and resulted in findings in direct contradiction with those reported by Papatryphon *et al.* (2004). Gross nutritional energy was deemed to be a more appropriate allocation criterion because it provides a value-neutral representation of resource flows that effectively expresses a common physical currency of food co-products both within and between production systems. Although mass-based allocation adequately represents material flows, gross nutritional energy-based allocation adds an additional level of sophistication by providing a first order accounting of the biological value of the materials, which is often distributed unequally between co-product streams.

A sensitivity analysis was conducted to evaluate the potential impacts of alternative allocation criteria on the results of this study. This involved running three successive iterations of the impact assessment phase to calculate the resultant life cycle impacts when environmental burdens were portioned between co-products using either gross nutritional energy content, mass, or economic value. Not surprisingly, it was found that allocation by economic value of co-products resulted in a lower attribution of impacts to feeds containing animal by-products compared to allocation by gross energy content because of the very low economic value of these by-products compared to the meat products for human consumption. Impacts were generally marginally higher using mass allocation, which suggests that animal feed ingredients are of lesser biological value than their co-products. This was largely influenced by the crop co-product ingredients because the high-energy oil fractions had been removed for human consumption.

4.2 Life Cycle Assessment of Feeds for Conventional and Organic Salmon Aquaculture

Maladaptive patterns of production and consumption underlie the majority of contemporary environmental problems. For this reason, redressing societal norms and assumptions that are incompatible with biophysical reality is prerequisite to achieving sustainability in human societies. Specifically, it is imperative that our industrial activities be premised on a recognition and respect for the finite nature of resources and the limited capacity of ecosystems to absorb waste and respond to change. Such recognition must necessarily be accompanied by dramatic reductions in material/energy throughput and the identification and preferential development of maximally efficient production technologies.

In a biophysically rational political economy, most industrial animal husbandry systems would be considered far too wasteful to justify due to the inefficiencies inherent to biological feed conversion (any animal husbandry system is inevitably bound to consume far more food energy than it furnishes). However, if as a society we are to continue to consume animal products then it is imperative that we choose production systems that maximize the return on resource investments, both biotic and abiotic, and minimize environmental harms.

Given that warm-blooded animals divert the majority of food energy consumed into maintaining body temperature, cold-blood organisms are by nature more efficient in feed-to-flesh conversion. Since aquaculture organisms are almost exclusively cold-blooded, it is logical to conclude that aquaculture holds considerable potential for efficient animal protein production. However, the relative efficiencies of various aquaculture production systems can vary widely, depending on the trophic level of the culture organism (or, more specifically, the trophic level at which we choose to rear them), the feed conversion ratio, and the material and energy intensity of both feed provision and the actual aquaculture production technology employed (Troell *et al.* 2004). From a perspective of biophysical efficiency, an ideal aquaculture production system would feature a low-trophic level organism, whose feed requirements could be met using low-intensity co-product materials truly not suitable for human consumption, cultured in an integrated system in which the nutrient emissions could be cycled directly into the production of other aquaculture species or used in agricultural production. Indeed, the majority of global finfish aquaculture production (largely comprised of carp and tilapia) meets many of these criteria (FAO 2006). Suffice to say that the current intensive salmon aquaculture production model falls short of this ideal, since salmon farms raise carnivorous fish using energy intensive concentrated feeds that contain high proportions of animal-derived ingredients, and discharge considerable amounts of biologically valuable nutrients directly into the marine environment.

A comparative Life Cycle Assessment of the environmental burdens associated with the production of conventional and alternative feeds for salmon aquaculture showed that animal-derived ingredients generate a disproportionate share of the impacts associated with feed provision. It was also found that these impacts can be reduced substantially by substituting vegetable-based ingredients in place of animal-derived ingredients such as fishmeals and oils and poultry by-product meal. As well as reducing impacts, such substitution also dramatically improved industrial energy and biotic resource use efficiency. However, the degree of substitution that can be achieved will ultimately depend on the physiology of the culture organism.

Organic aquaculture standards seek to improve the environmental performance of salmon farming by regulating allowable inputs and emissions. However, too date, the efficacy of these specifications in improving the environmental performance of salmon aquaculture generally, and salmon feeds specifically, has not previously been assessed in a rigorous manner.

With respect to feeds, organic standards generally require that all agricultural ingredients be certified organic. The results of this study suggest that organic crop production systems perform considerably better relative to conventional crop production systems in almost all impact categories considered. These findings are consistent with other LCA research, as well as numerous energy analyses that have been undertaken.

In a Life Cycle Assessment of intensive, extensive, and organic grassland farming in southern Germany Hass *et al.* (2001) found that organic farms consumed less energy and water, and generated lower greenhouse gas emissions. This was primarily due to foregoing the use of nitrogen fertilizer, which has an energy-intensive production process. The organic farms also scored better in terms of lower contributions to acidification, eutrophication, and impacts to biodiversity.

Mattsson (1999) reported tradeoffs in environmental impacts between organic and integrated carrot production systems. Organic production created higher acidification and eutrophication impacts due to the use of manure as fertilizer, and required more arable land. However, energy use was higher in the integrated system due to production costs of pesticides and fertilizers, both of which also contributed to greater toxicity impacts.

Studies of milk production on conventional and organic farms demonstrated that the import of concentrated feeds for conventional production resulted in higher energy consumption, largely due to significant inputs of synthetic fertilizers (Cederberg and

Mattsson 2000; Refsgaard *et al.* 1998). Toxicity impacts due to use of pesticides and herbicides in the production of ingredients for concentrate feeds were also substantial (Cederberg and Mattsson 2000). Overall contributions to global warming were similar between systems, with higher nitrous oxide emissions from the conventional system and higher methane emissions attributed to the organic system (due to cattle consuming roughage fodder).

Pimentel *et al.* (2005) found that industrial energy use in organic cropping systems were 28-32% less than in conventional rotations. Hoeppner (2001) reported that crop rotations using green manures in place of synthetic nitrogen fertilizers were up to twice as energy efficient as conventional rotations. In general, it appears that organic crop production is considerably more energy-efficient than conventional production and that this is largely a function of nitrogen procurement strategies.

However, even though organic crop production generates substantially lower life cycle impacts, the substitution of organic for conventional crop ingredients in salmon feeds results in only very minor improvements in environmental performance. This is because the gains in efficiency are watered down by the addition of (sometimes considerable) processing and transportation impacts and, more importantly, because the remaining margin is simply overwhelmed by the much larger impacts associated with the production of animal-derived ingredients.

With respect to animal-derived ingredients, organic aquaculture standards generally disallow by-product meals from terrestrial animal husbandry and stipulate that any fishmeals and oils used be derived from the processing trimmings of fisheries for human consumption. The rationale is that it is preferable to use by-products from existing food production systems as opposed to conducting dedicated reduction fisheries for virgin resources. This position has been adopted by numerous organic aquaculture certifying bodies and embodied in most major organic aquaculture standards (see Soil Association 2005; Naturland 2005). However, from a life cycle perspective, these products are even more environmentally costly, because they derive from less fuel-efficient fisheries and require more raw material to produce due to the low fishmeal/oil yield rates typically achieved from processing trimmings.

Using co-products from food production is logical if the co-products are not, themselves, suitable for human consumption. That said, regardless of whether or not a product is suitable for human consumption, it is never “free” in terms of biophysical costs. Rather, it carries a price tag commensurate with the environmental impacts generated by its production. However, if a co-product used as a feed ingredient in animal husbandry has a high biophysical price tag, it is not sufficient to simply forego use of the product. Rather, it suggests that we should carefully examine the efficiency of the production system from which it is derived. The biophysical rationality of supporting industries that generate large co-product streams not suitable for human consumption, such as energy-intensive fisheries or the sugar and vegetable oil industries, should be questioned as much as the sense of using their environmentally costly co-products to produce animal protein. This systemic perspective would be constructive in policy-maker and industry circles generally.

For example, Nonhebel (2004) estimates that the production of sufficient sugar and soy oil in the Netherlands to support average consumption of these products results in the generation of enough co-products to produce 33 kg of pork per capita, which represents 75% of per capita meat consumption. If sugar and oil were deemed to be essential contributors to the human diet and their production was relatively efficient when compared to competing products, then the generation and use of their co-products to produce meat would be a somewhat justifiable allocation of resources. However, since neither sugar nor oil figure prominently in a healthy diet (and, more pointedly, figure prominently only in the diets of the affluent and in remarkable parallel with obesity and various related health problems) this argument for efficiency holds little merit. Moreover, many co-product such as soy meal are, in fact, suitable for human consumption. In such cases, preference should be given to maintaining these products in the human food stream rather than using them to produce meat, which greatly reduces the amount of available food energy.

Overall, it appears that current standards for organic salmon aquaculture fall considerably short of ensuring improved environmental performance in feed provision according to the impact categories considered in this study. Clearly, options do exist for dramatically reducing the feed-related impacts of salmon farming and these measures may prove relatively easy to implement. However, it must be recognized that this mode of food production represents a single link in a highly complex and interconnected global food system which is, itself, fraught with inefficiencies. It is therefore insufficient to hang our banners on the merits or shortcomings of specific production technologies. Rather, what is required is a reconceptualization of how we produce food, for what purposes, by what means, and with respect to which performance measures.

Food is a fundamental human requirement. Although much of the material and energy throughput of contemporary society is non-essential and can be pared away as resource constraints require, we cannot change the basic human physiological requirement for a minimum amount and quality of food energy. However, as with salmon aquaculture, what we can change is the efficiency of our food production systems, such that we satisfy human needs in ways that minimize environmental harms. Such considerations must necessarily be central to the development of a sustainable human society.

4.3 Recommendations for Further Research

Achieving a sustainable food supply requires that we identify production technologies that are resource efficient and environmentally benign. Continued LCA research of food production systems can and should play an important role in this process. The efficacy of LCA in this respect would be much improved by the further development of additional (and refinement of existing) environmental impact categories, as well as the standardization of allocation procedures and reporting mechanisms. In turn, this will facilitate the development and application of global databases that will allow regional, national and international resource flow accounting. Given the increasingly global nature of food production and consumption, and the associated environmental impacts, such flows in the food system certainly bear scrutiny. Comparative evaluations of competing food production technologies would be much enhanced by the availability of regionally specific data sets, since certain life cycle impacts are contingent on local conditions.

Normalization data for specific production sectors is also desirable to facilitate more meaningful intra-sectoral comparisons of efficiency.

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Appendix A

Key Assumptions

In the interest of arriving at broadly representative models of the various production systems considered, the following key assumptions were made:

Inputs to Crop Agriculture

The use of one standard mid-sized tractor and a single farm implement having 50% of the mass of the tractor was assumed for all possible tilling, seeding, spraying, harvesting, and drying operations. For each crop production system, specific machinery inputs were calculated using average lifespan data for farm machinery (Nagy 1999) and fuel intensity per kg of crop produced (Appendix B). Although organic systems can be expected to use tillage equipment more frequently for green manure production and mechanical weed control while conventional systems will use sprayers and other applicators more frequently for fertilizers, pesticides, and herbicides, previous research has shown similar inputs to farm machinery between conventional and organic systems (Hoeppner 2001). Inputs to farm machinery/ha were therefore assumed to be similar between the conventional and organic crop production systems considered in this study.

Nutrients and Nutrient Emissions

All nitrogen inputs to organic crops were assumed to be derived from green manure rotations. Nitrogen inputs to crops produced in Alberta and Saskatchewan (canola and wheat) were calculated based on a three year crop-crop-lentil green manure rotation and were charged the input costs of producing the green manure on the basis of yield (Appendix D). For crops produced in Ontario, a four-year crop-crop-crop-clover green manure rotation was assumed for soy, and a two-year crop-clover green manure rotation for corn. Inputs to green manure production followed Zentner *et al.* (in Hoeppner 2001).

Although several researchers have reported nutrient emissions for conventional and organic agriculture (Drinkwater *et al.* 1998; Robertson *et al.* 2000; Stolze *et al.* 2000; Haas *et al.* 2001; Kirchmann and Bergstrom 2001; Biro *et al.* 2005; Valli *et al.* 2006) and numerous others for agricultural systems using fertilizers or green manure production to satisfy nitrogen requirements (Granstedt 1991; Magdoff 1991; Norris and Shabman 1992; Campbell *et al.* 1995) a review of the literature suggests that clear consensus has not been reached regarding comparative emission potentials. Moreover, given the sensitivity of nutrient emission potentials to a variety of factors including climate, soil type, rate and form of nutrient application, type of tillage used, nutrient uptake by crops, and volume and carbon:nitrogen ratio of crop residues, it is unlikely that such a consensus is achievable (Eichner 1990). However, for the purpose of the broad-stroke comparisons employed in this study, comparative emission potentials were chosen for nitrate and phosphorous leaching, ammonia volatization, and NOx emissions to air according to best available information in published literature. For all organic crop production systems, a 50% nitrate leaching potential/ha relative to the equivalent conventional system was assumed. Similarly, a 75% phosphorous leaching potential/ha, 75% ammonia volatization potential/ha, and 100% gaseous nitrous oxide emission potential/ha was assumed.

Yields

Numerous studies report comparative crop yields in conventional and organic systems (Drinkwater *et al.* 1995; Pimentel *et al.* 2005). In addition, several reviews report average comparative yields across studies for specific crops (Stanhill 1990; Entz *et al.* 2001; Pretty and Hine 2001; Surridge 2004; Badgley *et al.* 2006). Based on a literature review of comparative yields, yields in organic systems were assumed to be 90% of conventional crop yields for wheat, 95% for corn, 100% for soy, and 90% for canola.

Appendix B

Methods and Assumptions for Calculating Agricultural Machinery Inputs

Tractor – all operations are assumed to employ the same tractor with characteristics:

model:	John Deere 8230
power:	200 hp
mass:	10771 kg
lifespan:	12000 hours
fuel consumption:	35 l/h (Nagy 1999)
composition;	90% steel, 5% rubber, 3% mixed plastics, 2% mixed metals
energy to manufacture:	14.6 MJ/kg (Doering *et al.* (1980) in Audsley *et al.* (1997))
energy for maintenance:	26% of manufacturing energy (Mughal (1994) in Audsley *et al.* (1997))

Implements – all farm implements used for cultivation, fertilizer/pesticide application, and harvesting etc. are assumed to have the following characteristics:

mass:	50% that of the tractor (estimate based on range of implement masses in Nagy (1999))
composition	95% steel, 5% rubber
lifespan:	25% of lifespan of tractor *(i.e.* 3000 hours) based on range of values in Nagy (1999)
energy to manufacture:	8.6 MJ/kg (Doering *et al.* (1980) in Audsley *et al.* (1997))
energy for maintenance:	30% of manufacturing energy (Mughal (1994) in Audsley *et al.* (1997))

Calculating Material Inputs – assuming 50% of original mass required for repairs throughout life cycle (as per Tyedmers (2000) for fishing vessels)

material inputs to tractor (manufacture)	= 10771 kg / 12 000 hours = .898 kg/hour
material inputs to tractor (repair)	= .5(10771 kg / 12 000 hours) = .449
total	= 1.35 kg/hour
material inputs to implements (manufacture)	= 5385 kg / 3000 hours = 1.95 kg/hour
material inputs to implements (repair)	= .5(5385 kg / 2500 hours) = .97 kg/hour
total	2.92 kg/hour

Example: Indirect Material and Energy Inputs to Machinery Canadian Wheat Production

yield	= 2700 kg/ha
fuel consumption	= 29.7 l/ha

If average fuel consumption for fieldwork is 35 l/hour and fuel consumption for wheat production is 29.7 l/ha then the average amount of machine-time investment for wheat production is:

29.71 l/ha / 35 l/h = .85 h/ha

Material inputs to tractor/kg wheat	= (.85 h/ha x 1.35 kg/h)/2700 kg/ha = .000426 kg/kg (90% steel, 5% rubber, 3% mixed plastics, 2% mixed metals)
Material inputs to implements/kg wheat	= (.85h/ha x 2.92 kg/h)/2700 kg/ha = .000919 kg/kg (95% steel, 5% rubber)
Energy to manufacture tractor/kg wheat	= (1.35 kg/h x .85 h/ha x 14.6 MJ/kg)/2700 kg/ha = .00622 MJ/kg
Energy to manufacture implements/kg wheat	= (2.92 kg/h x .85 h/ha x 8.6 MJ/kg)/2700 kg/ha = .00791 MJ/kg
Energy to maintain tractor	= .00622 MJ/kg x .26 = .00162 MJ/kg
Energy to maintain implements	= .00791 MJ/kg x .30 = .00237 MJ/kg

Appendix C

Life Cycle Inventory Data for Conventional Crop Production

Table C1. Life Cycle Inventory data for the production of 1 kg of soybeans in Canada.

Input Type	Input	Unit	Amount
Industrial Energy for Field Operations and Drying [a]	Diesel Fuel	kJ	504.000
	Gasoline	kJ	126.000
	Liquid Propane Gas	kJ	24.400
	Electricity	kJ	25.200
	Natural Gas	kJ	24.400
Fertilizer [a]	Nitrogen	g	3.337
	Phosphorus	g	13.350
	Potassium	g	23.360
	Sulphur	g	1.668
Seed [a]	Soy Beans	g	45.000
Crop Protection [a]	Pesticide	g	0.501
Production Equipment [b]	Material Inputs to Tractor Manufacture and Repair	g	0.719
	Energy Inputs to Tractor Manufacture	kJ	10.497
	Energy Inputs to Tractor Repair	kJ	2.729
	Material Inputs to Farm Implmnt Man. and Repair	g	1.554
	Energy Inputs to Farm Implmnt Man.	kJ	13.360
	Energy Inputs to Farm Implements Repair	kJ	4.008
Agricultural Land [a]	Land Area	ha	4.20×10^{-4}
Output Type	**Output**		
Emissions to Air [c]	Nitrous Oxide	g	0.298
	Ammonia	g	0.241
Emissions to Water	Nitrate [c]	g	25.414
	Phosphorus [d]	g	0.150

a) from $(S\&T)^2$ (2005)
b) see Appendix B
c) see Appendix H
d) Cederberg and Flysjö (2004)

Table C2. Life Cycle Inventory data for the production of 1 kg of corn in Canada.

Input Type	Input	Unit	Amount
Industrial Energy for Field Operations and Drying [a]	Diesel Fuel	kJ	951.330
Fertilizer [a]	Nitrogen (assumed 100% chemical fertilizer)	g	19.490
	Phosphorus	g	6.270
	Potassium	g	6.360
Seed [a]	Corn Seed	g	0.045
Crop Protection [a]	Pesticide	g	0.390
Production Equipment [b]	Material Inputs to Tractor Manufacture and Repair	g	1.005
	Energy Inputs to Tractor Manufacture	kJ	14.673
	Energy Inputs to Tractor Repair	kJ	3.815
	Material Inputs to Farm Implmnt Man. and Repair	g	2.174
	Energy Inputs to Farm Implmnt Man.	kJ	18.696
	Energy Inputs to Farm Implements Repair	kJ	5.609
Agricultural Land [a]	Land Area	ha	1.39×10^{-4}
Output Type	**Output**		
Emissions to Air [c]	Nitrous Oxide	g	0.241
	Ammonia	g	0.196
Emissions to Water	Nitrate [c]	g	7.617
	Phosphorus [d]	g	0.030

a) from Levelton Engineering Ltd. and $(S\&T)^2$ (1999)
b) see Appendix B
c) see Appendix H
d) Cederberg and Flysjö (2004)

Table C3. Life Cycle Inventory data for the production of 1 kg of canola in Canada.

Input Type	Input	Unit	Amount
Industrial Energy for Field Operations and Drying [a]	Diesel Fuel	kJ	1260.000
Fertilizer [a]	Nitrogen	g	46.100
	Phosphorus	g	22.000
	Potassium	g	10.020
	Sulphur	g	8.020
Seed [a]	Canola Seed	g	4.718
Crop Protection [a]	Pesticide (active ingredient)	g	1.403
Production Equipment [b]	Material Inputs to Tractor Manufacture and Repair	g	1.357
	Energy Inputs to Tractor Manufacture	kJ	19.812
	Energy Inputs to Tractor Repair	kJ	5.151
	Material Inputs to Farm Implmnt Man. and Repair	g	2.936
	Energy Inputs to Farm Implements Manufacture	kJ	25.250
	Energy Inputs to Farm Implements Repair	kJ	7.575
Agricultural Land [a]	Land Area	ha	7.80×10^{-4}
Output Type	**Output**		
Emissions to Air [c]	Nitrous Oxide	g	0.579
	Ammonia	g	0.461
Emissions to Water	Nitrate [c]	g	19.961
	Phosphorus [d]	g	0.120

a) from $(S\&T)^2$ (2005)
b) see Appendix B
c) see Appendix H
d) Cederberg and Flysjö (2004)

Table C4. Life Cycle Inventory data for the production of 1 kg of wheat in Canada.

Input Type	Input	Unit	Amount
Industrial Energy for Field Operations and Drying [a]	Diesel Fuel	kJ	396.000
Fertilizer [a]	Nitrogen (assumed 100% chemical fertilizer)	g	24.418
	Phosphorus	g	12.260
	Sulphur	g	8.020
Seed [a]	Wheat Seed	g	45.720
Crop Protection [a]	Pesticide (active ingredient)	g	1.617
Production Equipment [b]	Material Inputs to Tractor Manufacture and Repair	g	0.426
	Energy Inputs to Tractor Manufacture	kJ	6.220
	Energy Inputs to Tractor Repair	kJ	1.620
	Material Inputs to Farm Implmnt Man. and Repair	g	0.919
	Energy Inputs to Farm Implements Manufacture	kJ	7.910
	Energy Inputs to Farm Implements Repair	kJ	2.370
Agricultural Land [a]	Land Area	ha	3.70×10^{-4}
Output Type	**Output**		
Emissions to Air [c]	Nitrous Oxide	g	0.302
	Ammonia	g	0.244
Emissions to Water	Nitrate [c]	g	2.528
	Phosphorus [d]	g	0.090

a) from $(S\&T)^2$ (2003)
b) see Appendix B
c) see Appendix H
d) Cederberg and Flysjö (2004)

Appendix D

Life Cycle Inventory Data for Organic Crop Production

Table D1. Life Cycle Inventory Data for the production of 1 kg of organic soybeans in Canada.

Input Type	Input	Unit	Amount
Industrial Energy for Field Operations and Drying [a]	Diesel Fuel	kJ	504.000
	Gasoline	kJ	126.000
	Liquid Propane Gas	kJ	24.400
	Electricity	kJ	25.200
	Natural Gas	kJ	24.400
Fertilizer [b]	Clover Green Manure Cultivation	ha	1.4×10^{-4}
	Phosphate Rock (ground, P_2O_5)	g	31.370
	Potassium (Potash)	g	23.360
	Sulphur	g	1.668
Seed [a]	Organic Soy Bean Seed	g	45.000
Production Equipment [c]	Material Inputs to Tractor Manufacture and Repair	g	0.719
	Energy Inputs to Tractor Manufacture	kJ	10.497
	Energy Inputs to Tractor Repair	kJ	2.729
	Material Inputs to Farm Implmnt Man. and Repair	g	1.554
	Energy Inputs to Farm Implements Manufacture	kJ	13.360
	Energy Inputs to Farm Implements Repair	kJ	4.008
Agricultural Land [d]	Land Area	ha	4.20×10^{-4}
Output Type	**Output**		
Emissions to Air [e]	Nitrous Oxide	g	0.298
	Ammonia	g	0.241
Emissions to Water [e]	Nitrate	g	12.707
	Phosphorus	g	0.113

a) assumed same as for conventional soy production, but yield adjusted ($(S\&T)^2$ 2005)
b) yield-adjusted amounts based on fertilizer inputs for conventional soy production ($(S\&T)^2$ 2005) using allowable inputs for organic agriculture as per the Canada Organic Production Systems Permitted Substances (2006)
c) assumed same as for conventional soy production, but yield adjusted (see Appendix B)
d) assumed same as for conventional soy production as per Badgely *et al.* (2006)
e) adjusted from emissions for conventional soy production as per Appendix A and H

Table D2. Life Cycle Inventory data for the production of 1 kg of organic corn in Canada.

Input Type	Input	Unit	Amount
Industrial Energy for Field Operations and Drying [a]	Diesel Fuel	kJ	1001.400
Fertilizer [b]	Clover Green Manure Cultivation	ha	1.466×10^{-4}
	Phosphate Rock (ground, P_2O_5)	g	15.507
	Potassium (Potash)	g	6.695
Seed [a]	Organic Corn Seed	g	0.047
Production Equipment [c]	Material Inputs to Tractor Manufacture and Repair	g	1.058
	Energy Inputs to Tractor Manufacture	kJ	15.451
	Energy Inputs to Tractor Repair	kJ	4.017
	Material Inputs to Farm Implmnt Man. and Repair	g	2.289
	Energy Inputs to Farm Implements Manufacture	kJ	19.687
	Energy Inputs to Farm Implements Repair	kJ	5.906
Agricultural Land [d]	Land Area	ha	1.466×10^{-4}
Output Type	**Output**		
Emissions to Air [e]	Nitrous Oxide	g	0.254
	Ammonia	g	0.154
Emissions to Water [e]	Nitrate	g	4.009
	Phosphorus	g	0.024

a) assumed same as conventional corn production but yield adjusted (Levelton Engineering Ltd. and $(S\&T)^2$ (1999)
b) yield adjusted amounts based on fertilizer inputs for conventional corn production (Levelton Engineering Ltd. and $(S\&T)^2$)(1999) but using allowable inputs for organic agriculture as per the Canada Organic Production Systems Permitted Substances (2006)
c) assumed same as for conventional corn production but yield adjusted (see Appendix B)
d) assumed 95% of conventional corn production as per Badgely *et al.* (2006)
e) adjusted from emissions for conventional corn production as per Appendix A and H

Table D3. Life Cycle Inventory data for the production of 1 kg of organic canola in Canada.

Input Type	Input	Unit	Amount
Industrial Energy for Field Operations and Drying [a]	Diesel Fuel	kJ	1400.000
Fertilizer [b]	Lentil Green Manure Cultivation	ha	4.355×10^{-4}
	Phosphate Rock (ground, P_2O_5)	g	57.444
	Potassium (from Potash)	g	11.133
	Sulphur	g	8.911
Seed [a]	Organic Canola Seed	g	5.242
Production Equipment [c]	Material Inputs to Tractor Manufacture and Repair	g	1.508
	Energy Inputs to Tractor Manufacture	kJ	22.011
	Energy Inputs to Tractor Repair	kJ	5.723
	Material Inputs to Farm Implmnt Man. and Repair	g	3.262
	Energy Inputs to Farm Implements Manufacture	kJ	28.053
	Energy Inputs to Farm Implements Repair	kJ	8.416
Agricultural Land [d]	Land Area	ha	8.67×10^{-4}
Output Type	**Output**		
Emissions to Air [e]	Nitrous Oxide	g	0.643
	Ammonia	g	0.384
Emissions to Water [e]	Nitrate	g	11.090
	Phosphorus	g	0.100

a) assumed same as for conventional canola production but yield-adjusted ((S&T)[2] 2005)
b) yield adjusted amounts based on fertilizer inputs for conventional canola production ((S&T)[2] 2005) but using allowable inputs for organic agriculture as per the Canada Organic Production Systems Permitted Substances (2006)
c) assumed same as for conventional canola production but yield adjusted (see Appendix B)
d) assumed 90% of conventional canola production
e) adjusted from emissions for conventional canola production as per Appendix A and H

Table D4. Life Cycle Inventory data for the production of 1 kg of organic wheat in Canada.

Input Type	Input	Unit	Amount
Industrial Energy for Field Operations and Drying [a]	Diesel Fuel	kJ	440.000
Fertilizer [b]	Lentil Green Manure Cultivation	ha	2.05×10^{-4}
	Phosphate Rock (ground, P_2O_5)	g	32.000
	Sulphur	g	8.911
Seed [a]	Organic Wheat Seed	g	50.800
Production Equipment [c]	Material Inputs to Tractor Manufacture and Repair	g	0.473
	Energy Inputs to Tractor Manufacture	kJ	6.910
	Energy Inputs to Tractor Repair	kJ	1.800
	Material Inputs to Farm Implmnt Man. and Repair	g	1.021
	Energy Inputs to Farm Implements Manufacture	kJ	8.788
	Energy Inputs to Farm Implements Repair	kJ	2.633
Agricultural Land [d]	Land Area	ha	4.10×10^{-4}
Output Type	**Output**		
Emissions to Air [e]	Nitrous Oxide	g	0.336
	Ammonia	g	0.203
Emissions to Water [e]	Nitrate	g	1.404
	Phosphorus	g	0.076

a) assumed same as for conventional wheat production but yield adjusted ((S&T)^2 2003)
b) yield adjusted amounts based on fertilizer inputs for conventional wheat production ((S&T)^2 2003) but using allowable inputs for organic agriculture as per the Canada Organic Production Systems Permitted Substances (2006)
c) assumed same as for conventional wheat production but yield adjusted (see Appendix B)
d) assumed 90% of conventional wheat production as per Badgely *et al.* (2006)
e) adjusted from emissions for conventional wheat production as per Appendix A and H

Appendix E

Life Cycle Inventory Data for Crop Processing

Table E1. Life Cycle Inventory data for the processing of 1 kg of soybeans.

Input Type	Input	Unit	Amount
Industrial Energy for Processing [a]	Electricity	kJ	244.800
	Natural Gas (for steam production)	kJ	812.000
Output Type	**Output**		
Processed Products [b]	Soy Meal	g	812.000
	Soy Oil	g	188.000

a) (S&T)[2] (2005)
b) Ahmed *et al*. 1994

Table E2. Life Cycle Inventory data for the wet-milling of 1 kg of corn.

Input Type	Input	Unit	Amount
Industrial Energy for Processing	Electricity	kJ	878.400
	Natural Gas (for steam production)	kJ	3770.000
Chemicals for Processing	Sulphur	g	0.930
	Peroxitan	g	1.770
Output Type	**Output**		
Processed Products	Corn Gluten Feed	g	238.000
	Corn Starch	g	610.000
	Corn Gluten Meal	g	67.000
	Corn Germ Meal	g	85.000
Emissions to Water	COD	g	0.610
	BOD	g	3.060

source: Cederberg (1998)

Table E3. Life Cycle Inventory data for the grinding of 1 kg of corn.

Input Type	Input	Unit	Amount
Industrial Energy for Processing	Electricity	kJ	418.000
Output Type	**Output**		
Processed Products	Ground Corn	kg	1.000

source: Southwell and Rothwell (1977)

Table E4. Life Cycle Inventory data for the processing of 1 kg of canola.

Input Type	Input	Unit	Amount
Industrial Energy for Processing	Electricity	kJ	162.000
	Natural Gas (for steam production)	kJ	844.000
Output Type	**Output**		
Processed Products	Canola Meal	g	580.000
	Canola Oil	g	420.000

source: $(S\&T)^2$ (2005)

Table E5. Life Cycle Inventory data for the grinding of 1 kg of wheat.

Input Type	Input	Unit	Amount
Industrial Energy for Processing [a]	Electricity	kJ	418.000
Output Type	**Output**		
Processed Products	Ground Wheat	kg	1.000

a) assumed same energy required to grind 1 kg of corn (Southwell and Rothwell 1977)

Appendix F

Life Cycle Inventory Data for Fishmeal and Oil Production

Table F1. Life Cycle Inventory data for material and energy inputs to producing and maintaining fishing boats and gear per tonne of fish landed (based on 18 m aluminum-hulled purse seiner).

Input Type	Input	Unit	Amount
Industrial Energy for Boat Construction	Electricity	MJ	447.000
Materials Used in Construction and Repair	Aluminum (80% recycled)	kg	14.300
	Steel (80% recycled scrap)	kg	1.490
	Steel	kg	11.400
	Plastic (gear)	kg	6.200
	Lead (gear)	kg	0.200

source: Tyedmers (2000)

Table F2. Life Cycle Inventory data for fuel inputs to fisheries contributing raw material to fishmeal and oil production.

Input Type	Input	Unit	Amount
Peruvian Reduction Fishery [a]	Diesel	MJ	1548.000
US Menhaden Reduction Fishery [b]	Diesel	MJ	1237.000
BC Herring Roe Fishery [c]	Diesel	MJ	4955.000

a) $(S\&T)^2$ (2004)
b) Tyedmers (2004)
c) Tyedmers (2000)

Table F3. Life Cycle Inventory data for the processing of one tonne of fish into fishmeal and oil in the Peruvian reduction fishery.

Input Type	Input	Unit	Amount
Industrial Energy for Processing	Electricity	MJ	108.000
	Fuel Oil	MJ	2104.000
Output Type	**Output**		
Processed Products	Fishmeal	kg	216.000
	Fish Oil	kg	76.000
Emissions to Water	COD	kg	42.000

source: $(S\&T)^2$ (2004)

Table F4. Life Cycle Inventory data for the processing of one tonne of fish into fishmeal and oil in the US menhaden reduction fishery.

Input Type	Input	Unit	Amount
Industrial Energy for Processing [a]	Electricity	MJ	128.900
	Bunker C Fuel Oil	MJ	0.850
	Natural Gas	MJ	1468.000
Output Type	**Output**		
Processed Products [a]	Fishmeal	kg	249.000
	Fish Oil	kg	120.000
Emissions to Water [b]	COD	kg	42.000

a) Anonymous (2003)
b) (S&T)^2 (2004)

Table F5. Life Cycle Inventory data for the processing of one tonne of fish into fishmeal and oil in the BC herring roe fishery.

Input Type	Input	Unit	Amount
Industrial Energy for Processing [a]	Natural Gas	MJ	1660.000
Output Type	**Output**		
Processed Products [a]	Fishmeal	kg	160.000
	Fish Oil	kg	20.000
Emissions to Water [b]	COD	kg	42.000

a) Tyedmers (2000)
b) (S&T)^2 (2004)

Appendix G

Life Cycle Inventory Data for Poultry By-product Meal Production

Table G1. Life Cycle Inventory Data for the production of one tonne of poultry feed.

Input Type	Input	Unit	Amount
Crop and Animal Product Inputs[a]	Ground Corn	kg	656.000
	Ground Wheat	kg	100.000
	Soybean Meal	kg	122.000
	Fishmeal	kg	50.000
	Fat (Poultry Grease)	kg	33.600
Feed Additives[a]	Salt	kg	4.900
	Ground Limestone	kg	12.200
	Dibasic Calcium Phosphate	kg	6.400
	Vitamin/Mineral Premix	kg	10.000
	DL-methionine	kg	1.000
	L-lysine	kg	2.100
Industrial Energy Inputs[b]	Electricity	MJ	49.320

a) Proudfoot *et al*. (1991)
b) Cederberg and Flysjö (2004)

Table G2. Life Cycle Inventory data for the production of one tonne of broiler poultry.

Input Type	Input	Unit	Amount
Industrial Energy Inputs [a]	Electricity	MJ	448.800
	Natural Gas	MJ	4802.700
Feed [b]	Poultry Feed	kg	1537.380
Output Type	**Output**		
Emissions to Air [c]	Nitrous Oxide	kg	1.084
	Ammonia	kg	14.645
	Methane	kg	0.570
Emissions to Water [c]	Nitrate	kg	41.270
	Phosphate	kg	0.228

a) Personal communication, Dr. Ulf Sonesson, Swedish Institute of Food and Biotechnology, 2006
b) Proudfoot *et al*. (1991)
c) LCA of Food Database (2006)

Table G3. Life Cycle Inventory data for the processing of one tonne of broiler poultry.

Input Type	Input	Unit	Amount
Industrial Energy Inputs [a]	Electricity	MJ	739.900
Output Type	**Output**		
Poultry Products [b]	Poultry Meat	kg	560.000
	Poultry Processing Wastes	kg	440.000
Emissions to Water [c]	BOD	kg	30.450
	Hexane Extractables	kg	15.880
	Nitrate	kg	0.970
	Phosphate	kg	0.220

a) Personal communication, Dr. Ulf, Swedish Institute of Food and Biotechnology, 2006
b) Ockerman and Hansen (2002)
c) EPA 2002

Table G4. Life Cycle Inventory Data for the processing of 1 tonne of poultry processing wastes into by-product meal and fat.

Input Type	Input	Unit	Amount
Industrial Energy Inputs [a]	Natural Gas	MJ	2125.000
	Electricity	MJ	288.000
Output Type	**Output**		
Poultry Products [a]	Poultry Fat	kg	100.000
	Poultry By-product Meal	kg	260.000
Emissions to Water [b]	BOD	kg	7.280
	Hexane Extractables	kg	5.940
	Nitrate	kg	0.660
	Phosphate	kg	0.130

a) $(S\&T)^2$ (2005)
b) EPA (2002)

Appendix H

Methods for Calculating Nitrogenous Emissions from Crop Production Systems

Nitrogenous emissions from agricultural crop production were modeled based on methodology and information adopted from Brentrup *et al.* (2000), Canada GHGenius, and the Intergovernmental Panel on Climate Change by Arsenault (2006). The following example calculations for nitrogenous emissions from corn production in Canada is representative of the methodology employed.

Table H1. Calculation of nitrogenous emissions from corn production in Canada.

Calculation Step	Unit	Amount
N from synthetic fertilizer[c]	g	19.4900
Percent Fertilizer N lost as ammonia[d]	%	1.0000
NH_3-N lost from fertilizer application	g	**.1949**
Fertilizer N remaining	g	19.2951
Total N entering the plant system	g	19.2951
Percent N lost as N_2O[e]	%	1.2500
N_2O-N lost to the atmosphere	g	**.2412**
Percent N lost as N_2[f]	%	9.0000
N_2-N lost to the atmosphere (not env. relevant)	g	1.7366
N remaining in the soil	g	17.3175
N removed with plant during harvest[g]	g	10.7000
Rate of atmospheric N deposition[h]	kg/ha	7.5000
Atmospheric N deposition	g	1.0000
N remaining in the soil	g	7.6173
Percent N leached as NO_3[i]	%	1.0000
NO_3-N lost to ground water	g	**7.6174**

Appendix I

Mass, Economic and Energy Allocation Data

Table I1. Allocation data.

Ingredient	Mass Allocation	Economic Allocation (mass adjusted)		Energy Allocation (mass adjusted)	
	% Mass	Value/ Tonne	% Value	Gross Energy MJ/kg	% Gross Energy
Peruvian Fishmeal	74[1]	883 USD[7]	77	20.3[14]	60
Peruvian Fish Oil	26[1]	750 USD[7]	23	38[15]	40
US Menhaden Fishmeal	67.5[2]	883 USD[7]	70	20.3[14]	52
US Menhaden Fish Oil	32.5[2]	750 USD[7]	30	38[15]	48
BC By-product Fishmeal	88.9[3]	883 USD[7]	90	20.3[14]	81
BC By-product Fish Oil	11.1[3]	750 USD[7]	10	38[15]	19
BC Herring Roe	20[3]		95[8]	6[16]	20
BC Herring Carcasses	80[3]		5[8]	6[17]	80
Poultry	56[4]		95[8]		56[8]
Poultry Processing Wastes	44[4]		5[8]		44[8]
Poultry By-product Meal	72[5]	225 USD[9]	88	22.5[14]	60
Poultry Fat	28[5]	100 USD[10]	12	39.4[18]	40
Corn Gluten Meal	6.7[6]	250[11]	8.3	21.01[19]	9
Corn Gluten Feed	23.8[6]	102[11]	11.4	8.44[20]	13
Corn Starch	61[6]	250[11]	75.4	17.38[21]	67
Corn Germ Meal	8.5[6]	123[11]	4.9	20.34[22]	11
Canola Meal	58[5]	207[12]	30	18.1[14]	39
Canola Oil	42[5]	665[12]	70	39.8[23]	61
Soybean Meal	81.2[5]	182 USD[13]	67	17.9[14]	66
Soybean Oil	18.8[5]	381 USD[13]	33	39.6[24]	34

1. $(S\&T)^2$ (2004)
2. Anonymous (2003)
3. Tyedmers (2000)
4. Ockerman and Hansen (2001)
5. $(S\&T)^2$ (2005)
6. Cederberg (1998)
7. FAO 2006

8. Assuming 95% value for primary product, 5% value for co-product and equivalent gross energy content/kg
9. Jacobsen Price Guide 2006
10. Shumaker *et al.* 2003
11. EPA 2003
12. Canola Council of Canada (2006)
13. United Soybean Board (2002)
14. Hajen *et al.* (1993)
15. Booth *et al.* (2005)
16. Paul *et al.* (1996)
17. Paul *et al.* (1998)
18. Kienzlie *et al.* (2002)
19. NAS (1971)
20. Shapouri *et al.* (1995)
21. D'Alfonso (2005)
22. Calculated assuming 8.54/6.38/47.35 g protein/fibre/fat per 100g @ 4/4/9 cal/g
23. Glencross 2003
24. Assumed same as canola oil.

Appendix J

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