Life Cycle Analysis of an Integrated Biogas-Based Agriculture Energy System

by

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ABSTRACT

Large quantity of manure is generated in the livestock industry in British Columbia (BC) and natural gas is being consumed intensively in BC's agriculture sector. We proposed to integrate the livestock farms and the greenhouses to promote waste-to-energy and waste-to-material exchanges following the principles of Industrial Ecology (IE). Natural gas consumptions on farms are replaced by renewable biogas generated from anaerobic digestion (AD) of farm wastes (mainly livestock manure). CO_2 for plant enrichment in greenhouses is supplied by biogas combustion flue gases and the residues (digestate) from digesters are used as animal bedding materials, plant growing media, and liquid fertilizers. An integrated dairy farm and greenhouse was first modeled. Co-digestion of manure with a variety of organic farm wastes was further evaluated with an aim to enhance the biogas production. To address the problems of too much digestate surplus and high CO_2 demand for greenhouse CO_2 enrichment, the mushroom farm was further introduced into the integrated system. In this way, the digestate surplus can be used as a growing media for growing mushrooms and the CO_2–rich ventilation air from the mushroom can be directed to the greenhouse for CO_2 enrichment. A Life Cycle Analysis (LCA) was conducted to quantify the environmental impacts of each of the proposed cases in comparison to the conventional agriculture practices.

The LCA results showed that the integrated dairy farm-greenhouse system reduces non-renewable energy consumption, climate change, acidification, respiratory effects from organic emissions, and human toxicity by more than 50% compared to conventional operations; among which the reductions in non-renewable energy consumption, climate change, and human toxicity are the most significant. If the digestate surplus is treated as a waste, the integrated system has a ~20% increase in eutrophication and respiratory effects from inorganic emissions. When other organic wastes are co-digested with dairy manure, all the impacts can be further reduced in all cases. If a mushroom farm is introduced to form an integrated dairy farm-greenhouse-mushroom farm system, a large greenhouse can be facilitated and the digestate can be largely reused; thus all the analyzed impacts are significantly reduced compared to the base scenario.

PREFACE

Some preliminary results of Chapter 2 were published on page 1 to 12 of supplement of Journal of Yunnan Normal University (Natural Science Edition) in 2011 under the title of “A Life Cycle Assessment Case Study on a Farm Scale Biogas Plant in British Columbia”. The co-author of this article is Xiaotao Bi. Also part of the content in Chapter 2 was presented in a poster in the Symposium on Industrial Ecology for Young Professionals (SIEYP II) on June 11th, 2011 at Berkeley under the title of “A case study on an integrated animal farm-greenhouse Eco-Industrial system in British Columbia”. The co-authors are Xiaotao Bi and Roland Clift. The method used in the preliminary studies was generally the same as in Chapter 2. But the system boundary, scenarios, and results were later modified and updated in Chapter 2.

Parts of the content in Chapters 2 and 4 were orally presented in the 62nd Canadian Chemical Engineering Conference on October 15th, 2012 at Vancouver under the title of “A Biogas Based Agriculture Eco-Industrial Park in British Columbia”. The scenarios and results were again modified and updated in this thesis.

Lastly, Appendix E, an article titled “Life Cycle Costing of an integrated animal farm-greenhouse Eco-Industrial system in British Columbia” was published on page 20 to 28 of Life Cycle Assessment XI Conference (Chicago 2011) Proceedings. The co-authors of this article are Xiaotao Bi, Anthony Lau, and Roland Clift.

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LIST OF ABBREVIATIONS

Abbreviation	Full name
AD	Anaerobic digestion
BC	British Columbia
CIHR	Canadian Institutes of Health Research
CSTR	Continuous Stirred-Tank Reactor
EPA	Environment Protection Agency
GHG	Greenhouse gas
GWP	Global Warming Potential
HDV	Heavy duty vehicle
IE	Industrial Ecology
IPCC	Intergovernmental Panel on Climate Changes
LCA	Life cycle analysis (also known as life cycle assessment)
LCC	Life Cycle Costing
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
MPF	Mixed plug-flow
NMVOC	Non-methane volatile organic compounds
PF	Plug-flow
PM	Particulate matters
PM_{10}	Particulate matters with diameter less than 10 µm
$PM_{2.5}$	Particulate matters with diameter less than 2.5 µm
UBC	University of British Columbia
VOC	Volatile organic compounds

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DEDICATION

I would like to dedicate this work to my entire family, especially my parents, Jimin Liu and Jianzhi Zhang, who have been giving me continuous and unconditional support. Also to my boyfriend, Yanjie Liu, for his love and support. Without them, this achievement would not have been possible.

1. INTRODUCTION

1.1 Problems and Opportunities

Manure is the major by-product of the livestock industry. In Canada, the livestock industry produces 150 million tonnes of manure each year (Statistics Canada, 2006). Livestock farms in British Columbia (BC) generate 2.7 million tonnes of animal manure each year which accounts for 82% of total organic waste (Electrigaz Technologies Inc., 2007). Dairy, swine, and poultry are the most common livestock in BC. Dairy production ranks first in BC's agriculture economy (BC Ministry of Agriculture, 2011). The 1.75 million tonnes of dairy manure generated each year accounts for 65% of total manure waste. Swine manure and poultry manure each account for 17% of the total manure waste (Electrigaz Technologies Inc., 2007).

Manure from different animals varies in density, water content, and nutrient content. Livestock farms conventionally store the manure to stabilize its properties and for use as a fertilizer at appropriate times. Stored manure will eventually be applied on land as fertilizer manually or mechanically. During storage, microorganisms in manure decompose the organic matter and release a number of air pollutants. If the storage facility is not well contained, leakage of manure to soil and ground water is also an environmental and health problem. Factors influencing manure storage emissions include animal species, storage system structures, and local environment. Specifically, the original nutrient content, ambient temperature, and aeration conditions directly influence the decomposition of the organic matter and thus the final emissions. Similar to manure storage, soil microorganisms along with manure microorganisms continue the decomposition process after manure land application. Soil conditions and local weather will further influence the micro-environment and therefore the reactions. Air emissions from manure application are released gradually for months and will eventually disperse. There are nearly 200 compounds reported to be emitted from livestock manure management (O'Neill and Phillips, 1992). The contamination and eutrophication impact on ground water and surface water bodies is one of the major hazards of manure land application. Therefore, the conventional practice of livestock manure management is identified as an environmental and public health problem.

Anaerobic digestion (AD) is a growing technology taking advantage of microorganisms to degrade organic matters into CH_4-rich biogas, a renewable energy source. The AD process is a biological decomposition of organic matters in the absence of oxygen. Unlike aerobic digestion, which completely oxidizes the carbon compounds into carbon dioxide, AD stops the reactions at the stage of intensive methane generation. AD includes the stages of hydrolysis, acidogenesis/acetogenesis and methanogenesis. In this process, biogas is generated with a typical composition of ~60% methane and ~40% carbon dioxide. AD has been widely used in manure treatment and waste water treatment. Manure treatment especially gets the most interest and application since the early times (Monnet, 2003). It is said that the Romans used organic waste to produce biogas for water heating thousands of years ago. Household scale digesters are widely used to provide cooking gas fuel in developing countries like China and India. In the US, utilizing biogas to fuel an engine was first introduced in the late 1970's partly due to the first energy crisis and the first on-farm digester was installed in Pennsylvania around 1980 (Badger et al., 1995).

Nowadays, AD technology has been extensively researched and improved. A number of digester configurations have been designed. Briefly they include Continuous Stirred-Tank Reactor (CSTR), Fed Batch Digester, Plug Flow Digester (PF), and Mixed Plug Flow Digester (MPF) (Baldwin et al., 2009). Raw biogas usually undergoes upgrading or purification depending on the end use, which may be heating fuel, generating electricity or transport fuel. The material remaining after digestion, known as digestate, is a kind of organic slurry that can be used as fertilizer or soil improver. The digestate can be wholly applied as fertilizer or separated into liquid and solid phases. If separated, the liquid phase is a nitrogen-rich fertilizer with mineralized nitrogen easily absorbed by plants; it is reported that more than 20% more of the total nitrogen becomes available for plants compared to raw manure (DEFRA, 2010; Sommer et al., 2001). The solid phase is the fibers that were not digested during AD, derived originally from the grass and other cellulose materials in the digestion feedstock. Similar to compost product, this can be used as a soil improver for horticulture and gardening. Noting that sawdust or straw are needed for livestock farms as bedding materials, using solid digestate as a substitute has also become common practice (Werner and Strehler, 2012). With renewable energy and waste recycling being strongly promoted all over the world, applications of AD have been increasing. Germany had nearly 80% of all the digesters in Europe in 2005 (AD Nett, 2005) with roughly 5,000 plants operational in Germany by 2010. The US had more than 160 digesters operating by late 2010. Currently, on-farm AD plants are the most widespread application in both Europe and the North America. Some countries such as Denmark, Sweden, and Germany also operate centralized plants where several farms supply livestock manure to one AD facility, sometimes with other agricultural waste (Hjort-Gregersen et al., 2011).

AD is advantageous to treat manure waste compared to conventional practices in terms of environmental protection. NH_3 and H_2S emissions are severe on farms with conventional manure management (Brunet, 2006). In AD, raw manure is fed into a sealed digester without storage requirement. After digestion, the gas pollutants containing in the raw biogas are removed by gas upgrading. The organic matter decomposed by AD also includes the sources of organic odor species, which are therefore eliminated. The non-volatile organic matters that normally lead to BOD hazard after manure spreading are also decomposed, thus lowering the BOD potential. In addition, AD is able to kill many weed seeds and pathogens as a result of the warm digester environment. Thus using digestate can avoid the usage of herbicides, and the health concern of pathogen contamination is largely relieved (AFBI, 2012). AD also retains the nutrients in manure, thus maintaining nutrient content and hence fertilizer quality. Studies have focused mainly on nitrogen (N), phosphorus (P), and potassium (K). It was discovered that since organic N is degraded into ammonium during digestion, the digestate contains more plant-available N nutrient compared to raw manure (Sommer et al., 2001). Contents of P and K generally remain the same as raw manure (Topper, 2012). If the digestate is separated, N will be left in the liquid phase while P and K are mostly adsorbed onto the solid materials (Lukehurst et al., 2010). When the liquid phase is used as fertilizer, there is N nutrient surplus compared to the same amount of raw manure while P and K are lost in the solid phase (Börjesson and Berglund, 2007). However, it is reported that P and K have been applied excessively on land in BC, leading to eutrophication impacts (Poon and Schmidt, 2010; Schmidt and Hughes-Games, 2010); under these circumstances, it may actually be beneficial to lose some P and K nutrients in the solid digestate which will no longer be applied on land.

Another major industrial scale agricultural activity in BC is the greenhouse industry. The greenhouse industry heavily relies on natural gas for heating and CO_2 enrichment. From 1993 to 2006, BC greenhouse production has grown from about 0.64 million m^2 to nearly 5.3 million m^2 in 2006 (Foster et al., 2011). It has been estimated that greenhouse farming contributes more than $600 million to BC's economy (BC Greenhouse Growers Association, 2012), accounting for over one-fifth of the total value of BC agriculture (Foster et al., 2011). BC is a high latitude area with a year-round average temperature of only 10^{o}C (Lynch, 2012). The required average temperature inside a greenhouse is around 21^{o}C (Luczynski and Portree, 2005). Therefore large amounts of natural gas have been consumed for heating the greenhouses. It was reported that in 1997, 913 TJ natural gas was consumed for greenhouse cultivation - higher than any other agriculture sector in BC, representing 36% of the total natural gas consumption of 2,510 TJ by BC's agriculture production (Khakbazan, 2000). In addition, CO_2 is considered as a fertilizer to enhance the photosynthesis of plants in greenhouses: 108 kg CO_2 per year is required for one square meter of greenhouse

(Luczynski and Portree, 2005). A common way to supply CO_2 is to apply flue gases from the natural gas boiler. Another way is to purchase manufactured CO_2 such as food quality CO_2. Food quality CO_2 is clean without impurities and thus good for the seedlings (Luczynski and Portree, 2005) but it is not widely used due to the high cost compared to the flue gases. In summer, heating demand is low while CO_2 demand is high as the plants grow robustly. Thus significant amounts of natural gas have to be combusted for the purpose of CO_2 enrichment.

As a fossil fuel, extraction and processing of natural gas involve significant environmental and health impacts. Burning natural gas is considered as cleaner than other types of fossil fuels in terms of air pollutant emissions but the large amount of CO_2 in the flue gas is fossil CO_2 which is a typical greenhouse gas (GHG). Like other non-renewable fuels, natural gas will reach a peak production and eventually run out of reserve (Goodstein, 2004). Conventional natural gas has become scarce in many areas in the world and efforts have been made to explore alternative energy sources. Extraction of non-conventional natural gas such as shale gas and coal bed methane has been growing fast to make up conventional gas deficiency and meet a growing energy demand (International Energy Agency, 2012). BC has released a “Natural Gas Strategy” to promote shale gas extraction and marketing (BC Ministry of Energy and Mines, 2011). However, these non-conventional gases are still fossil fuel and thus exhaustible. Debates about the environmental impacts of extraction, manufacturing, and transport have been heated (Palmer, 2012). It is not sustainable to rely on such type of alternative fuels; renewable energy should be truly explored and promoted. Due to the continuing crisis and uncertainty in the natural gas sector, the price of natural gas has been increasing with frequent fluctuations. Reviewing the price history of natural gas in the past 10 years (starting from 2002) in BC reveals an increasing trend: dramatic rises occurred in 2005 and 2007 due to the natural gas crises in the USA resulting from growing demand for natural gas for industrial production as well as electricity generation. In the recent forecast of natural gas price in BC, the moderate scenario predicts 25% increase in 2020 compared to 2012 while the high increase scenario predicts 41% increase (Fortis BC, 2012). Therefore, another problem for the BC agricultural industry is identified as high natural gas consumption and reliance.

1.2 Industrial Ecology for System Integration

Industrial Ecology is defined as “The concept which requires that an industrial system be viewed not in isolation from its surrounding systems, but in concert with them. It is a systems view in which one seeks to optimize the total materials cycle from virgin material, to finished material, to component, to product, to obsolete product, and to ultimate disposal. Factors to be optimized include resources, energy, and capital (Graedel and Allenby, 2003). Put simply, Industrial Ecology introduces natural ecology concepts to industrial activities. In a natural ecosystem, creatures form a food web system by some units feeding the others with food and energy. Unlike human society, there is no permanent waste in a natural eco-system because all the materials can eventually be consumed and decomposed. Such a sustainable and efficient natural design inspires industrial designs that also deal with material and energy flows. Individual industrial players also consume and generate materials and energy just like creatures in nature. When some units' waste is potentially useful as other units' raw material, they can be connected just like a food chain in nature. Interconnecting more units in this way can probably save energy and materials as well as reduce net waste production. Such an integrated industrial system is called an eco-industrial system or Eco-Industrial Park (EIP) when the scale of the system is large. The earliest and most famous EIP is at Kalundborg in Denmark, with more than 20 players involved (Ehrenfeld and Gertler, 1997). Furthermore, an eco-industrial system is actually not limited to industrial partners. A typical EIP can include other sectors and players as long as there are mutually beneficial material and energy interactions. Therefore, an EIP is defined as "a community of manufacturing and service businesses located together on a common property. Members seek enhanced environmental, economic, and social performance through collaboration in managing environmental and resource issues" (Lowe, 2001).

The existing eco-industrial systems mostly deal with industrial players while agriculture-based systems are mainly still at the design stage (The Center for an Agricultural Economy, 2012). One agriculture-based eco-industrial system is the Guitang Group in China formed from a sugar refinery;now it includes an alcohol plant, a fertilizer plant, a pulp plant, a paper mill, and a cement mill (Geng et al., 2006). One eco-system is proposed in Hardwick Vermont, US, to integrate the production of maple syrup, dairy products, whey-based wood finish, and organic soy products (Gilbert et al., 2007). More agricultural eco-industrial systems are designed to include bio-energy or bio-fuels from biomass. Examples are the Riverside Eco-park in Vermont, US (Skiff, 1997), an agro-EIP in rural China (Indigo Development, 2005), a bio-based eco-industrial cluster in Dambulla, Sri Lanka (Herath and Ratnayake, 2009), and an environmental "green" energy park in Prince Albert, Canada (Canadian Bio-energy Management Inc., 2009). In BC, an agriculture-based eco-industrial system was proposed for the City of Richmond with potential partners being explored (Eco-Industrial Solutions Ltd., 2012).

1.3 Life Cycle Analysis for Impact Assessments

A comprehensive assessment is required to evaluate the environmental performance of a system for information and decision-making. There are a number of environmental assessment approaches, including state of the environment (SOE), Integrated Environmental Assessment and reporting (IEA), environmental impact assessment (EIA), and strategic environmental assessment (SEA). Starting from the 1990s, LCA has been growing fast and gained widespread applications. According to the ISO 14000 series on environmental management, LCA is defined as a systematic tool for identifying and evaluating the environmental aspects of products and services from extraction of resource inputs to the eventual disposal of the product or its waste (ISO 14044, 2006). The key feature of LCA is that the products and services are analyzed throughout their whole "life cycle" or supply chain, which is usually referred to as "cradle to grave" analysis. This avoids misleading conclusions from limited scope of analysis by offering a complete picture and comprehensive description of and information on the environmental impacts of providing a product or service. LCA is also a powerful tool for comparison studies as the unit differences between scenarios can be analyzed. It is gaining wide applications in recent years as a popular assessment tool for industrial design and various decision-making processes. The ISO 14044 standard, *Environmental management-Life cycle assessment-Requirements and guidelines*, specifies requirements and provides guidelines on the execution of an LCA (ISO, 2006). Typically, there are four phases in an LCA study: 1) goal and scope definition; 2) life cycle inventory (LCI) analysis; 3) life cycle impact assessment (LCIA); and 4) life cycle interpretation.

1) Goal and scope definition:

The goal of the study is defined at the outset; the system function and functional unit are defined; the boundary for analysis is established and the processes to be studied are described.

2) Life Cycle Inventory analysis:

The material and energy inputs and outputs of each process are compiled to form an LCI. Inputs focus on resources and energy and outputs cover products, by-products, wastes and contaminant emissions, etc. Calculations and conversions are frequently required after the data are collected and should be conducted correctly.

3) Life Cycle Impact Assessment:

With the LCI compiled, potential impacts that associated with the identified inputs and outputs are assessed. This phase translates the interactions between the product/system and the environment into classified and detailed environmental burdens. The LCIA phase is a very important phase and, therefore, the detailed LCIA methods for this study are described in Section 1.4.2.

4) LCA Interpretation:

In the LCA interpretation phase, the LCA work is discussed and qualified with the uncertainties and limitations reviewed so that the implications of the study can be communicated and recommendations proposed. Interpretation is mostly conducted after the impact assessment phase but is not limited to this phase in the study: important conclusions may emerge before the study is completed. Interpretation may also be needed during the inventory analysis and impact assessment phases.

Several LCA studies have been reported on analyzing the environmental benefits of conducting AD for manure treatment, generally confirming the benefits from producing biogas as well as using digestate to substitute conventional fertilizers and materials. However, specific results cannot be directly compared between any two studies due to different system boundaries, data sources, uncertainties, and assumptions. Thus the conclusions can be inconsistent. Börjesson and Berglund (2006a, 2006b, 2007) reported a series of LCA studies on AD of various organic wastes using different technologies, comparing the results with a group of more conventional practices. They concluded that biogas systems have the potential to be an effective strategy in combating several serious environmental problems such as climate change, acidification, eutrophication, and air pollution. The emission reductions come more from indirect or background processes than from direct or foreground processes. However, the specific results vary significantly with different raw materials digested, the energy services provided and the reference systems replaced. Another conclusion is that there are various uncertainties regarding the availability of accurate input data and assumptions about technologies and geographical conditions. Crolla et al. (2007) reported an LCA study based on monitoring data from four on-farm digesters in Ontario, Canada. They observed a 95 % emission reduction of volatile fatty acids and 70-95% reduction of pathogens in digestate compared to raw manure. The life cycle CH_4 emissions also dropped from 245 kg CH_4/cow/year in the base scenario to 56 kg CH_4/cow/year with AD. De Vries et al. (2010) from Wageningen University conducted an LCA study comparing dairy manure digestion and dairy manure-silage maize co-digestion with non-digestion baseline. They observed that the climate change impact of digestion and co-digestion are 101 kg CO_2-eq and -3.9 kg CO_2-eq per ton digestate respectively compared to 149 kg CO_2-eq/ton raw manure in the base scenario. The significant reduction in greenhouse gas emissions in co-digestion is due to the high biogas generation rate for the replacement of fossil electricity. Co-digestion showed the potential to reduce acidification and eutrophication impacts significantly while not much difference was observed between the base scenario and pure manure digestion. Venczel and Powers (2010) studied the environmental impacts of conventional processing and AD on a medium sized dairy farm in the US. The study looked into fossil fuel consumption and climate change impact. Results showed clear benefits on fossil energy use. For GHG emission accounting, the authors reported difficulty in accurate quantification and suggested further studies. Mezzullo et al. (2012) reported an LCA study on the AD system of a cattle farm with a herd of 130 in UK; the biogas was used to displace kerosene heating oil as a fuel, whilst the digestate was considered for substitution of chemical fertilizers. This study analyzed only the AD system and thus does not constitute a full LCA comparison study. It observed net fossil energy savings from biogas production but found that the usage of biogas and digestate caused impacts on human respiratory systems and increased acidification and eutrophication effects. Lansche and Müller (2012) modeled four biogas plants digesting cattle manure with maize silage and grass silage in various proportions. The LCA study concluded that climate change impacts are reduced in all scenarios compared to the reference due to avoidance of manure storage and replacement of fossil electricity. Acidification and eutrophication were reduced with pure manure digestion whereas co-digestion showed increased impacts. Overall, it has been consistently revealed in previous studies that converting animal manure to biogas can reduce non-renewable energy consumption and climate change impact but it is uncertain whether acidification and eutrophication impacts can be reduced.

1.4 Objectives and Proposed Approaches

1.4.1 Objectives and research goals

In the previous sections two problems in BC's agriculture industry were identified: intensive manure generation with environmental burdens in its handling and high natural gas consumption associated with the greenhouse operation. Following the industrial ecology principles, there appears to exist an opportunity to integrate the animal farm and the greenhouse so that the waste manure can be converted to biogas in a digester, which is then used in the greenhouse for heating and CO_2 enrichment, replacing natural gas. The objective of this study is to explore and quantify the performance of an eco-industrial system in which livestock farming and greenhouse production are integrated to deal with manure waste and produce renewable energy for the greenhouse.

With AD as a good option for manure treatment and energy and resource production as explained above, the specific system studied achieves integration by building an anaerobic digester on the dairy farm. A dairy farm is selected because it has the highest manure generation in BC. The biogas produced is used to replace natural gas consumed for the two farms. Digestate is separated into liquid and solid phases. The solid phase is used as cow bedding material and greenhouse growing media. The liquid digestate is applied as fertilizer. In this way, we apply the principles of industrial ecology and realize the energy and material self-sufficiency. The expected benefits from such an eco-industrial system are reduced waste and pollution, protected local air quality, and abated greenhouse gas emissions.

Establishing AD systems on livestock farms is now a widespread practice, but integrating them with other agricultural activities to match energy and material demands and supplies has rarely been studied. The biogas product is usually fed into natural gas pipelines after purification (e.g. ADIAC, 2009) or used to generate electricity for the farm or the grid (e.g. Crolla et al., 2007; Gooch and Inglis, 2008; Gooch and Pronto, 2008a; Gooch and Pronto, 2008b; Pronto and Gooch, 2008; Shayya, 2008; Gooch and Pronto, 2009; Pronto and Gooch, 2009; Morris et al., 2010; Frost and Gilkinson, 2011). Heat is usually a by-product from the electricity generation facilities (Wright and Ma, 2003a; Wright and Ma, 2003b; Wright, 2004). Using biogas to generate electricity requires complex pretreatment of the raw biogas which impairs the economic feasibility. Some small scale digesters may rather flare the biogas, leading to wasted energy and air pollution (Torenvliet, 2011). The digestate is usually consumed on farms, but the surplus is occasionally sold to other users or disposed. The potential for integrating a farm AD system with other activities has been briefly mentioned by Row and Neabel (2005). So far, to the author's knowledge, only two studies by Huang and Bi (2008) and Hagen et al. (2011) have proposed an integrated livestock farm and greenhouse system for energy exchange and/or CO_2 enrichment in the greenhouse. In the study of Huang and Bi (2008), the livestock manure is combusted with the heat and CO_2 from the flue gas supplied to the greenhouse. In the study by Hagen et al. (2011), AD is applied and the biogas is primarily used for combined heat and power (CHP) generation. The CO_2 from the CHP facilities can substitute the conventional CO_2 from burning propane in the greenhouse. No further interactions were discussed, such as digestate usage as proposed in this study.

1.4.2 Proposed approaches

To achieve the objectives, life cycle analysis method is applied to quantify the environmental benefits associated with such an eco-industrial system.

The impact categories and the LCIA methods chosen in this study are further explained here. The categories are related to resource and energy use and to the impacts of the various emissions which specifically include non-renewable energy consumption, climate change, aquatic acidification, aquatic eutrophication, respiratory effects and human toxicity. These are selected to address the important concerns and to reflect the significant impacts from this system.

Non-renewable energy consumption is calculated by the Cumulative Energy Demand (CED) method version 1.04 developed by EcoInvent and updated by PR é Consultants. The version used in this study allows distinguishes three classes of non-renewable energy: fossil energy, nuclear energy, and energy from primary forests (Frischknecht et al., 2007). More than 99.5% of the non-renewable energy consumption in this study is fossil energy with a majority as natural gas. Therefore the total non-renewable energy consumptions are analyzed without further descriptions of specific types of non-renewable energy. The characterization unit is GJ primary energy.

Climate change impact is characterized using IPCC's 2007 global warming potential (GWP) factors with a timeframe of 100 years (Solomon et al., 2007). This impact category is also frequently termed global warming but, since it is uncertain whether the local effect will be warming, the more general term "climate change" is used. This method only includes the direct GWP of air emissions. Indirect GWP of emissions such as CO and transformation of N_2O from nitrogen is not included. Biogenic CO_2 is counted as carbon neutral and therefore no GWP is assigned. However for biogenic CH_4, a GWP is assigned since before the CH_4 is transformed into CO_2, it acts as a GHG in the atmosphere (Varshney and Attri, 1998). The detailed GWP values can be found in the IPCC Fourth Assessment Report (Solomon et al., 2007). CO_2 is universally used as the benchmark compound, and the climate change impact is expressed as mass of CO_2-equivalent.

Aquatic acidification and aquatic eutrophication are analyzed using the method CML 2001 version 2.05. CML 2001 is an LCA methodology developed by the Center of Environmental Science (CML) of Leiden University in the Netherlands (Frischknecht et al., 2007). It characterizes acidification and eutrophication potentials of all relevant N, S and P species (Huijbregts, 1999). The benchmark substances for acidification and eutrophication are SO_2 and PO_4, respectively. This method was developed in Europe with the background models representing European conditions. However, since the alternative methods are very limited, this method is also widely used for regions other than Europe.

Impacts on human health are analyzed using IMPACT 2002+ method version 2.10. These include respiratory inorganic emissions, respiratory organic emissions, carcinogen emissions, and non-carcinogen emissions. Specifically, the generic potential impact factors are calculated at a continental level for Western Europe (Jolliet et al., 2003). Respiratory effects from emissions distinguish between inorganic and organic species. Dose-response relations derived from epidemiological studies are used to determine the hazard potential of each substance (Goedkoop and Spriensma, 1999). Epidemiological studies have shown that several inorganic substances are related to respiratory effects in humans. The primary inorganic pollutants include particulate matters (PM), NOx, NH_3, CO, and SOx. $PM_{2.5}$, which is the most harmful pollutant for respiratory system because it is small enough to invade even the smallest airways (USEPA, 2012), is selected as the benchmark substance for inorganic respiratory pollutants. A number of volatile organic compounds (VOCs) are observed to have respiratory effects; C_2H_4 is taken as the benchmark substance for organic respiratory pollutants (Goedkoop and Spriensma, 2000). Human toxicity is determined as the sum of carcinogenic and non-carcinogenic impacts. The characterization factors for carcinogen and non-carcinogen emissions are calculated by dividing the human damage factor of the considered substance by that of the benchmark substance. For both groups, the benchmark substance is C_2H_3Cl; this is a declared human carcinogen with well-defined fate data and its main impact pathway is air inhalation (Jolliet et al., 2003).

Monte Carlo stochastic method is used to analyze the significance of difference between the base scenario and the integrated system. In a Monte Carlo analysis, the computer randomly picks up the input data from a probability distribution to calculate the results thus the range of the results reflects the uncertainty (Fishman, 1995). In this study, the Monte Carlo approach is used to present the probability that the impact of one scenario is higher/lower than that of another (Goedkoop, 2010).

This study is considered as an attributional LCA according to the classification of attributional and consequential LCA. An attributional LCA aims at describing the environmental properties of the life cycle of a product or a system whereas a consequential LCA aims at describing the effects of changes within the life cycle. Accordingly, an attributional LCA tends to use average data to evaluate the specific product or system while a consequential LCA usually includes indirectly affected processes and uses marginal data (Ekvall, 2003). An attributional LCA is usually the first step for providing information followed by a consequential LCA for decision-making. The goal of this LCA study is to systematically assess the potential advantages of eco-industrial integration of livestock farm and greenhouse. The environmental benefits which can thereby be achieved are quantified with the key contributions identified. The intended application is to provide knowledge on the environmental impacts of an integrated livestock farm-greenhouse system and to compare it to the conventional livestock manure management and greenhouse cultivation. The scope of this study covers the proposed eco-industrial system along with the corresponding conventional operations. The system boundary will be presented in the specific sections below. The data in this study are mainly secondary, obtained from literature reviews, reports, and LCA databases in SimaPro. The data are selected preferably relevant to BC. Interpretation of this study covers both the LCI and LCIA phases.

In this study, the collected environmental data are first organized in Excel and the life cycle system is built up and modeled in a licensed LCA software, SimaPro version 7.3.3. SimaPro is one of the leading LCA software packages; it has been developed by PRé Consultants. It has 14 databases containing unit or life cycle inventories of products and services in a number of sectors; the user can select these inventories or create new ones to build up the systems to be analyzed. SimaPro also provides popular and up-to-date LCIA methods on a variety of categories. Impact assessment is carried out using the above mentioned methods which are built in SimaPro. The software is able to assign the LCI data to the selected impact categories (classification) and calculate the impact results (characterization). The results for each category are therefore shown as a total value in units of the equivalent quantity of the benchmark substance.

1.5 Structure of Thesis

The thesis consists of three case studies in which three integrated eco-industrial systems are evaluated and presented in Chapters 2 to 4, respectively. In Chapter 2, a simple dairy farm-greenhouse eco-industrial system is presented with animal waste digested to provide heat and CO_2 for the greenhouse. In Chapter 3, the co-digestion of other organic waste is proposed and examined to increase the biogas production for the improvement of the performance of the eco-industrial system. In Chapter 4, a mushroom farm is introduced into the eco-industrial system to provide CO_2 to the greenhouse in the summer and also to use surplus solid digestate as the mushroom growing media. Each chapter starts with an introduction of the ideas and reasons for putting up or modifying an eco-industrial system, followed by a systematic life cycle assessment of the environmental performance of each eco-industrial system, with results and discussions presented in detail.

Conclusions for each specific study are given at the end of the relevant chapter. Chapter 5 draws the overall conclusions, states the limitations of this study and recommends some future work to improve the current study.

2. LCA OF AN INTEGRATED DAIRY FARM AND GREENHOUSE

2.1 Introduction

Dairy manure is the major manure waste generated in BC, so installing AD on a dairy farm is proposed and examined in this study. A typical dairy farm is selected for examining the feasibility of on-farm digestion and eco-industrial integration in this study defined as Case 1, although all livestock manure is suitable for AD. There are wide applications of dairy manure digestion alone or along with other organic wastes (Braun and Wellinger, 2003). In dairy manure, methanogens naturally exist which can digest the organic matters even after the manure is excreted. Thus dairy manure is especially a preferred feedstock for AD. However, it is worth noting that the biogas yield from dairy manure digestion is not as high as the yield from other common manures, i.e. swine and poultry manure (Werner and Strehler, 2012). The greenhouse modeled in this study is not strictly specified as any particular type. The majority of the data are from pepper growing since pepper is the most widely grown greenhouse crop in BC. However, some data used in the analysis are generic or averaged across different types of greenhouses.

The calculations to be presented show that there will be a surplus of solid digestate after meeting the demand for bedding material. The growing medium in the greenhouse, which is conventionally rockwool, is therefore proposed to be substituted by the surplus solid digestate. Also since the N nutrient in the digestate is higher than that in raw manure, part of the liquid digestate can be spared for greenhouse cultivation in addition to land spreading. As an organic fertilizer substitute for chemical fertilizer, the liquid digestate is potentially beneficial for organic greenhouses.

2.2 Framing of LCA study

2.2.1 Functional unit

The functional unit of this LCA study is selected as the “disposal of 1,100 tonnes of dairy manure”, corresponding to the amount of manure generated by 500 cows in one month (Electrigaz Technologies Inc., 2007). The size of the greenhouse is determined by matching the biogas generated from AD with the total gas demand in the summer season; it is calculated to be 5330 m^2. The gas demand in the summer is set as the base demand in a year and is primarily for CO_2 enrichment; the heat generated from burning the biogas actually exceeds what is needed in the summer and the excess heat is assumed to be dissipated along with the exhaust gases. In other seasons, natural gas supplementation is required.

2.2.2 System boundary

Figure 2-1 below shows the system boundaries of the base scenario (left) and the integrated eco-industrial system (right) in this study.

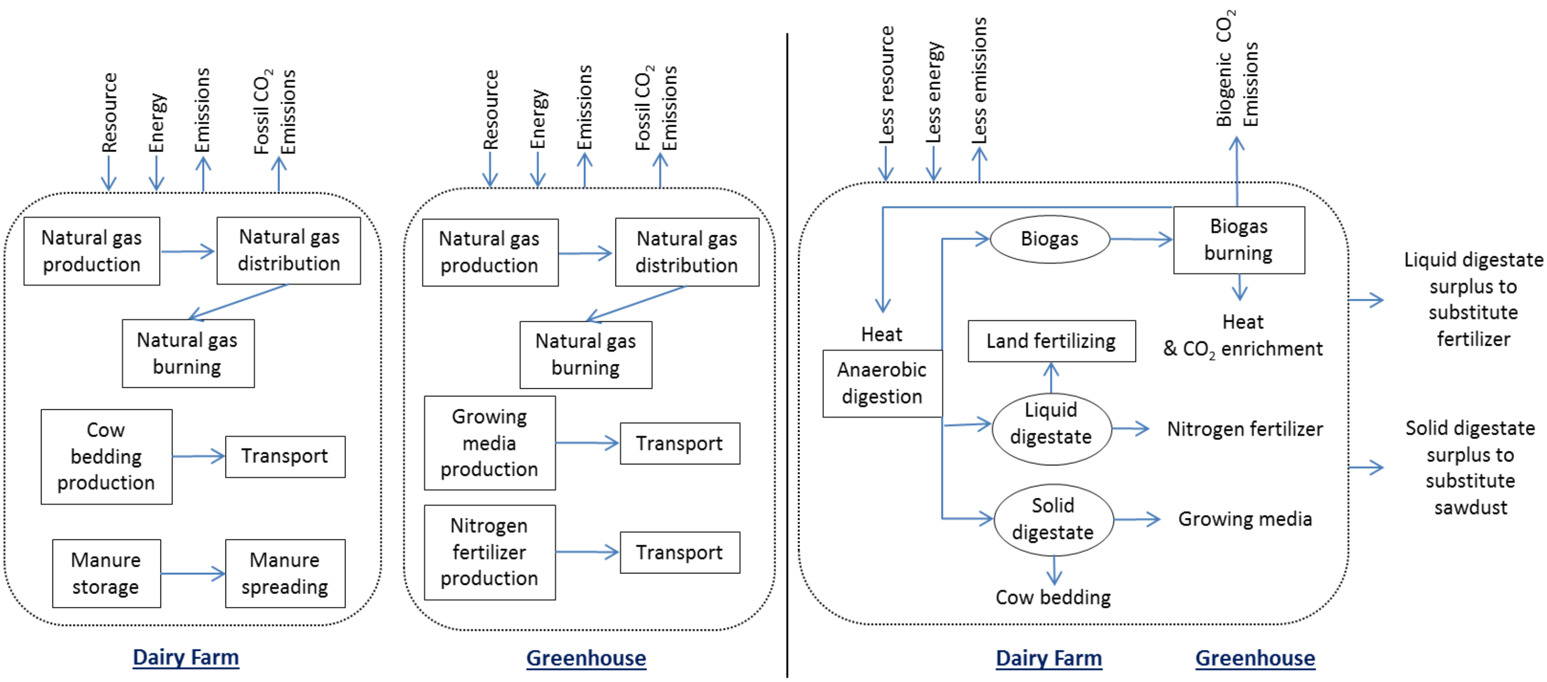


Figure 2-1 System boundaries of Case 1

As the LCA is intended to assess the advantages of the proposed eco-industrial system over the conventional system, the system boundaries for the conventional operation and the integrated system are kept strictly consistent and comparable, although the components may be different in the two systems.

The foreground life cycle stages for conventional operation are mainly dairy manure storage and land spreading, and natural gas combustion for heat and CO_2. Energy is used in the form of natural gas for heat and CO_2, diesel for vehicles, and electricity for farm operations. The main resources used in the conventional operation are sawdust for bedding, and rockwool substrate and nitrogen fertilizer for cultivation in greenhouses. Distribution of energy and transport of purchased resources are included. The crop land for spreading stored manure is assumed to be adjacent to the livestock farms so that no transport is needed. In the eco-industrial system, gas fuels and resources are substituted by biogas and digestates from on-site so that the production and transport are avoided. Dairy manure is digested directly on the dairy farm with no need for transport, storage, or pre-treatment. Biogas from AD is upgraded, and digestate is separated into liquid digestate and solids digestate with the solids digestate being further dried. Biogas is burned in boilers, replacing some of the natural gas used in the conventional operation. Liquid digestate spreading on crop land is also included so as to be consistent with the conventional manure spreading practice. Infrastructure data are only available for some of the agricultural facilities. Since the impacts from infrastructures are insignificant when allocated over the entire life time in this particular system, infrastructures are excluded from the current analysis.

In some scenarios, the surpluses of digestate in the eco-industrial system after substituting the base demand are counted as still useful for other agriculture producers so they can potentially be used to replace materials elsewhere to avoid production of those materials. The avoided products are included in the system boundary to generate environmental savings (negative impacts). Specifically, the avoided products in this study are sawdust and nitrogen fertilizers (modeled as urea ammonium nitrate), replaced by solid and liquid digestates, respectively. However, the surplus digestate is not necessarily useful in reality if local users cannot be found. It is thus possible that the surpluses have to be disposed as waste.

2.2.3 Scenarios

Table 2-1 defines three scenarios in this case study. Table 2-2 presents the basic data of the system. Table 2-3 shows the data at different seasons over a time horizon of one month.

Table 2-1 Scenarios for Case 1

Number	Scenario	Bio-waste composition
Scenario 1-a	Base scenario, conventional	100% dairy manure
Scenario 1-b	Eco-industrial system, optimistic	
Scenario 1-c	Eco-industrial system, moderate	

Table 2-2 Basic performance data for Case 1

Item	Value	Unit
Greenhouse area	5,330	m^2
Biogas generation	36,900	m^3/month
Dried solid digestate generation	114	tonne/month
Liquid digestate generation	791	tonne/month
Sawdust bedding demand in dairy farm	42	tonne/month

Item	Value	Unit
Growing media demand in greenhouse	557	kg/month
Dried solid digestate surplus	72	tonne/month
N fertilizer demand in greenhouse	40	kg N/month
Liquid digestate surplus	1.1	tonne N/month

Table 2-3 Seasonal performance data for Case 1

Season	Heat demand for dairy farm	Heat demand for greenhouse	CO_2 demand for greenhouse	Natural gas use in base cases	Heat demand for digester	Biogas generation	Heat supply from biogas	CO_2 supply from biogas	Natural gas supplement for heating
	GJ/month	GJ/month	tonne/month	GJ/month	GJ/month	m^3/month	GJ/month	tonne/month	GJ/month
Spring	114	762	55	988	134	36905	710	74	346
Summer	4	309	74	1,318	93	36905	710	74	0
Fall	97	586	55	988	127	36905	710	74	149
Winter	185	927	7	1,112	158	36905	710	74	596

Four seasons are analyzed for each scenario with calculations based on average data for a month in each season. One should note that one could also match the heat demand for winter operation, leaving surplus biogas in other seasons for export. The integrated system modeled here tries to avoid the export of biogas because CO_2 and other inert gases need to be removed before the biogas can be exported to the natural gas pipeline; the small quantity of biogas produced from the single digester does not seem to justify such an extra biogas purification effort. Since there is a surplus of digestate in the eco-industrial system, two scenarios are created to account for the difference in handling the surplus. In the optimistic scenario the liquid digestate surplus is treated to substitute urea ammonium nitrate fertilizer and the solid digestate surplus is used as a sawdust substitute. Whether this is possible depends on whether the digestate surplus can be successfully sold to other farms; this is actually happening in many places in the US (Wright, 2004; Gooch and Pronto, 2008a; Gooch and Inglis, 2008; Gooch and Pronto, 2009). In the other scenario, called the moderate scenario, the surplus is not utilized but has to be disposed as a waste. In this case, liquid digestate surplus is dumped and solid digestate surplus is composted; this can be necessary if the digestate surplus finds no value locally.

2.3 Description of Inventory Model

The main life cycle stages are described in this section starting with energy and material flows, followed by the foreground operations of livestock farm, greenhouse, and the integrated eco-industrial system. This section mainly explains the development of the system inventory with the specific statistical data and data sources presented in Appendixes A and B.

2.3.1 Inventory for Energy

- Natural gas

LCI data for natural gas are obtained from GHGenius software version 4.01 (Delucchi and Levelton, 2012). GHGenius differentiates natural gas produced for industrial use and for secondary energy (electricity) production, respectively. The data are not very different but are used for the relevant cases here: data for industry natural gas are selected for greenhouse heating while data for energy production are used for electricity usage in the LCI. The data are specifically for BC for the year 2012.

- Diesel

Diesel is consumed in heavy duty vehicles (HDV) for transport and tractors for manure and liquid digestate spreading. Data over the whole life cycle are used, covering feedstock extraction, fuel production, distribution and use. Off-road vehicles, i.e. tractors, generally consume more diesel than highway vehicles. In the current study, the diesel fuel consumption associated with tractors is quite small compared to highway vehicles. Therefore, diesel fuel use in this system is all modeled as highway vehicles as provided in the GHGenius software (Delucchi and Levelton, 2012). Data are for the year 2012.

- Electricity

Electricity is consumed in a number of processes. Direct consumption in this system is for manure storage, gas boiler operation, and digester operation. Other consumptions are associated with manufacturing processes such as sawdust production. Electricity mix data for BC are from Pa (2010), which are originally based on GHGenius version 4.01 and modified with the most updated data (Environment Canada, 2011). In 2011, BC's electricity mix was reported to consist of hydro, natural gas, biomass and fuel oil in the proportions of 93.35%, 5.1%, 1.45% and 0.1%, respectively (Environment Canada, 2011). We used this electricity mix in this study; it has generally been stable in the past few years but may change in the future. For all sources of electricity, the datasets cover the whole system from upstream extraction through transmission to end use (Delucchi and Levelton, 2012). GHGenius differentiates electricity generation from natural gas between steam turbine and gas turbine generators, with different efficiency. No details are available on the proportions of the two practices in BC so an average efficiency is used.

Datasets for some stages in this study are adopted from EcoInvent database which is developed in Europe based on European electricity mix data (Swiss Centre for Life Cycle Inventories et al., 2010). The electricity mix was changed to BC mix data because BC's electricity has much lower environmental impacts, specifically carbon intensity, than the European average.

2.3.2 Inventory for Materials

- Sawdust

Sawdust is widely used as bedding material for cows and cattle in BC. The high precipitation requires barns be used in all BC dairy farms; the bedding material consumption is typically 2.8 kg/cow/day (Torenvliet, 2011). The sawdust data in this study are obtained from a BC specific dataset developed by Pa (2010). It covers the life cycle from logging to sawmilling as well as transport using HDVs. Dried solid digestate can be used to supplement sawdust as bedding material. Thus the solid digestate surplus is modeled as a substitute for sawdust in the optimistic scenarios.

- Rockwool growing media

The growing medium for the greenhouses in the base scenario is assumed to be commercial rockwool products. Rockwool is not the most commonly used growing medium in BC, but it is selected due to data availability. A cradle to gate dataset for rockwool growing media product available from the LCA FOOD DK database in SimaPro is used in this study, which is based on production by the Grodan company, one of the largest rockwool manufacturers in the world. The application rate of growing medium is obtained from an interview with a local greenhouse owner (Yakel, 2011). In this study it is assumed that the properties of dried solid digestate are similar to rockwool so that the same application rate is used in the integrated eco-industrial system; in practice further investigation and experiment are recommended as they actually have different characteristics and additional sterilization might be necessary for solids digestate although in principle AD should provide sufficient sterilization.

- Fertilizers

Nitrogen (N), phosphorus (P), and potassium (K) contents of digestate from AD are analyzed to quantify fertilizer use. Digestate has a higher ammonium content due to the significant degradation of organic-bound N in the high pH digestion environment (Börjesson and Berglund, 2007; Edelmann et al., 2005). The ammonium is readily taken up by plants. Thus the digestate is considered as a more nutrient N-rich fertilizer compared to raw manure. It was reported that about 62.5% of the total N is plant available (in ammonium) in raw manure, while in digestate this percentage is 85% (DEFRA, 2010; Sommer et al., 2001). Compared to composting of plant waste, food waste and FOG waste, digestion also produces more plant-available nitrogen. Taking composting as reference, digestion produces an N surplus of 2.8 kg from 1 tonne plant waste feedstock and 1.6 kg from 1 tonne food waste or FOG waste (Börjesson and Berglund, 2007). These N fertilizer surpluses from the integrated system are considered in all scenarios. Also, nitrogen was reported to be partitioned into the liquid phase if the digestate was separated into a liquid and a solids phase, as in this study (Börjesson and Berglund, 2007). Therefore, the liquid digestate is treated here as an organic N fertilizer. In the optimistic eco-industrial scenarios, urea ammonium nitrate is assumed to be substituted by recovered surplus N from liquid digestate with the surplus amount being calculated by mass balance. The data for urea ammonium nitrate production are from EcoInvent database (Swiss Centre for Life Cycle Inventories et al., 2010).

P and K contents usually do not change during digestion, and they tend to stay in the solid digestate after separation (Börjesson and Berglund, 2007); thus only applying liquid digestate as fertilizer would initially lose these nutrients compared to the base scenario. About 64% of the P and 83% of the K are lost into the solid phase after digestion (Gooch and Pronto, 2008a; Pronto and Gooch, 2009). In this study, as the solid digestate is utilized as cow bedding and recycled to digester, the digestate will be eventually saturated with P and K and the P and K contents will increase in the liquid phase. Therefore the loss of P and K is expected to be much less than 64% and 83%, respectively. However, phosphorus and potassium are applied excessively on land in BC when raw manure is used (Poon and Schmidt, 2010; Schmidt and Hughes-Games, 2010), which causes problems including eutrophication. Management to limit fertilization with P and K has therefore been required. Under such circumstances, losing some P and K in the liquid manure can be beneficial. Even with reduced P and K content, the liquid digestate is expected to still meet the fertilizing requirement more than raw manure. Therefore, no chemical P and K fertilizers are required to compensate the losses in P and K in the integrated system. Subject to more detailed information and data available in the future, further calculations should be conducted to quantify the P and K fertilizer balance.

2.3.3 Inventory for System operations

- Transport

Transport is involved for the purchased sawdust, rockwool, and chemical fertilizers in the base scenario. In the integrated eco-industrial system, transport is avoided since the substitute materials are generated onsite. A diesel-fuelled lorry with 20-28 tonnes load capacity is modeled for the transport. LCI data for transport are local data from Pa (2010) with the average distance modified to 10 km which is an estimate of a reasonable distance within the agriculture area in Lower Mainland of BC (Booy et al., 2011). The return trip is assumed to be empty.

- Manure storage

Raw dairy manure is a liquid slurry with a dry matter content of around 8% (Jungbluth et al., 2007). Conventionally it is stored in lagoons on farms. Storage of raw dairy manure in this study is modeled as in a tank on the dairy farm (Jungbluth, 2007). Electricity is consumed for operations associated with pumping and is thus proportional to the amount of manure to be stored. Emission factors for manure storage are all obtained from literature. CH_4 and N_2O emissions are derived from IPCC. Some data were originally reported in unit of kg/head/year, which were converted to kg/tonne using manure generation rate by the NERC manure generation calculator (NERC). NH_3 emissions are calculated based on the total ammonia nitrogen with proper allocation between storage and spreading (Stucki et al., 2011). Data for manure composition were resourced from ASAE (2003). The H_2S emission factor was taken from a study by Alberta Agriculture (Atia et al., 1999).

- Manure spreading

Stored manure is applied to crop land near the dairy farm as fertilizers. Manure application is done by vacuum tank tractors (Nemecek and Kagi, 2007). There is a high uncertainty on emission factors for manure spreading because the available measurements differ in their scope and measurement method and refer to different environments. Most studies report emission rates at a certain time rather than cumulative emissions over the entire life span for a given amount of manure. As a result, there are limited data on emission factors over the whole life cycle of spread manure. Data selected for use here are not the average of all values reviewed because the original emission factors vary widely.

Instead, data from intensively cited papers are used. Specifically, NH_3 emissions are calculated based on the total ammonia nitrogen with allocation between storage and spreading (Stucki et al., 2011). N_2O emissions are based on total nitrogen content while CH_4 emissions are based on biogas potential. The H_2S emission factor is again obtained from a study by Alberta Agriculture (Atia et al, 1999).

- Composting

Composting is used to dispose solid digestate surplus in the moderate scenario of the eco-industrial system. The composting technology is modeled as windrow composting in which the organic matters are piled in long rows in the open air. This is a common practice to produce large amount of compost. The temperature and air flow control are usually realized by mechanical agitation (Composting Council of Canada, 2012). The inventory data are developed from the average values of two life cycle datasets for yard waste composting (RTI International, 2000; van Haaren et al., 2010) because solid digestate is similar to yard wastes as both originate from ligno-cellulosic plant materials. The data cover the whole chain from feedstock piling to end use of compost products. To be consistent, only the emission species reported by both sources are included, covering the most common and important emission compounds. The dataset was reported as a whole without stage-wise breakdowns such as on-site emissions or energy consumption for operations. Therefore, the composting stage will be analyzed as a whole on its environmental impacts.

- Anaerobic digestion

The AD stage is modeled using EcoInvent process data for the main reactor and emissions data obtained from the literature. The digester is modeled as a CSTR with the data sampled from 20 biogas plants in Switzerland (Jungbluth et al., 2007). The process is modified with no CHP on-site, unlike the original EcoInvent dataset. Electricity supply is modeled as from the local grid and heat is generated from biogas fuelled boilers. Heat is self-supplied by burning upgraded biogas. Raw manure is assumed to be digested without any storage. The dry matter content is assumed to be controlled at 12% for proper digestion; this falls within the recommended 10%~15% wet digestion range (Cant, 2006). The temperature inside the digester is modeled as stable at $35^{o}C$, corresponding to mesophilic digestion (Cant, 2006).

The biogas yield is assumed to be stable with a constant organic waste feed rate at a constant digester temperature. Data on biogas generation potential of raw manure are averaged values of literature data. Raw biogas is assumed to contain 60% CH_4, 37.9% CO_2, 2% H_2O, and 0.1% H_2S; this is a typical composition with other trace impurities neglected (BC Ministry of Environment, 2010). All emissions from the digestion process are assumed to be fugitive emissions; these are reported to comprise typically 0.5% of the generated biogas (Börjesson and Berglund, 2007; Swiss Centre for Life Cycle Inventories et al., 2010).

- Biogas upgrading

Raw biogas from the digester is piped into two adsorbent columns at elevated pressure, where H_2O and H_2S are removed by limestone and iron milling filings, respectively (Torenvliet, 2011). CO_2 is not removed as the biogas is used for combustion rather than for engines or grid distribution. No energy consumption is accounted for in this process. The production of the two materials is included as background processes with the dataset cited from EcoInvent (Swiss Centre for Life Cycle Inventories et al., 2010). Direct emissions from biogas upgrading are the fugitive emissions, calculated as 0.2% of the raw biogas produced (Börjesson and Berglund, 2007). Water vapour and H_2S are assumed to be completely removed in the upgrading process.

- Digestate separation and solid digestate drying

Separation of raw digestate into solid and liquid phases is done by a press-screw separator. Electricity consumption was reported to be 4.3 MJ primary energy/tonne digestate corresponding to 3.58 MJ electricity /tonne digestate (Pöschl et al., 2010). Solid digestate is assumed to be dried by natural drying, like the curing phase of composting, without consuming any energy. No emission data for digestate separation and drying are available; therefore emission rates are estimated, following Börjesson and Berglund (2006), by assuming that 20% of the total land application emissions occur during digestate separation and drying. This is justified by the fact that the emissions from digestate come from the remaining reactions of the digested matter, which are expected to take place during separation, drying, and usage. In addition, the separation process is modeled as taking place in an enclosed space, so that the emission can be further reduced to 20% (Stucki et al., 2011). Specifically, the NH_3 emission is calculated as 11.7% of the original total ammonia nitrogen in the manure, as suggested by Stucki et al. (2011).

- Digestate application

In the integrated system scenarios, spreading of liquid digestate is modeled using the same vacuum tank tractor as for manure application. The crop land for liquid digestate application is assumed to be the same as the land on which raw manure and compost are applied in the base scenarios so that no land change impact needs to be considered. Consistent with the assumption mentioned above, 80% of the total digestate emissions are allocated to digestate application. In addition, no data are available for emissions from solid digestate and liquid digestate, separately; the same emission factors are therefore applied to both solid and liquid digestate application. Specifically, NH_3 emissions are calculated as 45.8% of the original total ammonia nitrogen in the manure, as suggested by Stucki et al. (2011). Values for N_2O emissions are taken from another paper by Börjesson and Berglund (2007).

- Heat and CO_2 supply from boiler

Heat for the dairy farm and the greenhouse in the base scenario and integrated systems is supplied by a modulating boiler (Swiss Centre for Life Cycle Inventories et al., 2010). In the base scenario the boiler in the greenhouse consumes natural gas while in the integrated system the boiler burns upgraded biogas. The higher heating values of natural gas and upgraded biogas are 38 MJ/m^3 and 22 MJ/m^3, respectively[1] (Bartok, 2004; Fortis BC, 2012). The energy efficiency of the boiler is 96% and 90% for burning natural gas and biogas, respectively (Börjesson and Berglund, 2006, Swiss Centre for Life Cycle Inventories et al., 2010). The boiler is equipped with a basic flue gas control device with no specific type of device being reported (Swiss Centre for Life Cycle Inventories et al., 2010). Monthly average heating demands are calculated for each season based on the difference between required greenhouse temperature and ambient temperature. Natural gas consumption varies between seasons while biogas generation is assumed to be continuous and stable year-round. As explained above, the biogas supply is designed to match the lowest heating demand, which occurs in the summer season, to avoid export of surplus biogas. Emissions from the boiler consist of CO_2 and gases from incomplete oxidation as well as other air pollutants. The CO_2 emission rate from the natural gas boiler is obtained from emission factors in EcoInvent (Swiss Centre for Life Cycle Inventories et al., 2010) while the CO_2 from biogas combustion is calculated by mass balance. Carbon emissions from natural gas combustion and from biogas combustion are of fossil and biogenic origin, respectively; the latter is considered as carbon neutral when analyzing global warming impact (Solomon et al., 2007).

[1] The higher heating value of 22 MJ/m^3 for biogas corresponds to a biogas with 58% CH_4, which is slightly different from the typical value of 60% CH_4 but is well within the range of variations encountered in practice.

2.4 Results and Discussions

2.4.1 Life Cycle Inventory data

There are up to 600 substances involved in the inputs and outputs of the system studied. The Life Cycle Inventories (LCI) of selected key air emissions are presented in this section. The selected gases are CO_2 fossil, CH_4 fossil, CH_4 biogenic, N_2O, NOx, SOx, NH_3, H_2S, and NMVOCs (Non-Methane Volatile Organic Compounds), and PMs (PM_{10}, $PM_{2.5}$, and other PMs). These emissions are common species and major contributors to climate change (CO_2 fossil, CH_4, N_2O), eutrophication and acidification (NOx, SOx, NH_2, H_2S) and human health (NH_3, H_2S, NMVOC, and PMs), respectively. NOx refers to NO and NO_2; the term is commonly used to represent the sum of the two species. SOx mainly refers to two major sulphur oxides, SO_2 and SO_3, most commonly formed from fossil fuel processing and burning (Delucchi and Levelton, 2012). NMVOC is a group of organic compounds that are volatile at room temperature. Typically they include alcohols, aldehydes, alkanes, aromatics, ketones and halogenated derivatives of these substances. PMs are further classified based on the diameters of the particles. PM_{10} and $PM_{2.5}$ refer to particulate matters with a diameter less than 10 μm and 2.5 μm, respectively. Larger particles and those reported without specified diameters are classified as "other PMs" in this study. Table 2-4 shows the LCIs of the three scenarios for the different seasons. Tables 2-5 and 2-6 respectively summarize the absolute and percentage emission reductions for the eco-industrial system compared to the base scenario. It is noted that not all emission species are reduced in the moderate scenarios of the integrated eco-industrial system. Figure 2-2 shows the emissions of each major air pollutant for the summer season, divided into life cycle stages to further illustrate the contributions of specific processes. Specifically, the three diagrams refer to Scenarios 1-a to 1-c from top to bottom. Differences for the other seasons arise only from natural gas supply and use; therefore stage-wise results for the other three seasons are not presented to avoid repetition.

Table 2-4 Selected LCI results for three scenarios of Case 1
(Scenario a: base scenario; Scenario b: optimistic eco-industrial system; Scenario c: moderate eco-industrial system)

		Spring			Summer			Fall			Winter		
	Unit	Scenario 1-a	Scenario 1-b	Scenario 1-c	Scenario 1-a	Scenario 1-b	Scenario 1-c	Scenario 1-a	Scenario 1-b	Scenario 1-c	Scenario 1-a	Scenario 1-b	Scenario 1-c
CO_2 fossil	tonne	65	17	23	86	-5	1	65	5	11	73	33	39
CH_4 fossil	kg	79	18	29	103	-7	4	79	4	15	88	36	47
CH_4 biogenic	tonne	1.98	0.15	0.16	1.98	0.15	0.16	1.98	0.15	0.16	1.98	0.15	0.16
N_2O	kg	74	3	14	74	3	14	74	3	14	74	3	14
NOx	kg	62	62	92	75	49	79	62	54	84	67	72	102
SOx	kg	14	-2	7	17	-5	4	14	-4	5	15	0	9
NH_3	kg	616	494	659	616	494	659	616	494	659	616	494	659
H_2S	kg	422	0.24	0.26	422	0.23	0.26	422	0.23	0.26	422	0.24	0.26
NMVOC	kg	6.18	-0.31	1.98	7.89	-2.10	0.20	6.18	-1.33	0.97	6.82	0.98	3.27
PM_{10}	kg	0.06	-0.15	0.41	0.07	-0.16	0.41	0.06	-0.16	0.41	0.07	-0.15	0.42
$PM_{2.5}$	kg	0.19	-1.14	0.19	0.23	-1.18	0.15	0.19	-1.16	0.17	0.20	-1.11	0.22
Other PM[1]	kg	3.65	1.09	6.72	3.82	0.91	6.54	3.65	0.99	6.62	3.71	1.21	6.84

[1]Other PM includes the particular matters with a diameter greater than 10 μm and the unspecified PMs. This applies to the following tables as well.

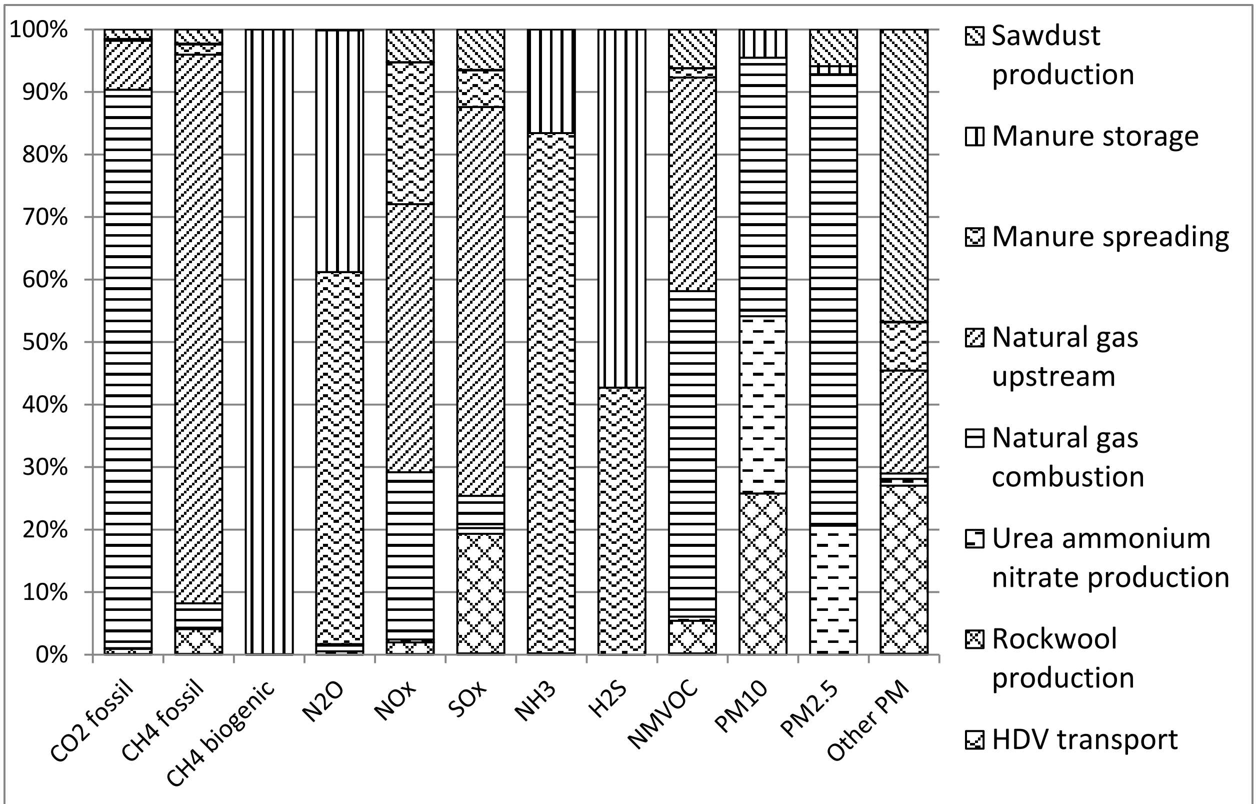
Table 2-5 Net emission reductions of the eco-industrial system over the base scenario in Case 1

		Spring		Summer		Fall		Winter	
	Unit	Scenario 1-b	Scenario 1-c	Scenario 1-b	Scenario 1-c	Scenario 1-b	Scenario 1-c	Scenario 1-b	Scenario 1-c
CO_2 fossil	tonne	48	42	91	85	60	55	40	34
CH_4 fossil	kg	61	51	110	99	75	65	52	42
CH_4 biogenic	tonne	1.83	1.83	1.83	1.83	1.83	1.83	1.83	1.83
N_2O	kg	71	60	71	60	71	60	71	60
NOx	kg	0	-30	27	-3	8	-22	-5	-35
SOx	kg	16	7	22	13	17	9	15	6
NH_3	kg	122	-43	122	-43	122	-43	122	-43
H_2S	kg	422	422	422	422	422	422	422	422

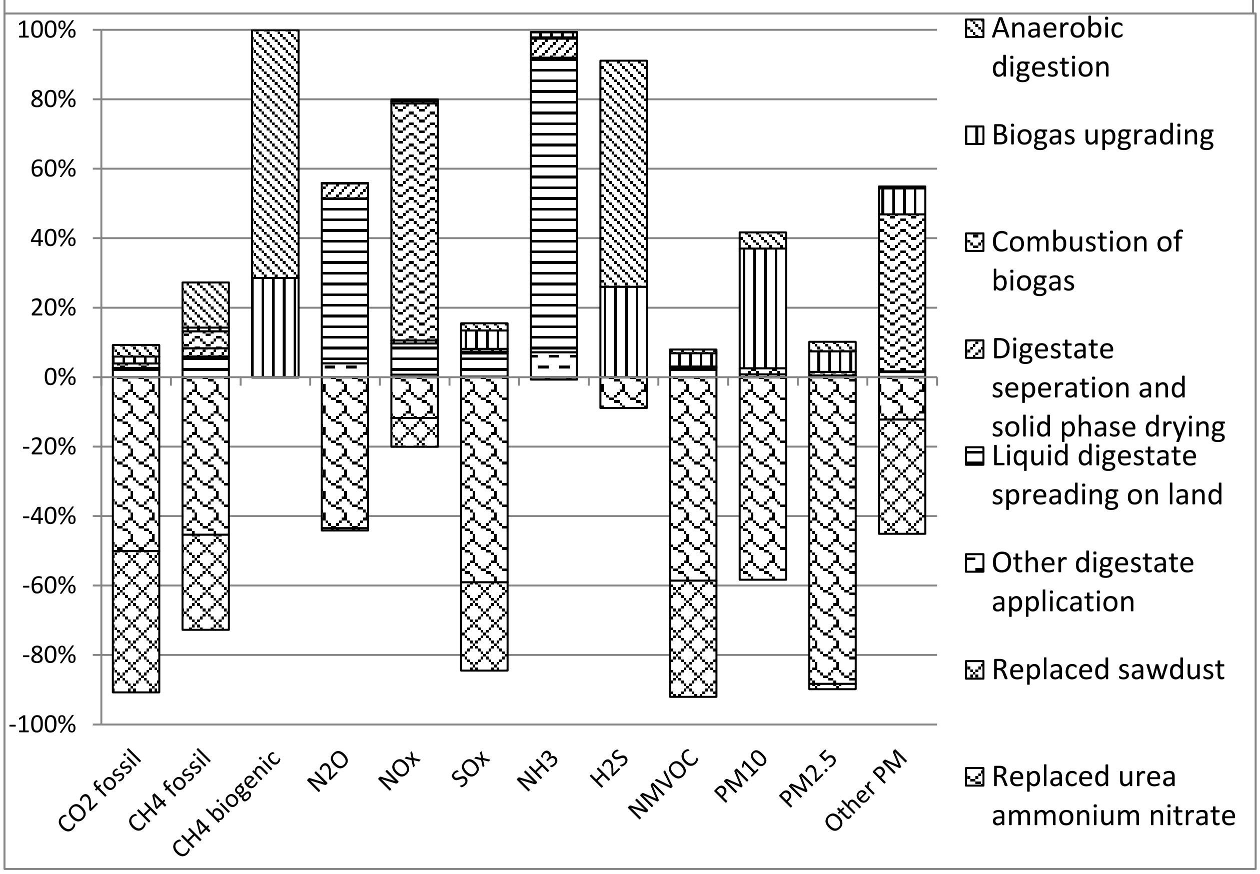
		Spring		Summer		Fall		Winter	
	Unit	Scenario 1-b	Scenario 1-c	Scenario 1-b	Scenario 1-c	Scenario 1-b	Scenario 1-c	Scenario 1-b	Scenario 1-c
NMVOC	kg	6.49	4.20	9.98	7.69	7.51	5.22	5.85	3.55
PM_{10}	kg	0.22	-0.35	0.23	-0.33	0.22	-0.35	0.22	-0.35
$PM_{2.5}$	kg	1.32	-0.01	1.41	0.08	1.35	0.02	1.31	-0.02
Other PM	kg	2.57	-3.06	2.91	-2.72	2.67	-2.96	2.50	-3.13

Table 2-6 Percentage emission reductions of the eco-industrial system over the base scenario in Case 1

	Spring		Summer		Fall		Winter	
	Scenario 1-b	Scenario 1-c	Scenario 1-b	Scenario 1-c	Scenario 1-b	Scenario 1-c	Scenario 1-b	Scenario 1.1-c
CO_2 fossil	73%	65%	105%	99%	92%	84%	54%	47%
CH_4 fossil	77%	64%	106%	96%	95%	82%	59%	47%
CH_4 biogenic	92%	92%	92%	92%	92%	92%	92%	92%
N_2O	96%	81%	96%	81%	96%	81%	96%	81%
NOx	0%	-49%	35%	-5%	12%	-36%	-8%	-53%
SOx	115%	53%	130%	78%	127%	65%	100%	42%
NH_3	20%	-7%	20%	-7%	20%	-7%	20%	-7%
H_2S	100%	100%	100%	100%	100%	100%	100%	100%
NMVOC	105%	68%	127%	97%	122%	84%	86%	52%
PM_{10}	341%	-545%	327%	-467%	348%	-538%	322%	-526%
$PM_{2.5}$	713%	-4%	621%	34%	727%	9%	651%	-12%
Other PM	70%	-84%	76%	-71%	73%	-81%	67%	-84%



100%
90%
80%
70%
60%
50%
40%
30%
20%
10%
0%
CO2 fossil
CH4 fossil
CH4 biogenic
N2O
NOx
SOx
NH3
H2S
NMVOC
PM10
PM2.5
Other PM
Sawdust production
Manure storage
Manure spreading
Natural gas upstream
Natural gas combustion
Urea ammonium nitrate production
Rockwool production
HDV transport



100%
80%
60%
40%
20%
0%
-20%
-40%
-60%
-80%
-100%
CO2 fossil
CH4 fossil
CH4 biogenic
N2O
NOx
SOx
NH3
H2S
NMVOC
PM10
PM2.5
Other PM
Anaerobic digestion
Biogas upgrading
Combustion of biogas
Digestate seperation and solid phase drying
Liquid digestate spreading on land
Other digestate application
Replaced sawdust
Replaced urea ammonium nitrate

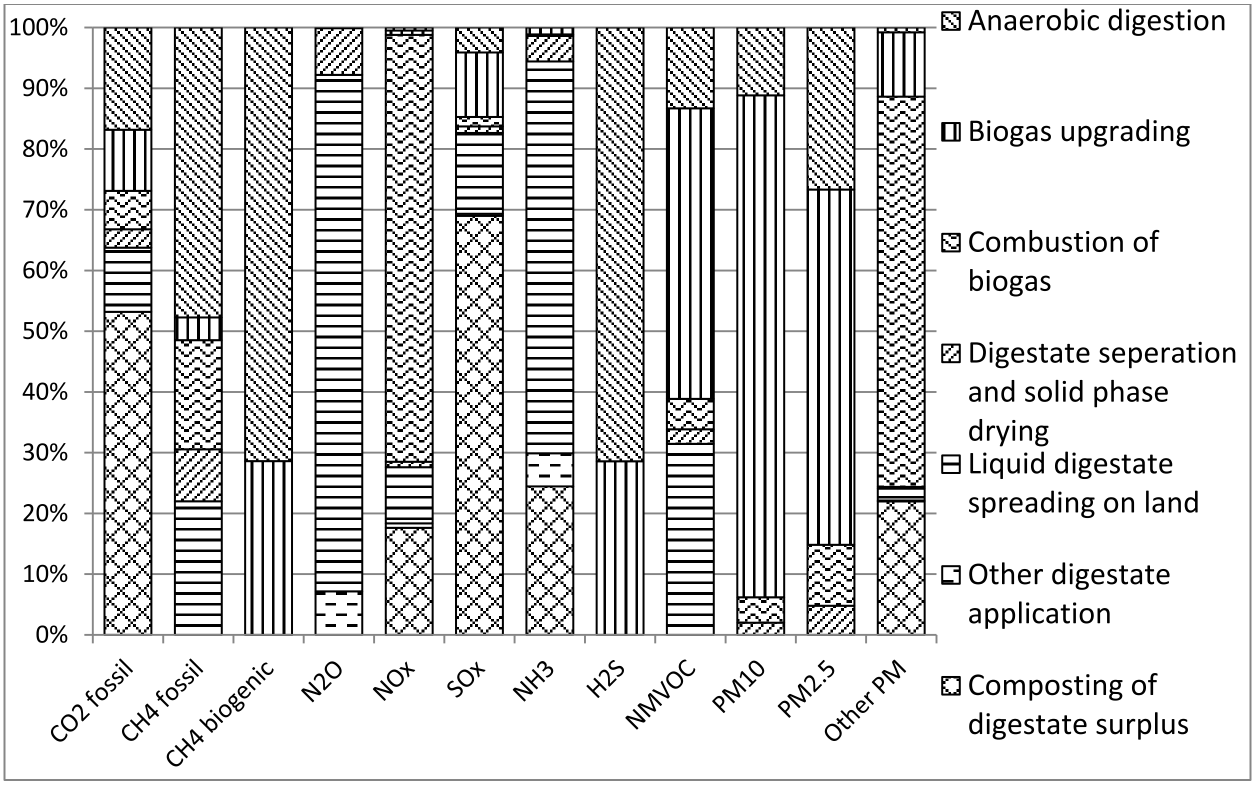


Figure 2-2 Stage-wise selected LCI results for Case 1

Overall, the LCI results show that under the optimistic circumstances, PMs, NMVOCs, SOx emissions have significant reductions (>100%) in all seasons. GHG emissions such as CO_2, CH_4, and N_2O are all reduced substantially (year-round average >80%) as well. In the moderate scenario where surplus digestate is composted instead of being used to replace resources, the reductions are less significant; some emissions such as NOx and PMs even show (much) higher values than the base scenario. The reasons will be explained in detail below. The seasonal variations are all related to the fluctuations of natural gas supplement and use. It is noted that reductions of biogenic CH_4 and N_2O emissions are the same in different seasons, because these two emissions are mainly associated with manure treatment rather than natural gas production or combustion. For the emissions that have notable seasonal variations, natural gas production and combustion must be significant contributors.

- Fossil CO_2 emissions

About 90% of the fossil CO_2 emissions in the base scenario come from natural gas combustion. The second significant fossil CO_2 source is the natural gas upstream processing. In the eco-industrial system where natural gas production and combustion are avoided, composting of solid digestate surplus appears to be the biggest contributor of fossil CO_2 emissions in the moderate scenario. Since the specific composting process is treated as a black box, as explained in Section 2.3.3, further investigation of the CO_2 source is not possible. However, it is reasonable to infer that diesel and electricity use in composting are responsible for most fossil CO_2 emissions. The data are cited from studies in the US, thus the actual emissions for BC are expected to be lower due to the hydro-dominant electricity mix. The operation of the biogas system accounts for the rest of the fossil CO_2 emissions in the eco-industrial scenarios, arising from the energy use in digestion, biogas upgrading, and digestate treatment. In the optimistic scenario, significant fossil CO_2 savings are observed from using surplus digestate to replace resources. Savings from replacement of sawdust and urea ammonium nitrate are both significant, and are higher than the sum of net emissions from biogas system operation. As a result, a net fossil CO_2 emission saving is achieved.

- Fossil CH_4 emissions

About 90% of the fossil CH_4 emissions in the base scenario results from production and distribution (upstream) of natural gas. Rockwool production and natural gas combustion also create some fossil CH_4 emissions. Further scrutiny of the rockwool production process shows that the emissions are mostly from consumption of fossil fuels such as coal and natural gas (Nielsen et al., 2003). In the eco-industrial system, the AD process turns out to be the biggest contributor followed by land spreading of liquid digestate, combustion of biogas, and digestate separation and drying. This follows a similar pattern to the fossil CO_2 emissions, with these emissions arising from electricity and diesel use. Different from fossil CO_2 emissions, composting of the digestate does not generate significant fossil CH_4 emissions but, again, the details on the data of composting process are not available. Also in the optimistic scenario, significant savings can be realized by substituting sawdust and nitrogen fertilizers with surplus digestate.

- Biogenic CH_4 emissions

Almost all the biogenic CH_4 emissions arise from manure storage in the base scenario, because of the anaerobic environment in the storage tank. CH_4 emission factors for land spreading of manure are reported to be one order of magnitude lower than for manure storage (Dong et al., 2006; Amon et al., 2006). In the eco-industrial scenarios, the CH_4 emissions are mainly from AD and biogas upgrading, in the form of leakages as explained in Section 2.3.3. No biogenic CH_4 emission is reported to be associated with rockwool production and only trace emissions occur during sawdust production; therefore, net savings in the optimistic scenario are negligible.

- N_2O emissions

Another GHG, N_2O is emitted intensively from manure storage and spreading in conventional practices. N_2O is generated as a by-product from combined nitrification and de-nitrification of nitrogen species as a consequence of changes in the aeration conditions. Generally N_2O production is more favoured in aerobic conditions, thus manure spreading creates relatively higher emissions. N_2O formation also takes place in the digestate so that digestate separation, drying and application account for the N_2O emissions in the eco-industrial scenarios. However, digestate provides less energy for the N_2O-forming microorganisms because the remaining organic matters are more difficult to decompose. This leads to a reduction of total N_2O emissions from digestate compared to raw manure (Börjesson and Berglund, 2007). N_2O emissions account for a significant proportion of GHG emissions for urea ammonium nitrate production because nitric acid is an intermediate product in the ammonium nitrate synthesis process (Wood and Cowie, 2004). Thus a large N_2O emission reduction is achieved from the substitution of nitrogen fertilizers by solid digestate surplus in the optimistic scenario.

- NOx emissions

NOx emissions in the base scenario arise mostly from natural gas upstream processing, natural gas combustion, and manure spreading, as a result of combustion of fuels directly or indirectly. Some NOx is also generated from the decomposition of the organic matters, but the quantity is insignificant (Hao and Chang, 2001). A small proportion of NOx emissions come from sawdust production; further investigations identified diesel use as the major source (Pa, 2010). In the eco-industrial systems, NOx emissions mainly arise from biogas combustion, digestate composting, and liquid digestate spreading, of which the biogas combustion is the dominant source. NOx emission factor of biogas combustion is higher than that of natural gas combustion. NOx emissions from digestate composting and digestate spreading are similar to the emissions from energy use in manure spreading. Overall, because of the high emissions from biogas combustion, the moderate eco-industrial scenario has a higher NOx emission than the base scenario (year-round average +36%). Even in the optimistic scenario where some reduction can be generated from substituted sawdust and nitrogen fertilizer production, the net reduction of NOx emissions is very limited.

- SOx emissions

Most of the SOx emissions in the base scenario are from natural gas upstream processing with fuel production and natural gas sweetening as the main contributors (Delucchi and Levelton, 2012). The second biggest source is rockwool production although detailed information on SOx generation in this specific rockwool dataset is not available (Nielsen et al., 2003). It is reported that SO_2 emissions from rockwool production mainly come from the use of coke in the melting process and from the sulphur content of the cement used in the waste recycling process (Rockwool France, 2012). Other SOx emissions in this system arise from combustion of fossil fuels. These include natural gas combustion, manure spreading, and sawdust production in the base scenario and surplus digestate composting, AD, and biogas upgrading in the eco-industrial system. In the optimistic scenario, significant SOx emissions are avoided from substitution of nitrogen fertilizer and sawdust; especially for urea ammonium nitrate, SOx emissions stem from fossil fuel consumptions during ammonia and nitric acid production. Overall, in the optimistic scenario, a negative emission inventory is observed due to the reuse of the surplus digestate.

- NH_3 emissions

NH_3 emissions in the base scenario arise predominantly from manure storage and spreading. Spreading has even higher emissions since it is cumulative over a long term compared to short-term storage. Rockwool production and urea ammonium nitrate production also generate NH_3 emissions but the amount is negligible within the life cycle. In the eco-industrial system, liquid digestate spreading is the most significant source of NH_3 emissions. As explained above, because of more ammonium in the digestate, NH_3 emission per tonne of spread digestate is higher than the NH_3 emission per tonne of spread raw manure (Petersen, 1999); however, less tonnage of digestate is required for a given land area compared to raw manure since the digestate contains more plant-available nitrogen (ammonium). Thus the overall NH_3 emission from digestate is lower than that from manure for a given land area. In addition, manure storage is not needed for AD so that the emissions during storage are avoided. Digestate separation and drying as well as other applications of digestate also release NH_3. Composting of surplus digestate in the moderate scenario of the integrated system contributes to the second largest NH_3 emission from decomposition of organic nitrogen compounds. NH_3 emissions from sawdust and urea ammonium nitrate production are very small, so that there are barely any NH_3 emission savings in the optimistic scenario.

- H_2S emissions

Similar to NH_3 emissions, almost all H_2S emissions in the base scenario come from manure storage and manure spreading. It is reported that emissions from the storage stage are higher than from land spreading since H_2S is produced under anaerobic conditions (Atia et al, 1999). In the biogas system, H_2S emission stems from the leakage of raw biogas and upgrading of biogas since H_2S is one of the main impurities of raw biogas. The results show a dramatic drop (almost 100%) of H_2S emissions when AD is used. This shows that a proper purification of biogas can remove hazardous H_2S emissions that would have occurred in conventional manure management. Nitrogen fertilizer production also produces some H_2S emissions, mostly from the ammonium production process, but the amount is not significant and the savings in the optimistic scenario are limited.

- NMVOC emissions

Most NMVOCs emissions in the base scenario of this system come from natural gas extraction, processing, and combustion. In the natural gas upstream stages, NMVOCs mostly stem from feedstock recovery and fuel production. Gas leakage and flaring also discharge NMVOCs (Delucchi and Levelton, 2012). Burning natural gas directly emits NMVOCs in the flue gas. The main NMVOC species are pentane, butane, and benzene (Swiss Centre for Life Cycle Inventories et al., 2010). Sawdust production (Pa, 2010) and rockwool production (Nielsen et al., 2003) also generate NMVOC emissions resulting from fuel refining and use. In the eco-industrial system, all NMVOCs are reported from indirect processes of energy and material production and use. No data are available for possible NMVOC emissions from AD, biogas use, or digestate treatment and use; therefore a bias may have been created due to the data gap. According to the results with currently available data, biogas upgrading and liquid digestate spreading account for most of the NMVOCs. Limestone milling is the major contributor in biogas upgrading stage. Diesel production and usage are the main emission sources in digestate spreading. NMVOCs from AD process and combustion of biogas are due to the electricity use. Urea ammonium nitrate production generates significant amount of NMVOC emissions, mainly from ammonia and nitric acid production (Swiss Centre for Life Cycle Inventories et al., 2010). Sawdust production also involves significant NMVOC emissions due to the energy consumption. In the optimistic scenario, total NMVOC emissions from the biogas system are fairly low compared to the savings from substituted sawdust and nitrogen fertilizers so that the net emission is far below zero.

- PM emissions

Emissions of PMs with different diameters show different attributional patterns in the inventories. It is worth noting that some data sources do not have detailed information for specific PM categories;

in these cases, the emissions are recorded as unspecified PM and included in the PM category with a diameter larger than 10 μm. In the base scenario, natural gas combustion emits one order of magnitude more $PM_{2.5}$ than PMs with a diameter ≥ 10 μm. Urea ammonium nitrate production produces similar amounts of $PM_{2.5}$, PM_{10}, and other PM; emissions of $PM_{2.5}$ and PM_{10} are reported to be equal, but account for only a tiny portion of the other PM since the other PM emission is much higher. Rockwool production also causes significant PM_{10} emissions, originating from fossil fuel processing (Nielsen et al., 2003). Sawdust production generates some $PM_{2.5}$ emissions from steam generation (Pa, 2010). Manure storage contributes to some PM_{10} emissions indirectly from electricity generation. For other PM, sawdust production is the biggest contributor, primarily from the process of wood sawing (Pa, 2010). PMs are also emitted from the upstream electricity and diesel production. Rockwool and natural gas production are the second significant contributors to other PM emissions with the majority of PM emissions from rockwool production arising from the manufacturing stage. PM emission from natural gas upstream is almost equally allocated to feedstock recovery, fuel production, and fuel storage and distribution (Delucchi and Levelton, 2012). Manure spreading also creates PM emissions as the process certainly gives rise to dust on the land. In the eco-industrial system, AD generates PM emissions indirectly from electricity production. Since no PM emission factors are available for the AD process, it is possible that some actual emissions are omitted in this study. Significant PM emissions stem from the manufacturing and disposal of milled limestone; therefore the biogas upgrading stage which consumes limestone contributes significantly to PM emissions. Combustion of biogas has direct emissions which are reported as unspecified PMs. Emissions of other PMs arise indirectly from upstream electricity production. In the moderate scenario, the PM emissions from biogas upgrading and combustion turn out to be tremendous, leading to a total inventory several times higher than the base scenario. The composting stage also reports unspecified PM emissions as dust generation from composting sites. Energy consumption for composting also produces some PM emissions. In the optimistic scenario, PM emission savings from substituted nitrogen fertilizer and sawdust are shown to be significant; especially for PM_{10} and $PM_{2.5}$ emissions, there are negative net emissions in the optimistic scenario. Some of the unspecified PM emissions are possibly small particles like PM_{10} or $PM_{2.5}$. Thus, the overall PM_{10} and $PM_{2.5}$ emissions are probably underestimated due to the limited data availability on this point.

2.4.2 Life Cycle Impact Assessment

The LCIA results are presented in Table 2-7 for each season, while Tables 2-8 and 2-9 summarize the net and percentage reductions in the eco-industrial systems compared to the base scenario. Stage-wise data are further shown in Figure 2-3 based on absolute characterization results. All numerical values for the charts are included in Appendix C for reference.

Table 2-7 Life Cycle Impact Assessment results for three scenarios of Case 1 (Scenario a: base scenario; Scenario b: optimistic eco-industrial system; Scenario c: moderate eco-industrial system)

		Spring			Summer			Fall			Winter		
		1-a	1-b	1-c	1-a	1-b	1-c	1-a	1-b	1-c	1-a	1-b	1-c
Non-renewable energy	GJ	1,270	346	441	1,677	-81	14	1,270	102	198	1,423	654	749
Climate change	tonne CO_2-eq	133	22	31	154	0	9	133	9	18	141	39	48
Aquatic acidification	kg SO_2-eq	1,827	820	1,108	1,837	809	1,098	1,827	814	1,103	1,831	827	1,116
Aquatic eutrophication	kg PO_4-eq	224	181	243	225	179	242	224	180	242	224	182	245
Respiratory inorganics	kg $PM_{2.5}$-eq	84	67	93	86	65	91	84	66	92	85	68	95
Respiratory organics	kg C_2H_4-eq	3.50	-0.13	1.09	4.41	-1.09	0.14	3.50	-0.68	0.55	3.85	0.55	1.78
Human toxicity	kg C_2H_3Cl-eq	227	-4	53	241	-18	39	227	-12	45	233	6	63

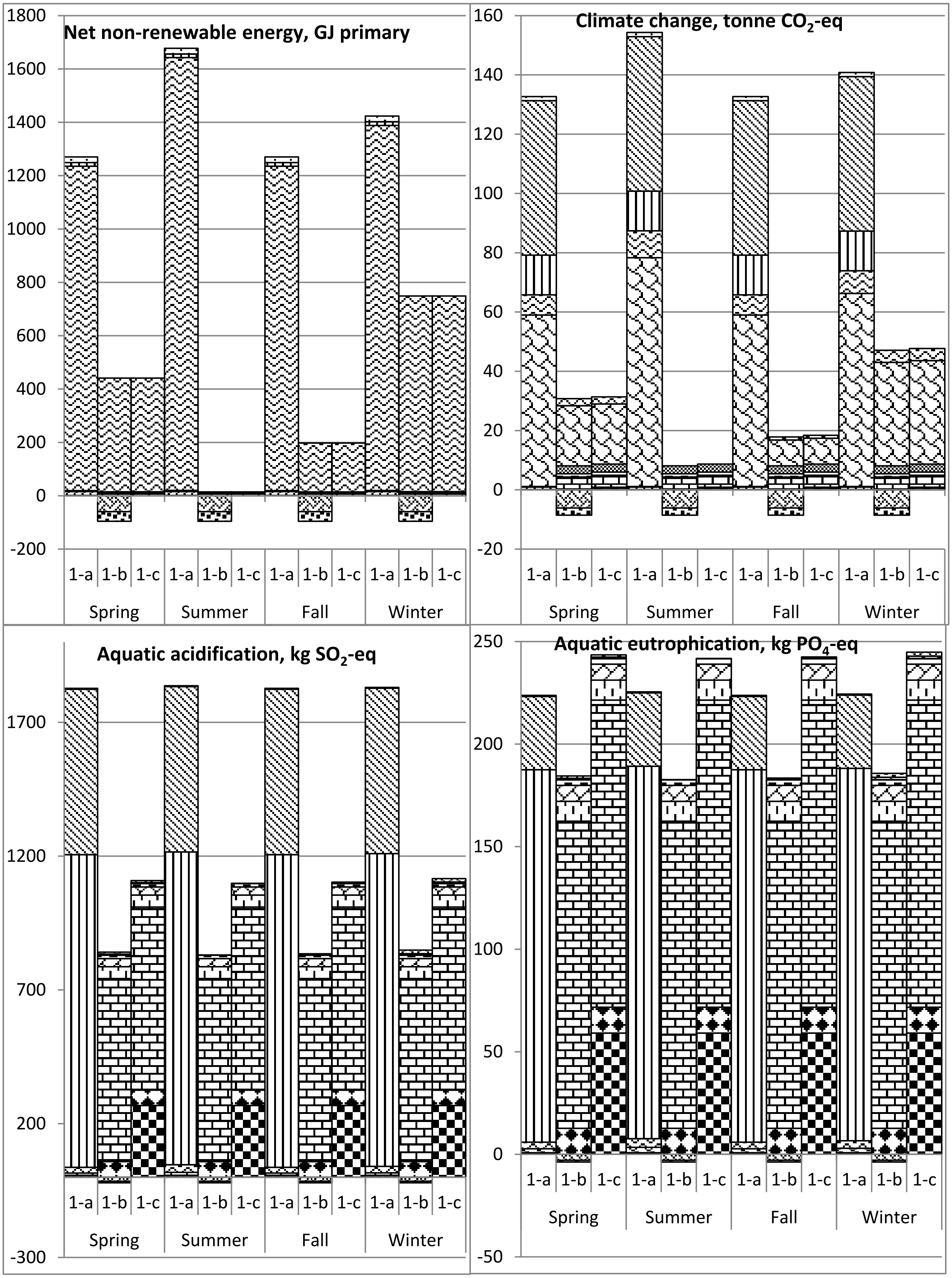
Table 2-8 Net impact reductions of the eco-industrial system over the base scenario in Case 1

		Spring		Summer		Fall		Winter	
		1-b	1-c	1-b	1-c	1-b	1-c	1-b	1-c
Non-renewable energy	GJ	925	829	1,759	1,663	1,168	1,073	770	674
Climate change	tonne CO_2-eq	110	101	155	146	123	114	102	93
Aquatic acidification	kg SO_2-eq	1,007	718	1,027	739	1,013	724	1,003	715
Aquatic eutrophication	kg PO_4-eq	43	-20	47	-16	44	-19	42	-20
Respiratory inorganics	kg $PM_{2.5}$-eq	18	-9	22	-5	19	-8	17	-10
Respiratory organics	kg C_2H_4-eq	3.64	2.41	5.50	4.27	4.18	2.95	3.29	2.06
Human toxicity	kg C_2H_3Cl-eq	231	175	259	202	239	183	226	169

Table 2-9 Percentage impact reductions of the eco-industrial system over the base scenario in Case 1

	Spring		Summer		Fall		Winter	
	1-b	1-c	1-b	1-c	1-b	1-c	1-b	1-c
Non-renewable energy	73%	65%	105%	99%	92%	84%	54%	47%
Climate change	83%	76%	100%	94%	93%	86%	73%	66%
Aquatic acidification	55%	39%	56%	40%	55%	40%	55%	39%
Aquatic eutrophication	19%	-9%	21%	-7%	20%	-8%	19%	-9%
Respiratory inorganics	21%	-11%	25%	-6%	22%	-9%	20%	-11%

	Spring		Summer		Fall		Winter	
	1-b	1-c	1-b	1-c	1-b	1-c	1-b	1-c
Respiratory organics	104%	69%	125%	97%	119%	84%	86%	54%
Human toxicity	102%	77%	107%	84%	105%	80%	97%	73%


Net non-renewable energy, GJ primary
1800
1600
1400
1200
1000
800
600
400
200
0
-200
1-a 1-b 1-c 1-a 1-b 1-c 1-a 1-b 1-c 1-a 1-b 1-c
Spring Summer Fall Winter
Climate change, tonne CO2-eq
160
140
120
100
80
60
40
20
0
-20
1-a 1-b 1-c 1-a 1-b 1-c 1-a 1-b 1-c 1-a 1-b 1-c
Spring Summer Fall Winter
Aquatic acidification, kg SO2-eq
1700
1200
700
200
-300
1-a 1-b 1-c 1-a 1-b 1-c 1-a 1-b 1-c 1-a 1-b 1-c
Spring Summer Fall Winter
Aquatic eutrophication, kg PO4-eq
250
200
150
100
50
0
-50
1-a 1-b 1-c 1-a 1-b 1-c 1-a 1-b 1-c 1-a 1-b 1-c
Spring Summer Fall Winter

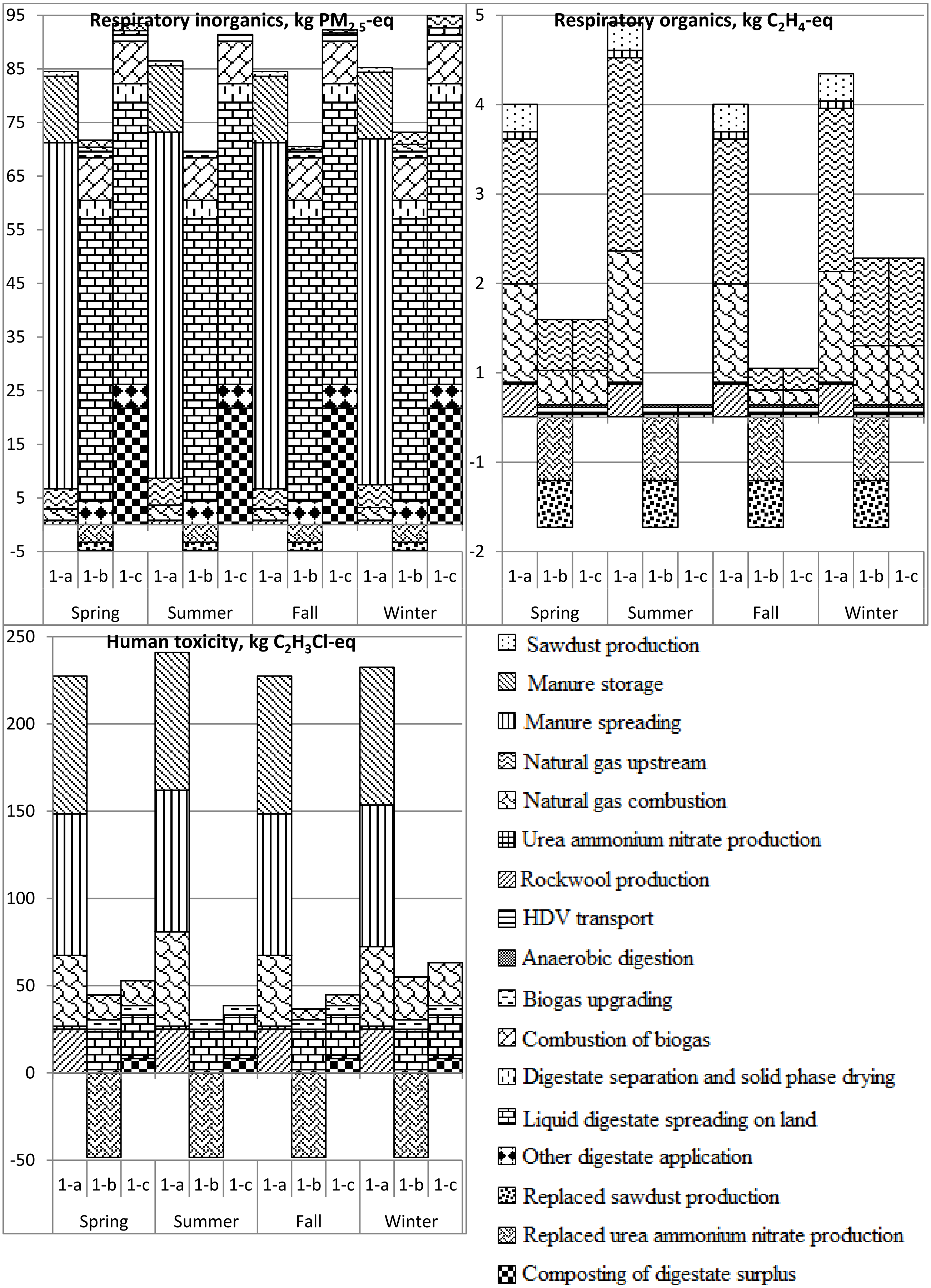

Respiratory inorganics, kg PM2.5-eq
Respiratory organics, kg C2H4-eq
Human toxicity, kg C2H3Cl-eq
1-a
1-b
1-c
Spring
Summer
Fall
Winter
Sawdust production
Manure storage
Manure spreading
Natural gas upstream
Natural gas combustion
Urea ammonium nitrate production
Rockwool production
HDV transport
Anaerobic digestion
Biogas upgrading
Combustion of biogas
Digestate separation and solid phase drying
Liquid digestate spreading on land
Other digestate application
Replaced sawdust production
Replaced urea ammonium nitrate production
Composting of digestate surplus

Figure 2-3 Seasonal stage-wise impact results for Case 1

- Non-renewable energy consumption

Natural gas accounts for majority of the non-renewable energy consumption in this system. The pattern of net non-renewable energy consumptions in different seasons is determined by the natural gas demand for heating and greenhouse CO_2 enrichment. Sawdust production, rockwool production, and manure spreading also consume some fossil fuels but are not significant compared to direct use of natural gas. For the base scenario, summer consumption of non-renewable energy is highest due to CO_2 enrichment demand. Winter shows the second highest non-renewable energy consumption for heating. Natural gas requirements in spring and fall are also determined by CO_2 enrichment demands, which are about 400 GJ/month lower than in the summer. The average CO_2 demands for spring and fall are the same (Luczynski and Portree, 2005). By conducting the eco-industrial integration, the overall non-renewable energy consumption is dramatically reduced by about 12,600 GJ/year (~74% reduction over the base scenario) in the moderate scenario. The most significant drop occurs in the summer, when all of the 1,622 GJ/month natural gas is avoided by the use of the 37,000 m^3 (~710GJ)/month biogas generated from the digester. For the other seasons, natural gas is supplemented to meet heating demand. CO_2 for greenhouse CO_2 enrichment can be supplied entirely by the biogas combustion flue gas because of the high CO_2 content (~40%) in the biogas. The amount of the required natural gas supplement is therefore proportional to the changes in heating demand over the year. Non-renewable energy associated with sawdust production, rockwool production, and manure spreading is also saved in the eco-industrial scenarios. AD and digestate spreading consume fossil energy as well but the amount is insignificant by comparison. In the optimistic scenario, non-renewable energy savings are obtained from substitution of sawdust and nitrogen fertilizer at an amount of about 1,060 GJ/month, as an extra credit.

- Climate change

Natural gas combustion is a significant source of GHG emissions. The climate change impact variations between different seasons are therefore largely correlated with natural gas consumption, and follow a similar pattern to non-renewable energy consumption. Fossil CO_2 and CH_4 emissions from natural gas upstream stages also induce climate change impact. Specifically, the emissions are mostly generated from feedstock recovery, fuel production, and gas leakage during storage and distribution (Delucchi and Levelton, 2012). Manure storage plus spreading is the biggest contributor to climate change impact in the base scenario. Referring to Section 2.4.1, biogenic CH_4 and N_2O

emissions are the primary GHG emissions in manure storage and N_2O is the primary GHG emission from manure spreading. These are two major GHGs that have high GWPs. The total climate change impact resulting from manure storage and spreading amounts to 65 tonne CO_2-eq/month. In the eco-industrial system, manure storage and spreading and part of the natural gas consumption are avoided, which leads to a significant abatement of climate change impact. The AD reactor produces a small impact due to CH_4 leakage as well as electricity use. Liquid digestate spreading also has some limited impact mainly as a result of N_2O emissions from the digestate spread on land, which overall is lower than that from raw manure spreading as explained in Section 2.4.1. In the optimistic scenario, reduction in climate change impact from reused surplus digestate is around 8 tonne CO_2-eq/month. The GHG emissions from urea ammonium nitrate are mainly fossil CO_2, CH_4 and N_2O emissions from the ammonia and nitric acid production (Swiss Centre for Life Cycle Inventories et al., 2010) while the GHG emissions from sawdust production are primarily fossil CO_2 and CH_4 associated with diesel consumption (Pa, 2010). Overall, the abatement of climate change impact is around 1,362 tonne CO_2-eq/year (81% reduction over the base scenario) in the moderate scenario of the integrated system.

- Aquatic acidification

There is barely any change in aquatic acidification impact in difference seasons because natural gas has very limited contributions. In the base scenario, manure spreading operation is the primary source of acidification impact. The emissions that lead to acidification are on-site and direct; these include H_2S, NH_3, SO_2, and NOx emissions. The second largest contributor to acidification impact is manure storage; similar to manure spreading, H_2S and NH_3 emissions stem from on-site storage. The reduction for the moderate scenario is about 724 kg SO_2-eq/month (40% lower than the base scenario). Digestate application generates lower emissions of H_2S, NH_3, SO_2, and NOx than manure spreading with an impact reduction corresponding to 427 kg SO_2-eq/month. The biogas combustion stage produces NOx and NH_3 emissions at 30 kg SO_2-eq/month. For the moderate scenario, composting of the surplus digestate also generates a significant acidification impact (268 kg SO_2-eq/month), mainly due to the relatively high NOx and SO_2 emissions. The reduction in acidification impact from substituted nitrogen fertilizer and sawdust is very limited, so that the optimistic scenario is not significantly better than the moderate scenario. In total, the decrease in acidification impact by setting up an eco-industrial system can be about 2,172 kg SO_2-eq/year.

- Aquatic eutrophication

The stages that have a significant aquatic eutrophication impact are the same as the major contributors to acidification. The major pollutants that lead to eutrophication impact in this system are NH_3 and NOx; the characterization factor of NH_3 is nearly double that of NOx (Guinée et al., 2002). In the base scenario, manure storage and spreading account for more than 97% of the total impacts; specifically, manure storage generates 36 kg PO_4-eq/month and manure spreading generates 182 kg PO_4-eq/month. The difference is due to the higher emissions of both NH_3 and NOx from manure spreading. Especially for NH_3 emissions, the emission factor of manure spreading is more than four times as high as manure storage (Stucki et al., 2011). In the eco-industrial system, eutrophication impact from digestate spreading and application is 163 kg PO_4-eq/month in total, which is a little less than from manure spreading. The overall eutrophication impact of digestate application is still significant but less than that caused by raw manure. The composting process in the moderate scenario creates high NH_3 emissions as explained in Section 2.4.1. In the moderate scenario, the total eutrophication impact of the eco-industrial system turns out to be higher than the base scenario with a net increase of 19 kg PO_4-eq/month. Similar to acidification impact, savings from the substitution with the surplus digestate are very limited. A reduction of eutrophication impact can be seen from the optimistic scenario not because of the surplus digestate reuse but due to the avoided composting of surplus digestate.

- Respiratory effects from inorganic emissions

For respiratory effects caused by inorganic emissions, manure storage and spreading appear to be the largest pollution source in the base scenario. Manure storage generates 12 kg $PM_{2.5}$-eq/month and manure spreading generates 65 kg $PM_{2.5}$-eq/month. The impact from manure storage is due to on-site NH_3 emission. The impact from manure spreading also mainly results from on-site emissions of NH_3, plus unspecified PMs, NOx, and SOx. Natural gas upstream stages generate 4 kg $PM_{2.5}$-eq/month mostly from the feedstock extraction, fuel production, and gas sweetening processes (Delucchi and Levelton, 2012). NOx, SOx, and PMs are the main emissions from these stages. It is also worth noting that as the fossil fuel upstream processing does not take place on the farm of this study, the receptors of this impact are not the same as those near the farm area. Further investigation and discussion of such geographically dispersed impacts are recommended. Natural gas combustion also imposes some impact from NOx, SOx, and PM emissions released on-site. In the eco-industrial scenario, the overall impact from digestate application is significant (53 kg $PM_{2.5}$-eq/month) yet not as high as manure spreading for the given functional unit. Biogas combustion has a higher respiratory inorganic impact than natural gas combustion for the same energy provided, due to the much lower heating values of the biogas in this study and the resultant increase in volume of gas burnt. The emissions from burning biogas are therefore higher than those from burning natural gas. The total impact of the moderate eco-industrial scenario turns out to be higher than the base scenario (~8 kg $PM_{2.5}$-eq/month more). Similar to the eutrophication impact, composting contributes significantly to respiratory impact from emitted NH_3, NOx, PMs, and SOx. The reduction in the impact from reused surplus digestate in the optimistic scenario is of a similar order as from natural gas upstream (5 kg $PM_{2.5}$-eq/month). Again, the impacts rising from sawdust manufacture and nitrogen fertilizer production are also from upstream processes; these respiratory impacts do not occur within the geographical boundary of the system.

- Respiratory effects from organic emissions

Organic emissions that have respiratory effects are from a number of VOCs which include the common NMVOCs. Most of these respiratory effects are associated with upstream productions of materials. In the base scenario, upstream processing of natural gas generates the highest respiratory effects from organic pollutants emissions. Emissions of CH_4 and NMVOCs from natural gas upstream were discussed in Section 2.4.1. Natural gas combustion is the second largest source of respiratory effects with the on-site emissions directly from the combustion process. The main species are pentane, butane, and benzene (Swiss Centre for Life Cycle Inventories et al., 2010). Sawdust production and rockwool production contribute about 0.35 kg C_2H_4-eq/month, respectively, due to NMVOC emissions. In the eco-industrial system, the impact resulting from natural gas production and consumption is still noticeable in spring, fall, and winter seasons but are much lower than the base scenario as less natural gas is required. The AD process and digestate treatment and usage do not generate significant respiratory effects and the composting process also has very small impacts. Therefore the difference between the optimistic and moderate scenarios comes from the reuse of surplus digestate. Production of sawdust and urea ammonium nitrate generates fossil CH_4 and NMVOCs (Section 2.4.1). About 1.2 kg C_2H_4-eq/month is reduced in the optimistic scenario by the use of surplus digestate which results in a net impact of -0.34 kg C_2H_4-eq/month. As noted in the above paragraph for respiratory effects from inorganic emissions, these impacts can be geographically dispersed since the material production processes are not limited to one area. The respiratory effects can be further analyzed based on specific receptors.

- Human toxicity

Human toxicity impact is associated with all stages: manure management, natural gas, and materials production. In the base scenario, manure storage and spreading contribute equally to non-carcinogenic toxicity (~80 kg C_2H_3Cl-eq/month) from emissions of H_2S and NH_3. Natural gas combustion generates both non-carcinogenic and carcinogenic toxicity, of which the carcinogenic toxicity is mainly due to the organic emissions (NMVOCs). Rockwool production has an impact of 25 kg C_2H_3Cl-eq/month which includes both non-carcinogenic and carcinogenic toxicity mainly due to the use of coal (Nielsen et al., 2003). In the eco-industrial scenarios, natural gas combustion remains part of the sources of toxicity impact in spring, fall, and winter but liquid digestate spreading is the most significant contributor (22 kg C_2H_3Cl-eq/month). Similar to the manure spreading process, spreading digestate only contributes to non-carcinogens from inorganic species released on-site. Biogas upgrading generates toxicity impacts from both on-site emissions and adsorbent production. On-site emissions from the leakage lead to non-carcinogenic toxicity. Adsorbent production, especially of limestone, results in both non-carcinogenic and carcinogenic toxicity. In the moderate scenario, composting of the surplus digestate creates non-carcinogenic toxicity. In the optimistic scenario, significant reduction is achieved from using digestate to substitute nitrogen fertilizer. Toxicity associated with the production of urea ammonium nitrate is both non-carcinogenic and carcinogenic, arising mostly from ammonia and nitric acid production. With the savings of substituted nitrogen fertilizer, the total toxicity impact of the optimistic scenario is on average 7 kg C_2H_3Cl-eq/month while the year-round reductions in the moderate and optimistic scenarios compared to the base scenario are 2,187 kg C_2H_3Cl-eq/year and 2,867 kg C_2H_3Cl-eq/year, respectively.

2.4.3 Normalization analysis

The results presented above are absolute impacts in each category. To assess their relative contributions to the regional/provincial and national impacts, the characterization results are normalized by the BC and Canada background inventories in this section. The year-round total impact reductions for the moderate eco-industrial scenario are used in the normalization analysis.

- Non-renewable energy consumption

The direct reduction of natural gas consumption in the eco-industrial system is about 9.95 TJ/year, for a dairy farm raising 500 head of cows while the biogas supports a 5,300 m^2 greenhouse. There were 105,000 cows in BC in 2006 (BC Ministry of Agriculture, 2007). If all the dairy manure is used for biogas production, it can supply heat and CO_2 demands to roughly 1.1 million m^2 greenhouses, out of the total of 5.3 million m^2 (Foster et al., 2011) in BC, leading to savings in natural gas consumption of 2,090 TJ/year. Total consumption of natural gas in the agriculture sector in 2010 was 5,716 TJ (Nyboer and Kniewasser, 2012). Thus the potential savings are up to 37% of the total consumption in the BC agriculture sector. For the total non-renewable primary energy consumption, the potential reduction is about 12.7 TJ /year for the system presented in Section 2.4.2; if all the dairy farms in BC are included, the primary energy reduction could be up to 2,667 TJ/year. The total consumption of energy products (including natural gas, petroleum products, electricity and so on) in BC's agriculture sector is reported as 17,964 TJ in 2010 (Nyboer and Kniewasser, 2012). Total life cycle non-renewable energy use is not available, but the total input primary energy to produce this amount of output primary energy must be higher because of the energy loss during extraction and upstream processing. An estimate is made using an energy upstream processing efficiency of 25%, which is close to many energy products used in BC (Delucchi and Levelton, 2012). The estimated total primary energy input to the BC economy is then about 22,455 TJ in 2010. In this case, the reduction of non-renewable energy consumption in BC's agriculture sector would be roughly 12% of the total consumption in the BC economy.

- Climate change

The climate change impact reduction in the eco-industrial system is 1,363 tonne CO_2-eq/year (Section 2.4.2). Scaling this climate change impact reduction to the entire dairy industry in BC, the total potential reduction is up to 286 kt CO_2-eq/year. The overall impact from agriculture sector in BC was reported as around 1 Mt CO_2-eq in 2010 (Nyboer and Kniewasser, 2012). Therefore the potential reduction accounts for 29% of all the impacts from the agricultural sector.

- Acidification and eutrophication

There have been studies concerning the acidification and eutrophication risks due to manure and fertilizer applications in BC but no thorough auditing of these risks has been conducted. To normalize the impact results of this study, Canadian average normalization factors are used. Lautier et al. (2010) reported the factors as 155 kg SO_2-eq/person/year and 15.8 kg PO_4-eq/person/year for acidification and eutrophication, respectively. The decrease of acidification impact by conducting eco-industrial integration can be about 8,688 kg SO_2-eq/year. Scaling up to the entire dairy industry comes to 1.8 million kg SO_2-eq/year. Given the population of BC as about 4.6 million (Statistics

Canada, 2012), the shared total acidification impact in BC is 713 million kg SO_2-eq/year. Thus the reduction from the eco-industrial system is around 0.25% of the total. For eutrophication, the life cycle impact results show a net increase for the moderate scenario of eco-industrial system: 226 kg PO_4-eq/year for the given functional unit. The entire dairy industry generates 0.047 million kg PO_4-eq/year. The total eutrophication impact in BC is 73 million kg PO_4-eq/year so that the increase is about 0.064%. The results show that the changes of the acidification and eutrophication impacts are insignificant; in fact, based on the national normalization factors, the impacts even from the base scenario of dairy industry appear to be insignificant. Using the national average factors can cause a high uncertainty, but the figures are so small that this conclusion is robust.

- Respiratory effects

Respiratory effects from inorganic emissions increase by 94 kg $PM_{2.5}$-eq/year in the eco-industrial system compared to the base scenario. For the whole dairy industry, the increase would be around 19.7 tonne $PM_{2.5}$-eq/year. Again, the national normalization factor, of 153 kg $PM_{2.5}$-eq /person/year, will be used because no local data are available. Multiplied by BC's population, the total for the province comes to 704 million kg $PM_{2.5}$-eq /year. Thus the respiratory effect increase of conducting Eco-Industrial integration is very small.

For the respiratory effect caused by organic emissions, Canadian normalization is not available. The factor developed for Western Europe is therefore used as a tentative alternative, leading to an estimate for BC's population of 15.3 million tonne C_2H_4-eq/year. The reduction from Eco-Industrial integration is 35 kg C_2H_4-eq/year for the specific farm and 7.4 tonne C_2H_4-eq/year for the entire dairy sector. This value is negligibly small compared to the total 15.3 million tonne C_2H_4-eq/year.

- Human toxicity

Human toxicity impact is also analyzed using Canadian normalization factors reported by Lautier et al. (2010) for human carcinogenic and non-carcinogenic toxicity, respectively. An average value of 200 kg C_2H_3Cl-eq/person/year is used in this study, leading to an estimate of up to 920 million kg/year as the total impact from BC. The year-round reduction in the moderate eco-industrial scenario compared to the base scenario for the current case study is 2,186 kg C_2H_3Cl-eq/year; scaling to the whole dairy sector comes to 0.46 million kg C_2H_3Cl-eq/year, corresponding to 0.05% of the total. This is also not significant, primarily because the agriculture sector does not create much toxicity impact.

Overall, the results show that the integrated system can result in substantial percentage reductions in regional and national non-renewable energy consumption and greenhouse gas emissions while the percentage reductions on local eco-system and human health impacts are insignificant. Thus the analysis of such an integrated greenhouse-animal farm system should be focused on non-renewable energy consumption and climate change impact.

2.4.4 Uncertainty analysis

Uncertainty analysis is conducted to test the significance of the difference between the eco-industrial system and the base scenario as mentioned in Section 1.4.2. Where the data sources report uncertainty, this has been included; where no uncertainty data are reported, a ±20% uncertainty was applied. A Monte Carlo analysis with 100 runs is performed to compare the base scenario and the moderate eco-industrial system, referred to as scenario 1-a and scenario 1-c, respectively. The results are shown in Table 2-10.

Table 2-10 Uncertainties for comparison of base scenario and moderate eco-industrial scenario in Case 1

	Probability of impact in scenario 1-a <impact in scenario 1-c	Probability of impact in scenario 1-a ≥impact in scenario1-c
Non-renewable energy	0%	100%
Climate change	0%	100%
Aquatic acidification	0%	100%
Aquatic eutrophication	80%	20%
Respiratory inorganics	80%	20%
Respiratory organics	0%	100%
Human toxicity	0%	100%

The Monte Carlo results show that the difference between systems is significant in all impact categories. For non-renewable energy consumption, climate change, acidification, respiratory organics, and human toxicity, the impact from base scenario is always (100%) higher than in integrated system. For eutrophication and respiratory inorganics, the impact from the integrated system has an 80% probability of being higher than in the base scenario. This confirms the small impact increase in the integrated system as stated in Section 2.4.2. Although uncertainty exists in the data sources, the conclusions from the current life cycle impact analysis seem to hold.

2.5 Conclusions

Overall, the integrated dairy farm-greenhouse eco-industrial system shows significant reductions in several environmental impact categories. If the surplus digestate can be all utilized to replace sawdust and urea ammonium nitrate as in the optimistic integration scenario, the impacts are reduced in all categories compared to the base case, with percentage reductions from about 20% to over 100%. If the surplus digestate has to be composted or disposed, the impacts on non-renewable energy consumption, climate change, aquatic acidification, respiratory effects from organic emissions, and human toxicity are reduced; but the impacts are increased in categories of aquatic eutrophication and respiratory effects from inorganic emissions. The most significant regional reductions in the base scenario were identified by the normalization analysis as non-renewable energy consumption and greenhouse gas emission; in the eco-industrial system, 12% and 29% of the total impacts in the entire BC's agriculture sector are reduced in the moderate scenario, respectively; while the reductions or increases in the other impact categories are all less than 1% of the total inventories in BC's agriculture sector. For the other impact categories, the contributions from animal farming and greenhouses were found to be insignificant in the context of the whole of BC, although it should be noted that the inventory data used for normalization of other impact categories are estimated based on the national average or European average values because of the lack of BC specific data.

Although the results cannot be directly compared with published studies as each LCA study has its own specific system boundaries and data sources with particular assumptions, these trends are generally in agreement with the literature. Reductions of non-renewable energy consumption and climate change are reported by other studies (Börjesson and Berglund, 2006a, 2006b, 2007; de Vries et al., 2010; Venczel and Powers, 2010; Mezzullo et al., 2012). Börjesson and Berglund (2006a, 2006b, 2007) while Lansche and Müller (2012) reported that AD also reduces acidification and eutrophication impacts if the digestate is all utilized for substituting other materials, consistent with the results of the optimistic case in this study. De Vries et al. (2010) observed similar impacts in the base scenario for acidification and eutrophication from dairy manure AD, while Mezzullo et al. (2012) found that the usage of biogas and digestate increased acidification and eutrophication effects. Mezzullo et al. (2012) also reported higher human respiratory effects from the use of biogas and digestate, consistent with the current finding that the inorganic emissions cause respiratory effects.

In this chapter, we showed that significant reductions in non-renewable energy consumption and GHG emissions can be achieved by integrating the animal farm and a greenhouse as linked by an anaerobic digester. In the next chapter, we will examine the potential of co-digesting other organic wastes in the digester to improve the performance of the integrated eco-industrial system.

3. LCA OF AN INTEGRATED DAIRY FARM AND GREENHOUSE WITH CO-DIGESTION

3.1 Introduction

In Case 1 presented in Chapter 1, we evaluated the performance of an integrated dairy farm and greenhouse. In this chapter, we will examine the potential improvement of the integrated dairy farm-greenhouse system if other organic waste is co-digested with animal manure in the anaerobic digester. Co-digestion in farm scale digesters as well as large scale centralized biogas plants has been widely practiced (Braun and Wellinger, 2003), and guidelines for managing and optimizing co-digestion operations have been well established (Hjort-Gregersen et al., 2011). Co-digestion of dairy manure with other organic wastes can first potentially enhance the biogas yield to better match the greenhouse with the dairy farm. The 500-cow dairy farm in Case 1 is more than three times as large as an average dairy farm in BC (150 cows) while the 5,300 m^2 greenhouse is smaller than the average scale of 8,000 m^2 in BC. Secondly, the organic waste audit in BC shows diverse organic waste sources being available (Electrigaz Technologies Inc., 2007). In the agriculture region in the lower mainland, there are a number of swine and poultry farms along with dairy farms (BC Ministry of Agriculture, 2011). Food processing waste and food waste are also generated from slaughtering, and from the residential and commercial sectors. With a digester established, it is natural to secure other nearby organic wastes either as a supplementary supply or a backup feedstock to the digester. Thirdly, some studies show the potential of co-digestion to reduce acidification and eutrophication impacts significantly (de Vries et al., 2010).

The other organic wastes to be introduced include swine manure, poultry manure, greenhouse plant waste, food waste and fats, oils and grease (FOG) waste. All these wastes are typical digestion feedstock and are commonly generated in the Lower Mainland of BC. Dairy manure remains the major component (80%~90% by mass) due to the fact that it is the most abundant organic waste in BC. The same LCA method as used in Chapter 2 is applied to the study of co-digestion.

3.2 Framing of LCA Study

3.2.1 Functional unit

The functional unit of this LCA study is “disposal of 1,100 tonnes of organic waste”. As mentioned in the introduction, five cases are developed to analyze the co-digestion of dairy manure with five types of other organic wastes; those organic wastes are added to dairy manure at 10 to 20% by mass. The ratios of the waste components to dairy manure are determined based on technical feasibility and supply availability. The size of the dairy farm matches the manure requirement in specific cases and the size of the greenhouse is determined by matching the biogas generation with the heat and CO_2 enrichment demand in the summer season, the same as in Case 1.

3.2.2 System boundary

The system boundaries for the five cases in this study are the same as in Case 1 with a few extensions associated with the additional organic wastes. The additional processes involved in the base scenarios are the storage and spreading of swine manure and poultry manure, in the same way as with dairy manure treatment, and composting of greenhouse plant waste, food waste, and FOG waste on-site. In the eco-industrial systems, transport of the other organic waste to the digester on the dairy farm is included. Case 4 is an exception as the greenhouse is assumed to be adjacent to the dairy farm so the plant waste does not need transport. Therefore, the system boundaries can mostly refer to Figure 2-1; all the descriptions and explanations in Section 2.2.2 are applicable to this study so will not be repeated.

3.2.3 Scenarios

Table 3-1 defines the scenarios in this study. Table 3-2 presents the basic data of the systems with Case 1 included for reference. Table 3-3 shows the data for different seasons with a time horizon of one month. In the same way as for Case 1, one base scenario and two (optimistic and moderate) eco-industrial scenarios are established for each case. For the LCA study, four seasons are analyzed specifically for each scenario.

Table 3-1 Scenarios for Cases 2 to 6

Scenario number	Description	Case
Scenario 2-a	Base case	2. 80% dairy manure + 20% swine manure
Scenario 2-b	Eco-industrial system, optimistic	
Scenario 2-c	Eco-industrial system, moderate	
Scenario 3-a	Base case	3. 80% dairy manure + 20% poultry manure
Scenario 3-b	Eco-industrial system, optimistic	
Scenario 3-c	Eco-industrial system, moderate	
Scenario 4-a	Base case	4. 80% dairy manure + 20% plant waste
Scenario 4-b	Eco-industrial system, optimistic	
Scenario 4-c	Eco-industrial system, moderate	

Scenario number	Description	Case
Scenario 5-a	Base case	5. 80% dairy manure + 20% food waste
Scenario 5-b	Eco-industrial system, optimistic	
Scenario 5-c	Eco-industrial system, moderate	
Scenario 6-a	Base case	6. 90% dairy manure + 10% FOG waste
Scenario 6-b	Eco-industrial system, optimistic	
Scenario 6-c	Eco-industrial system, moderate	

Table 3-2 Basic performance data for Cases 1 to 6

Item	Unit	Case 1	Case 2	Case 3	Case 4	Case 5	Case 6
Greenhouse area	m^2	5,330	5,440	7,240	6,740	8,150	8,370
Biogas generation	m^3/month	36,900	37,700	50,150	46,660	56,440	57,970
Dried solid digestate generation	tonne/month	114	114	113	113	112	112
Liquid digestate generation	tonne/month	791	790	781	784	777	775
Sawdust bedding demand in dairy farm	tonne/month	42	34	34	34	34	38
Growing media demand in greenhouse	kg/month	557	569	757	705	852	875
Dried solid digestate surplus	tonne/month	72	80	79	79	78	74
N fertilizer demand in greenhouse	kg N/month	40	41	54	51	61	63
Liquid digestate surplus	tonne N/month	1.12	1.16	1.45	1.43	1.18	1.14

Table 3-3 Seasonal performance data for Cases 2 to 6

Scenarios	Season	Heat demand for dairy farm	Heat demand for greenhouse	CO_2 demand for greenhouse	Natural gas use in base cases	Heat demand for digester	Biogas generation	Heat supply from biogas	CO_2 supply from biogas	Natural gas supplement for heating
		GJ/month	GJ/month	tonne/month	GJ/month	GJ/month	m^3/month	GJ/month	tonne/month	GJ/month
Case 2	Spring	91	779	57	1,009	137	37,700	725	75	331
	Summer	4	316	75	1,346	95	37,700	725	75	0
	Fall	77	599	57	1,009	130	37,700	725	75	133
	Winter	148	947	8	1,095	161	37,700	725	75	572
Case 3	Spring	91	1,036	75	1,342	182	50,150	965	100	413
	Summer	4	420	100	1,791	127	50,150	965	100	0
	Fall	77	797	75	1,342	173	50,150	965	100	153
	Winter	148	1,260	10	1,408	214	50,150	965	100	717
Case 4	Spring	91	964	70	1,249	169	46,660	898	93	390
	Summer	4	391	93	1,667	118	46,660	898	93	79
	Fall	77	741	70	1,249	161	46,660	898	93	148
	Winter	148	1,172	9	1,320	199	46,660	898	93	677
Case 5	Spring	91	1,166	85	1,511	205	56,440	1,086	113	376
	Summer	4	473	113	2,016	143	56,440	1,086	113	0
	Fall	77	897	85	1,511	195	56,440	1,086	113	82
	Winter	148	1,418	11	1,566	241	56,440	1,086	113	721
Case 6	Spring	103	1,197	87	1,552	210	57,970	1,116	116	395
	Summer	4	486	116	2,070	147	57,970	1,116	116	0
	Fall	87	921	87	1,552	200	57,970	1,116	116	92
	Winter	166	1,457	12	1,623	248	57,970	1,116	116	755

An ideal carbon to nitrogen (C:N) ratio for AD has been reported to be between 20:1 and 30:1 (AgStart, 2012). Dairy manure has a low C:N ratio of about 12:1. C:N ratios of the other organic waste, excluding FOG waste, range from 15:1 to 35:1 while the C:N ratio of FOG manure is as high as 90:1 (lleleji et al., 2012). Therefore, a proportion of 10 to 20% of the other wastes is suitable for co-digestion. Biogas yield for the digestion of different organic wastes mixture varies significantly. Up to 57% increase is seen in co-digestion with FOG waste compared to pure dairy manure digestion. Digestate generation is expected to decrease with the increase of biogas production based on the mass balance.

3.3 Description of Inventory Model

Most of the stages remain the same as in Case 1. Only the modified and new processes are described in this section. Detailed statistical data and data sources are also presented in Appendixes A and B.

- Manure management and digestate application

Swine manure has a dry matter content of 8% (Berglund and Börjesson, 2006) while poultry manure has a higher dry matter content (Electrigaz Technologies Inc., 2007) and can quickly become dry if not collected and properly stored. Storage of swine manure and poultry manure is modeled as in tanks in this study, as for dairy manure. Vacuum tanker, which is modeled for dairy manure spreading, is also chosen for swine and poultry manure spreading. Since poultry manure is dry, proper dilution is assumed before it can be spread using vacuum tank tractors. Emission factors are available or converted for specific livestock species. NH_3 emissions are still calculated based on the total ammonia nitrogen content, with an allocation between storage and spreading (Stucki et al., 2011); N_2O emissions are based on total nitrogen content (Dong et al., 2006; Börjesson and Berglund, 2007), and CH_4 emissions are based on biogas potential from AD. No specific emission data for digestate produced from different feedstock are available; thus data from studies with dairy manure digestion are used for all the digestate in each case (Börjesson and Berglund, 2006; Stucki et al., 2011).

- Composting

Greenhouse plant waste, food waste, and FOG waste are composted at the place where they are generated in the base scenarios. Composting of plant waste is modeled as the same process as for surplus solid digestate in the moderate eco-industrial scenarios since they are similar materials. Composting of both FOG waste and food waste are modeled with a dataset for food waste composting (RTI International, 2000). However, it should be noted that food waste and FOG waste cannot be composted alone: the hydrogen content in these wastes is so high that the C:N ratio does not meet the requirement for effective composting. Therefore, food waste always has to be composted with high carbon content materials such as woody waste (Composting 101, 2012). And the environmental data for composting are for mixed (e.g. food waste + wood waste) composting. Data for food waste are calculated as proportional values based on the composition. For example, if food waste accounts for 50% of the total waste it is assumed that 50% of the total emissions are from food waste composting.

- Anaerobic digestion

The AD technology and the product treatments for co-digestion are modeled in the same way as for dairy manure digestion. No extra processes are needed since mixing is already available even for single feedstock digestion. Biogas yield from co-digestion is calculated as the summation of the biogas yield from the two respective organic wastes. Composition of the co-digestion biogas is assumed to be the same: 60% CH_4, 37.9% CO_2, 2% H_2O, and 0.1% H_2S. The biogas generation rates of all the six types of organic waste are summarized in Table 3-4 below.

Table 3-4 Biogas generation rates of different types of organic matters

Feedstock	Biogas rate, m^3/ tonne	Source
Dairy manure	34	Swiss Centre for Life Cycle Inventories, 2010; Berglund and Börjesson, 2006
Swine manure	37	Swiss Centre for Life Cycle Inventories, 2010; Berglund and Börjesson, 2006
Poultry manure	94	Electrigaz Technologies Inc., 2007
Plant waste	78	Electrigaz Technologies Inc., 2007
Food waste	122	Viswanath et al., 1992; Electrigaz Technologies Inc., 2008; Geomatrix Consultants and Engineers Inc., 2008
FOG waste	225	Werner and Strehler, 2012

3.4 Results and Discussions

3.4.1 Life Cycle Inventory data

The same emission species as in Case 1 are selected for discussion. The LCIs are analyzed for each season with the seasonal variations similar to Case 1. Differences in the material and energy demands and supplies are reflected at the scales of the farm and greenhouse; the most significant factor is the gas fuel demand and supply. The monthly average LCI results of Cases 1 to 6 are presented in Table 3-5. Tables 3-6 and 3-7 show the absolute and percentage emission reductions for the eco-industrial systems compared to the corresponding base scenarios. Specifically, stage-wise percentage results for Case 6 are presented in Figure 3-1 (the three diagrams refer to scenarios 3-a to 3-c from top to bottom); together with the stage-wise results presented in Figure 2-2, comparison can be made between digestion of dairy manure alone and digestion of the mixed waste.

Table 3-5 Selected LCI results for three scenarios of Cases 1 to 6
(Scenario a: base scenario; Scenario b: optimistic eco-industrial system; Scenario c: moderate eco-industrial system)

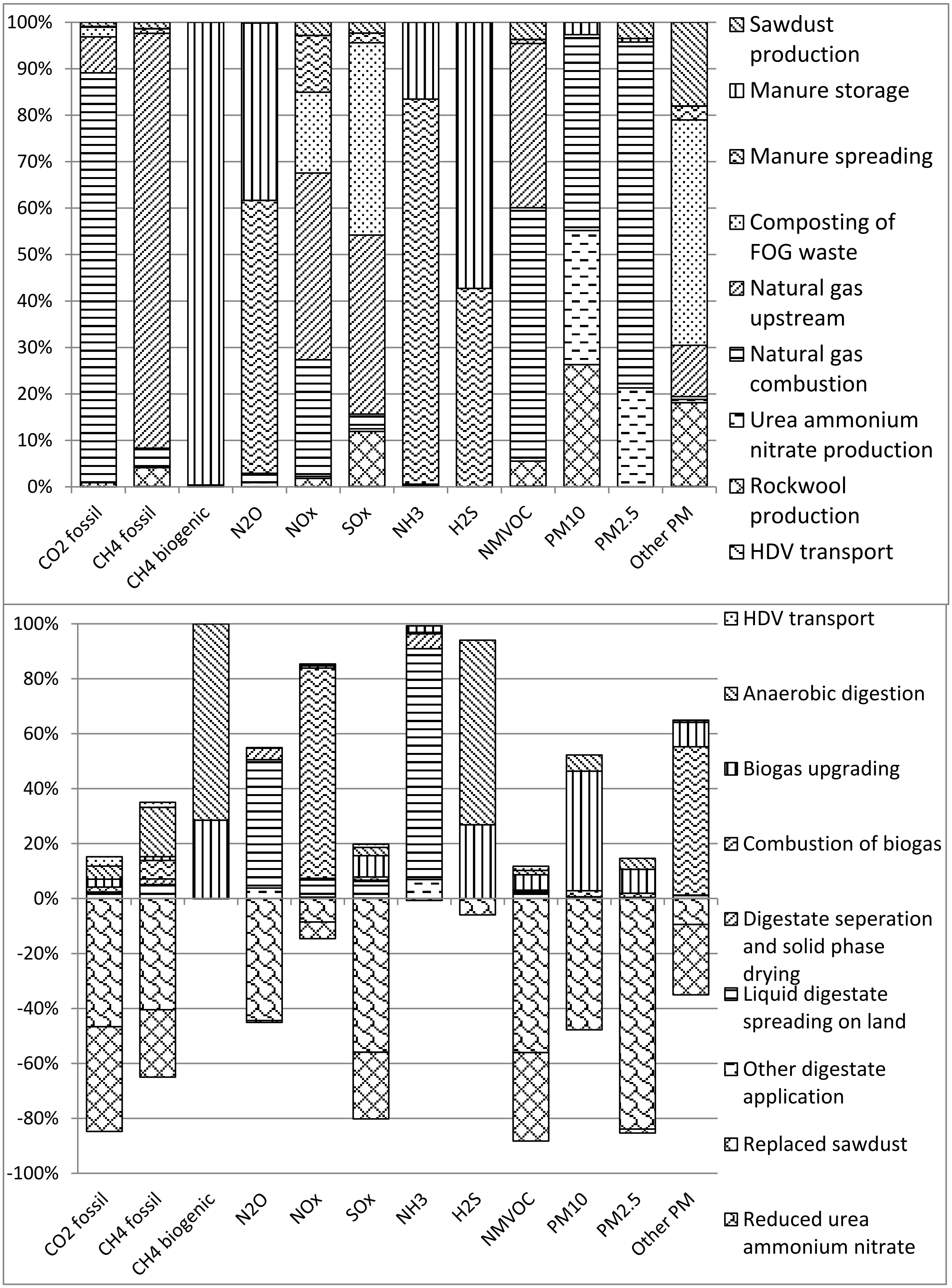
		Scenario 1-a	Scenario 1-b	Scenario 1-c	Scenario 2-a	Scenario 2-b	Scenario 2-c	Scenario 3-a	Scenario 3-b	Scenario 3-c
CO_2 fossil	tonne	72	13	18	73	12	18	96	15	22
CH_4 fossil	kg	88	13	24	88	12	23	115	16	29
CH_4 biogenic	tonne	1.98	0.15	0.16	2.91	0.16	0.16	5.02	0.21	0.21
N_2O	kg	74	3	14	77	2	14	96	0	14
NOx	kg	67	59	89	66	59	92	81	78	113
SOx	kg	15	-3	6	15	-3	6	19	-3	7
NH_3	kg	616	494	659	954	488	671	934	485	666
H_2S	kg	422	0.24	0.26	422	0.24	0.27	422	0.32	0.35
NMVOC	kg	6.77	-0.69	1.61	6.75	-0.83	1.62	8.74	-0.83	1.98
PM_{10}	kg	0.07	-0.16	0.41	0.07	-0.17	0.42	0.09	-0.18	0.56
$PM_{2.5}$	kg	0.20	-1.15	0.18	0.20	-1.19	0.19	0.26	-1.48	0.24
Other PM	kg	3.7	1.0	6.7	3.4	0.8	7.0	3.9	2.3	8.6
		Scenario 4-a	Scenario 4-b	Scenario 4-c	Scenario 5-a	Scenario 5-b	Scenario 5-c	Scenario 6-a	Scenario 6-b	Scenario 6-c
CO_2 fossil	tonne	91	14	21	113	14	20	113	15	21
CH_4 fossil	kg	107	14	26	128	16	27	132	17	28
CH_4 biogenic	tonne	1.59	0.20	0.20	1.60	0.24	0.24	1.79	0.24	0.24
N_2O	kg	60	0	14	60	2	13	67	3	14
NOx	kg	116	72	107	129	89	121	111	93	123
SOx	kg	25	-4	7	55	-2	7	39	-2	7
NH_3	kg	986	485	667	496	480	658	556	489	657
H_2S	kg	338	0.30	0.33	338	0.37	0.40	380	0.38	0.41
NMVOC	kg	8.16	-0.99	1.80	9.72	-0.58	1.87	10.05	-0.43	1.91
PM_{10}	kg	0.08	-0.21	0.52	0.10	0.02	0.62	0.10	0.06	0.64
$PM_{2.5}$	kg	0.24	-1.47	0.23	0.29	-1.14	0.26	0.30	-1.08	0.27
Other PM	kg	8.1	1.7	8.1	12.8	3.4	9.5	8.7	3.8	9.6

Table 3-6 Net emission reductions of the eco-industrial system over the base scenario in Cases 2 to 6

		Scenario 1-b	Scenario 1-c	Scenario 2-b	Scenario 2-c	Scenario 3-b	Scenario 3-c	Scenario 4-b	Scenario 4-c	Scenario 5-b	Scenario 5-c	Scenario 6-b	Scenario 6-c
CO_2 fossil	tonne	60	54	61	55	80	74	77	71	99	93	98	92
CH_4 fossil	kg	75	64	76	65	100	86	94	81	113	101	115	104
CH_4 biogenic	tonne	1.83	1.83	2.76	2.76	4.81	4.81	1.39	1.39	1.36	1.36	1.55	1.55
N_2O	kg	71	60	74	63	96	82	60	46	58	46	65	54
NOx	kg	7	-23	7	-26	3	-32	44	9	40	8	19	-12
SOx	kg	17	9	18	8	22	12	29	19	58	49	41	32
NH_3	kg	122	-43	466	282	449	268	500	319	16	-162	67	-101
H_2S	kg	422	422	422	422	422	422	338	338	338	338	380	380
NMVOC	kg	7.46	5.16	7.58	5.13	9.57	6.76	9.15	6.36	10.30	7.86	10.48	8.14
PM_{10}	kg	0.22	-0.35	0.24	-0.35	0.27	-0.47	0.29	-0.44	0.07	-0.52	0.04	-0.54
$PM_{2.5}$	kg	1.34	0.02	1.39	0.01	1.73	0.02	1.71	0.02	1.43	0.03	1.38	0.03
Other PM	kg	2.7	-3.0	2.6	-3.6	1.7	-4.7	6.4	0.0	9.4	3.3	4.9	-0.8

Table 3-7 Percentage emission reductions of the eco-industrial system over the base scenario in Cases 2 to 6

	Scenario 1-b	Scenario 1-c	Scenario 2-b	Scenario 2-c	Scenario 3-b	Scenario 3-c	Scenario 4-b	Scenario 4-c	Scenario 5-b	Scenario 5-c	Scenario 6-b	Scenario 6-c
CO_2 fossil	82%	75%	84%	75%	84%	77%	85%	77%	87%	82%	86%	81%
CH_4 fossil	85%	73%	87%	74%	86%	75%	87%	75%	88%	79%	87%	79%
CH_4 biogenic	92%	92%	95%	95%	96%	96%	88%	88%	85%	85%	86%	86%
N_2O	96%	81%	97%	82%	100%	86%	100%	77%	97%	78%	96%	80%
NOx	11%	-34%	11%	-39%	4%	-39%	38%	8%	31%	6%	17%	-11%
SOx	118%	60%	120%	57%	118%	63%	114%	74%	104%	88%	105%	83%
NH_3	20%	-7%	49%	30%	48%	29%	51%	32%	3%	-33%	12%	-18%
H_2S	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%	100%
NMVOC	110%	76%	112%	76%	109%	77%	112%	78%	106%	81%	104%	81%
PM_{10}	334%	-517%	348%	-519%	302%	-525%	352%	-529%	75%	-530%	40%	-528%
$PM_{2.5}$	675%	8%	699%	7%	669%	7%	707%	7%	494%	9%	463%	9%
Other PM	72%	-80%	77%	-106%	42%	-121%	78%	0%	74%	26%	56%	-9%


100%
90%
80%
70%
60%
50%
40%
30%
20%
10%
0%
CO2 fossil
CH4 fossil
CH4 biogenic
N2O
NOx
SOx
NH3
H2S
NMVOC
PM10
PM2.5
Other PM
Sawdust production
Manure storage
Manure spreading
Composting of FOG waste
Natural gas upstream
Natural gas combustion
Urea ammonium nitrate production
Rockwool production
HDV transport
100%
80%
60%
40%
20%
0%
-20%
-40%
-60%
-80%
-100%
CO2 fossil
CH4 fossil
CH4 biogenic
N2O
NOx
SOx
NH3
H2S
NMVOC
PM10
PM2.5
Other PM
HDV transport
Anaerobic digestion
Biogas upgrading
Combustion of biogas
Digestate seperation and solid phase drying
Liquid digestate spreading on land
Other digestate application
Replaced sawdust
Reduced urea ammonium nitrate

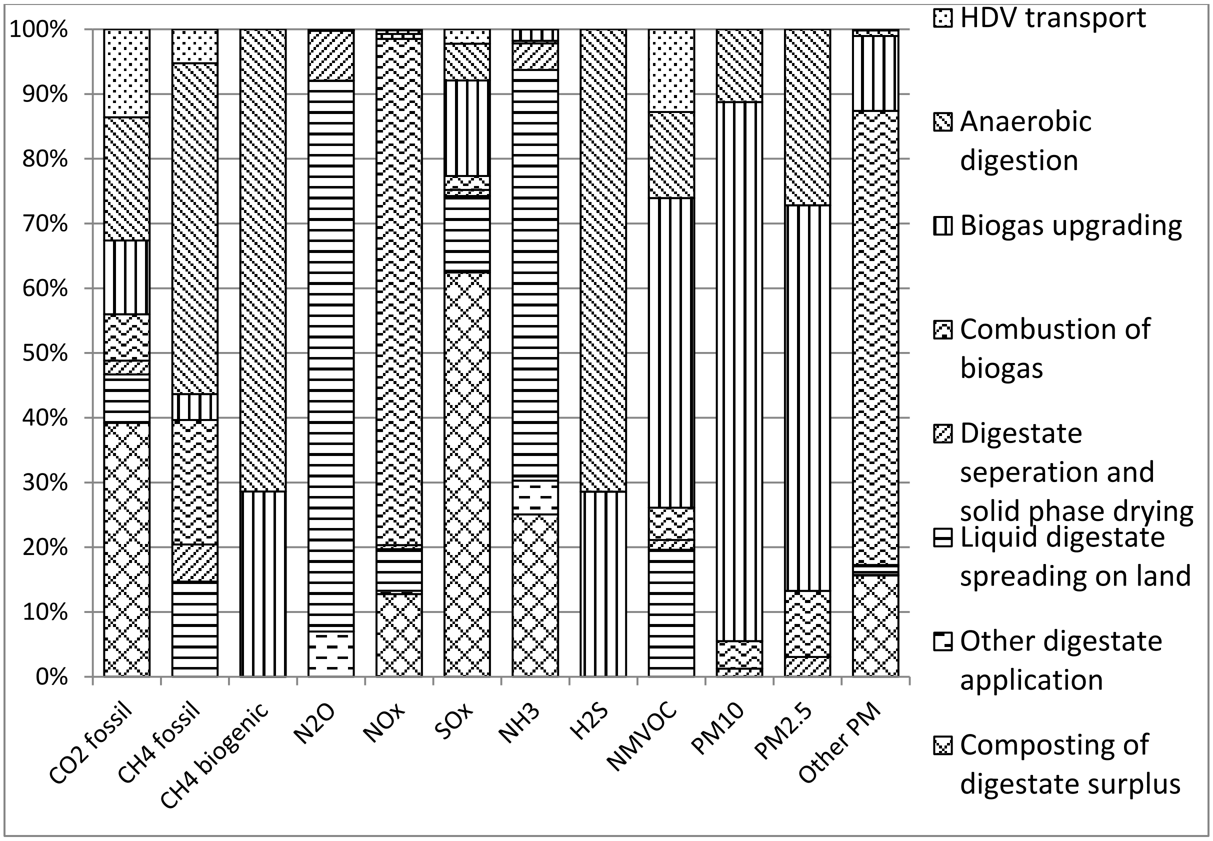


Figure 3-1 Stage-wise selected LCI results for Case 6

Overall, all the eco-industrial systems reduce emissions of fossil CO_2, fossil and biogenic CH_4, N_2O, SOx, H_2S, $PM_{2.5}$, and NMVOC. Moderate eco-industrial scenarios of manure digestions and FOG waste co-digestion show a net increase of NOx emissions while the systems with plant waste and food waste co-digested generate lower NOx emissions compared to the base scenario. For NH_3 emissions, the moderate eco-industrial scenarios of dairy manure digestion, food waste co-digestion, and FOG waste co-digestion result in an increase over the base scenarios. All the moderate eco-industrial scenarios have higher PM_{10} emissions than the base scenarios. For the other PMs, only the systems with food waste and FOG waste co-digestion show emission reductions. The results and comparisons will be discussed for each emission species in the following paragraphs since the variation does not follow straightforward patterns between the cases. Because the base scenario levels of different systems can vary widely, percentage reductions should not be compared directly across cases.

- Fossil CO_2 and fossil CH_4 emissions

Fossil CO_2 and fossil CH_4 emissions arise mostly from natural gas combustion and natural gas upstream processing. The more natural gas is replaced in the eco-industrial system, the more emission reductions can be achieved; therefore, a gradual increase of the emission reductions can be observed from Case 1 to Case 6.

- Biogenic CH_4 emissions

Biogenic CH_4 emissions almost all arise from manure storage in the base scenarios and from AD and biogas upgrading in the eco-industrial systems. Emission reductions can be obtained by avoiding manure storage. By contrast, higher biogas production will lead to increased CH_4 emissions from digestion and biogas upgrading operations. Methane emissions reduce from Case 1 to Case 3, because poultry and swine manure generate more CH_4 than dairy. For Cases 4 to 6, less manure is digested in the base case which shrinks the potential for emission reductions while, in the integrated systems, more biogas is produced so that more CH_4 is emitted; therefore, the reductions of biogenic CH_4 emissions in Cases 4 to 6 are even lower than in Case 1.

- N_2O emissions

N_2O emissions show a similar pattern as biogenic CH_4 emissions. The main contributor in the base scenarios is also manure management; the N_2O emission increases from Case 1 to Case 3 but decreases in Cases 4 to 6 due to less manure involved. In the eco-industrial systems, liquid digestate spreading is the main source of emissions while liquid digestate generation does not vary much among the six cases. Overall, the emission rate is mainly determined by the reduction potentials from manure storage and spreading.

- NOx emissions

NOx emissions in the base scenarios arise mostly from natural gas upstream processing, natural gas combustion, manure spreading, and composting if any. The base scenarios of Cases 4 to 6 have significantly higher NOx emissions compared to the first three cases as a combined result of higher natural gas consumption and composting of the non-manure organic wastes; amongst them, Case 6 is the lowest as the FOG waste accounts for only 10% of the total digestion feedstock. Less manure in these cases lowers the emissions to a limited extent but the overall results still show increased NOx. In the eco-industrial systems, NOx emissions arise mainly from biogas combustion, digestate composting, and liquid digestate spreading. Generation of liquid digestate and solid digestate surplus do not vary much across the systems; thus the overall emissions from the eco-industrial systems are determined by biogas generation rate. As in Case 1, the moderate scenarios of Cases 2, 3, and 6 generate higher NOx emissions than the base scenarios due to biogas combustion. However, the eco-industrial systems in Cases 4 and 5 show a small reduction of NOx emissions because the emissions in the base scenarios are so high, leaving room for reduction although combustion of a large amount of biogas leads to significant emissions of NOx.

- SOx emissions

In Cases 1 to 3, most of the SOx emissions in the base scenarios arise from natural gas upstream processing. As shown in Figure 3-1, food composting also contributes significant SOx emissions, four times the emissions from plant waste composting. Therefore, the gradual increase of SOx emissions in the base scenarios of Cases 1 to 5 primarily results from the increasing natural gas demand. Cases 5 and 6 have higher SOx emissions because of food waste composting (FOG waste composting is also modeled as food composting); specifically, much higher emissions are created in Case 5 than Case 6 because the food waste in Case 5 is twice the amount of the FOG waste in Case 6. In the optimistic scenarios of the eco-industrial systems, net SOx emissions are insignificant while savings from the replaced urea ammonium nitrate and sawdust are high, resulting in negative total emissions in all cases. In the moderate scenarios, composting of the surplus digestate emits more SOx than the other stages, which leads to positive life cycle results, unlike the optimistic scenarios. Variations between the cases are small since the amounts of surplus digestate do not differ much; therefore the total emissions in the moderate scenarios are all similar, and the reductions in the eco-industrial systems are all determined by the base scenarios.

- NH_3 emissions

NH_3 emissions in the base scenarios almost all arise from manure storage and spreading (Figure 3-1). Increased NH_3 emissions can be observed in Cases 2 and 3 because the emission factors for poultry and swine manure are higher than that for dairy manure. Food waste composting does not emit significant amount of NH_3; thus Cases 5 and 6 have lower NH_3 emissions than Cases 1 to 3 because less manure waste is involved. However, plant waste composting leads to very high NH_3 emissions. In Case 4, the NH_3 emissions from plant waste composting are as significant as from manure spreading, which leads to the highest emissions over the life cycle in all cases. In the eco-industrial systems, the main source of NH_3 emissions is digestate treatment and application. For the moderate scenarios, another contributor is composting of the surplus digestate but since the digestate generation and application levels do not vary much across the cases, the total emissions in the moderate and optimistic scenarios are similar among the six cases. Therefore, the emission reductions in the eco-industrial systems are determined by the base scenarios. Significant reductions are realized in the cases where emissions from base scenarios are high, i.e. Cases 2 to 4.

- H_2S emissions

H_2S emissions follow relatively simple patterns. Manure management is the biggest contributor in the base scenarios. Since the available data do not indicate variation in H_2S emissions between the different types of manure, the estimated emissions are only determined by the quantities of manure handled. In the eco-industrial systems, the emissions mainly arise from leakage from AD and biogas upgrading but are insignificant compared to the base scenarios. In the optimistic scenarios, emission reductions from the replaced nitrogen fertilizer are also insignificant. Therefore, there is little difference between the two types of scenarios for the eco-industrial systems. Overall, the emissions from the eco-industrial systems are all three orders of magnitude lower than those from the base scenarios; thus the reductions are mainly determined by the emission levels in the base scenarios.

- NMVOC emissions

NMVOCs emissions arise mainly from biogas upstream processing and combustion in the base scenarios. In the eco-industrial systems, NMVOCs are mainly generated from AD, biogas upgrading and combustion, and liquid digestate spreading. The emission levels are determined by the amount of biogas produced in Cases 1 to 4 while the emissions in Cases 5 and 6 are lower than in Case 3 due to less usage of liquid digestate. Significant emission savings can be obtained from substitution of sawdust and nitrogen fertilizer in the optimistic scenarios, which lead to negative results over the life cycle.

- PM emissions

Natural gas combustion, urea ammonium nitrate production, and rockwool production are the main contributors to PM_{10} and $PM_{2.5}$ emissions in the base scenarios. A system that requires more natural gas corresponds to a larger greenhouse which accordingly requires more fertilizers and rockwool. Therefore, the pattern of PM_{10} and $PM_{2.5}$ emissions in the base scenarios is consistent with the natural gas demands. In the moderate scenarios of the eco-industrial systems, AD and biogas upgrading are the major sources of PM_{10} and $PM_{2.5}$ so that the gross emissions are proportional to biogas production. In the optimistic scenarios, emission savings can be achieved from replacement of nitrogen fertilizers; net $PM_{2.5}$ emissions are negative in all cases and the net PM_{10} emissions are negative in Cases 1 to 4. The net emissions in Cases 5 and 6 are still positive because of their high biogas production rates.

Other PM emissions arise mostly from sawdust production, rockwool production, and natural gas upstream processing in the base scenarios of Cases 1 to 3. In Cases 4 to 6, composting generates the most significant emissions. Therefore, the emissions in Cases 1 to 3 are at a similar level while a significant increase can be observed in Cases 4 to 6. Specifically, emissions from Case 5 are much higher than from Case 4 since food waste composting generates about twice the emissions from plant waste composting. Case 6 has lower emissions because the waste amount (FOG waste) is only half the food waste in Case 5. In the moderate scenarios of the eco-industrial systems, biogas upgrading and combustion and composting of the surplus digestate are the major sources of other PM emissions. The surplus digestate does not vary much across different cases; thus the total emissions are mainly determined by the biogas production. In the optimistic scenarios, the substituted sawdust and nitrogen fertilizers generate emission savings but the net life cycle emissions still follow the biogas production rate. Case 2 is an outlier because its biogas production increase is small compared to Case 1 but with the highest solid digestate surplus; therefore, the total emissions from Case 2 are lower than Case 1, and are also the lowest among all the cases.

3.4.2 Life Cycle Impact Assessment

Following the same order as the LCI results presented above, year-round monthly average impacts in selected impact categories are presented and compared in this section. Stage-wise results are not presented here but they follow similar patterns as presented in Figure 2-3. Tables 3-8 to 3-10 present the LCIA results, showing the percentage reductions of the eco-industrial systems over the base scenarios. For reference and comparison, Case 1 is also included. Again, the absolute characterization results are averaged over a one-month time horizon. The impact results are closely associated with the selected emission species covered in the LCI results.

Table 3-8 Life Cycle Impact Assessment results for three scenarios of Cases 1 to 6
(Scenario a: base scenario; Scenario b: optimistic eco-industrial system; Scenario c: moderate eco-industrial system)

		Scenario 1-a	Scenario 1-b	Scenario 1-c	Scenario 2-a	Scenario 2-b	Scenario 2-c	Scenario 3-a	Scenario 3-b	Scenario 3-c
Non-renewable energy	GJ	1,410	255	351	1,423	238	340	1,867	302	418
Climate change	tonne	140	17	27	162	16	26	238	20	31
Aquatic acidification	kg SO_2-eq	1,830	818	1,106	2,371	807	1,128	2,352	812	1,131
Aquatic eutrophication	kg PO_4-eq	224	180	243	343	178	248	338	179	249
Respiratory inorganics	kg $PM_{2.5}$-eq	85	66	93	126	66	95	126	68	97
Respiratory organics	kg C_2H_4-eq	4	0	1	4	0	1	5	0	1
Human toxicity	kg C_2H_3Cl-eq	232	-7	50	250	-9	50	273	-18	54
		Scenario 4-a	Scenario 4-b	Scenario 4-c	Scenario 5-a	Scenario 5-b	Scenario 5-c	Scenario 6-a	Scenario 6-b	Scenario 6-c
Non-renewable energy	GJ	1,740	275	390	2,089	285	387	2,153	306	403
Climate change	tonne	146	18	30	169	21	30	176	22	31
Aquatic acidification	kg SO_2-eq	2,301	809	1,129	1,560	810	1,122	1,707	827	1,122
Aquatic eutrophication	kg PO_4-eq	363	179	248	191	179	247	210	183	247
Respiratory inorganics	kg $PM_{2.5}$-eq	138	67	97	83	69	97	86	71	98
Respiratory organics	kg C_2H_4-eq	5	0	1	5	0	1	6	0	1
Human toxicity	kg C_2H_3Cl-eq	243	-18	53	237	-7	53	256	-3	54

Table 3-9 Net impact reductions of the eco-industrial system over the base scenario in Cases 1 to 6

		Scenario 1-b	Scenario 1-c	Scenario 2-b	Scenario 2-c	Scenario 3-b	Scenario 3-c	Scenario 4-b	Scenario 4-c	Scenario 5-b	Scenario 5-c	Scenario 6-b	Scenario 6-c
Non-renewable energy	GJ	1,155	1,060	1,185	1,083	1,566	1,449	1,465	1,350	1,804	1,703	1,847	1,749
Climate change	tonne	123	114	146	136	217	206	128	117	149	139	154	145
Aquatic acidification	kg SO_2-eq	1,013	724	1,564	1,243	1,540	1,221	1,492	1,172	750	438	880	585

		Scenario 1-b	Scenario 1-c	Scenario 2-b	Scenario 2-c	Scenario 3-b	Scenario 3-c	Scenario 4-b	Scenario 4-c	Scenario 5-b	Scenario 5-c	Scenario 6-b	Scenario 6-c
Aquatic eutrophication	kg PO_4-eq	44	-19	164	95	158	89	184	115	12	-56	27	-37
Respiratory inorganics	kg $PM_{2.5}$-eq	19	-8	61	31	58	29	71	41	14	-14	16	-11
Respiratory organics	kg C_2H_4-eq	4	3	4	3	5	4	5	4	6	4	6	5
Human toxicity	kg C_2H_3Cl-eq	239	182	260	200	291	219	261	190	243	184	259	202

Table 3-10 Percentage impact reductions of the eco-industrial systems over the base scenario in Cases 1 to 6

	Scenario 1-b	Scenario 1-c	Scenario 2-b	Scenario 2-c	Scenario 3-b	Scenario 3-c	Scenario 4-b	Scenario 4-c	Scenario 5-b	Scenario 5-c	Scenario 6-b	Scenario 6-c
Non-renewable energy	82%	75%	83%	76%	84%	78%	84%	78%	86%	81%	86%	81%
Climate change	88%	81%	90%	84%	91%	87%	87%	80%	88%	82%	87%	82%
Aquatic acidification	55%	40%	66%	52%	65%	52%	65%	51%	48%	28%	52%	34%
Aquatic eutrophication	20%	-8%	48%	28%	47%	26%	51%	32%	6%	-29%	13%	-18%
Respiratory inorganics	22%	-9%	48%	25%	46%	23%	52%	30%	17%	-17%	18%	-13%
Respiratory organics	109%	77%	111%	76%	108%	77%	110%	78%	105%	81%	103%	81%
Human toxicity	103%	79%	104%	80%	106%	80%	107%	78%	103%	77%	101%	79%

- Non-renewable energy consumption

As mentioned in Chapter 2, natural gas use represents almost all the total non-renewable energy consumption in these systems. The eco-industrial systems that have higher biogas productions are thus expected to save more non-renewable energy, giving rise to a gradual increase in non-renewable energy reduction from Case 1 to Case 6 with Case 3 slightly higher than Case 4.

- Climate change

Climate change impact results from GHG emissions which are fossil CO_2, CH_4, and N_2O as analyzed in the LCI results. These emissions are mostly associated with manure managements and natural gas production and combustion. Figure 3-2 illustrates the relations between the GHG emission reductions and the climate change impact reductions of the eco-industrial systems. It can be observed that the climate change results are dominated by N_2O emissions. This is reasonable since N_2O is a strong GHG with a GWP of 298 kg CO_2-eq/kg N_2O. Therefore, the highest climate change impact reduction is achieved in Case 3, being consistent with the N_2O emission pattern (Section 3.4.1).

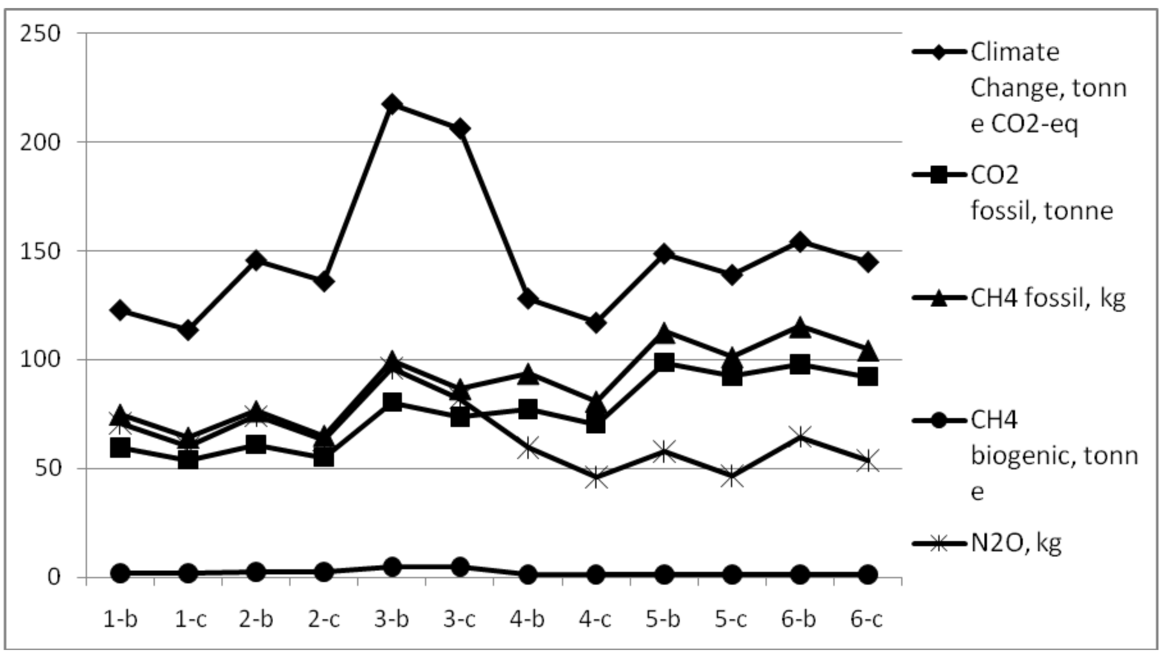


Figure 3-2 Reductions of climate change impacts and GHG emissions in the eco-industrial systems in Cases 1 to 6

- Aquatic acidification and eutrophication

Aquatic acidification mainly results from H_2S, NH_3, SO_2, and NOx emissions in this study. NH_3 and NOx emissions also contribute to aquatic eutrophication. The major life cycle stages that contribute to these impacts are manure storage and spreading in the base scenarios and liquid digestate spreading and surplus digestate composting in the eco-industrial systems. For the plant waste or food waste composting in the base scenarios in Cases 4 to 6, NH_3, SOx, and NOx emissions also occur (Section 2.4.2). The acidification and eutrophication potentials of these characterized species are in the same order of magnitude; thus the impact pattern follows the sum of all these emissions in particular cases. Figures 3-3 and 3-4 illustrate the relations between the gas emissions and the acidification and eutrophication results based on the net reductions in the eco-industrial systems. Cases 2 to 4 have the highest reductions in acidification and eutrophication impacts because the corresponding base scenarios have significant NH_3 emissions compared to the other cases (Section 3.4.1).

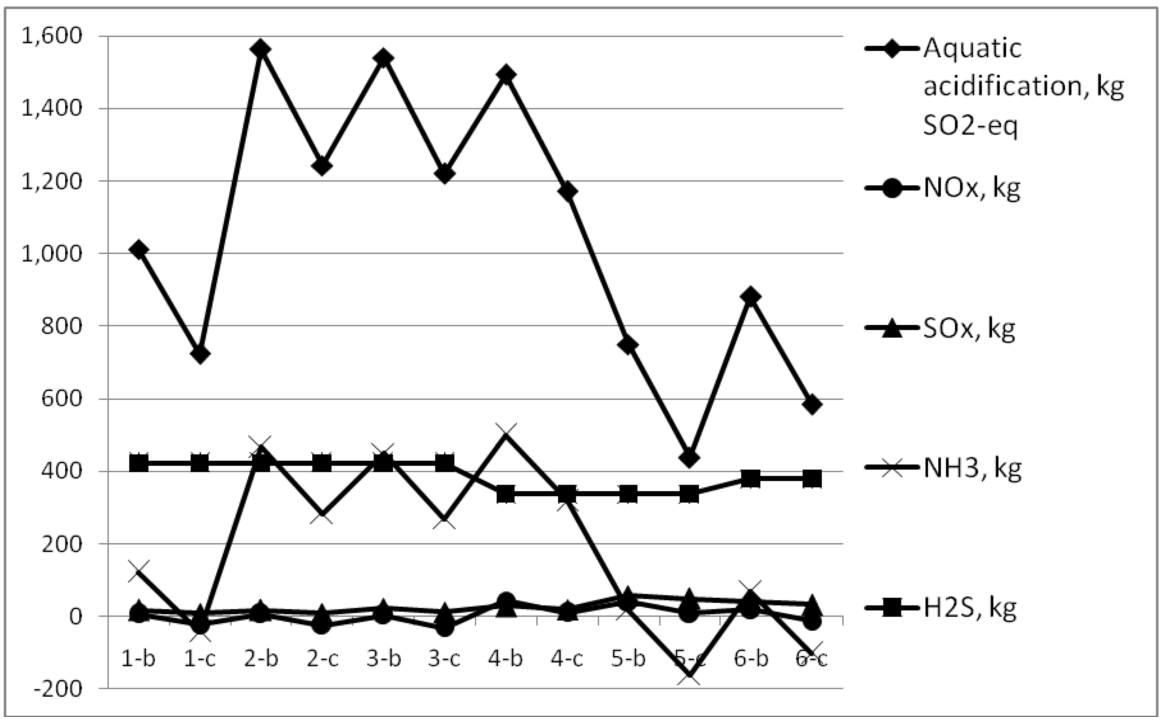


Figure 3-3 Reductions of acidification impacts and the related emissions in the eco-industrial systems in Cases 1 to 6

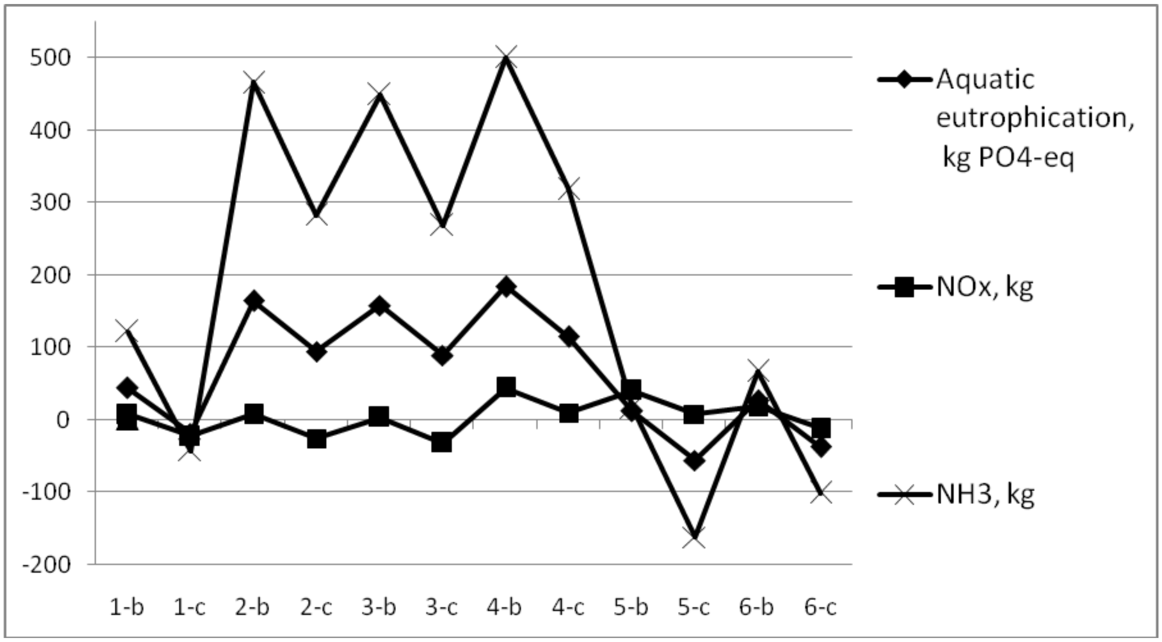


Figure 3-4 Reductions of eutrophication impacts and the related emissions in the eco-industrial systems in Cases 1 to 6

- Respiratory effects from inorganic emissions

In this study, respiratory effects due to inorganic emissions arise mainly from PMs, NOx, NH_3, and SOx. The impact potentials of these species are in the same order of magnitude, with NH_3 being the dominant species because of its highest emission over the life cycle. Referring to the LCI results in Section 3.4.1, the main life cycle stages that contribute to the respiratory effects are the same as those that contribute to acidification and eutrophication: manure management and composting of other organic wastes cause respiratory effects in the base scenarios and liquid digestate spreading is the major contributor in the eco-industrial systems. Figure 3-5 illustrates the relations between the inorganic emission and the respiratory effects. It can be observed that the impacts are correlated to NH_3 emissions; thus again, highest reductions are achieved by the eco-industrial systems in Cases 2 to 4.

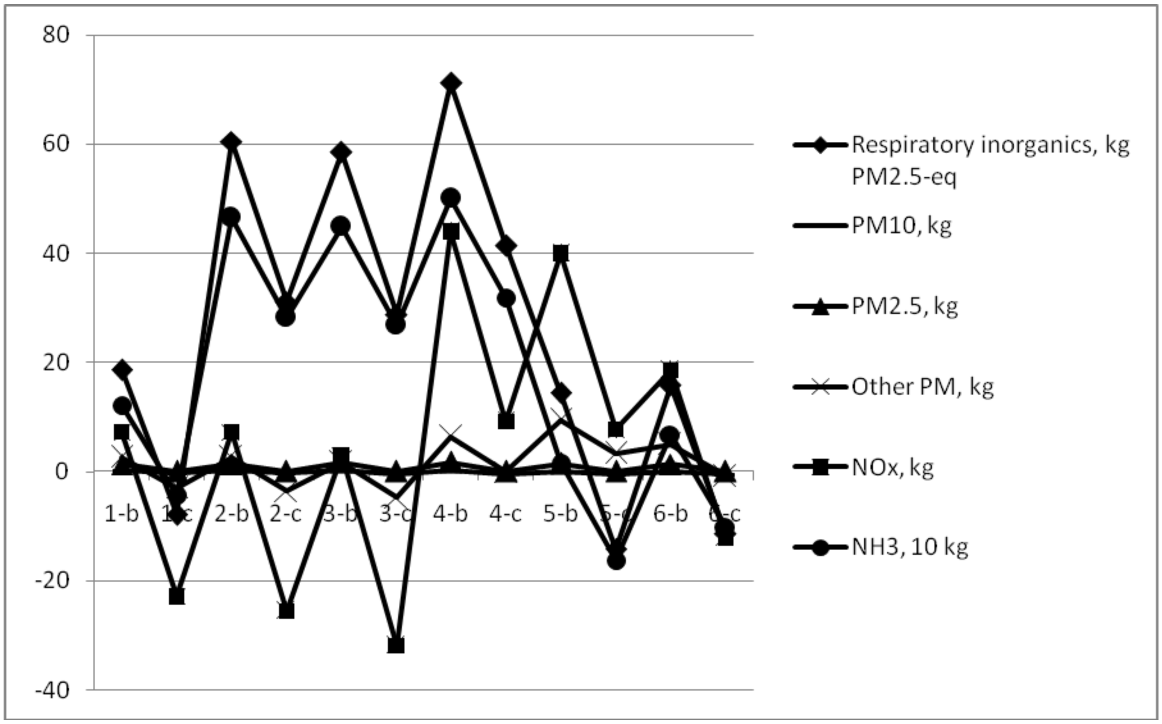


Figure 3-5 Reductions of inorganic-caused respiratory effects and the related emissions in the eco-industrial systems in Cases 1 to 6

- Respiratory effects from organic emissions

Respiratory effects due to NMVOC emissions mainly arise from natural gas upstream processing and combustion, so that results for this impact reflect those of non-renewable energy consumption: the eco-industrial systems leads to greatest impact reductions in the cases that have higher natural gas consumptions in the base scenarios.

- Human toxicity

Human toxicity impact in the base scenarios mainly comes from manure management, natural gas combustion, and rockwool production (Section 2.4.2). Composting is not associated with significant human toxicity impact. The emission intensities from manure management increase from Case 1 to Case 3, while in Cases 5 and 6 the available manure quantities are lower and more natural gas is consumed. Overall, Case 3 has the highest impact due to manure management. In the eco-industrial systems, human toxicity mostly arises from digestate application and surplus digestate composting along with natural gas combustion; digestate generation is similar in different cases so the differences come from the natural gas supplements. The highest impact reduction is achieved in Case 3 for the moderate eco-industrial systems as a result of the high impact in the base scenario, although it also has the highest impact in the eco-industrial system among the six cases. In the optimistic scenarios, impact reductions from the replaced urea ammonium nitrate are significant, which leads to a net negative life cycle emissions in all the cases. Case 3 still shows the highest impact reduction as it has the most liquid digestate surplus.

3.4.3 Normalization analysis

Similar to Case 1, the moderate eco-industrial scenario is used in the normalization analysis. As described in Section 3.4.2, all eco-industrial systems with co-digestion create greater reductions in all impact categories than Case 1 with the most significant reductions in non-renewable energy consumption and GHG emissions. In the co-digestion cases, 400 cows (Cases 2 to 5) or 450 cows (Case 6) can support a greenhouse up to more than 8,000 m^2 (in Cases 5 and 6). The total number of dairy cows in BC is 105,000 (BC Ministry of Agriculture, 2007); thus the total area of greenhouses that can be potentially supported by eco-industrial integration with the available co-digestion feedstock is over 2.1 million m^2. The total area of greenhouses in BC is about 5.3 million m^2; therefore the overall integration potential depends on the availability of dairy manure and the other organic wastes. In BC, swine and poultry manure each account for 17% of the total manure waste (dairy manure accounts for 65%), so that it is appropriate to use a 4:1 ratio for co-digestion of other manure with dairy manure (Electrigaz Technologies Inc., 2007). Unfortunately, information on the total amounts of other modeled wastes is not available. Thus, a scaling factor of 210, which is the same as used in Case 1, is used to examine the total potentials of the co-digestion cases, instead of 262 (105,000/400=262).

- Non-renewable energy consumption

The year-round reduction of non-renewable energy consumption in the eco-industrial system in Case 1 is about 12.7 TJ/year while the reductions in Cases 2 to 6 range from 13 to 21 TJ/year. Scaling up to the entire agriculture sector gives rise to a total non-renewable energy reduction potential of 2,730 to 4,410 TJ/year. The highest case (co-digestion with FOG waste) can potentially achieve nearly twice the reduction as in Case 1 with a smaller dairy farm and 10% FOG waste. With the estimated primary energy input to BC economy being about 22,455 TJ in 2010 (Section 2.4.3), the potential reductions from eco-industrial integration for the whole agricultural sector in BC are in the range 12% to 20%.

- Climate change

The climate change impact reductions in the different cases range from 1,362 to 2,472 tonnes CO_2-eq/year. Scaling up to the whole dairy industry leads to an estimate of 286 to 519 kt CO_2-eq/year. The overall GHG emissions from the agriculture sector in BC were reported as around 1 Mt CO_2-eq in 2010 (Nyboer and Kniewasser, 2012); thus the potential reduction from the eco-industrial systems can be 28.6% to 51.9%.

- Acidification and eutrophication

As analyzed in Section 2.4.3, the acidification and eutrophication impacts from the dairy industry in BC are insignificant. Therefore, normalizations of potential acidification and eutrophication impact reductions for the whole of BC are not analyzed. Only the relative variations between the cases are discussed here which can be considered as internal normalization. This method is also used for the normalization analysis of respiratory effects and human toxicity as follows.

The LCIA results for acidification show that co-digestion with food and FOG waste could create reductions around 30% less than Case 1 while the other co-digestion cases generate reductions more than 65% greater than Case 1. The reduction in scenario 2 is the highest at 15 tonne SO_2-eq/year and 3 kt SO_2-eq/year after scaling up. The eutrophication LCIA results show significant differences between the different cases. Co-digestion with food waste and with FOG waste in the eco-industrial system generate higher impacts with the net increases two to three times as high as the increase in Case 1. On the contrary, co-digestion with swine manure, poultry manure, and plant waste create significant impact reductions; three to four times greater than in Case 1.

- Respiratory effects

The respiratory effects from inorganic emissions follow a pattern similar to eutrophication. The eco-industrial systems of co-digestions with food and FOG waste generate an impact increase greater than in Case 1 while the other co-digestion systems show a reduction even in the moderate scenarios. The highest reduction is achieved for co-digestion with plant waste: more than four times greater than in Case 1. Reductions respiratory effects from organic emissions increase from Case 1 to Case 6. The highest reduction, which is for co-digestion with FOG waste, is about 67% greater than the reduction in Case 1.

- Human toxicity

All the eco-industrial systems reduce human toxicity impacts, while the reductions in Cases 2 to 6 are greater than in Case 1. The highest reduction occurs in co-digestion with poultry manure, which is 20% greater than that in Case 1. The reductions in co-digestion with swine manure and FOG waste are around 10% greater than in Case 1 and the reductions in co-digestion with plant waste and food waste are only slightly greater (1%~4%) than in Case 1.

3.5 Conclusions

An integrated eco-industrial system with co-digestions of dairy manure with other organic wastes would be more advantageous compared to digestion of dairy manure alone in terms of non-renewable energy reduction and environment protection, because of the increased biogas yields. In the impact categories analyzed, the eco-industrial systems with co-digestion can achieve greater impact reductions in natural gas usage and other resources for a larger greenhouse as well as from avoiding further processing of manure and other organic wastes. If the surplus digestate can be used to substitute resources outside the analyzed system boundary, even more significant impact reductions can be achieved. Co-digestion with FOG waste gives the greatest reductions in non-renewable energy consumption and the respiratory effects from organic emissions. Co-digestion with poultry manure creates most reductions in greenhouse emissions and human toxicity. For acidification, eutrophication, and respiratory effects from inorganic emissions, the most significant reductions come from co-digestion with swine manure, poultry manure, and food waste. Since the most significant potential contributions to the reduction in regional impacts are in non-renewable energy consumption and climate change, co-digestion of dairy manure with FOG waste and poultry manure are recommended to improve the integrated system. Co-digestion of dairy manure with plant waste is also favoured as the plant waste is readily available from the greenhouse. In general, the design of the co-digestion system should depend on the waste available in close proximity and the reduction of target environment impacts. As large areas of greenhouses are operating in BC, it would be feasible and economical to introduce co-digestion of multiple wastes for the proposed agriculture eco-industrial systems.

In this chapter, improvement is observed in experimental performance of integrated systems with co-digestion compared to the eco-industrial system in Case 1. In the following chapter, possible further improvement of the integrated eco-industrial system is examined by incorporating a mushroom farm.

4. LCA OF AN INTEGRATED ANIMAL FARM, GREENHOUSE, AND MUSHROOM FARM

4.1 Introduction

In Case 1, analysis of eco-industrial integration of dairy farms and greenhouses revealed significant reductions in a group of environmental impact categories. However, the performance can be further improved if two remaining problems can be addressed. One problem is the high natural gas demand associated with CO_2 enrichment for plant growth stimulation in the greenhouse. In the base scenario representing the conventional operation, natural gas is combusted to provide heating but also CO_2. The highest natural gas demand occurs in the summer when a large amount of CO_2 is required to meet the fast growth rate of greenhouse crops. Since heating demand is the lowest in the summer, extra natural gas is burned for CO_2 enrichment only. In some greenhouse operations, heat from burning extra natural gas in the summer is stored by hot water in the day and used for heating during the night, resulting in low energy efficiency. Alternatively, compressed CO_2 from cylinders has been used for CO_2 enrichment in the summer to reduce the natural gas consumption. In the integrated eco-industrial system, the biogas consumption can be reduced if alternative sources of CO_2 can be used for CO_2 enrichment so that a larger greenhouse can be matched to a given dairy farm for the same amount of biogas produced. The other problem is that the surplus digestate has to be composted if users cannot be found for substituting sawdust and fertilizers, which can cause significant environmental impacts. In some impact categories, the disposal of surplus digestate leads to a net impact increase in the eco-industrial system compared to the base scenario. Therefore, other potential players were explored who might be introduced to either provide alternative CO_2 sources or /and consume the surplus digestate.

Mushroom growing emits CO_2 continuously from biological respiration and composting of the organic growing medium. Contrary to green plants, mushroom is a fungus which does not consume CO_2 for photosynthesis but extracts carbohydrates and proteins from a medium of decaying organic matter media (compost). The growing cycle of mushrooms typically includes mycelium development, spawning, casing, pinning, and harvesting. CO_2 emissions are observed to be especially high after the formation of spawns. To avoid the CO_2 emissions being accumulated undesirably in the mushroom farm, ventilation is required regularly to maintain a balanced air composition. The ventilation rate varies throughout a growing cycle but on average, 7.2 kg CO_2 is released from one square meter of mushroom growing area in a month at an average concentration of ~815 ppm. The CO_2 enrichment requirement in the greenhouse is around 500~1000 ppm (Luczynski and Portree, 2005). Therefore, the mushroom ventilation air is potentially a biogenic CO_2 source for greenhouses.

The growing media for mushrooms are prepared to contain rich and balanced nutrients with compost as the main body. Nitrogen-rich material such as horse or chicken manure is usually required to improve the nitrogen content. A typical mushroom compost "recipe" contains about 94% straw-bedded horse manure, 2% dried poultry manure, 2% dried brewer's grain, and 2% gypsum (Hardy et al., 2010). Omoanghe et al. (2009) conducted two studies to prove the feasibility of using solid digestate as part of the mushroom compost. They concluded that substrates containing 10–20% solid digestate, 70–80% wheat straw, and 10–20% millet gave the highest mushroom yield (Isikhuemhen et al., 2009a, 2009b). This result was verified by experiments on two mushroom species. Therefore, mushroom farming can also be a potential player to be introduced to consume the solid digesate surplus generated from the previously proposed eco-industrial system.

As a mushroom farm can potentially supply biogenic CO_2 for greenhouse and, at the same time, consume solid digestate surplus from the digester, we propose to introduce a mushroom farm into the dairy farm-greenhouse eco-industrial system. The expanded eco-industrial system is expected to reduce the energy consumption associated with the greenhouse CO_2 enrichment and to avoid the off-site transportation and composting of solids digestate surplus, thereby, further reducing the environmental impacts.

4.2 Framing of LCA Study

4.2.1 Functional unit

The functional unit of this study is again selected as the "disposal of 1,100 tonnes of dairy manure" which is the amount of manure generated by 500 cows in one month (as in Case 1). The sizes of the greenhouse and the mushroom farm are determined based on the following criteria:

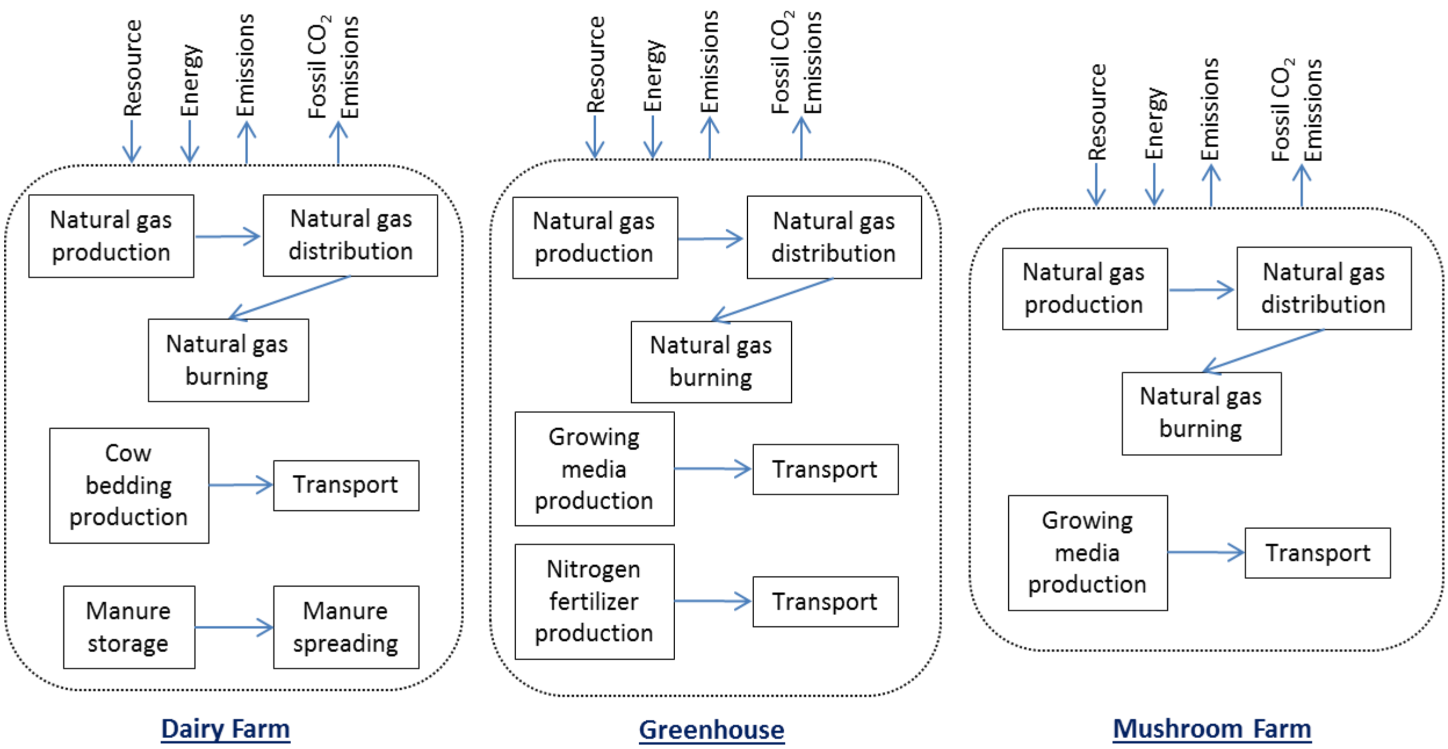
1) The heating potential in the biogas exactly meets the heating demand of dairy farm, digester, greenhouse, and mushroom farm in the summer season. In the other seasons natural gas has to be supplemented so the biogas is intended to meet the base demand only.
2) Besides the CO_2 supplies from burning biogas and natural gas, the remaining CO_2 demand in the greenhouse is met by CO_2 in the ventilated air from the mushroom farm. The mushroom farm provides adequate amount of CO_2 in the summer to meet the peak demand; for the other seasons,

CO_2 in the combustion flue gases is sufficient for greenhouse CO_2 enrichment, thus no extra CO_2 from the mushroom farm is required with the mushroom air being ventilated without use.

Based on these criteria, we come up with a greenhouse of 10,400 m^2 and a mushroom farm with 9,750 m^2 mushroom growing area. The size of the mushroom farm is assumed to be 1/3 of the mushroom growing area which is 3,250 m^2 since multi-layer growing is commonly practiced.

4.2.2 System boundary

Figure 4-1 illustrates the system boundaries of the base scenario (top) and the integrated eco-industrial system (bottom) in this study.



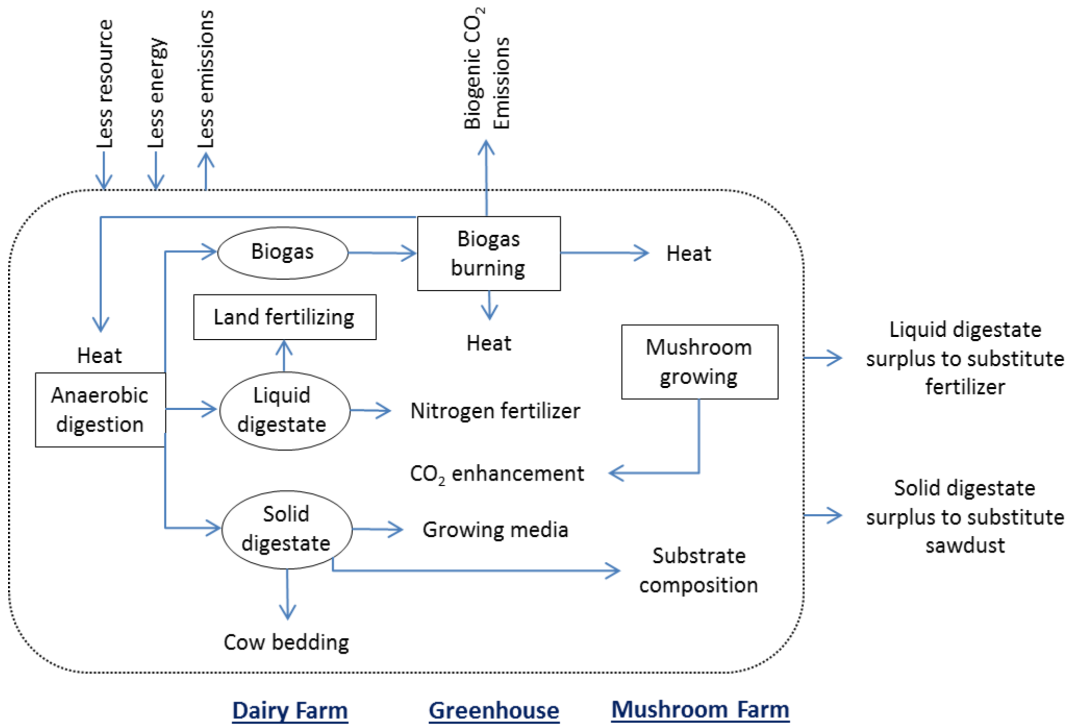


Figure 4-1 System boundaries of Case 7

The system boundaries in this case are very similar to the previous cases with a few extensions associated with the mushroom farm. The additional processes involved in the base scenario are wheat straw production and transport for mushroom growing media preparation. Natural gas consumption for the mushroom farm and the associated upstream processing are also included. For the integrated eco-industrial system, the figure mainly illustrates the operation in the summer when the CO_2-rich ventilation stream from the mushroom farm is supplied to the greenhouse for CO_2 enrichment, as explained in Section 4.2.1.

4.2.3 Scenarios

In the same way as in the previous cases, one base scenario plus one optimistic and one moderate scenario for the eco-industrial system are analyzed designated as Scenario 7-a, Scenario 7-b, and Scenario 7-c, respectively. Table 4-1 presents the basic data of the system. Table 4-2 shows the data at different seasons over a time horizon of one month.

Table 4-1 Basic performance data for Case 7

Item	Value	Unit
Greenhouse area	10,400	m^2
Mushroom farm area	3,250	m^2

Item	Value	Unit
Biogas generation	36,900	m^3/month
Dried solid digestate generation	114	tonne/month
Liquid digestate generation	791	tonne/month
Sawdust bedding demand in dairy farm	42	tonne/month
Growing media demand in greenhouse	1.1	tonne/month
Straw demand in mushroom farm	53	tonne/month
Dried solid digestate surplus	19	tonne/month
N fertilizer demand in greenhouse	78	kg N/month
Liquid digestate surplus	1.1	tonne N/month

Table 4-2 Seasonal performance data for Case 7

Item	Unit	Spring	Summer	Fall	Winter
Heat demand for dairy farm	GJ/month	114	4	97	185
Heat demand for greenhouse	GJ/month	1,487	603	1,144	1,809
Heat demand for mushroom farm	GJ/month	254	10	215	410
CO_2 demand for greenhouse	tonne/month	108	144	108	14
Natural gas use in base scenario	GJ/month	1,928	2,572	1,928	2,561
Heat demand for digester	GJ/month	134	93	127	158
Biogas generation	m^3/month	36,900	36,900	36,900	36,900
Heat supply from biogas	GJ/month	710	710	710	710
Natural gas supplement for heating	GJ/month	1,279	0	872	1,851
CO_2 supply from flue gas	tonne/month	119	74	100	127
CO_2 required from mushroom farm	tonne/month	0	70	8	0

As Table 4-2 shows, a significant amount of natural gas is required for the CO_2 enrichment when a larger greenhouse is integrated; especially in the summer, the total natural gas demand is almost four times the demand for heating. With the CO_2 supplied from the mushroom farm, no extra biogas needs to be burned for CO_2-enrichment in the greenhouse, and, as a result, the same amount of biogas as in Case 1 is now able to support a double sized greenhouse.

4.3 Description of Inventory Model

The descriptions in Section 2.3 remain applicable for Case 7. Only the new stages involved in the mushroom farm are described here. Detailed statistical data and data sources also refer to Appendixes A and B.

- CO_2 supply from mushroom farm ventilation

As mentioned in Section 4.1, 7.2 kg CO_2/month is released from one square meter of mushroom growing area and is ventilated at a rate of 12 m^3 air/m^2/hour; thus the average CO_2 concentration in the ventilation air is about 815 ppm. During a short time during spawning, the CO_2 concentration can be up to 10,000 ppm and the maximum tolerable level is 30,000 ppm (van Griensven, 1988). In this preliminary design and assessment study, the ventilation air is assumed to be piped to the greenhouse with no pre-treatment. In fact, the mushroom growing medium is a compost product which emits various pollutants; together with suspended mushroom spores during spawning, the ventilation air needs to be purified before it is used in the greenhouse, and the CO_2 concentration may also need to be adjusted occasionally. Once more information becomes available, the analysis can be updated.

- Mushroom growing substrate

Mushroom growing substrate is prepared from composting products and the application rate was reported as 36 kg/m^2 growing area/month on average (van Griensven, 1998). As recommended by Omoanghe et al. (2009a, 2009b), we model the substrate composition as containing 15% of the solid digestate in the integrated eco-industrial system, which replaces the same amount of wheat straw used in the base scenario. Production of wheat straw is included in the system boundary and the dataset is from EcoInvent database as a by-product from the wheat growing operation on the farm (Swiss Centre for Life Cycle Inventories et al., 2010).

4.4 Results and Discussions

4.4.1 Life Cycle Inventory data

The same group of air emissions as in the previous cases is selected for examination. Table 4-3 shows the monthly average LCIs of the three scenarios in each season. Tables 4-4 and 4-5 respectively summarize the absolute and percentage emission reductions for the eco-industrial system over the base scenario. In the same way as in Case 1, the LCI results for the summer season are divided into life cycle stages (Figure 4-2). Specifically, the three diagrams refer to Scenarios 7-a to 7-c from top to bottom. Major contributors of specific air emissions are identified and discussed. Compared to Case 1, the newly involved straw production generates significant emissions of certain species, and differences mainly result from more natural gas and resources being replaced in the eco-indust rial system.

Table 4-3 Selected LCI results for three scenarios of Case 7

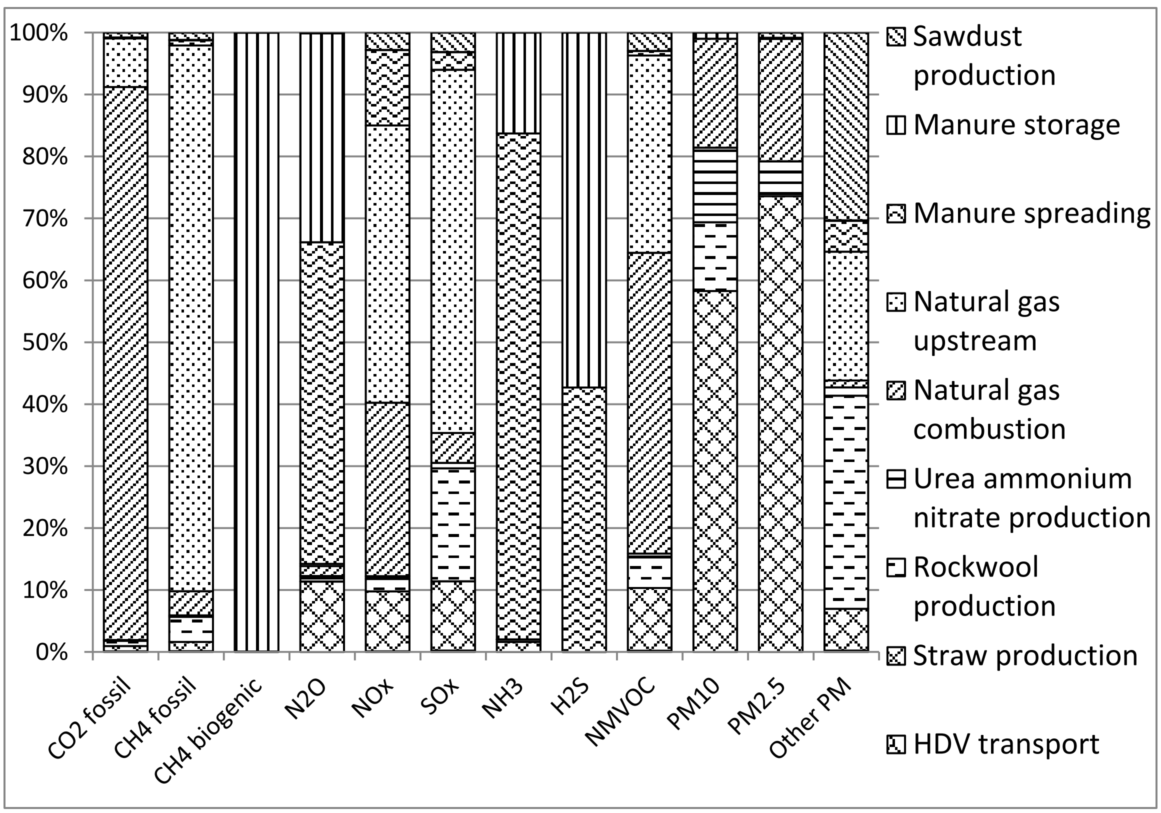
		Spring			Summer			Fall			Winter		
		Scenario 7-a	Scenario 7-b	Scenario 7-c	Scenario 7-a	Scenario 7-b	Scenario 7-c	Scenario 7-a	Scenario 7-b	Scenario 7-c	Scenario 7-a	Scenario 7-b	Scenario 7-c
CO_2 fossil	tonne	127	78	82	168	-3	1	127	53	56	167	115	118
CH_4 fossil	kg	154	88	96	200	-3	4	154	59	66	200	129	137
CH_4 biogenic	tonne	1.98	0.15	0.16	1.98	0.15	0.16	1.98	0.15	0.16	1.98	0.15	0.16
N_2O	kg	84	5	16	85	5	15	84	5	15	85	6	16
NOx	kg	115	105	120	140	55	69	115	89	104	140	128	143
SOx	kg	29	7	13	34	-3	2	29	4	9	34	12	17
NH_3	kg	627	538	584	627	538	584	627	538	584	627	538	584
H_2S	kg	422	0.25	0.27	422	0.23	0.26	422	0.24	0.27	422	0.25	0.28
NMVOC	kg	13	5	7	16	-1	0	13	3	5	16	8	10
PM_{10}	kg	0.31	-0.11	0.44	0.33	-0.14	0.41	0.31	-0.12	0.43	0.33	-0.10	0.45
$PM_{2.5}$	kg	1.53	-0.96	0.31	1.61	-1.12	0.15	1.53	-1.01	0.26	1.61	-0.89	0.38
Other PM	kg	5.6	3.9	6.1	5.9	3.2	5.5	5.6	3.6	5.9	5.9	4.1	6.4

Table 4-4 Net emission reductions of the eco-industrial system over the base scenario in Case 7

		Spring		Summer		Fall		Winter	
		Scenario 7-b	Scenario 7-c	Scenario 7-b	Scenario 7-c	Scenario 7-b	Scenario 7-c	Scenario 7-b	Scenario 7-c
CO_2 fossil	tonne	49	45	171	167	75	71	53	49
CH_4 fossil	kg	66	59	204	196	95	88	70	63
CH_4 biogenic	tonne	1.83	1.83	1.83	1.83	1.83	1.83	1.83	1.83
N_2O	kg	79	69	80	70	79	69	79	69
NOx	kg	9	-5	86	71	26	11	12	-3
SOx	kg	21	16	38	32	25	20	22	17
NH_3	kg	89	43	89	43	89	43	89	43
H_2S	kg	422	422	422	422	422	422	422	422
NMVOC	kg	8	6	18	16	10	8	8	7
PM_{10}	kg	0.43	-0.12	0.47	-0.08	0.44	-0.11	0.43	-0.12
$PM_{2.5}$	kg	2.49	1.22	2.73	1.46	2.54	1.27	2.50	1.23
Other PM	kg	1.7	-0.6	2.7	0.4	1.9	-0.4	1.7	-0.5

Table 4-5 Percentage emission reductions of the eco-industrial system over the base scenario in Case 7

	Spring		Summer		Fall		Winter	
	Scenario 7-b	Scenario 7-c	Scenario 7-b	Scenario 7-c	Scenario 7-b	Scenario 7-c	Scenario 7-b	Scenario 7-c
CO_2 fossil	38%	36%	102%	100%	59%	56%	31%	29%
CH_4 fossil	43%	38%	102%	98%	62%	57%	35%	32%
CH_4 biogenic	92%	92%	92%	92%	92%	92%	92%	92%
N_2O	94%	81%	95%	82%	94%	82%	93%	81%
NOx	8%	-4%	61%	51%	22%	10%	9%	-2%
SOx	74%	56%	110%	95%	86%	68%	64%	49%
NH_3	14%	7%	14%	7%	14%	7%	14%	7%
H_2S	100%	100%	100%	100%	100%	100%	100%	100%
NMVOC	61%	48%	109%	99%	77%	64%	51%	41%
PM_{10}	136%	-38%	143%	-24%	139%	-36%	131%	-36%
$PM_{2.5}$	162%	80%	169%	91%	166%	83%	155%	76%
Other PM	31%	-10%	45%	7%	34%	-7%	30%	-9%



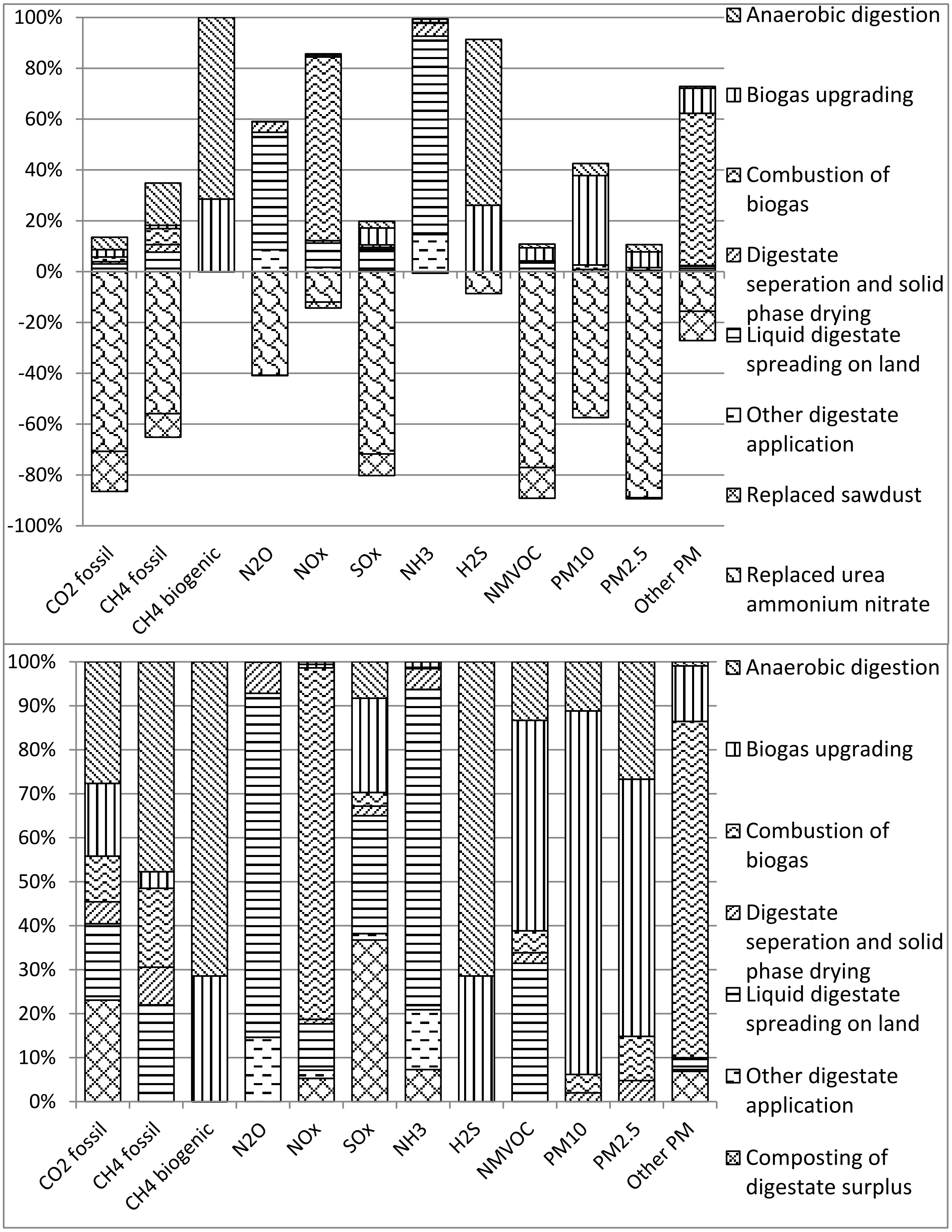


Figure 4-2 Stage-wise selected LCI results for Case 7

The LCI results are discussed mainly by comparison with Case 1 with net emission reductions as a focus. Overall, the differences mainly result from the significant reduction in natural gas use due to substitution by CO_2 from the mushroom farm. Also, straw production creates significant emissions of certain species; thus the replacement of straw with solid digestate surplus leads to noticeable reductions of these emissions.

- Fossil CO_2 and fossil CH_4 emissions

Fossil CH_4 and CO_2 emissions are mainly generated from natural gas production and combustion. In this eco-industrial system, natural gas is largely replaced with biogas, supplying heat and CO_2, and with the CO_2 containing ventilation stream from the mushroom farm. Compared to the results in Case 1, the fossil CO_2 and CH_4 emission reductions are much greater with all the natural gas demand for a larger greenhouse being substituted with biogas and mushroom-generated CO_2. However, the emissions in the base scenario of Case 7 are higher than in Case 1 as a result of a larger greenhouse plus a mushroom farm; hence the percentage reductions in the eco-industrial system in Case 7 are overall smaller than in Case 1. Another point to note is that composting of solid digestate surplus creates a relatively significant amount of fossil CO_2, as shown in Figure 4-2. As the majority of solid digestate surplus is utilized in the mushroom farm as the growing substrate, the emissions from composting the remaining digestate surplus in the moderate scenario are less than in Case 1. In summary, the eco-industrial system in Case 7 shows greater net emission reductions compared to Case 1.

- Biogenic CH_4 and H_2S emissions

Almost all the biogenic CH_4 emissions and H_2S emissions arise from manure storage. Since manure generation in Case 7 is exactly the same as in Case 1, the associated emissions in Case 7 are also the same. Because the manure generation is constant throughout a year, there are no variations between different seasons.

- N_2O emissions

N_2O emissions mainly arise from manure storage, manure spreading, and straw production. A small amount is also recognizable from natural gas combustion. The avoided straw production contributes to the additional emission reduction in the eco-industrial systems compared to Case 1, which amounts to about 9 kg per month. In Case 1, no seasonal variations are observed but a slightly larger reduction is observed in Case 7 over the summer which results from the further reduction of natural gas usage due to the alternative CO_2 source from the mushroom farm.

- NOx emissions

NOx emissions arise mostly from natural gas production and combustion, manure spreading, and straw production in the base scenario. In the eco-industrial system, NO_x emission reductions from using biogas and mushroom CO_2 to substitute natural gas show similar results to CO_2 and CH_4 emissions as explained in the earlier paragraph. Greater reductions are observed in the summer and fall when the CO_2 stream from mushroom production is utilized. Emission reduction from the avoided manure spreading is the same as in Case 1. Substituting straw with solid digestate contributes to a constant emission reduction in all seasons. Overall, NOx emission reductions in Case 7 are larger in all seasons than in Case 1. The moderate scenario of Case 7 in the summer and fall also achieves emission reductions, unlike in Case 1, because the emissions from composting of the surplus digestate are largely abated although the emissions from biogas combustion and liquid digestate spreading remain the same as in Case 1.

- SOx and NMVOC emissions

In the base scenario, SOx and NMVOCs are mainly emitted from natural gas production and combustion. Rockwool production and straw production are the second significant contributors; thus in the eco-industrial system, greater emission reductions than in Case 1 can be observed by replacing these materials with digestate. In the moderate scenarios of the eco-industrial system, the smaller amount of surplus digestate to be composted further reduces SOx emissions. Overall, reductions of SOx and NMVOC emissions in Case 7 are larger than in Case 1.

- NH_3 emissions

NH_3 emissions are mostly generated from manure storage and spreading in the base scenario and from digestate application in the eco-industrial system. Since it is barely associated with energy consumption, no seasonal variations are identified. It is found that the NH_3 emission reduction in the optimistic scenario of the eco-industrial system is smaller than in Case 1 because more digestate is used in the mushroom farm and the larger greenhouse in Case 7. In the moderate scenario, emissions are significantly reduced because less digestate has to be composted; therefore, net overall emission reductions are also observed in the moderate scenario, unlike in Case 1.

- PM emissions

PM_{10} and $PM_{2.5}$ emissions arise mainly from production of material including straw, rockwool, and urea ammonium nitrate. The remainder of the emissions mostly come from natural gas combustion. In the optimistic scenarios of the eco-industrial system, all the straw, rockwool, and urea ammonium nitrate are substituted by digestate. For liquid digestate, a higher amount is utilized for a larger greenhouse to substitute nitrogen fertilizer. For solid digestate, the substitution of sawdust does not significantly reduce emissions of PM_{10} and $PM_{2.5}$, but the substitution of straw and rockwool creates high emission reductions. Thus overall, the emission reductions in the eco-industrial system in Case 7 are larger than in Case 1. In the moderate scenario, the life cycle PM_{10} emissions are still significant and on average higher than the base scenario due to biogas upgrading and combustion operations (Section 2.4.1), but the difference is smaller than that in Case 1. The $PM_{2.5}$ results in the moderate scenario are now lower than the base scenario in all seasons.

Reduction of other PM emissions in the optimistic eco-industrial scenario in Case 7 is lower than in Case 1 because less solid digestate surplus is generated. Sawdust production generates significant other PM emissions (Figure 4-2); although emission reductions can be achieved from replacing more rockwool and straw, reductions from using the digestate surplus to substitute sawdust are not as high as in Case 1. In the moderate scenario, lower emissions arise from composting since less solid digestate surplus is generated. Although the life cycle emissions are still higher than the base scenario on average, the difference is significantly decreased compared to Case 1.

4.4.2 Life Cycle Impact Assessment

The monthly average LCIA results are presented in Table 4-6 for each season, while Tables 4-7 and 4-8 summarize the net and percentage reductions, respectively, in the eco-industrial system compared to the base scenario. Stage-wise data are further shown in Figure 4-3 based on absolute characterization results. All numerical values for the charts are included in Appendix D for reference.

Table 4-6 Life Cycle Impact Assessment results for three scenarios of Case 7

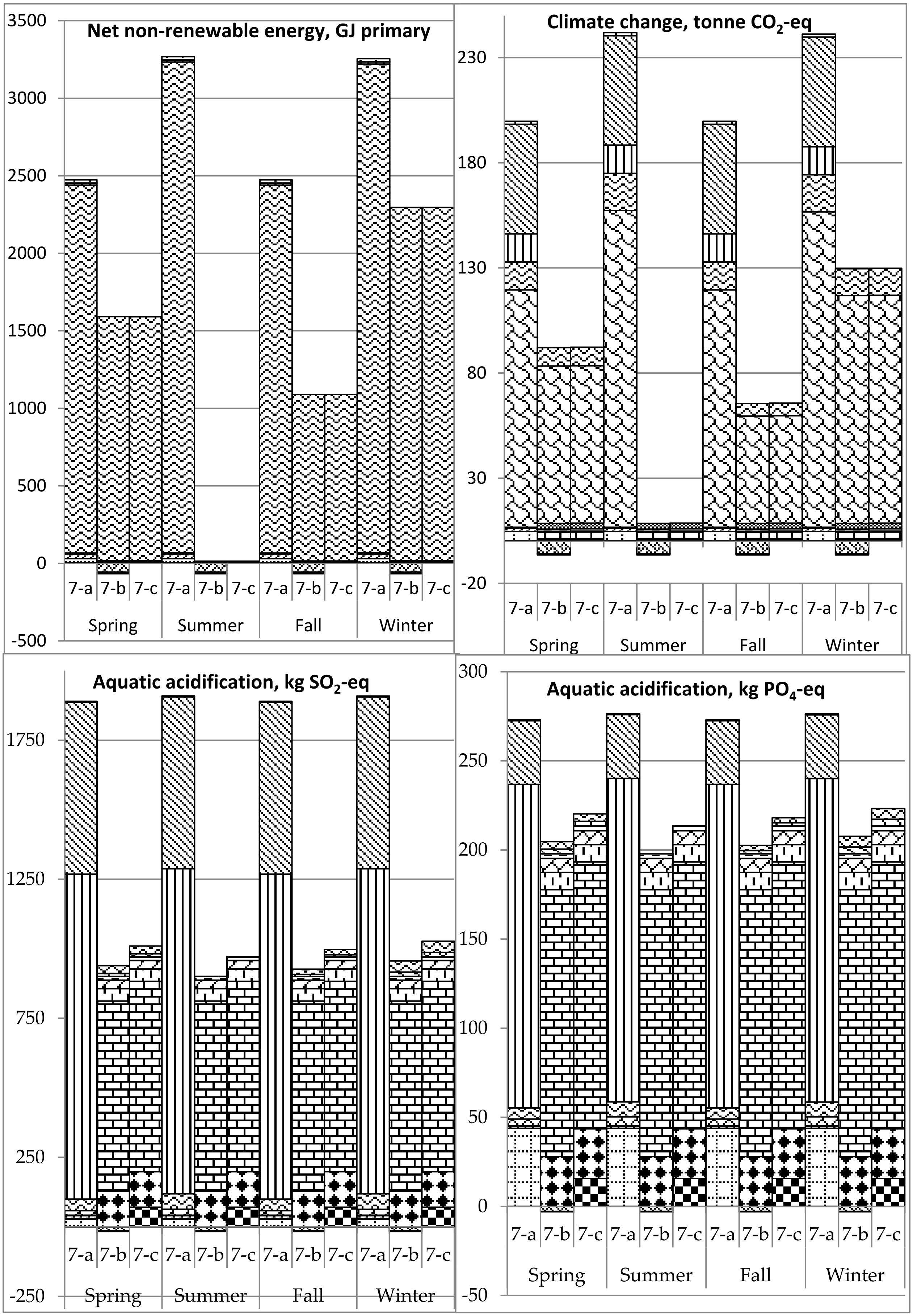
		Spring			Summer			Fall			Winter		
		7-a	7-b	7-c	7-a	7-b	7-c	7-a	7-b	7-c	7-a	7-b	7-c
Non-renewable energy	GJ	2474	1524	1590	3268	-52	14	2474	1023	1089	3255	2229	2296
Climate Change	tonne CO_2-eq	200	86	92	242	2	9	200	59	66	241	123	130
Aquatic acidification	kg SO_2-eq	1890	923	1010	1909	884	971	1890	910	998	1909	940	1027
Aquatic eutrophication	kg PO_4-eq	273	202	220	276	195	214	273	200	218	276	205	223
Respiratory inorganics	kg $PM_{2.5}$-eq	95	79	88	99	71	81	95	77	86	99	82	92
Respiratory organics	kg C_2H_4-eq	7.44	2.84	3.66	9.22	-0.68	0.14	7.44	1.72	2.54	9.19	4.42	5.24
Human toxicity	kg C_2H_3Cl-eq	867	38	87	894	-14	35	867	22	71	893	62	111

Table 4-7 Net impact reductions of the eco-industrial system over the base scenario in Case 7

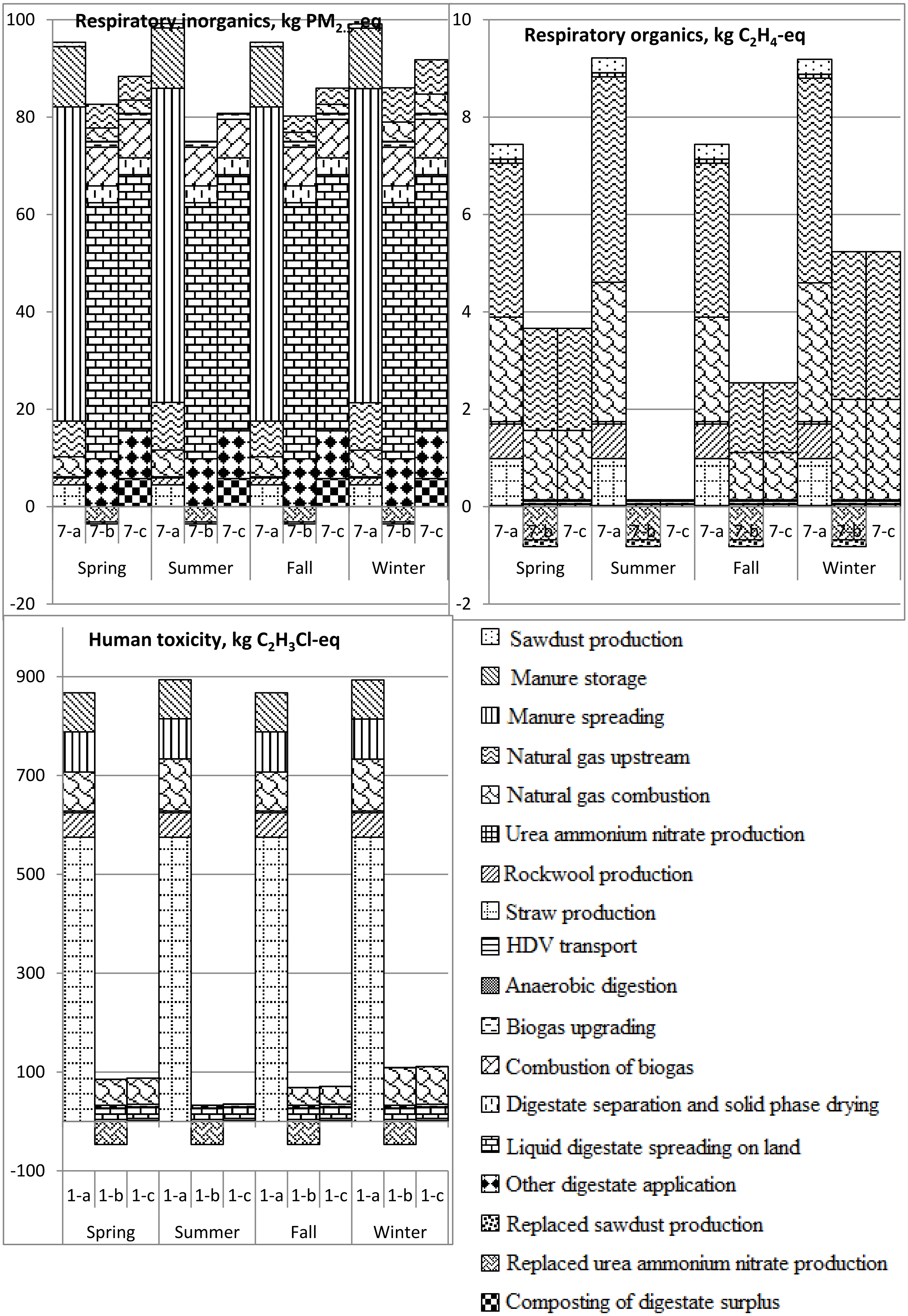
		Spring		Summer		Fall		Winter	
		7-b	7-c	7-b	7-c	7-b	7-c	7-b	7-c
Non-renewable energy	GJ	950	884	3,321	3,254	1,452	1,385	1,026	960
Climate Change	tonne CO_2-eq	114	107	240	233	141	134	118	111
Aquatic acidification	kg SO_2-eq	967	880	1025	938	979	892	969	882
Aquatic eutrophication	kg PO_4-eq	71	53	81	63	74	55	72	53
Respiratory inorganics	kg $PM_{2.5}$-eq	16	7	28	18	19	9	17	7
Respiratory organics	kg C_2H_4-eq	4.60	3.78	9.90	9.08	5.72	4.90	4.77	3.95
Human toxicity	kg C_2H_3Cl-eq	829	780	908	859	845	797	831	782

Table 4-8 Percentage impact reductions of the eco-industrial system over the base scenario in Case 7

	Spring		Summer		Fall		Winter	
	7-b	7-c	7-b	7-c	7-b	7-c	7-b	7-c
Non-renewable energy	38%	36%	102%	100%	59%	56%	32%	29%
Climate change	57%	54%	99%	96%	71%	67%	49%	46%
Aquatic acidification	51%	47%	54%	49%	52%	47%	51%	46%
Aquatic eutrophication	26%	19%	29%	23%	27%	20%	26%	19%
Respiratory inorganics	17%	7%	28%	18%	20%	9%	17%	7%
Respiratory organics	62%	51%	107%	98%	77%	66%	52%	43%
Human toxicity	96%	90%	102%	96%	97%	92%	93%	88%



Net non-renewable energy, GJ primary
3500
3000
2500
2000
1500
1000
500
0
-500
7-a 7-b 7-c 7-a 7-b 7-c 7-a 7-b 7-c 7-a 7-b 7-c
Spring Summer Fall Winter
Climate change, tonne CO_2-eq
230
180
130
80
30
-20
7-a 7-b 7-c 7-a 7-b 7-c 7-a 7-b 7-c 7-a 7-b 7-c
Spring Summer Fall Winter
Aquatic acidification, kg SO_2-eq
1750
1250
750
250
-250
7-a 7-b 7-c 7-a 7-b 7-c 7-a 7-b 7-c 7-a 7-b 7-c
Spring Summer Fall Winter
Aquatic acidification, kg PO_4-eq
300
250
200
150
100
50
0
-50
7-a 7-b 7-c 7-a 7-b 7-c 7-a 7-b 7-c 7-a 7-b 7-c
Spring Summer Fall Winter



Respiratory inorganics, kg PM2.5-eq
100
80
60
40
20
0
-20
7-a 7-b 7-c 7-a 7-b 7-c 7-a 7-b 7-c 7-a 7-b 7-c
Spring Summer Fall Winter
Respiratory organics, kg C2H4-eq
10
8
6
4
2
0
-2
7-a 7-b 7-c 7-a 7-b 7-c 7-a 7-b 7-c 7-a 7-b 7-c
Spring Summer Fall Winter
Human toxicity, kg C2H3Cl-eq
900
700
500
300
100
-100
1-a 1-b 1-c 1-a 1-b 1-c 1-a 1-b 1-c 1-a 1-b 1-c
Spring Summer Fall Winter
Sawdust production
Manure storage
Manure spreading
Natural gas upstream
Natural gas combustion
Urea ammonium nitrate production
Rockwool production
Straw production
HDV transport
Anaerobic digestion
Biogas upgrading
Combustion of biogas
Digestate separation and solid phase drying
Liquid digestate spreading on land
Other digestate application
Replaced sawdust production
Replaced urea ammonium nitrate production
Composting of digestate surplus

Figure 4-3 Seasonal stage-wise impact results for Case 7

- Non-renewable energy consumption

Non-renewable energy consumption is still mainly associated with natural gas consumption. With a larger greenhouse and an additional mushroom farm involved, much more natural gas is required in the base scenario. This demand is significantly reduced by biogas production and the use of the CO_2 from the mushroom farm. In total, over 19 TJ non-renewable energy is saved in the moderate scenario of the eco-industrial system in one year.

- Climate change

Climate change impact arises from natural gas production and consumption, as explained in Section 2.4.2. The total reduction of climate change impact in the moderate eco-industrial scenario in this case is 1,758 tonne CO_2-eq/year, which is greater than the reduction achieved in Case 1 (1,362 tonne CO_2-eq/year).

- Aquatic acidification and eutrophication

Acidification and eutrophication impacts are mostly attributed to manure storage and spreading in the base scenario and to digestate application and composting of surplus digestate in the moderate scenario of the eco-industrial system. The impacts in the base scenario are not very different from Case 1 since the manure to be managed is the same; a slight increase is due to the increased natural gas consumption. In the moderate scenario of the eco-industrial system, much less digestate surplus is generated compared to Case 1; thus a significant reduction is achieved from reduced composting. As a result, the total reduction of acidification impact in Case 7 is up to 10.8 tonne SO_2-eq/year.

- Respiratory effects from inorganic emissions

The base scenario in Case 7 has higher inorganics respiratory effects than the base scenario in Case 1 as a result of higher natural gas consumption and straw production. In the eco-industrial system, the significant impact from composting of surplus digestate in Case 1 is largely abated in Case 7 because much less digestate surplus is left. Therefore, an impact reduction of 127 kg $PM_{2.5}$-eq/year is observed in Case 7, by contrast with the net increase in Case 1.

- Respiratory effects from organic emissions

Respiratory effects from organic emissions arise mainly from natural gas production and consumption, rockwool production, and straw production in the base scenario. In the eco-industrial system in Case 7, natural gas use is significantly reduced, more rockwool is replaced, and straw use is substituted so that a larger impact reduction is achieved compared to Case 1, amounting to 65 kg C_2H_4-eq/year in the moderate scenario.

- Human toxicity

Straw production shows enormous human toxicity impact which arises from the energy use, fertilizer use, and processing over the life cycle of crop production (wheat in this case). In the base scenario of Case 7, the total human toxicity impact is up to around 870 kg C_2H_3Cl-eq/month, which is almost four times as high as in Case 1. The impact from straw production can be avoided in the eco-industrial system and a further reduction can be obtained from substituting more natural gas and rockwool. Overall, the moderate scenario of the eco-industrial system achieves a human toxicity reduction of about 10 tonne C_2H_3Cl-eq/year.

4.4.3 Normalization analysis

The method for normalization analysis in this study is the same as in Chapter 3. Non-renewable energy consumption and climate change impact are significant when normalized with respect to the averages for BC. For the other impact categories, normalization by the equivalent Canadian or European averages shows that the contributions from the dairy industry in BC are small (Section 2.4.3); therefore only the results relative to Case 1 are discussed. As in earlier chapters, the reduction results in the moderate scenario are discussed.

The non-renewable energy saving in the eco-industrial system in this study is 19 TJ /year, corresponding to 3,990 TJ/year if such an integration were applied to the whole industry in BC; this is 18% of the 22,455 TJ energy which is the estimated total non-renewable energy consumption in the BC economy in 2010 (Section 2.4.3). The climate change reduction from this system is 1,758 tonne CO_2-eq/year and the potential reduction from the entire dairy industry can be up to 0.37 Mt CO_2-eq/year which amounts to 37% of the total GHG emissions in BC's agriculture sector (Nyboer and Kniewasser, 2012).

The reduction of acidification impact in Case 7 is up to 10.8 tonne SO_2-eq/year which is 25% greater than in Case 1 (8.7 tonne SO_2-eq/year). The eutrophication impact in Case 7 is lower than the corresponding base scenario, which is contrary to Case 1; the net reduction is 671 kg PO_4-eq/year while in Case 1 the impact increase is 228 kg PO_4-eq/year. Similar to eutrophication, an impact reduction of 127 kg $PM_{2.5}$-eq/year in respiratory effects due to inorganic emissions is observed in Case 7 rather than the net increase found in Case 1. The 65 kg C_2H_4-eq/year reduction of respiratory effects from organic emissions is nearly twice as high as in Case 1 (35 kg C_2H_4-eq/year). The reduction in human toxicity impact is 10 tonne C_2H_3Cl-eq/year, which is five times as high as in Case 1 (2 tonne C_2H_3Cl-eq/year).

4.5 Conclusions

In the integrated dairy farm-greenhouse-mushroom farm system, environmental impacts are reduced compared to the base scenario for all the categories studied, even in the moderate scenario. Non-renewable energy consumption, climate change, aquatic acidification, respiratory effects from organic emissions, and human toxicity were all reduced in the integrated dairy farm-greenhouse system, but the reductions in this enhanced eco-industrial system are even greater. For the aquatic eutrophication and respiratory effects caused by inorganic emissions that are increased in the integrated dairy farm-greenhouse system, this enhanced eco-industrial system lowers the impacts below the base scenario. Among the impact categories, non-renewable energy consumption and climate change remain the most significant impacts in this system. Therefore, improvement is realized by avoiding more environmental impact sources and by fully utilizing the renewable energy and resources for substitution. With mushroom growing as an active agricultural sector in BC, incorporating the mushroom farm into the eco-industrial system reduces the natural gas consumption for the greenhouse operation so that a larger greenhouse can be supplied by the biogas and digestate; meanwhile, CO_2 from the mushroom farm is biogenic that has no global warming impacts. Based on the results in Chapter 3, the dairy farm in this eco-industrial system can install a co-digestion unit using other organic waste available locally to improve biogas yields. It is expected that the environmental impacts would be further reduced with the use of the biogenic CO_2-rich stream to replace the natural gas derived CO_2.

5. CONCLUSIONS AND FUTURE WORK

The large amount of manure generated in BC's livestock industry has been causing environmental problems to local air quality, human health and greenhouse emissions. The high natural gas consumption in the greenhouse industry and other farms is also not sustainable as natural gas is a kind of fossil fuel. Therefore, an environmentally friendly approach to manage livestock manure and heating demand in greenhouses is necessary. One potential waste-to-energy approach is to convert manure to biogas for the substitution of natural gas use in the greenhouse. Previous studies have shown that AD is an advantageous technology to treat manure and produce renewable energy. However, no study has investigated the comprehensive integration of livestock farms with other agriculture producers to establish an eco-industrial system for minimization of material and non-renewable energy uses. In this study, we identified the opportunity to integrate livestock farming and greenhouses in BC by installing an anaerobic digester. The on-site generated biogas and digestate are used to substitute natural gas and materials within the system, thereby further avoiding impacts from materials transport. After a preliminary assessment, we further proposed to incorporate co-digestion of other organic wastes to enhance biogas production, and to introduce a mushroom farm into the integrated system to supplement CO_2 to the greenhouse. In the enhanced integration of dairy farm, greenhouse and mushroom farm, material and energy uses become more self-sufficient with CO_2 from mushroom growing being utilized in the greenhouse and more digestate being used to replace straw usage in the mushroom farm.

A comprehensive environmental assessment using LCA was conducted on a dairy farm-greenhouse eco-industrial system and compared with the conventional practices of the two farms separately. Seven indicators were selected to evaluate the environmental impacts, namely non-renewable energy consumption, climate change, aquatic acidification, aquatic eutrophication, respiratory effects caused by both inorganic and organic pollutants, and human toxicity. In the eco-industrial system where surplus digestate is not utilized and has to be disposed, non-renewable energy consumption, climate change, aquatic acidification, respiratory effects caused by organic pollutants, and human toxicity all have a reduction of more than 50% compared to the base scenario while aquatic eutrophication and respiratory effects from inorganic emissions show an increase of about 20%. If the surplus digestate can be used to substitute sawdust and nitrogen fertilizers in other farms beyond the investigated system, the eutrophication impacts and inorganic caused respiratory effects from the eco-industrial system can be lower than the base scenario. This suggests that it is important to identify local users of the surplus digestate in order to maximize the benefits of such an integrated agricultural system.

Considering the availability of a variety of organic wastes in BC and the potential high biogas yield from co-digestion, co-digestion of dairy manure with five types of organic waste - swine manure, poultry manure, greenhouse plant waste, food waste, and FOG waste - have been evaluated. In general, because of the high biogas yield, the introduction of co-digestion can improve the environmental performance of the eco-industrial system even when the surplus digestate cannot be utilized. All co-digestion cases show larger reductions than the digestion of dairy manure only in non-renewable energy consumption, climate change, respiratory effects caused by organic pollutants, and human toxicity. Co-digestion with plant waste and FOG waste shows smaller percentage impact reductions in aquatic acidification, aquatic eutrophication, and respiratory effects from inorganic pollutants. Co-digestion with swine manure, poultry manure, and plant waste creates larger reductions in all impact categories than the eco-industrial system with only dairy manure digestion.

LCA was also conducted on an advanced eco-industrial system consisting of a dairy farm, a greenhouse, and a mushroom farm. The results showed that the advanced integrated system achieves larger reductions in all impact categories compared to the dairy farm-greenhouse integrated system. In the integrated dairy farm-greenhouse system, some impacts are slightly increased in the scenario in which digestate surplus cannot find a use, but in the advanced dairy farm-greenhouse-mushroom farm eco-industrial system, all categories of impacts are reduced compared to its base scenario.

The results in this study support the hypothesis that environmental benefits are gained from the biogas-based eco-industrial integration of dairy farms and greenhouses. This is consistent with similar studies on the benefit of AD, although no direct comparisons with the other studies are made because of the lack of consistent data and system boundaries. The major environmental impact reductions result from avoided conventional manure management practice and substituted non-renewable energy (natural gas). For some impact categories, substituting materials such as chemical fertilizers, rockwool and sawdust also generate significant environmental benefits. Thus the overall changes resulting from applying AD and eco-industrial integration largely depend on the impacts in the base scenarios. If the biogas is used to replace electricity, for example, the benefit of producing biogas may be more limited than for replacing natural gas, because the BC electricity mix (>90% hydro power) used in the base scenario does not have high environmental impacts. Overall, non-renewable energy consumption and climate change impact from the conventional practices of the studied system are identified as the most significant impacts compared to the regional inventories. By applying the eco-industrial integration, reductions of 1,000~1,800 GJ in non-renewable energy use and 100~200 tonne CO_2-eq in GHG emissions can be achieved per month, which account for up to 20% of the total non-renewable energy use in BC and 50% of the total GHG emissions in BC's agriculture sector, respectively. Impacts of acidification, eutrophication, respiratory effects, and human toxicity from the studied system are not significant in the context of BC as a whole, but the inventories are estimated based on the Canadian or European normalization factors which still need to be verified once BC specific data become available. As a general guideline to local policy makers and practitioners, the deployment and promotion of such an integrated system should focus on its benefits to the reduction in regional fossil fuel use and greenhouse gas emissions.

There are limitations in this study that need to be noted for future improvement. First, the study relies largely on secondary data. Data for the agriculture sector normally have high uncertainties due to the variability of operations and the high uncertainty in the effect of emissions on the natural environment. Taking the emission factors of manure spreading as an example, the real emissions from a certain type and amount of manure can vary significantly due to particular livestock diet, manure processing facilities, local weather after spreading, and the soil and crop conditions etc. Studies to quantify the emissions also used different measurement techniques and approaches under different time horizons. A review of emission factors revealed that the data could vary by more than one order of magnitude. Although the qualitative conclusions are clear and reliable based on the Monte Carlo uncertainty analysis, quantitative results of the emission inventories and the environmental impacts should be improved once primary data are available for conditions specific to BC. Secondly, this study only assessed the environmental performance of the eco-industrial system. In reality, economic feasibility is another important aspect to evaluate. An anaerobic digester normally has higher capital and operating costs than conventional practices. Sound process maintenance is crucial to continuous and stable production of biogas and accordingly economic profitability. Once the economic aspect is taken into account, a better functional unit to compare different scenarios may be selected based on the profits of the eco-industrial system over the base scenario. A preliminary Life Cycle Costing (LCC) study was conducted for the integrated dairy farm-greenhouse eco-industrial system. Results given in Appendix D showed that the integrated system could be cost-effective if it is appropriately designed and planned. Sustained clean energy promotion and market incentives like carbon offsets will further enhance the feasibility. This LCC study was published in the proceeding of the LCA XI conference in Chicago, US in 2011, and is given in Appendix E. Third, there is currently a barrier to actually build such an eco-industrial system in BC as the agriculture zoning regulations do not allow a greenhouse to be built in the proximity of a livestock farm. To include more players such as the mushroom farm, proper siting may impose a challenge as the existing farms are sparsely distributed. This is a common barrier to building eco-industrial systems based on existing facilities, especially agricultural production, unless new players can be introduced. Therefore, changes in policy and regulations are necessary in order to accommodate the introduction of new players into the proposed integrated eco-industrial system.

To address the limitations as discussed above, the following future work is recommended.

1. BC has been enthusiastic in installing ADs at livestock farms especially dairy farms. Currently there are two digesters running on two dairy farms. However, their operations have not been stable so that no primary on-site data are available so far. Once they are well established or new ones put in place, primary data can be collected to fill data gaps and replace the secondary data from the literature as used in this study. The uncertainties from assumptions and the use of secondary data can thus be reduced to some extent.

2. Since the average scale of livestock farms in BC is small, centralized large AD plants may be a better choice as being implemented in some parts of Europe. In that way, the potential of co-digestion can be extended by including multiple organic waste streams rather than the co-digestion of two types of wastes analyzed in this study. Therefore, more cases can be assessed to find an optimal eco-industrial system.

3. It is important to conduct a comprehensive economic assessment to evaluate the feasibility of turning the modeled system into practice. The assessment should be based on updated technologies, local market status, and evolving environmental incentive programs.

4. Based on the advantages of an eco-industrial system revealed in this study, policy analysis should be conducted to evaluate the feasibility and barriers for the deployment of such eco-industrial systems in BC and elsewhere in Canada.

REFERENCES

AD Nett (The European Anaerobic Digestion Network). (2005). Farm Biogas Plants in EU. Retrieved from: http://www.adnett.org/

AFBI (Agri-Food and Biosciences Institute). (2012). Benefits of Anaerobic Digestion. Northern Ireland. Retrieved from: http://www.afbini.gov.uk/index/services/specialist-advice/renewable-energy/re-anaerobic-digestion-intro/re-anaerobic-digestion-benefits.htm

AgStart. (2012). Increasing Anaerobic Digester Performance with Codigestion. USEPA. September 2012. Retrieved from: http://www.epa.gov/agstar/documents/codigestion.pdf

Amon B., Kryvoruchko V., Amon T., Zechmeister-Boltenstern S. (2006). Methane, nitrous oxide and ammonia emissions during storage and after application of dairy cattle slurry and influence of slurry treatment. Agriculture, Ecosystems and Environment. 112 (2006): 153-162

ASAE (American Society of Agricultural Engineers). (2003). Manure Production and Characteristics. ASAE D384.1 FEB03

Atia A., Haugen-Kozyra K.,and Amrani M. Ammonia and Hydrogen Sulfide Emissions from Livestock Production. (1999). Alberta Agriculture, Food and Rural Development. Retrieved from:
http://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/epw8313/$FILE/chapter7.pdf

Badger P. C., Lindsey J. K., Vietch J. D. (1995). Energy Production from Animal Wastes, In Animal Waste and the Land Water Interface, Steele, K. (Ed.); Lewis Publishers-CRC: New York

Baldwin S., Lau A., and Wang M. (2009). Development of a Calculator for the Techno-economic Assessment of Anaerobic Digestion Systems. Report Submitted to BC Ministry of Agriculture and Land and BC Life Sciences. December 9, 2009.

Bartok J. W. (2004). Approximate Heating Value of Common Fuels. University of Connecticut. December 2004. Retrieved from:
http://www.hrt.msu.edu/energy/pdf/heating%20value%20of%20common%20fuels.pdf

BC Greenhouse Growers Association. (2012). Quick Facts in BC Greenhouse Growers Association Website. Retrieved from: http://www.bcgreenhouse.ca/quick_facts.htm

BC Hydro. (2012). Draft Integrated Resource Plan 2012. Chapter 4 - Market Price Forecasts. May 2012.

BC Ministry of Agriculture. (2007). Fast Stats 2010- Agriculture, Aquaculture and Food. 2007.

BC Ministry of Agriculture. (2011). Fast Stats 2010- Agriculture, Aquaculture and Food. June 1 2011

BC Ministry of Energy and Mines. (2011). British Columbia Natural Gas Strategy. Fuelling B.C.'s Economy for the Next Decade and Beyond. Retrieved from: http://www.gov.bc.ca/ener/popt/down/natural_gas_strategy.pdf

BC Ministry of Environment. (2010). On-farm Anaerobic Digestion Waste Discharge Authorization Guideline. BC Ministry of Environment. May 14, 2010. Retrieved from: http://www.env.gov.bc.ca/epd/industrial/agriculture/pdf/anaerobic-digestion-guideline.pdf

Berglund M. and Börjesson P. (2006). Assessment of energy performance in the life-cycle of biogas production. Biomass and Bioenergy. 30 (3): 254–266

Bi T. and Lo V. (2001). CO_2 Recovery and Utilization from Combustion Flue Gases of Alternative Fuels. Submitted to BC Hot House Growers Association. July 31, 2001.

Booy G. et al. (2011). B.C. Holstein Breeders' & Dairy Producers' Directory. BC Holstein News 2011/2012. Page 57-73

Börjesson P. and Berglund M. (2006) Environmental systems analysis of biogas systems—Part I: Fuel-cycle emissions. Biomass and Bioenergy. 30 (2006): 469–485

Börjesson P. and Berglund M. (2007). Environmental systems analysis of biogas systems—Part II: The environmental impact of replacing various reference systems. Biomass and Bioenergy. 31 (2007). 326–344

Braun R. and Wellinger A. (2003). Potential of Co-digestion. IEA Bioenergy. Task 37-Energy from Biogas and Landfill Gas. Retrieved from: http://www.iea-biogas.net/_download/publi-task37/Potential%20of%20Codigestion%20short%20Brosch221203.pdf

Brunet L. (2006). Hazardous Gases. Ontario Ministry of Agriculture, Food, and Rural Affairs.

Canadian Bio-energy Management Inc. (2009). Prince Albert Environmental "Green" Energy Park. Phase II Report. Retrieved from: http://www.citypa.ca/Portals/0/Green%20Energy%20Park/Green%20Energy%20Park%20Study.pdf

Cant M. (2006). Municipal Solid Waste (MSW) Options: Integrating Organics Management and Residual Treatment/Disposal. Municipal Waste Integration Network / Recycling Council of Alberta. April 2006. Page 50. Retrieved from: http://www.recycle.ab.ca/uploads/File/pdf/MSWworkshop/MSW_Options_Report.pdf

Composting 101. (2012). A Composting Guide for the Home Gardener. A Balancing Act (Carbon-to-Nitrogen Ratios). Retrieved from: http://www.composting101.com/c-n-ratio.html

Composting Council of Canada. Composting Processing Technologies. A report produced for the Ontario Ministry of Environment. Retrieved in 2012 from: http://www.compost.org/pdf/compost_proc_tech_eng.pdf

Crolla A., Kinsley C., Désilets É, et al. (2007). Assessment of the Production and Land Application of Anaerobically Digested Manure from Medium Sized Livestock Farms. Conference presentation. Growing the Margins: Energy Conservation and Generation for Farms and Food Processors. London, Ontario. April 11-14, 2007.

de Vries J.W., Corré W.J., van Dooren H.J.C. (2010). Environmental assessment of untreated manure use, manure digestion and co-digestion with silage maize. Deliverable for the 'EU-AGRO-BIOGAS' project. Wageningen UR Livestock Research. June 2010. Retrieved from: http://edepot.wur.nl/146789

DEFRA (Department for Environment, Food and Rural Affairs, UK). (2010). Fertiliser Manual, 8th Edition. June 2010. Retrieved from: http://www.defra.gov.uk/publications/files/rb209-fertiliser-manual-110412.pdf

Delucchi, M., and Levelton. (2012). GHGenius v4.01. Retrieved from http://www.ghgenius.com/

Dong h., Mangino J. and McAllister T. A. (2006). 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 4: Agriculture, Forestry and Other Land Use. Chapter 10: Emissions from Livestock and Manure Management. Retrieved from: http://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/4_Volume4/V4_10_Ch10_Livestock.pdf

Eco-Industrial Solutions Ltd. (2012). Eco-Industrial Network Possibilities in Richmond BC. Retrieved in 2012 from: http://www.ecoindustrial.ca/projects/Richmond_EIN_Assessment.pdf

Edelmann W., Baier U., Engeli H. (2005). Environmental aspects of the anaerobic digestion of the organic fraction of municipal solid wastes and of solid agricultural wastes. Water Science & Technology. 52(1-2): 203-208

Ehrenfeld J. and Gertler N. (1997). "Industrial Ecology in Practice. The Evolution of Interdependence at Kalundborg". Journal of Industrial Ecology. 1(1): 67-79

Ekvall T. (2003). Attributional and consequential LCI modelling. InLCA/LCM 2003, Seattle, September 22-25 2003. Retrieved from: http://www.lcacenter.org/InLCA-LCM03/Ekvall-presentation.pdf

Electrigaz Technologies Inc. (2007). Feasibility Study – Anaerobic Digester and Gas Processing Facility in the Fraser Valley, British Columbia. A report prepared for BC BioProducts Association. November 2007

Electrigaz Technologies Inc. (2008). Feasibility Study – Biogas upgrading and grid injection in the Fraser Valley, British Columbia. A report prepared for BC Innovation Council. June 2008

Environment Canada. (2011). National Inventory Report 1990-2009: Greenhouse Gas Sources and Sinks in Canada. Retrieved from: https://www.ec.gc.ca/Publications/default.asp?lang=En&xml=A07097EF-8EE1-4FF0-9AFB-6C392078D1A9

Fishman G. S. (1995). Monte Carlo: Concepts, Algorithms, and Applications. New York: Springer. ISBN 0-387-94527-X

Fortis BC. (2012). Heat content values. Retrieved from: http://www.fortisbc.com/NaturalGas/Business/PriceAndMarketInformation/Pages/Heat-content-values.aspx

Foster L. T., Keller C. P., McKee B., & Ostry A. (2011). BC Atlas of Wellness 2nd Edition. The Geography of Food Security and Nutrition. Retrieved from: http://www.geog.uvic.ca/wellness/wellness2011/Atlas%20of%20Wellness%202011%20Final.pdf

Frischknecht R., Jungbluth N., et al. (2007). Implementation of Life Cycle Impact Assessment Methods. Ecoinvent report No. 3. December 2007. Retrieved from: http://www.ecoinvent.org/fileadmin/documents/en/03_LCIA-Implementation.pdf

Frost P. and Gilkinson S. (2011). 27 Months Performance Summary for Anaerobic Digestion of Dairy Cow Slurry at Afbi Hillsborough. Interim Technical Report. June 2011. Retrieved from: http://www.afbini.gov.uk/afbi-ad-hillsborough-27-months-june-11.pdf

Geng Y, Haight M, and Zhu Q. (2006). Empirical Analysis of Eco-Industrial Development in China. Sustainable Development. Retrieved from: http://environment.uwaterloo.ca/planning/publications/Geng_Yong_Eco_industrial.pdf

Geomatrix Consultants and Engineers Inc. (2008). Final Report for Study of Food-Based Inputs for Biogas Systems in Ontario. A report prepared for Ontario Ministry of Agriculture, Food and Rural Affairs. May 9, 2008

Gilbert T. Meyer A., and Stearns T. (2007). Hardwick Area Agro-Eco Industrial Park-Adding Value to Vermont's Agriculture. Concept Paper. December 2007. Retrieved from: http://www.uvm.edu/~susagctr/Documents/HardwickAgro-EcoIndustrialPark.pdf

Goedkoop M., de Schryver A., Oele M., Durksz S., and de Roest D. (2010). Introduction to LCA with SimaPro 7. PRé Consultants. November 2010.

Goedkoop M. and Spriensma R. (2000). The Eco-indicator 99 A damage oriented method for Life Cycle Impact Assessment. Methodology Report. Second edition. 17 April 2000. Retrieved from: http://teclim.ufba.br/jsf/indicadores/holan%20ecoindicator%2099.pdf

Gooch C. and Inglis S. (2008). Anaerobic Digestion at Patterson Farms, Inc.: Case Study. Manure Management Program. Cornell University.

Gooch C. and Pronto J. (2008a). Anaerobic Digestion at AA Dairy: Case Study. Manure Management Program. Cornell University.

Gooch C. and Pronto J. (2008b). Anaerobic Digestion at Emerling Farm, Inc.: Case Study. Manure Management Program. Cornell University.

Gooch C. and Pronto J. (2009). Anaerobic Digestion at Aurora Ridge Farm: Case Study. Manure Management Program. Cornell University.

Goodstein D. L. (2004). Out of Gas: The End of the Age of Oil. W. W. Norton & Company.

Graedel T. E. and Allenby B R. (2003). Industrial ecology. Upper Saddle River, New Jersey. Prentice Hall, c2003.

Guinée, J.B., Gorrée, M., Heijungs, R., et al., (2002). Handbook on LCA. Instititute of Environmental Sciences (CML). University Leidon. Retrieved from: http://cml.leiden.edu/research/industrialecology/researchprojects/finished/new-dutch-lca-guide.html#reports-in-the-english-language-last-update-3-july-2002

Hagen Ø, Hung C, Josefsen K. D., and Pettersen J. (2011). A manure-based eco-park: how society and technology affects the environmental performance of biogas from manure. MiSA (Miljøsystemanalyse - Environmental Systems Analysis), Norway.

Hao X. and Chang C. (2001). Gaseous NO, NO_2, and NH_3 Loss during Cattle Feedlot Manure Composting. Agriculture and Agri-Food Canada. Retrieved from: https://mail-attachment.googleusercontent.com/attachment/u/0/?ui=2&ik=153483f80f&view=att&th=13c53bef3436c4c2&attid=0.1&disp=inline&realattid=f_hc5037mv0&safe=1&zw&saduie=AG9B_P-Qj9Cx7X9TQx0O7PRqhSj0&sadet=1358709327082&sads=5IgqzIiCPEidqzBy-7n7RPh1jv8

Hardy C., Hedges S., Simonds D. (2000). Integrated Bio-Systems: Mushrooming Possibilities. Developing Industrial Ecosystems. Yale F&ES Bulletin

Herath G. and Ratnayake U. (2009). Bio-Based Eco Industrial Clustering in Dambulla Sri Lanka. University of Peradeniya, Sri Lanka. Retrieved from http://www.iges.or.jp/en/be/pdf/activity11/Sri_Lanka.pdf

Hjort-Gregersen K., Blandford D. and Gooch C. A. (2011). Biogas from Farm-based Biomass Sources, Developments in Europe and the US. EuroChoices. 2011. 10(3).

Huijbregts M. (1999). Life-cycle assessment of acidifying and eutrophying air pollutants. University of Amsterdam. December 1999. Retrieved from: http://media.leidenuniv.nl/legacy/Life-cycle%20impact%20assessment.pdf

Indigo Development. (2005). Developing Agricultural Eco-Industrial Parks in China. An Indigo Industrial Ecology Paper. Retrieved from: http://www.indigodev.com/AEIPwhitepaper.html

International Energy Agency. (2012). Golden Rules for a Golden Age of Gas. World Energy Outlook Special Report on Unconventional Gas. 12 November 2012. Retrieved from: http://www.worldenergyoutlook.org/media/weowebsite/2012/goldenrules/WEO2012_GoldenRulesReport.pdf

Isikhuemhen O. S. and Mikiashvili N. A. (2009b). Lignocellulolytic enzyme activity, substrate utilization, and mushroom yield by Pleurotus ostreatus cultivated on substrate containing anaerobic digester solids. J Ind Microbiol Biotechnol. (2009) 36:1353-1362

Isikhuemhen O. S., Mikiashvili N. A., Kelkar V. (2009a). Application of solid waste from anaerobic digestion of poultry litter in Agrocybe aegerita cultivation: mushroom production, lignocellulolytic enzymes activity and substrate utilization. Biodegradation. (2009) 20:351-361

ISO 14044:2006. Environmental management-Life cycle assessment-Requirements and guidelines.

Jolliet, O., Margni, M., Charles, R., Humbert, S., Payet, J., Rebitzer, G., & Rosenbaum, R. (2003). IMPACT 2002+: A new life cycle impact assessment methodology. The International Journal of Life Cycle Assessment, 8(6), 324-330. Retrieved from http://dx.doi.org/10.1007/BF02978505

Jungbluth N. et al. (2007). Life Cycle Inventories of Bioenergy. EcoInvent report no. 17. Swiss Centre for Life Cycle Inventories. December 2007.

Khakbazan M. (2000). Descriptive Analysis of On-Farm Energy Use in Canada. A Report to Natural Resources Canada (NRCan). February 2000. Retrieved from: http://www.usask.ca/agriculture/caedac/pubs/Energy.PDF

Lansche J. and Müller J. (2012). Life cycle assessment of energy generation of biogas fed combined heat and power plants: Environmental impact of different agricultural substrates. Eng. Life Sci. 2012, 12, No. 3, 313–320

Lau A., Baldwin S., Wang M. (2010): Techno-Economic Assessment Of Anaerobic Digestion Systems For Agri-Food Wastes. XVIIth World Congress of the International Commission of Agricultural and Biosystems Engineering (CIGR). Québec City, Canada

Lleleji K. E., Martin C., and Jones D. Basics of energy Production through Anaerobic digestion of Livestock Manure. Purdue University. Retrieved in 2012 from: http://www.extension.purdue.edu/extmedia/ID/ID-406-W.pdf

Lowe, E. A. (2001). Eco-industrial Park Handbook for Asian Developing Countries. A Report to Asian Development Bank, Environment Department, Indigo Development, Oakland, CA

Luczynski A. and Portree J. (2005). Growing Greenhouse Peppers in British Columbia. A production guide for commercial growers. Published by BC Greenhouse Growers' Association and BC Ministry of Agriculture, Food and Fisheries.

Lukehurst C.T., Frost P., Al Seadi T. (2010). Utilisation of digestate from biogas plants as biofertiliser. IEA Bioenergy Task 37. Retrieved from: http://www.iea-biogas.net/_download/publi-task37/Task37_Digestate_brochure9-2010.pdf

Lynch D. (2012). Vancouver Weather - Vancouver Average Monthly Temperatures. Retrieved from: http://vancouver.about.com/od/vancouverevents/a/Vancouver-Weather.htm

Mezzullo, W. G., Mcmanus, M. C., Hammond, G. P. (2012). Life cycle assessment of a small-scale anaerobic digestion plant from cattle waste. Applied Energy. University of Bath.

Monnet F. (2003). An Introduction of Anaerobic Digestion of Organic Wastes. Remade Scotland. November 2003. Retrieved from: http://www.biogasmax.co.uk/media/introanaerobicdigestion__073323000_1011_24042007.pdf

Morris C., Jorgenson M., and Snellings S. (2010). Carbon and Energy Life-Cycle Assessment for Five Agricultural Anaerobic Digesters in Massachusetts on Small Dairy Farms. International Food and Agribusiness Management Review. Volume 13, Issue 3, 2010.

Nemecek T.and Kagi T. (2007). Life Cycle Inventories of Agriculture Production Systems. EcoInvent report no. 15. Swiss Centre for Life Cycle Inventories. December 2007.

NERC. Manure Production and Bedding Used Calculator. Retrieved from: www.nerc.org/.../manure.../manure_generation_calculator.xls

Nielsen P.H., Nielsen A.M., Weidema B.P., Dalgaard R, and Halberg N. (2003). LCA food database. www.lcafood.dk

Nyboer J. and Kniewasser M. (2012). Energy and GHG Emissions in British Columbia 1990-2010. The Pacific Institute for Climate Solutions (PICS). June 2012. Retrieved from: http://pics.uvic.ca/sites/default/files/uploads/publications/Energy_Data_Report_2010_1.pdf

O'Neil, D.H. and Phillips V. R. A. (1992). Review of the control of odour nuisance from livestock buildings: Part 3: Properties of the odourous substances which have been identified in livestock wastes or in the air around them. Journal of Agricultural Engineering Research 53(1): 23 - 50.

Pa A. A. (2009). Development of British Columbia Wood Pellet Life Cycle Inventory and its Utilization in the Evaluation of Domestic Pellet Applications. M.A.Sc. thesis. University of British Columbia, Canada. December, 2010

Palmer V. (2012). Uncertainties make it hard to predict when first LNG terminal will be built. The Vancouver Sun. October 1, 2012. Retrieved from: http://www.vancouversun.com/business/energy/Palmer+Uncertainties+make+hard+predict+when+first/6908045/story.html

Pennington D. W., Margni M., Ammann C., and Jolliet O. (2005). Multimedia Fate and Human Intake Modeling: Spatial versus Nonspatial Insights for Chemical Emissions in Western Europe. Environ. Sci. Technol. 39, pp. 1119-1128.

Petersen S. Nitrous oxide emissions from manure and inorganic fertilisers applied to spring barley. Journal of Environmental Quality. 1999;28: 1610–1618.

Poon D. and Schmidt O. (2010). Phosphorus Considerations for Nutrient Management. BC Ministry of Agriculture and Lands. Retrieved from: http://www.agf.gov.bc.ca/resmgmt/publist/600Series/631500-4_Phosphorus_Considerations.pdf

Pöschl M., Ward S., Owende P. (2010). Evaluation of energy efficiency of various biogas production and utilization pathways. Applied Energy. 87 (2010): 3305-3321

Pronto J. and Gooch C. (2008). Anaerobic Digestion at New Hope View Farm: Case Study. Manure Management Program. Cornell University.

Pronto J. and Gooch C. (2009). Anaerobic Digestion at Noblehurst Farms, Inc.: Case Study. Manure Management Program. Cornell University.

Rockwool France. (2012). Acid rain and the Rockwool impact. Retrieved in 2012 from: http://www.rockwool-coresolutions.com/environment/sustainability/acid+rain

Row J. and Neabel D. (2005). Integrated Manure Utilization System Life-Cycle Value Assessment-IMUS LCVA. Pembina Institute. October 2005. Retrieved from: http://www.pembina.org/pub/208

RTI International. (2000). Life Cycle Inventory and Cost Model for Mixed Municipal and Yard Waste Composting. A report produced for U.S. Environmental Protection Agency. July 2000. Retrieved from: https://mswdst.rti.org/docs/Compost_Model_OCR.pdf Also with some of the data obtained from personal contact with Dr. Eric Solano. 2012.

Schmidt O. and Hughes-Games G. (2010). Potassium Considerations for Nutrient Management. BC Ministry of Agriculture and Lands. Retrieved from: http://www.agf.gov.bc.ca/resmgmt/publist/600Series/631500-5_Potassium_Considerations.pdf

Shayya W. H. (2008). Anaerobic Digestion at Morrisville State College: A Case Study. November 30 2008.

Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor and H.L. Miller (eds.) (2007). Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, 2007. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Skiff R. (1997). Eco-Industrial Park Workshop Proceedings. Cape Charles, Virginia. February 1997. Retrieved from: http://clinton2.nara.gov/PCSD/Publications/Eco_Workshop.html#v-b

Sommer SG, Møller HB, Petersen SO. (2001). Reduktion af drivhusgasemission fra gylle og organisk affald ved biogasbehandling [in Danish, The reduction of greenhouse gases from manure and organic waste using digestion and biogas production]. DJF-rapport no 31. Tjele, Denmark: Husdyrbrug, Danish Institute of Agricultural Sciences; 2001.

Statistics Canada. (1996). 1996 Agricultural Operations National and Provincial Highlights Tables. British Columbia Highlights. Retrieved from: http://www.statcan.gc.ca/ca-ra/bc-cb-eng.pdf

Statistics Canada. (2006). 2006 Census of Agriculture. Retrieved from: http://www.statcan.gc.ca/ca-ra2006/index-eng.htm

Statistics Canada. (2012). British Columbia's population clock. Retrieved from: http://www.statcan.gc.ca/ig-gi/pop-bc-eng.htm

Stucki M., Jungbluth N., Leuenberger M. (2011). Life Cycle Assessment of Biogas Production from Different Substrates. ESU-services Ltd. Retrieved from: http://www.esu-services.ch/fileadmin/download/publicLCI/stucki-2011-biogas-substrates.pdf

Swiss Centre for Life Cycle Inventories. (2010). Ecoinvent Database v 2.2.

Topper P. A. (2012). The Fate of Nutrients and Pathogens During Anaerobic Digestion of Dairy Manure. Penn State Digester Projects. Retrieved from: http://extension.psu.edu/energy/waste-to-energy/biogas/projects/g-71

Torenvliet P. Bakerview EcoDairy Farm. Personal Contact. 2011

USEPA. (2000). Emission Factors AP-42, Vol. I, CH1.4: Natural Gas Combustion. August 2000. Retrieved from: http://www.epa.gov/ttn/chief/ap42/ch03/final/c03s02.pdf

USEPA. (2012). Particulate Matter (PM). Retrieved from: http://www.epa.gov/pm/

van Griensven L. J. L. D. (1988). The Cultivation of Mushrooms. Darlington Mushroom Lab.

van Haaren R., Themelis N., Barlaz M. (2010). LCA comparison of windrow composting of yard wastes with use as alternative daily cover (ADC). Waste Management. 30 (2010): 2649-2656

Varshney C. K. and Attri A. K. (1998). Global warming potential of biogenic methane. Tellus. (1999). 51B: 612-615.

Venczel M. and Powers S. E. (2010). Anaerobic Digestion and Related Best Management Practices: Utilizing Life Cycle Assessment. Written for presentation at the 2010 ASABE Annual International Meeting. Pittsburgh, Pennsylvania. June 20 – June 23, 2010

Viswanath P., Devi S. S., Nand K. (1992). Anaerobic Digestion of Fruit and Vegetable Processing Wastes for Biogas Production. Bioresource Technology. 40 (1992): 43-48

Werner E. and Strehler B. (2012). British Columbia On-Farm Anaerobic Digestion Benchmark Study. A report prepared for BC Agricultural Research and Development Corporation CH-Four Biogas Inc. Retrieved from: http://www.bcfarmbiogas.ca/files/file/AD%20Benchmarking%20Study.pdf

Wong R. (2012). OriginO-Origin Organic Farms. Personal contacts.

Wood S. and Cowie A. (2004). A Review of Greenhouse Gas Emission Factors for Fertiliser Production. Cooperative Research Centre for Greenhouse Accounting. New South Wales. For IEA Bioenergy Task 38. June 2004

Wright P. (2004). Anaerobic Digester at EL-VI Farms: Case Study. Manure Management Program. Cornell University.

Wright P. and Ma J. (2003a). Anaerobic Digester at Freund Dairy: Case Study. Manure Management Program. Cornell University.

Wright P. and Ma J. (2003b). Anaerobic Digester at Farber Dairy Farm: Case Study. Manure Management Program. Cornell University.

Yakel G. (2011). Personal contact.

APPENDIX A – DATA AND DATA SOURCES FOR MODELING

Table A-1 Average data for energy consumptions

Energy type	Process	Consumption rate	Source
Natural gas	Dairy farming	200 MJ/month/cow	Khakbazan, 2000; Statistics Canada, 1996
	Greenhouse cultivation	1,455 MJ/m^2 greenhouse/year	Average of 1) Luczynski and Portree, 2005 and 2) Bi and Lo, 2001
	Rockwool production	0.13 m^3/kg produced	Nielsen et al, 2003
	Boiler combustion	1.04 MJ/MJ heat produced	Swiss Centre for Life Cycle Inventories, 2010
	Electricity generation from steam turbine	2.44 MJ/MJ electricity produced	GHGenius v 4.01
	Electricity generation from gas turbine	2.22 MJ/MJ electricity produced	GHGenius v 4.01
Biogas	Boiler combustion	0.05 m^3 upgraded biogas/ MJ heat produced	BC Hydro, 2012; Börjesson and Berglund, 2007
Diesel	HDV transport	2.09 MJ/tkm	GHGenius v 4.01
	Slurry and liquid digestate land spreading	0.217 kg/m^3 slurry	Swiss Centre for Life Cycle Inventories, 2010
	Sawdust production from sawmill operation	26.8 MJ/tonne produced	Pa, 2010
	Electricity generation	2.5 MJ/MJ electricity produced	GHGenius v 4.01
Hydro	Electricity generation	1.05 MJ/MJ electricity produced	Swiss Centre for Life Cycle Inventories, 2010
Biomass[1]	Electricity generation	3.57 MJ/MJ electricity produced	GHGenius v 4.01
Electricity	Livestock manure storage	0.375 kWh/m^3 manure	Swiss Centre for Life Cycle Inventories, 2010
	Anaerobic digestion	0.158 kWh/m^3 biogas generated	Stucki et al., 2011
	Digestate separation	3.58 GJ/ tonne digestate	Pöchl et al., 2010
	Gas combustion in boiler	2.78 kWh/MJ heat generated	Swiss Centre for Life Cycle Inventories, 2010
	Sawdust production from sawmill operation	116 MJ/ tonne produced	Pa, 2010
	Urea ammonium nitrate produced	0.157 kWh/ kg produced	Swiss Centre for Life Cycle Inventories, 2010

[1]The biomass used to generate electricity is assumed to be mixture of softwood and hardwood waste.

Table A-2 Data sources for energy consumptions

Energy type	Primary energy requirement	Upstream emission	Downstream emission
Natural gas for industry	Swiss Centre for Life Cycle Inventories, 2010	GHGenius v.4.01	GHGenius v.4.01
Natural gas for electricity	Swiss Centre for Life Cycle Inventories, 2010	GHGenius v.4.01	GHGenius v.4.01
Biogas	Swiss Centre for Life Cycle Inventories, 2010 with calculations	Not applicable	Börjesson and Berglund, 2007
Middle distillate (diesel)	GHGenius v.4.01	GHGenius v.4.01	GHGenius v.4.01
Hydro	Swiss Centre for Life Cycle Inventories, 2010	Not applicable	GHGenius v.4.01
Biomass (wood waste) to electricity	Swiss Centre for Life Cycle Inventories, 2010	GHGenius v.4.01	Swiss Centre for Life Cycle Inventories, 2010

Table A-3 Average data and data sources for material consumptions

Material type	Process	Consumption rate	Source	Emission factor source
Sawdust bedding material	Dairy farming	2.8 kg/cow/day	Torenvliet, 2011	Pa, 2010
Rockwool growing media	Greenhouse cultivation	6.5 L/m^2/year	Yakel, 2011	Nielsen et al, 2003
Wheat straw	Mushroom growing	5.4 kg/m^2/month	van Griensven, 1998).	Swiss Centre for Life Cycle Inventories, 2010
N fertilizer	Greenhouse cultivation	7.5 g N/m^2/month	Wong, 2012	Swiss Centre for Life Cycle Inventories, 2010

Table A-4 Data sources for emission factors of the system operations

Process	Source
Manure storage	Dong et al, 2006; Stucki et al, 2011; Atia et al, 1999
Manure spreading	Amon et al, 2006, Börjesson and Berglund, 2007, Stucki et al, 2011, Atia et al, 1999
Natural gas combustion	Swiss Centre for Life Cycle Inventories, 2010
HDV transport	GHGenius v.4.01
Composting	RTI International, 2000; Haaren et al., 2010
Anaerobic digestion	Börjesson and Berglund, 2007
Biogas upgrading	Börjesson and Berglund, 2007
Biogas combustion	Electrigaz Technologies Inc., 2007
Digestate separation and solid phase drying	Börjesson and Berglund, 2006; Stucki et al, 2011
Digestate application[1]	Börjesson and Berglund, 2006; Börjesson and Berglund, 2007; Stucki et al, 2011

[1]Digestate application includes liquid digestate as fertilizer, solid digestate as cow bedding, and solid digestate as growing media.

APPENDIX B – EMISSION FACTORS

Table B-1 Emission factors for electricity generation and distribution, in kg/ MJ of electricity distributed

	Natural gas (boiler)			Natural gas (turbine)			Fuel oil			Hydro			Biomass		
	Up-stream	Down-stream	Total	Up-stream	Down-stream	Total	Up-stream	Down-stream	Total	Up-stream	Down-stream	Total	Up-stream	Down-stream	Total
CO_2 fossil	1.25E-02	1.33E-01	1.46E-01	1.14E-02	1.21E-01	1.32E-01	5.12E-02	1.93E-01	2.44E-01	-	1.51E-03	1.51E-03	1.15E-03	-	1.15E-03
CH_4 fossil	1.18E-04	2.60E-06	1.21E-04	1.08E-04	8.93E-06	1.17E-04	3.89E-04	2.18E-06	3.91E-04	-	9.05E-05	9.05E-05	1.27E-06	-	1.27E-06
CH_4 biogenic	-	-	-	-	-	-	-	-	-	-	-	-	1.76E-09	3.50E-05	3.50E-05
N_2O	3.01E-07	9.40E-08	3.95E-07	2.74E-07	3.11E-06	3.39E-06	1.51E-06	3.60E-08	1.55E-06	-	-	-	1.12E-07	1.55E-05	1.56E-05
NOx	5.96E-05	1.92E-04	2.51E-04	5.43E-05	2.02E-04	2.56E-04	1.31E-04	2.13E-04	3.44E-04	-	-	-	1.16E-05	3.67E-04	3.78E-04
SOx	1.90E-05	1.41E-06	2.04E-05	1.73E-05	1.27E-06	1.86E-05	1.50E-04	6.26E-04	7.76E-04	-	-	-	2.25E-06	3.59E-04	3.62E-04
NH_3	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
H_2S	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
NMVOC	4.53E-06	9.82E-06	1.43E-05	4.12E-06	1.58E-06	5.70E-06	1.31E-04	2.13E-04	3.44E-04	-	-	-	1.92E-05	3.00E-05	4.92E-05
PM_{10}	-	8.58E-06	8.58E-06	-	6.85E-06	6.85E-06	-	2.65E-05	2.65E-05	-	-	-	-	1.20E-04	1.20E-04
$PM_{2.5}$	-	8.58E-06	8.58E-06	-	6.85E-06	6.85E-06	-	1.94E-05	1.94E-05	-	-	-	-	1.03E-04	1.03E-04
Other PM	1.16E-06	8.58E-06	9.74E-06	1.06E-06	6.85E-06	7.91E-06	9.68E-06	3.75E-05	4.72E-05	-	-	-	1.21E-06	1.33E-04	1.35E-04

Table B-2: Emission factors for key material manufacturing

	Sawdust bedding material	Rockwool growing media	N fertilizer
	kg/kg sawdust produced	kg/kg rockwool produced	kg/kg N produced
CO_2 fossil	5.59E-05	1.30E+00	2.48E+00
CH_4 fossil	5.59E-05	7.39E-03	5.14E-05
CH_4 biogenic	1.59E-06	-	8.56E-06

	Sawdust bedding material	Rockwool growing media	N fertilizer
	kg/kg sawdust produced	kg/kg rockwool produced	kg/kg N produced
N_2O	2.27E-06	2.27E-05	9.51E-03
NOx	9.35E-05	2.53E-03	8.49E-03
SOx	2.55E-05	5.69E-03	3.98E-06
NH_3	-	2.00E-03	3.12E-03
H_2S	-	2.33E-05	2.26E-05
NMVOC	1.16E-05	6.71E-04	8.45E-04
PM_{10}	-	2.61E-05	1.22E-06
$PM_{2.5}$	3.17E-07	-	1.17E-03
Other PM	4.25E-05	1.01E-03	1.35E-06

Table B-3: Emission Factors for system operations-1 (Manure management)

	Dairy manure storage	Swine manure storage	Poultry manure storage	Dairy manure spreading	Swine manure spreading	Poultry manure spreading
	kg/tonne stored	kg/tonne stored	kg/tonne stored	kg/tonne spread	kg/tonne spread	kg/tonne spread
CO_2 fossil	1.20E-05	1.20E-05	1.20E-05	1.99E-04	1.99E-04	1.99E-04
CH_4 fossil	1.24E-07	1.24E-07	1.25E-07	1.51E-06	1.51E-06	1.51E-06
CH_4 biogenic	1.80E-03	6.04E-03	1.56E-02	1.31E-06	1.45E-06	5.50E-06
N_2O	2.60E-05	3.10E-05	6.50E-05	4.00E-05	4.77E-05	1.00E-04
NOx	2.56E-08	2.56E-08	2.58E-08	1.55E-05	1.55E-05	1.55E-05
SOx	9.60E-09	9.60E-09	9.65E-09	5.85E-07	5.85E-07	5.85E-07
NH_3	9.27E-05	3.48E-04	3.33E-04	4.66E-04	1.75E-03	1.67E-03
H_2S	2.20E-04	2.20E-04	2.20E-04	1.64E-04	1.64E-04	1.64E-04
NMVOC	1.72E-09	1.72E-09	1.73E-09	1.07E-07	1.07E-07	1.07E-07
PM_{10}	2.95E-09	2.95E-09	2.96E-09	-	-	-

	Dairy manure storage	Swine manure storage	Poultry manure storage	Dairy manure spreading	Swine manure spreading	Poultry manure spreading
	kg/tonne stored	kg/tonne stored	kg/tonne stored	kg/tonne spread	kg/tonne spread	kg/tonne spread
$PM_{2.5}$	2.61E-09	2.61E-09	2.62E-09	-	-	-
Other PM	3.34E-09	3.34E-09	3.36E-09	2.68E-07	2.68E-07	2.68E-07

Table B-4: Emission Factors for system operations-2

	Natural gas combustion	HDV transport	Composting of yard waste	Composting of food waste	Anaerobic digestion	Biogas upgrading	Biogas combustion	Digestate separation and solid phase drying	Digestate application
	kg/MJ heat produced	kg/tkm	kg/tonne feedstock	kg/tonne feedstock	kg/m3 biogas generated	kg/m3 biogas to be upgraded	kg/m3 biogas combusted	kg/tonne digestate	kg/tonne digestate
CO_2 fossil	5.83E-02	1.88E-01	8.10E-03	2.62E-02	4.99E-03	2.98E-03	1.93E-03	3.14E-05	-
CH_4 fossil	2.08E-06	2.80E-04	-	-	5.17E-05	4.05E-06	2.00E-05	3.26E-07	-
CH_4 biogenic	5.29E-09	-	1.11E-06	4.99E-05	3.00E-03	1.20E-03	1.12E-07	2.22E-09	8.00E-09
N_2O	5.23E-07	8.09E-06	-	-	1.85E-07	1.03E-07	7.14E-08	1.00E-06	2.00E-05
NOx	1.53E-05	1.63E-04	1.93E-04	1.99E-04	1.07E-05	1.49E-05	1.54E-03	6.67E-07	1.20E-05
SOx	7.32E-08	8.21E-05	3.46E-05	1.58E-04	4.00E-06	1.04E-05	1.55E-06	2.52E-08	2.40E-07
NH_3	-	-	2.24E-03	1.07E-05	1.00E-06	2.00E-04	4.84E-05	2.61E-05	7.31E-04
H_2S	-	-	-	-	5.00E-06	2.00E-06	-	-	-
NMVOC	1.31E-08	3.63E-05	-	-	7.16E-07	2.28E-06	2.77E-07	4.51E-09	-
PM_{10}	2.25E-08	-	-	-	1.23E-06	-	4.76E-07	7.74E-09	-
$PM_{2.5}$	1.24E-07	-	-	-	1.09E-06	2.38E-06	4.21E-07	6.85E-09	-
Other PM	2.55E-08	1.48E-05	2.00E-05	3.94E-05	1.39E-06	2.79E-05	1.17E-04	1.80E-08	1.84E-07

APPENDIX C – NUMERICAL VALUES FOR LCIA FIGURES IN CHAPTER 2

Table C-1: Stage-wise non-renewable energy consumptions for three scenarios of Case 1, unit: GJ/month

	Spring			Summer			Fall			Winter		
	1-a	1-b	1-c	1-a	1-b	1-c	1-a	1-b	1-c	1-a	1-b	1-c
Composting of digestate surplus	-	-	0	-	-	0	-	-	0	-	-	0
Replaced urea ammonium nitrate production	-	-59	-	-	-59	-	-	-59	-	-	-59	-
Replaced sawdust production	-	-36	-	-	-36	-	-	-36	-	-	-36	-
Other digestate application	-	0	0	-	0	0	-	0	0	-	0	0
Liquid digestate spreading on land	-	7	7	-	7	7	-	7	7	-	7	7
Digestate separation and solid phase drying	-	0.6	0.6	-	0.6	0.6	-	0.6	0.6	-	0.6	0.6
Combustion of biogas	-	1.3	1.3	-	1.3	1.3	-	1.3	1.3	-	1.3	1.3
Biogas upgrading	-	2	2	-	2	2	-	2	2	-	2	2
Anaerobic digestion	-	3	3	-	3	3	-	3	3	-	3	3
HDV transport	1.2	-	-	1.2	-	-	1.2	-	-	1.2	-	-
Rockwool production	15	-	-	15	-	-	15	-	-	15	-	-
Urea ammonium nitrate production	2	-	-	2	-	-	2	-	-	2	-	-
Natural gas combustion	2	0.6	0.6	2	0	0	2	0.3	0.3	2	1.0	1.0
Natural gas upstream	1,216	426	426	1,622	0	0	1,216	183	183	1,369	734	734
Manure spreading	14	-	-	14	-	-	14	-	-	14	-	-
Manure storage	0.2	-	-	0.2	-	-	0.2	-	-	0.2	-	-
Sawdust production	21	-	-	21	-	-	21	-	-	21	-	-
Total	1,270	346	441	1,677	-81	14	1,270	102	198	1,423	654	749

Table C-2: Stage-wise climate change impact for three scenarios of Case 1, unit: tonne CO_2-eq/month

	Spring			Summer			Fall			Winter		
	1-a	1-b	1-c	1-a	1-b	1-c	1-a	1-b	1-c	1-a	1-b	1-c
Composting of digestate surplus	-	-	0.6	-	-	0.6	-	-	0.6	-	-	0.6
Replaced urea ammonium nitrate production	-	-6	-	-	-6	-	-	-6	-	-	-6	-
Replaced sawdust production	-	-2	-	-	-2	-	-	-2	-	-	-2	-
Other digestate application	-	0.3	0.3	-	0.3	0.3	-	0.3	0.3	-	0.3	0.3
Liquid digestate spreading on land	-	4	4	-	4	4	-	4	4	-	4	4
Digestate separation and solid phase drying	-	0.4	0.4	-	0.4	0.4	-	0.4	0.4	-	0.4	0.4
Combustion of biogas	-	0.1	0.1	-	0.1	0.1	-	0.1	0.1	-	0.1	0.1
Biogas upgrading	-	1.1	1.1	-	1.1	1.1	-	1.1	1.1	-	1.1	1.1
Anaerobic digestion	-	3	3	-	3	3	-	3	3	-	3	3
HDV transport	0.1	-	-	0.1	-	-	0.1	-	-	0.1	-	-
Rockwool production	0.8	-	-	0.8	-	-	0.8	-	-	0.8	-	-
Urea ammonium nitrate production	0.2	-	-	0.2	-	-	0.2	-	-	0.2	-	-
Natural gas combustion	58	20	20	77	0	0	58	9	9	65	35	35
Natural gas upstream	7	2	2	9	0	0	7	1	1	8	4	4
Manure spreading	13	-	-	13	-	-	13	-	-	13	-	-
Manure storage	52	-	-	52	-	-	52	-	-	52	-	-
Sawdust production	1.4	-	-	1.4	-	-	1.4	-	-	1.4	-	-
Total	133	22	31	154	0	9	133	9	18	141	39	48

Table C-3: Stage-wise aquatic acidification impact for three scenarios of Case 1, unit: kg SO_2-eq/month

	Spring			Summer			Fall			Winter		
	1-a	1-b	1-c	1-a	1-b	1-c	1-a	1-b	1-c	1-a	1-b	1-c
Composting of digestate surplus	-	-	268	-	-	268	-	-	268	-	-	268
Replaced urea ammonium nitrate production	-	-16	-	-	-16	-	-	-16	-	-	-16	-

	Spring			Summer			Fall			Winter		
	1-a	1-b	1-c	1-a	1-b	1-c	1-a	1-b	1-c	1-a	1-b	1-c
Replaced sawdust production	-	-6	-	-	-6	-	-	-6	-	-	-6	-
Other digestate application	-	58	58	-	58	58	-	58	58	-	58	58
Liquid digestate spreading on land	-	684	684	-	684	684	-	684	684	-	684	684
Digestate separation and solid phase drying	-	44	44	-	44	44	-	44	44	-	44	44
Combustion of biogas	-	30	30	-	30	30	-	30	30	-	30	30
Biogas upgrading	-	13	13	-	13	13	-	13	13	-	13	13
Anaerobic digestion	-	0.8	0.8	-	0.8	0.8	-	0.8	0.8	-	0.8	0.8
HDV transport	0.1	-	-	0.1	-	-	0.1	-	-	0.1	-	-
Rockwool production	6	-	-	6	-	-	6	-	-	6	-	-
Urea ammonium nitrate production	0.6	-	-	0.6	-	-	0.6	-	-	0.6	-	-
Natural gas combustion	8	3	3	11	0	0	8	1.3	1.3	9	5	5
Natural gas upstream	21	8	8	29	0	0	21	3	3	24	13	13
Manure spreading	1,169	-	-	1,169	-	-	1,169	-	-	1,169	-	-
Manure storage	618	-	-	618	-	-	618	-	-	618	-	-
Sawdust production	3	-	-	3	-	-	3	-	-	3	-	-
Total	1,827	820	1,108	1,837	809	1,098	1,827	814	1,103	1,831	827	1,116

Table C-4: Stage-wise aquatic eutrophication impact for three scenarios of Case 1, unit: kg PO_4-eq/month

	Spring			Summer			Fall			Winter		
	1-a	1-b	1-c	1-a	1-b	1-c	1-a	1-b	1-c	1-a	1-b	1-c
Composting of digestate surplus	-	-	59	-	-	59	-	-	59	-	-	59
Replaced urea ammonium nitrate production	-	-3	-	-	-3	-	-	-3	-	-	-3	-
Replaced sawdust production	-	-0.9	-	-	-0.9	-	-	-0.9	-	-	-0.9	-
Other digestate application	-	13	13	-	13	13	-	13	13	-	13	13
Liquid digestate spreading on land	-	150	150	-	150	150	-	150	150	-	150	150

	Spring			Summer			Fall			Winter		
	1-a	1-b	1-c	1-a	1-b	1-c	1-a	1-b	1-c	1-a	1-b	1-c
Digestate separation and solid phase drying	-	10	10	-	10	10	-	10	10	-	10	10
Combustion of biogas	-	8	8	-	8	8	-	8	8	-	8	8
Biogas upgrading	-	3	3	-	3	3	-	3	3	-	3	3
Anaerobic digestion	-	0.1	0.1	-	0.1	0.1	-	0.1	0.1	-	0.1	0.1
HDV transport	0	-	-	0	-	-	0	-	-	0	-	-
Rockwool production	0.6	-	-	0.6	-	-	0.6	-	-	0.6	-	-
Urea ammonium nitrate production	0.1	-	-	0.1	-	-	0.1	-	-	0.1	-	-
Natural gas combustion	2	0.7	0.7	3	0	0	2	0.3	0.3	2	1.2	1.2
Natural gas upstream	3	1.1	1.1	4	0	0	3	0.5	0.5	4	1.9	1.9
Manure spreading	182	-	-	182	-	-	182	-	-	182	-	-
Manure storage	36	-	-	36	-	-	36	-	-	36	-	-
Sawdust production	0.5	-	-	0.5	-	-	0.5	-	-	0.5	-	-
Total	224	181	243	225	179	242	224	180	242	224	182	245

Table C-5: Stage-wise inorganic respiratory effect for three scenarios of Case 1, unit: kg $PM_{2.5}$-eq/month

	Spring			Summer			Fall			Winter		
	1-a	1-b	1-c	1-a	1-b	1-c	1-a	1-b	1-c	1-a	1-b	1-c
Composting of digestate surplus	-	-	22	-	-	22	-	-	22	-	-	22
Replaced urea ammonium nitrate production	-	-3	-	-	-3	-	-	-3	-	-	-3	-
Replaced sawdust production	-	-2	-	-	-2	-	-	-2	-	-	-2	-
Other digestate application	-	4	4	-	4	4	-	4	4	-	4	4
Liquid digestate spreading on land	-	53	53	-	53	53	-	53	53	-	53	53
Digestate separation and solid phase drying	-	3	3	-	3	3	-	3	3	-	3	3
Combustion of biogas	-	8	8	-	8	8	-	8	8	-	8	8
Biogas upgrading	-	1.1	1.1	-	1.1	1.1	-	1.1	1.1	-	1.1	1.1

	Spring			Summer			Fall			Winter		
	1-a	1-b	1-c	1-a	1-b	1-c	1-a	1-b	1-c	1-a	1-b	1-c
Anaerobic digestion	-	0	0	-	0	0	-	0	0	-	0	0
HDV transport	0	-	-	0	-	-	0	-	-	0	-	-
Rockwool production	0.7	-	-	0.7	-	-	0.7	-	-	0.7	-	-
Urea ammonium nitrate production	0.1	-	-	0.1	-	-	0.1	-	-	0.1	-	-
Natural gas combustion	2	0.7	0.7	3	0	0	2	0.3	0.3	2	1.3	1.3
Natural gas upstream	4	1.3	1.3	5	0	0	4	0.6	0.6	4	2	2
Manure spreading	65	-	-	65	-	-	65	-	-	65	-	-
Manure storage	12	-	-	12	-	-	12	-	-	12	-	-
Sawdust production	0.9	-	-	0.9	-	-	0.9	-	-	0.9	-	-
Total	84	67	93	86	65	91	84	66	92	85	68	95

Table C-6: Stage-wise organic respiratory effect for three scenarios of Case 1, unit: kg C_2H_4-eq/month

	Spring			Summer			Fall			Winter		
	1-a	1-b	1-c	1-a	1-b	1-c	1-a	1-b	1-c	1-a	1-b	1-c
Composting of digestate surplus	-	-	0	-	-	0	-	-	0	-	-	0
Replaced urea ammonium nitrate production	-	-0.71	-	-	-0.71	-	-	-0.71	-	-	-0.71	-
Replaced sawdust production	-	-0.52	-	-	-0.52	-	-	-0.52	-	-	-0.52	-
Other digestate application	-	0	0	-	0	0	-	0	0	-	0	0
Liquid digestate spreading on land	-	0.04	0.04	-	0.04	0.04	-	0.04	0.04	-	0.04	0.04
Digestate separation and solid phase drying	-	0.00	0.00	-	0.00	0.00	-	0.00	0.00	-	0.00	0.00
Combustion of biogas	-	0.01	0.01	-	0.01	0.01	-	0.01	0.01	-	0.01	0.01
Biogas upgrading	-	0.06	0.06	-	0.06	0.06	-	0.06	0.06	-	0.06	0.06
Anaerobic digestion	-	0.03	0.03	-	0.03	0.03	-	0.03	0.03	-	0.03	0.03
HDV transport	0.01	-	-	0.01	-	-	0.01	-	-	0.01	-	-
Rockwool production	0.36	-	-	0.36	-	-	0.36	-	-	0.36	-	-

	Spring			Summer			Fall			Winter		
	1-a	1-b	1-c	1-a	1-b	1-c	1-a	1-b	1-c	1-a	1-b	1-c
Urea ammonium nitrate production	0.03	-	-	0.03	-	-	0.03	-	-	0.03	-	-
Natural gas combustion	1.10	0.39	0.39	1.47	0	0	1.10	0.17	0.17	1.24	0.66	0.66
Natural gas upstream	1.62	0.57	0.57	2.16	0	0	1.62	0.24	0.24	1.82	0.98	0.98
Manure spreading	0.08	-	-	0.08	-	-	0.08	-	-	0.08	-	-
Manure storage	0.00	-	-	0.00	-	-	0.00	-	-	0.00	-	-
Sawdust production	0.31	-	-	0.31	-	-	0.31	-	-	0.31	-	-
Total	3.50	-0.13	1.09	4.41	-1.09	0.14	3.50	-0.68	0.55	3.85	0.55	1.78

Table C-7: Stage-wise human toxicity impact for three scenarios of Case 1, unit: kg C_2H_3Cl-eq/month

	Spring			Summer			Fall			Winter		
	1-a	1-b	1-c	1-a	1-b	1-c	1-a	1-b	1-c	1-a	1-b	1-c
Composting of digestate surplus	0	0	8	0	0	8	0	0	8	0	0	8
Replaced urea ammonium nitrate production	0	-48	0	0	-48	0	0	-48	0	0	-48	0
Replaced sawdust production	0	0	0	0	0	0	0	0	0	0	0	0
Other digestate application	0	2	2	0	2	2	0	2	2	0	2	2
Liquid digestate spreading on land	0	22	22	0	22	22	0	22	22	0	22	22
Digestate separation and solid phase drying	0	1.4	1.4	0	1.4	1.4	0	1.4	1.4	0	1.4	1.4
Combustion of biogas	0	0	0	0	0	0	0	0	0	0	0	0
Biogas upgrading	0	5	5	0	5	5	0	5	5	0	5	5
Anaerobic digestion	0	0	0	0	0	0	0	0	0	0	0	0
HDV transport	0	0	0	0	0	0	0	0	0	0	0	0
Rockwool production	25	0	0	25	0	0	25	0	0	25	0	0
Urea ammonium nitrate production	2	0	0	2	0	0	2	0	0	2	0	0
Natural gas combustion	41	14	14	54	0	0	41	6	6	46	24	24
Natural gas upstream	0	0.0	0.0	0	0	0	0	0.0	0.0	0	0.0	0.0

	Spring			Summer			Fall			Winter		
	1-a	1-b	1-c	1-a	1-b	1-c	1-a	1-b	1-c	1-a	1-b	1-c
Manure spreading	81	0	0	81	0	0	81	0	0	81	0	0
Manure storage	79	0	0	79	0	0	79	0	0	79	0	0
Sawdust production	0	0	0	0	0	0	0	0	0	0	0	0
Total	227	-4	53	241	-18	39	227	-12	45	233	6	63

APPENDIX D – NUMERICAL VALUES FOR LCIA FIGURES IN CHAPTER 4

Table D-1: Stage-wise non-renewable energy consumptions for three scenarios of Case 7, unit: GJ/month

	Spring			Summer			Fall			Winter		
	7-a	7-b	7-c	7-a	7-b	7-c	7-a	7-b	7-c	7-a	7-b	7-c
Composting of digestate surplus	-	-	0	-	-	0	-	-	0	-	-	0
Replaced urea ammonium nitrate production	-	-57	-	-	-57	-	-	-57	-	-	-57	-
Replaced sawdust production	-	-10	-	-	-10	-	-	-10	-	-	-10	-
Other digestate application	-	0	0	-	0	0	-	0	0	-	0	0
Liquid digestate spreading on land	-	7	7	-	7	7	-	7	7	-	7	7
Digestate separation and solid phase drying	-	0.6	0.6	-	0.6	0.6	-	0.6	0.6	-	0.6	0.6
Combustion of biogas	-	1.3	1.3	-	1.3	1.3	-	1.3	1.3	-	1.3	1.3
Biogas upgrading	-	2	2	-	2	2	-	2	2	-	2	2
Anaerobic digestion	-	3	3	-	3	3	-	3	3	-	3	3
HDV transport	3	-	-	3	-	-	3	-	-	3	-	-
Straw production	28	-	-	28	-	-	28	-	-	28	-	-
Rockwool production	29	-	-	29	-	-	29	-	-	29	-	-
Urea ammonium nitrate production	4	-	-	4	-	-	4	-	-	4	-	-
Natural gas combustion	3	2	2	4	0	0	3	1.5	1.5	4	3	3
Natural gas upstream	2,372	1,574	1,574	3,166	0	0	2,372	1,074	1,074	3,152	2,278	2,278
Manure spreading	14	-	-	14	-	-	14	-	-	14	-	-
Manure storage	0.2	-	-	0.2	-	-	0.2	-	-	0.2	-	-
Sawdust production	21	-	-	21	-	-	21	-	-	21	-	-
Total	2,474	1,523	1,590	3,268	-52	14	2,474	1,023	1,089	3,255	2,229	2,296

Table D-2: Stage-wise climate change impact for three scenarios of Case 7, unit: tonne CO_2-eq/month

	Spring			Summer			Fall			Winter		
	7-a	7-b	7-c	7-a	7-b	7-c	7-a	7-b	7-c	7-a	7-b	7-c
Composting of digestate surplus	-	-	0.2	-	-	0.2	-	-	0.2	-	-	0.2
Replaced urea ammonium nitrate production	-	-6	-	-	-6	-	-	-6	0	-	-6	-
Replaced sawdust production	-	-0.6	-	-	-0.6	-	-	-0.6	0	-	-0.6	-
Other digestate application	-	0.6	0.6	-	0.6	0.6	-	0.6	0.6	-	0.6	0.6
Liquid digestate spreading on land	-	4	4	-	4	4	-	4	4	-	4	4
Digestate separation and solid phase drying	-	0.4	0.4	-	0.4	0.4	-	0.4	0.4	-	0.4	0.4
Combustion of biogas	-	0.1	0.1	-	0.1	0.1	-	0.1	0.1	-	0.1	0.1
Biogas upgrading	-	1.1	1.1	-	1.1	1.1	-	1.1	1.1	-	1.1	1.1
Anaerobic digestion	-	3	3	-	3	3	-	3	3	-	3	3
HDV transport	0.2	-	-	0.2	-	-	0.2	-	-	0.2	-	-
Straw production	4	-	-	4	-	-	4	-	-	4	-	-
Rockwool production	2	-	-	2	-	-	2	-	-	2	-	-
Urea ammonium nitrate production	0.4	-	-	0.4	-	-	0.4	-	-	0.4	-	-
Natural gas combustion	113	75	75	151	0	0	113	51	51	150	108	108
Natural gas upstream	13	9	9	18	0	0	13	6	6	18	13	13
Manure spreading	13	-	-	13	-	-	13	-	-	13	-	-
Manure storage	52	-	-	52	-	-	52	-	-	52	-	-
Sawdust production	1.4	-	-	1.4	-	-	1.4	-	-	1.4	-	-
Total	200	86	92	242	2	9	200	59	66	241	123	130

Table D-3: Stage-wise aquatic acidification impact for three scenarios of Case 7, unit: kg SO_2-eq/month

	Spring			Summer			Fall			Winter		
	7-a	7-b	7-c	7-a	7-b	7-c	7-a	7-b	7-c	7-a	7-b	7-c
Composting of digestate surplus	-	-	71	-	-	71	-	-	71	-	-	71
Replaced urea ammonium nitrate production	-	-15	0	-	-15	0	-	-15	0	-	-15	0
Replaced sawdust production	-	-1.5	0	-	-1.5	0	-	-1.5	0	-	-1.5	0
Other digestate application	-	128	128	-	128	128	-	128	128	-	128	128
Liquid digestate spreading on land	-	684	684	-	684	684	-	684	684	-	684	684
Digestate separation and solid phase drying	-	44	44	-	44	44	-	44	44	-	44	44
Combustion of biogas	-	30	30	-	30	30	-	30	30	-	30	30
Biogas upgrading	-	13	13	-	13	13	-	13	13	-	13	13
Anaerobic digestion	-	0.8	0.8	-	0.8	0.8	-	0.8	0.8	-	0.8	0.8
HDV transport	0.2	-	-	0.2	-	-	0.2	-	-	0.2	-	-
Straw production	28	-	-	28	-	-	28	-	-	28	-	-
Rockwool production	13	-	-	13	-	-	13	-	-	13	-	-
Urea ammonium nitrate production	1.1	-	-	1.1	-	-	1.1	-	-	1.1	-	-
Natural gas combustion	16	11	11	22	0	0	16	7	7	22	16	16
Natural gas upstream	42	28	28	56	0	0	42	19	19	55	40	40
Manure spreading	1,169	-	-	1,169	-	-	1,169	-	-	1,169	-	-
Manure storage	618	-	-	618	-	-	618	-	-	618	-	-
Sawdust production	3	-	-	3	-	-	3	-	-	3	-	-
Total	1,890	923	1010	1,909	884	971	1,890	910	998	1,909	940	1,027

Table D-4: Stage-wise aquatic eutrophication impact for three scenarios of Case 7, unit: kg PO_4-eq/month

	Spring			Summer			Fall			Winter		
	7-a	7-b	7-c	7-a	7-b	7-c	7-a	7-b	7-c	7-a	7-b	7-c
Composting of digestate surplus	-	-	16	-	-	16	-	-	16	-	-	16
Replaced urea ammonium nitrate production	-	-3	-	-	-3	-	-	-3	-	-	-3	-
Replaced sawdust production	-	-0.2	-	-	-0.2	-	-	-0.2	-	-	-0.2	-
Other digestate application	-	28	28	-	28	28	-	28	28	-	28	28
Liquid digestate spreading on land	-	150	150	-	150	150	-	150	150	-	150	150
Digestate separation and solid phase drying	-	10	10	-	10	10	-	10	10	-	10	10
Combustion of biogas	-	8	8	-	8	8	-	8	8	-	8	8
Biogas upgrading	-	3	3	-	3	3	-	3	3	-	3	3
Anaerobic digestion	-	0.1	0.1	-	0.1	0.1	-	0.1	0.1	-	0.1	0.1
HDV transport	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-
Straw production	44	-	-	44	-	-	44	-	-	44	-	-
Rockwool production	1.2	-	-	1.2	-	-	1.2	-	-	1.2	-	-
Urea ammonium nitrate production	0.2	-	-	0.2	-	-	0.2	-	-	0.2	-	-
Natural gas combustion	4	3	3	5	0	0	4	2	2	5	4	4
Natural gas upstream	6	4	4	8	0	0	6	3	3	8	6	6
Manure spreading	182	-	-	182	-	-	182	-	-	182	-	-
Manure storage	36	-	-	36	-	-	36	-	-	36	-	-
Sawdust production	0.5	-	-	0.5	-	-	0.5	-	-	0.5	-	-
Total	273	202	220	276	195	214	273	200	218	276	205	223

Table D-5: Stage-wise inorganic respiratory effect for three scenarios of Case 7, unit: kg $PM_{2.5}$-eq/month

	Spring			Summer			Fall			Winter		
	7-a	7-b	7-c	7-a	7-b	7-c	7-a	7-b	7-c	7-a	7-b	7-c
Composting of digestate surplus	-	-	6	-	-	6	-	-	6	-	-	6
Replaced urea ammonium nitrate production	-	-3	-	-	-3	-	-	-3	-	-	-3	-
Replaced sawdust production	-	-0.4	-	-	-0.4	-	-	-0.4	-	-	-0.4	-
Other digestate application	-	10	10	-	10	10	-	10	10	-	10	10
Liquid digestate spreading on land	-	53	53	-	53	53	-	53	53	-	53	53
Digestate separation and solid phase drying	-	3	3	-	3	3	-	3	3	-	3	3
Combustion of biogas	-	8	8	-	8	8	-	8	8	-	8	8
Biogas upgrading	-	1.1	1.1	-	1.1	1.1	-	1.1	1.1	-	1.1	1.1
Anaerobic digestion	-	0.1	0.1	-	0.1	0.1	-	0.1	0.1	-	0.1	0.1
HDV transport	0.0	-	-	0.0	-	-	0.0	-	-	0.0	-	-
Straw production	4	-	-	4	-	-	4	-	-	4	-	-
Rockwool production	1.4	-	-	1.4	-	-	1.4	-	-	1.4	-	-
Urea ammonium nitrate production	0.2	-	-	0.2	-	-	0.2	-	-	0.2	-	-
Natural gas combustion	4	3	3	5	0	0	4	2	2	5	4	4
Natural gas upstream	7	5	5	10	0	0	7	3	3	10	7	7
Manure spreading	65	-	-	65	-	-	65	-	-	65	-	-
Manure storage	12	-	-	12	-	-	12	-	-	12	-	-
Sawdust production	0.9	-	-	0.9	-	-	0.9	-	-	0.9	-	-
Total	95	79	88	99	71	81	95	77	86	99	82	92

Table D-6: Stage-wise organic respiratory effect for three scenarios of Case 7, unit: kg C_2H_4-eq/month

	Spring			Summer			Fall			Winter		
	7-a	7-b	7-c	7-a	7-b	7-c	7-a	7-b	7-c	7-a	7-b	7-c
Composting of digestate surplus	-	-	0	-	-	0	-	-	0	-	-	0
Replaced urea ammonium nitrate production	-	-0.68	-	-	-0.68	-	-	-0.68	-	-	-0.68	-
Replaced sawdust production	-	-0.14	-	-	-0.14	-	-	-0.14	-	-	-0.14	-
Other digestate application	-	0	0	-	0	0	-	0	0	-	0	0
Liquid digestate spreading on land	-	0.04	0.04	-	0.04	0.04	-	0.04	0.04	-	0.04	0.04
Digestate separation and solid phase drying	-	0.00	0.00	-	0.00	0.00	-	0.00	0.00	-	0.00	0.00
Combustion of biogas	-	0.01	0.01	-	0.01	0.01	-	0.01	0.01	-	0.01	0.01
Biogas upgrading	-	0.06	0.06	-	0.06	0.06	-	0.06	0.06	-	0.06	0.06
Anaerobic digestion	-	0.03	0.03	-	0.03	0.03	-	0.03	0.03	-	0.03	0.03
HDV transport	0.02	-	-	0.02	-	-	0.02	-	-	0.02	-	-
Straw production	0.96	-	-	0.96	-	-	0.96	-	-	0.96	-	-
Rockwool production	0.71	-	-	0.71	-	-	0.71	-	-	0.71	-	-
Urea ammonium nitrate production	0.05	-	-	0.05	-	-	0.05	-	-	0.05	-	-
Natural gas combustion	2.15	1.42	1.42	2.86	0.00	0.00	2.15	0.97	0.97	2.85	2.06	2.06
Natural gas upstream	3.16	2.10	2.10	4.22	0.00	0.00	3.16	1.43	1.43	4.20	3.04	3.04
Manure spreading	0.08	-	-	0.08	-	-	0.08	-	-	0.08	-	-
Manure storage	0.00	-	-	0.00	-	-	0.00	-	-	0.00	-	-
Sawdust production	0.31	-	-	0.31	-	-	0.31	-	-	0.31	-	-
Total	7.44	2.84	3.66	9.22	-0.68	0.14	7.44	1.72	2.54	9.19	4.42	5.24

Table D-7: Stage-wise human toxicity impact for three scenarios of Case 7, unit: kg C_2H_3Cl-eq/month

	Spring			Summer			Fall			Winter		
	7-a	7-b	7-c	7-a	7-b	7-c	7-a	7-b	7-c	7-a	7-b	7-c
Composting of digestate surplus	-	-	2	-	-	2	-	-	2	-	-	2
Replaced urea ammonium nitrate production	-	-47	-	-	-47	-	-	-47	-	-	-47	-
Replaced sawdust production	-	0	-	-	0	-	-	0	-	-	0	-
Other digestate application	-	4	4	-	4	4	-	4	4	-	4	4
Liquid digestate spreading on land	-	22	22	-	22	22	-	22	22	-	22	22
Digestate separation and solid phase drying	-	1.4	1.4	-	1.4	1.4	-	1.4	1.4	-	1.4	1.4
Combustion of biogas	-	0.1	0.1	-	0.1	0.1	-	0.1	0.1	-	0.1	0.1
Biogas upgrading	-	5	5	-	5	5	-	5	5	-	5	5
Anaerobic digestion	-	0.1	0.1	-	0.1	0.1	-	0.1	0.1	-	0.1	0.1
HDV transport	0	-	-	0	-	-	0	-	-	0	-	-
Straw production	575	-	-	575	-	-	575	-	-	575	-	-
Rockwool production	50	-	-	50	-	-	50	-	-	50	-	-
Urea ammonium nitrate production	3	-	-	3	-	-	3	-	-	3	-	-
Natural gas combustion	79	52	52	105	0	0	79	36	36	105	76	76
Natural gas upstream	0.0	0.0	0.0	0.0	0	0	0.0	0.0	0.0	0.0	0.0	0.0
Manure spreading	81	-	-	81	-	-	81	-	-	81	-	-
Manure storage	79	-	-	79	-	-	79	-	-	79	-	-
Sawdust production	0	-	-	0	-	-	0	-	-	0	-	-
Total	867	38	87	894	-14	35	867	22	71	893	62	111

Life Cycle Costing of an integrated animal farm-greenhouse Eco-Industrial system in British Columbia

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Abstract:

Anaerobic digestion (AD) is a promising technology for converting livestock manure into methane-rich biogas, a renewable energy. The fast expanding greenhouse industry has been relying on natural gas for heating and CO_2 enrichment which has a high carbon footprint and is vulnerable to price fluctuations. We propose to integrate animal farms and greenhouses by replacing natural gas for greenhouse boilers with the AD biogas. Meanwhile, the residues from digesters are used as animal beddings or greenhouse growing media and liquid fertilizers, respectively. A preliminary Life Cycle Impact Assessment (LCIA) study showed that the integrated Eco-Industrial system has much less impact than the conventional farm practice in almost all impact categories especially on non-renewable energy consumption and global warming. In this study, a Life Cycle Costing (LCC) study was conducted to evaluate the economic feasibility of adopting such a system. Results showed that an integrated system could be cost-effective if appropriately designed and planned. Sustained clean energy promotion and market incentives like carbon offsets will further enhance the feasibility.

Key words: LCC, economics, animal farm, greenhouse, industrial ecology, energy integration

1. Introduction

Manure is the major by-product of the livestock industry. Anaerobic digestion (AD) is a promising technology taking advantage of microorganisms to degrade organic matters in manure into methane-rich biogas, a renewable energy source. Its potential to significantly reduce emissions and to generate renewable energy attracts favorable attention in British Columbia (BC) (ADIAC, 2012). On the other hand, the fast expanding greenhouse industry in BC has been relying on natural gas for heating and CO_2 enrichment, consuming substantial amount of natural gas due to BC's relatively low year-round temperature. As a fossil fuel, natural gas is vulnerable to price rise and its combustion emits massive fossil CO_2 which is a typical greenhouse gas.

There is essentially no waste released from a natural ecosystem because all the substances can be consumed or digested by living creatures or organisms (Odum, 1971). If we manage industrial units in a similar way we can make substantial achievements on waste reduction and sustainable development. This cross-disciplinary concept has been evolved into a discipline called Industrial Ecology (IE), and a designed industrial system with a number of interactive units is called an Eco-Industrial system or Eco-Industrial Park

(Lowe, 2001). We noticed that the biogas generated from AD possesses the potential to replace natural gas for greenhouses uses. Meanwhile, the digestion residues (digestate) can be separated into solid and nutrient-rich liquid, which can be used as organic fibers and liquid fertilizers, respectively. Thus we see the potential to ecologically integrated material and energy flows of animal farms and greenhouses to build a regional Eco-Industrial system to reduce air emissions, enhance sustainability, and protect public health.

To systematically evaluate the proposed animal farm-greenhouse Eco-Industrial system, a LCA was conducted to assess the performance in terms of environmental effects, human health, and economic feasibility. A preliminary LCIA compared the environmental and health impacts over the whole life cycle of a proposed dairy farm-greenhouse Eco-Industrial system to a base case with a conventional livestock manure management system. Results showed that the integrated system has much lower impacts, with non-renewable energy use, global warming, and terrestrial ecotoxicity potentials being reduced by as much as 95%, 77% and 71%, respectively (Zhang and Bi, 2011a, 2011b). However, to bring an environmentally sound project into commercial reality, economic analysis should be considered to inform the industry and market. A LCC study was therefore conducted to evaluate the economic feasibility of an animal farm-greenhouse Eco-Industrial system. The study was performed on a model case with various scenarios.

2. Case set-up

2.1 Animal farm-greenhouse system

The animal farm in this study was modeled as a dairy farm in BC as used in the preliminary LCIA study (Zhang and Bi, 2011a, 2011b). A dairy farm is chosen because BC has a prosperous dairy industry with the production ranked third nationwide. Besides, cow manure is particularly an ideal feedstock for AD with suitable carbon-to-nitrogen ratio, balanced pH, and is rich in anaerobic bacteria (Werner, 2011). Both of the two current pilot on-farm digesters in BC use dairy manure as the main feedstock (ADIAC, 2012). The functional unit in this study is managing a dairy farm with 1000 head of cows. In some scenarios, pig manure and poultry manure from other farms are also included as additional feedstock. The greenhouse analyzed in this study is assumed to be a greenhouse growing peppers in BC, with its size being determined by matching its average energy demand under normal weather conditions in BC to the biogas production rate from the digester on the dairy farm.

2.2 Anaerobic digestion

There are three common AD configuration modes: completely mixed (CSTR), plug flow (PF), and mixed plug flow (MPF). MPF is a new technology designed to compromise between the high efficiency of a PF digester and the stability of a CSTR digester, and is especially popular and efficient as farm-size digesters (Lau, 2010, USEPA, 2011). It was therefore selected as the model reactor in the preliminary LCIA study. But we also modeled one scenario for PF digester in this study. Biogas produced from anaerobic digestion is made up of approximately 60~70% CH_4, 30~40% CO_2, and some trace pollutants like H_2S and N_2O (BC Ministry of Environment, 2010). Raw biogas needs to be upgraded to have the impurities removed. Biogas quality requirements are less strict for boiler combustion compared to electricity generation or chemical processing so a water scrubber is adequate for biogas purification and was selected in this study.

3. Case modeling

3.1 AD Calculator

The digester plays a key role in the farm-greenhouse Eco-Industrial system. In this study it was modeled by using an *AD Calculator* software developed by Lau et al. (2010). Based on a comprehensive review and examination, the software was coded in Microsoft Excel with build-in anaerobic reaction kinetics. The software is capable of modeling different types of digesters operated at customized conditions, with biogas generation and economic information of digester as the output. In this study, the Excel program was revised by incorporating the costs of the biogas water scrubber and digestate separator as well as the revenues from the solid digestate and carbon offsets whereas the other algorithms remained unchanged.

3.2 Economic analysis method

Cash flows were calculated for each year within the 20 years' typical digester life time (Enahoro and Gloy, 2008). The calculation of capital costs of a digester system is regressed from a number of data available in the open literature and industrial reports. A digester system refers to a set of facilities that a farm has to install to start a digestion project. Its capital cost is estimated as a function of the maximum power output where power output refers to the electricity power potential in the biogas (Lau, 2010). Capital costs of the biogas water scrubber and the digestate separator are not originally included in the *AD Calculator*. Capital cost of a non-compression water scrubber as in this study was assumed to be $15,000, which is discounted from reported advanced upgraders (Ofori-Boateng, 2009). Reported digestate separator capital costs ranges from $48/cow-$104/cow (Wright and Ma, 2003; Gooch and Pronto 2008; Pronto and Gooch, 2009; Pronto and Gooch, 2010). An average value of $70,000 was selected for a press-screw separator applicable for a dairy farm with 1000 head capacity. Capital cost of the manure storage tank was not included since animal farms usually have manure storage tanks in place before a digester is installed.

The annual operating cost analyzed in this calculation was assumed to be 5% of the total capital cost. It covers labor, maintenance, and insurance. Utility cost which is the cost of electricity to operate the facilities was counted as an additional cost item, proportional to the scale of the facilities (Lau, 2010). Debt financing was taken into consideration in this study. The loan portion was set as 30% of the total capital cost which is the highest possible cost share by the BC Environmental Farm Plan (EFP) Program to support manure management projects in combination with AD (BC Ardcorp, 2011). The loan period was assumed to be 10 years. Annual loan repayment was calculated as the cash outflow.

Sales of biogas to the greenhouse generate most of the revenue. Another source of cash inflows is the utilization of the solid digestate, resulting from the savings from bedding materials for animal farms and growing media for greenhouses. Also, greater nutrient availability for plant uptake is achieved with the use of liquid digestate, leading to fertilizer savings (Werner, 2011). And there is less cost in applying liquid digestate than applying raw manure. Carbon offset credits were also considered in one of the scenarios.

To measure the profitability of the project, the indicators analyzed include the net present value (NPV), discounted payback period (PP), and the internal rate of return (IRR). Usually the IRR is expected to be greater than the minimum acceptable rate of return (MARR) to show the project is economically feasible. Table 1 lists the economic values used in this study.

Table 1: Economic values used in this study

Item	**Value**
Digester system capital cost (PF, MPF) (Lau, 2010)	7570×MaximumPowerOutput$^{0.8722}$.
Digester system capital cost (CSTR) (Lau, 2010)	26920×MaximumPowerOutput$^{0.7388}$
Digestate separator capital cost	\$70,000/ 1000cows
Biogas upgrader capital cost	\$15,000/ 1000cows
Utility (electricity) cost (Werner, 2011)	\$0.06/kWh
Equity/debt financing structure	70%/30%
Loan period	10 years
Loan compounded interest rate	6% per annum
Biogas revenue(Werner, 2011)	\$0.58/m^3 CH_4
Solid digestate savings (Wright and Ma, 2003)	\$28,000/ 1000cows/yr
Tipping fee revenue (Werner, 2011)	\$15/tonne off-farm waste
Manure spreading savings (Electrigaz Inc., 2007)	\$20/cow/yr
Fertilizer savings (BC ADIAC, 2012)	\$13/cow/yr
MARR	10%
Tax rate (federal and provincial)	13.5%
Carbon offset (BC Carbon Tax, 2012)	\$25/tonne CO_2-eq.

--The values were converted based on AD feedstock load, biogas yield, or digestate amount in specific scenarios.

--Sources of the values are referred to in the text if not indicated in this table.

3.3 Scenarios

11 scenarios with varied parameters were modeled to analyze the economic feasibility of an animal farm-greenhouse Eco-Industrial system. Details of each scenario are shown in Table 2.

Table 2: Details of the 12 analyzed scenarios

Scenario	Manure feedstock	Non-Manure feedstock	Digester type	HRT	Others
1	Dairy (1,000 head)	None	PF	25d	Feedstock temperature[a]=10 ^oC
2	Dairy (1,000 head)	None	MPF	25d	Feedstock temperature=10 ^oC
3	Dairy (1,000 head)	None	MPF	20d	Feedstock temperature=10 ^oC
4	Dairy (1,000 head)	None	MPF	25d	Feedstock temperature=15 ^oC
5	Dairy (1,000 head)	None	MPF	25d	Feedstock temperature=25 ^oC
6	Dairy (1,000 head)	None	MPF	25d	Feedstock temperature=10 ^oC With carbon offsets[b]
7	Dairy (1,000 head) + Pig[c] (2,000 head)	None	MPF	25d	Feedstock temperature=10 ^oC
8	Dairy (1,000 head) + Pig[c] (4,000 head)	None	MPF	25d	Feedstock temperature=10 ^oC
9	Dairy (1,000 head) + Chicken (50,000 head)	None	MPF	25d	Feedstock temperature=10 ^oC
10	Unspecified[d]	20% food waste[e]	MPF	25d	Feedstock temperature=10 ^oC
11	Unspecified[d]	10% food waste+10% fat and oil waste[e]	MPF	25d	Feedstock temperature=10 ^oC

[a]Assume feedstock is heated before entering the digester. The additional energy required for this part is from other sources

instead of from the internal energy / economic budget.

[b]Carbon emission reduction value is available for scenario 2 in the preliminary LCIA study which is 66.2 tonnes CO_2-eq per month(Zhang and Bi, 2011a, 2011b). This comes to an annual carbon credit revenue of $19,860 (BC Carbon Tax, 2012).

[c]Pigs are specified as boars and sows with quantity under a typical ratio of 1:19 (NSW, 2006).

[d]Modeling in these scenarios are in different routes where specification of manure type is not available.

[e]Food waste has a total solid content of 23% (Baldwin, 2009).

4. Results

Table 3 shows the economic LCA results.

Table 3: LCC results

Scenario	Biogas production	Net CH_4 production	Capital cost	Annual cost (excl. capital)	Annual gross revenue	NPV	IRR	PP
Unit	m^3/day	m^3/day	$	$/year	$/year	$	%	year
1	1,374	459	505,844	32,278	124,461	488,300	27.6	4
2	1,888	767	604,808	42,970	178,174	773,746	31.9	3
3	1,318	433	490,730	31,168	120,166	469,323	27.4	4
4	1,888	877	604,808	44,233	204,030	967,043	37.2	3
5	1,888	976	604,808	45,380	227,536	1,142,767	42.1	3
6	1,888	767	604,808	42,970	198,034	922,216	36.0	3
7	2,202	857	741,633	49,465	211,427	934,471	32.9	3
8	2,516	947	841,111	55,888	244,681	1,096,360	33.6	3
9	2,473	1,093	794,072	54,876	260,394	1,475,401	43.5	3
10	5,134	2,633	1,443,612	107,217	518,434	2,618,641	42.6	3
11	7,353	4,370	2,078,996	157,622	827,469	4,374,488	47.7	3

The results show that all scenarios have a payback period of less than 5 years and a positive NPV, with some of them fairly attractive. Since scenario set-up is intended to investigate the economic feasibility of the proposed farm-greenhouse Eco-Industrial system as well as the influence of different variables, the results are further discussed in groups to illustrate the effects of different design and operating parameters.

4.1 Digester type

Scenarios 1 and 2 compare the performance of a PF digester and a MPF digester under the same feed load. MPF digester has a higher biogas yield but also higher capital and operating costs. The PF digester is also acceptable on biogas yield but the NPV is low. While the MPF is designed to compromise between the stability of a CSTR digester and the high efficiency of a PF digester, its advantage of high gas production rate makes it economically competitive as well.

4.2 Hydraulic retention time

Twenty to 30 days is a typical hydraulic retention time (HRT) for an anaerobic digester under the analyzed capacity. Longer HRT leads to better decomposition of the solids and sterilization. But when solid digestate is intended to be sold for reuse like in this study, the solid digestion should not be maximized. Considering the efficiency factor at the same time, 25 days is an optimal time span (Baldwin, 2009). In this study, HRT has a relatively narrow adjustable range and thus only one scenario with a HRT of 20 days was modeled to compare with scenario 2, both for a MPF digester. Although scenario 3 at a shorter HRT is able to achieve acceptable NPV and IRR, its biogas yield and revenue are less competitive than a digester with 25 days HRT.

4.3 Feed temperature

Because the annual temperature in BC is generally low, scenarios 4 and 5 are modeled to reflect the impact of feed temperature on the overall economic performance. In contrast to scenario 2, the digester feed temperature was raised from ambient to 15 ^{o}C and 25 ^{o}C in scenarios 4 and 5, respectively. This means that some of the energy and associated cost for feed pre-heating to 35 ^{o}C are saved from the baseline scenario. Improvements on net CH_4 production and revenues are significant. However, the relatively high feed temperature has to be achieved from other efforts like effective heat preservation of raw manure or the use of waste heat.

4.4 Carbon offset

Scenario 6 includes carbon offset credits as part of the revenue. The price of carbon offsets was selected as $25/tonne CO_2-eq which is in accordance with current BC carbon tax level (BC Carbon Tax, 2012). From the preliminary LCIA study, the total carbon emission reduction of scenario 2 is 794 tonnes CO2-eq/year. This gives an annual revenue increase of $19,860 in scenario 7, which accounts for 10% of the total annual gross revenue of scenario 3. With the development of the carbon offset system regionally and globally, carbon offset credits have become an important source of revenue.

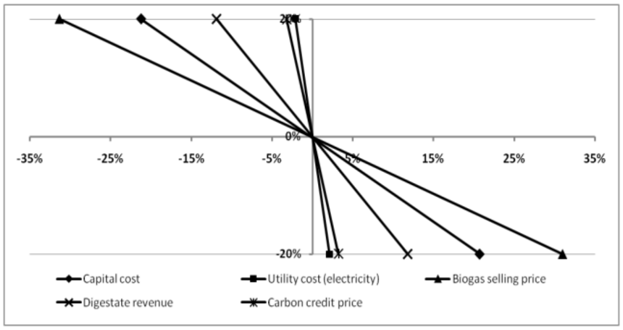
4.5 Co-digestion

Including other feedstock for co-digestion was analyzed in scenarios 7 to 11. The feedstock was modeled to maintain the base amount of manure from 1000 cows plus from other sources. When the biogas production is normalized by the base feedstock volume, blending with pig manure shows a decrease of 10% (scenario 7) and 16% (scenario 8) in biogas production rate. However, adding poultry manure and food waste improves the biogas production rate by 22% (scenario 9), 126% (scenario 10), and 239% (scenario 11), respectively. Hence co-digestion with poultry manure especially food waste and fat and oil waste can dramatically increase profits. These results can be explained by that food wastes have much higher organic nutrient content than animal manure. Besides, tipping fee revenue was generated in these scenarios which further enhances the profits. Although the gross revenue and NPV see substantial improvement, the capital cost increases as well. So further evaluations are needed for real applications. Also, costs of transportation of feedstock were not included in this study because of high uncertainties. If farms and the food industry are sparsely distributed, the cost on transportation can be substantial. Therefore a site-specific analysis is needed to improve the analysis.

5. Sensitivity analysis

Sensitivities of the factors of capital cost, utility cost, biogas selling price, digestate revenue, and carbon credit price were analyzed. Each factor was scaled by ±20% and the corresponding NPV results were plotted into a spider diagram as shown below.

Figure 1. Sensitivity analysis results



The results show that among the analyzed factors, the economic performance is most sensitive (31%) to biogas selling price. Influence from capital cost fluctuations is about 20%, followed by from digestate revenue at 12%. The system is only slightly sensitive (<5%) to factors of carbon credit price and utility costs. These indicate that biogas sale as the main revenue source dominates the overall profitability while capital cost is also a very important factor.

6. Conclusions

Integrating animal farm and greenhouse as an Eco-Industrial system shows sound capability to mitigate environmental and health impacts. Biogas and digestate as renewable resource can contribute to substantial economic profits as demonstrated in this study. Mixed Plug Flow digester with properly designed parameters is the most cost-effective configuration. Preheated warm feedstock can significantly reduce internal cost. Incentives like carbon offsets can also improve the economic feasibility of the proposed integrated system. Conditions permitted, co-digestion of multiple feedstock especially food industry waste will significantly increase the revenue.

References

Anaerobic Digestion Initiative Advisory Committee (ADIAC) (2012): http://www.bcfarmbiogas.ca/

Baldwin S., Lau A., and Wang M. (2009): Development of a Calculator for the Techno-economic Assessment of Anaerobic Digestion Systems. Report Submitted to BC Ministry of Agriculture and Land and BC Life Sciences.

BC Ardcorp. (2011): BC Environmental Farm Plan. Growing Forward Beneficial Management Practices 2011/2012. Available from: http://www.ardcorp.ca/userfiles/file/efp/BMP%2011-12%20MC%20approved%20list.pdf

BC Carbon Tax. (2011): Available from: http://www.fin.gov.bc.ca/tbs/tp/climate/A4.htm

BC Ministry of Environment. (2010): On-farm Anaerobic Digestion Waste Discharge Authorization Guideline. Available from: http://www.env.gov.bc.ca/epd/industrial/agriculture/pdf/anaerobic-digestion-guideline.pdf

Chadwick D. R., Sneath R. W., Phillips V. R. et al. (1999): A UK inventory of nitrous oxide emissions from farmed livestock. Atmospheric Environment. 1999. 33(20): 3345-3354

Dong H., Mangino J., McAllister T. A. et al. (2006): Emissions from Livestock and Manure Management. IPCC Guidelines for National Greenhouse Gas Inventories.

Electrigaz Technologies Inc. (2007): Feasibility Study – Anaerobic Digester and Gas Processing Facility in the Fraser Valley, British Columbia. Final report prepared for BC Bioproducts Association.

Enahoro D. K. and Gloy B. A. (2008): Economic analysis of anaerobic digestion systems and the financial. Report No. EB 2008-04.Department of Applied Economics and Management, Cornell University, Ithaca, New York

Goddard T. (2008): The Alberta Carbon Offset System. Alberta Agriculture and Food. Available from: http://www.seed.ab.ca/pdfs/AGM2008/The%20Alberta%20Carbon%20Offset%20System.pdf

Gooch C. and Pronto J. (2008): Anaerobic Digestion at AA Dairy: Case Study. Cornell University Manure Management Program.

Huang Z. Y. and Bi X. T. (2006): Regional Energy Integration to Reduce GHG Emissions and Improve Local Air Quality. Proc. Climate Change Technologies Conference, Ottawa, May 10-12

Lau A., Baldwin S., Wang M. (2010): Techno-Economic Assessment Of Anaerobic Digestion Systems For Agri-Food Wastes. XVIIth World Congress of the International Commission of Agricultural and Biosystems Engineering (CIGR). Québec City, Canada

Lowe, Ernest A. (2001): Eco-industrial Park Handbook for Asian Developing Countries.

Lynch D. Vancouver Average Monthly Temperatures. Retrieved from: http://vancouver.about.com/od/vancouverevents/a/Vancouver-Weather.htm

NSW Department of Primary Industries.(2006): Basic Pig Husbandry - The Boar.State of New South Wales. Available from: http://www.thepigsite.com/articles/1593/basic-pig-husbandry-the-boar

Odum E., Brewer R., Barrett G.W. (1971): Fundamentals of Ecology.Brooks Cole: 5 edition

Ofori-Boateng C. and Kwofie E.M. (2009): Water Scrubbing: A Better Option for Biogas Purification for Effective Storage. World Applied Sciences Journal, 5 (Special Issue for Environment): 122-125

Pronto J. and Gooch C. (2010): Anaerobic Digestion at New Hope View Farm: Case Study. Cornell University Manure Management Program.

Pronto J. and Gooch C. (2008): Anaerobic Digestion at Ridgeline Dairy Farm: Case Study. Cornell University Manure Management Program.

USEPA AgSTAR (2011): U.S. Anaerobic Digester Status: A 2011 Snapshot. Available from:http://www.epa.gov/agstar/documents/2011_digester_update.pdf

Werner E. and Strehler B. (2011): British Columbia On-Farm Anaerobic Digestion Benchmark Study. For B.C. Agricultural Research and Development Corporation.

Wright P. and Ma J. (2003): Anaerobic Digester at Freund Dairy: Case Study. Cornell University Manure Management Program.

Zhang S., Bi T. X. (2011a): A Case Study on an Integrated Animal Farm-Greenhouse Eco-Industrial System in British Columbia. Poster presentation.The 2011 Symposium on Industrial Ecology for Young Professionals. Berkeley, US.

Zhang S., Bi T. X. (2011b): A Life Cycle Assessment Case Study on a Farm Scale Biogas Plant in British Columbia. Journal of Yunnan Normal University (Natural Science Edition). 31(Supplement): 1-12