Dedicated to and in loving memory of

Ray A.B. Jacobsen sr. , my dad
Farfar Per “Jaks” Jacobsen, my grandpa
Opa en oma

Promoter: Prof. dr. Xavier Gellynck

Department of Agricultural Economics

Faculty of Bioscience Engineering, Ghent University

Dean: Prof. dr. ir. Marc Van Meirvenne

Rector: Prof. dr. Anne De Paepe

Faculty of Bioscience Engineering

ir. RAY JACOBSEN

Tools for sustainability assessment and management: Food chains and household waste case studies

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Members of the Jury

Promoter

Prof. dr. Xavier Gellynck
Ghent University
Department of Agricultural Economics

Reading Committee

Prof. dr. ir. Jacqueline Bloemhof
Wageningen University
Operations Research and Logistics

Prof. dr. Cesare Zanasi
University of Bologna
Dipartimento di Scienze e Tecnologie Agro-Alimentari

Prof. dr. Tom Verbeke
KU Leuven
Onderzoekseenheid Centrum voor Economische Studiën

Examination Committee

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Ghent University
Department of Animal Production

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Ghent University
Department of Biosystems Engineering

Prof. dr. Xavier Gellynck
Ghent University
Department of Agricultural Economics

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Wageningen University
Operations Research and Logistics

Prof. dr. ir. Stijn Speelman
Ghent University
Department of Agricultural Economics

Prof. dr. Cesare Zanasi
University of Bologna
Dipartimento di Scienze e Tecnologie Agro-Alimentari

Prof. dr. Tom Verbeke
KU Leuven
Onderzoekseenheid Centrum voor Economische Studiën

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Thank you for already reading this much of my PhD, a few more pages to go. Enjoy reading !

Ray M. L. Jacobsen
April 2016

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CF	Carbon Footprint
TBL	Triple Bottom Line
SAFA	Sustainability Assessment for Food & Agriculture
FAO	Food & Agriculture Organization
SA	Sustainability Assessment
EIA	Environment Impact Assessment
PV	Photovoltaic
UNEP	United Nations Environment Program
CFC	Chlorofluorocarbon
GWP	Global Warming Potential
GHG	Greenhouse Gases
SEA	Strategic Environmental Assessment
IPCC	Intergovernmental Panel on Climate Change
ETS	Emissions Trading System
LCA	Life Cycle Assessment
ISO	International Organization for Standardization
LCSA	Life Cycle Sustainability Assessment
KBBE	Knowledge Based Bio-Economy
ESD	Effort Sharing Decision
NIS	National Institute of Statistics
LA	Latin America
PCR	Product Category Rules
LUC	Land Use Change
IDF	International Dairy Federation
GE	Gross Energy
DE	Digestible Energy
FU	Functional Unit
OECD	Organization for Economic Cooperation & Development
FP7	Seventh Framework Program
GM	Genetically Modified

Preface

The world population clock is ticking with a current number of over 7 billion people living on our planet. This number is ever increasing. This brings our way of living in complex situations with our rising needs and demands leading to further complications. There is a rising demand for drinking water, food, energy sources and space of any kind. If mankind cannot fulfill these needs, it might lead to other indirect problems such as population pressure, diseases, poverty, inequality and so on. Over the coming decades, the world will witness increased competition for natural resources, which are finite and limited. The growing global population will need to have access to a safe and secure food supply.

Many words in the paragraph above are hot topics today, threatening ecosystems and potentially bringing our future living in danger. Something has to change and the word 'Sustainability' has become more important than ever before, as it involves every one of us. "Sustainability is something everyone can work towards" (Global Footprints, 2013); whether it is picking up garbage you see on the street or bringing into consideration harmful business methods, everybody can make a difference.

A transition is needed towards a more optimal use of resources. There is a clear need to move towards more sustainable primary production and processing systems able to produce more food with fewer inputs, lower environmental impacts and reduced greenhouse gas emissions.

This PhD dissertation will include tools for sustainability assessments and sustainability management applied to cases. The following sections of the introduction further elaborate on this matter. The manuscript is a compilation of several research papers. Sustainability assessment is helpful to aid in the progress towards higher sustainability.

The greatest threat to our planet is the belief that someone else will save it

(Robert Swan)

1 General Introduction: framework, research questions & outline of the thesis

1.1 General introduction

During the last years sustainable development became one of the buzzwords, used in many different contexts and settings. However, it remains extremely difficult to precisely define the concept. If decent indicators of sustainable development need to be produced, there should be an agreement on what needs to be indicated (Dahl, 1995).

In many policies, sustainability has to be one of the most frequently used terms during the last years (Dahl, 1995). Gieryn (1999) calls it a "boundary term", one where science and politics are meeting each other. It is not the aim to provide an overview of all existing definitions. There should be a working consensus on what is actually being measured. From that perspective, it is more handy to start with the main sustainability dimensions to be included in any assessment. Moreover this basis can help to create a working definition able to focus on the configuration and meaning of indicators and indices. This should be seen in the light of communicating sustainability indicators and the best way to do so (Voß, 2002).

During the last years, the development of indicators and indices at any geographical level, has become a common approach for meeting the need for assessment tools, as a means for communication to a wide audience (Moldan et al., 2012). These tools are a condition for implementing the concept of sustainability and sustainable development (Hansen, 1996). Distinction should be made between the two concepts (Voß, 2002). Sustainability refers to a system property (or quality), whereas the core of sustainable development is made by the Brundtland definition and Article 1 of the Rio Declaration (UNCED, 1992):

Sustainable development is about using resources in such a way that it meets the needs of the present generations without compromising the possibility for future generations to meet theirs (see also section 1.2). Given the fact that this definition is somewhat vague, several different interpretations have emerged to operationalize sustainability (Pope et al., 2004; Linton et al., 2007; Moldan et al., 2012).

Sustainable development emphasizes people and their wellbeing and is hence an anthropocentric view as human needs form the base. Maslow's pyramid presumes that basic needs (physiological, survival, love, esteem) should be met before a person can act unselfishly (Moldan et al., 2012).

The meaning of the sustainability concept has been explored further, and today, it is perceived as having three dimensions or pillars: the socio-cultural, the economic and the environmental dimension (Giudice et al., 2006; Van der Vorst, 2010). In order to truly accomplish sustainable development all three dimensions of sustainability have to be taken into account (Ness et al., 2007). At this point, it is noteworthy to indicate that sustainability is a very difficult concept to define in a meaningful and practical way in order to allow it to be operationalized (Pope et al., 2004; Hacking and Guthrie, 2008).

The separation of sustainability into 3 pillars tends to stress potentially competing interests instead of emphasizing linkages and interdependencies between them (Scoones et al., 2007).

This makes integration very difficult and challenging and to some extent it is creating trade-offs among dimensions, mostly degrading the environment (Sheate et al., 2003; Jenkins et al., 2003; Gibson, 2001, Pope et al., 2004). Most of the time independent sustainability goals are set for each of the dimensions. This is however somewhat in contradiction with the integrated sustainability approach of covering three dimensions (Ness et al., 2007; Van der Vorst, 2010). It is noteworthy that sustainability is a difficult concept to define in a way that is meaningful and practical to allow it to be operationalized (Pope et al., 2004). The definition of a suitable sustainability metric for supporting sustainability assessments is still an open issue within literature (Cucek et al., 2012). Thus, the question rises whether sustainability is only about monitoring a certain dimension over time, or whether it can be quantified in a single, complex index (aggregated, composite indicator). This will be the main objective of this PhD dissertation, namely to examine the possibility and necessity to express sustainability as 1 integrated, composite indicator.

For the transition towards sustainability, goals must be assessed which remains a challenge in providing efficient and reliable tools. In response to this challenge, sustainability assessment became a fast developing area (Ness et al., 2007). **Sustainability assessment** might help decision-makers and policy-makers in deciding what actions should and should not be taken in an attempt to make society more sustainable (Devuyst, 2001; Ness et al., 2007). It is the aim to ensure that “plans and activities make an optimal contribution to sustainable development” (Verheem, 2002). Sustainability assessment is often considered to be the next generation of environmental assessment as it more or less evolved from it (Sadler, 1999, Pope et al., 2004; Ness et al., 2007).

1.2 The sustainability concept

Due to the growing world population and industrialization taking place all over the globe, impact of mankind on the natural environment is increasing. At the same time this natural environment plays a vital role in the lives of human beings. The ecosystem fulfills several important functions related to the source-sink concept: it provides resources, absorbs and recycles waste and performs life support functions (e.g. UV-absorption by the stratospheric ozone layer) (Perman et al., 2003). According to the weak degree of sustainability (see section 1.3) it is possible to substitute environmental services by man-made capital. However, human society will always depend upon the natural environment. Despite this dependency humanity itself jeopardizes the fulfillment of the ecosystem functions by conducting activities with a negative impact on the environment. First, natural resources are extracted at a high rate, leading to resource depletion if extraction rates exceed renewal rates. Second, emissions are released as a result of human activities changing the state of the environment.

The first awareness about the fact that resources are not infinite on one hand and that human consumption impacts future generations on the other hand can be found back in ancient times, in forest management (Sylvicultura oeconomica (Von Carlowitz, 1713)). With the rise of the environmental movements in the 1960s and 70s, and the debates about the ‘Limits to growth’

(Meadows, 1972) related to the club of Rome, environmentalists were trying to draw attention towards the link between environmental issues and the major questions of development. It lasted until the 1980s however before sustainability was more widely debated. The centre of this debate was formed by the United Nations Commission chaired by Gro Brundtland in 1987. This commission came up with a landmark report 'Our Common Future' (Brundtland, 1987) mentioning the modern definition of sustainable development:

Sustainable development is development that meets the needs of the present without compromising the ability of future generations to meet their own needs (1987:43).

The core words in this definition are *needs* and *future generations.* Needs should not only be seen on a subsistence level but also considered in the context of the aspirations communities may have of the development process, and how this should be met in a sustainable way (Voß, 2005). Strategies regarding sustainable development are trying to improve the quality of human life as measured by key indicators such as health, housing, income, education, without surpassing the carrying capacities of the underlying resource-base. As such, **resource management** is placed at the centre of sustainable development. It involves the usage of resources in such a way that does not degrade the resource-base nor reduce the range of development options for future generations. In short, resource management, quality of life and inter-generational equity make up the foundations of sustainable development. The sustainable development definition implies that the ecosystem should be maintained in order to continue fulfilling its functions in the future.

More recent work regarding sustainability and the environment is found in works of economists and philosophers (Parsons, 1997; Linton et al., 2007). From the 1980s on, there was a huge increase in the academic debate on sustainability and sustainable development. At the same time, sustainability became a societal issue and entered the popular culture through books and films. The general, societal impact created an extra leverage in attracting extra attention and funding for research. Consideration of the sustainability concept is increasingly perceived in management literature as it is a hot topic both for science and the public: Linton et al. (2007) found a 5 times increase in articles regarding sustainability over a 15 years period of time (1992-2007). At the same time, the concepts and terms were transferred to the main stage of policy debates all over the globe, with a good example being the World Conference on Environment and Development held in Rio 1992.

Ever since, a plethora of definitions emerged (Johnston et al., 2007). However, the plurality of definitions creates a lot of confusion about the usage of them. Some terms are somewhat similar or just little different from each other (Glavic and Lukman, 2007). There is hence a need for dialogue across several fields, as the concept absorbs different meanings and tends to loose its meaning. The fact that sustainability is not precisely defined, does not imply that it cannot be used or assessed as a concept (Glavic and Lukman, 2007).

Sustainability describes a system quality. Human society is not a static system, but a dynamic one. As such, sustainability is a matter of seeking balance to be managed over a long period of time. The concept hence cannot be scaled and measured in an easy and uniform way, as it can be seen as a quality of motion instead of a fixed point. This can be better seen in practice, as there are plenty of forces able to disorder a balance over time. For this reason, many indicators can be seen as degrees of unsustainability, where the size and amount of imbalances are measured (UNEP, 1995). In general, sustainability can be achieved by reducing or even eliminating all forces disturbing the balance. A system is hence moving from stable state to stable state, seeking for equilibrium.

This goes hand in hand with the challenge to be able to manage a transition ahead by anticipating potential changes and attempt to act proactively rather than being forced to react (Haberl et al., 2004).

Observing the progress in either way is hindered by the large amount of perspectives on sustainability, originating from several fields. Progress might even be impossible to observe when certain goals cannot be made more clear in operational terms (Haberl et al., 2004).

Any society or economic system is confronted with several types of unsustainability: overuse of limited resources, bad supply inputs, redundant output demand, pollution pressures, …Sometimes, the system is able to control these forces, while other systems undergo more external pressures. The equilibrium is hence not ever lasting and undergoes pressures, shocks and stresses, both internally and externally (Scoones et al., 2007).

Sustainability is the ability of a system to overcome shocks and stresses in seeking a balance upon the interactions of ecological, social and economic aspects. In the quest for sustainability, a pathway should be followed in order to move from one stable state to another and remove the factors causing unsustainability.

Emphasizing pathways brings about a discussion on contested goals and objectives decided by politics. Constructing such pathways is very dynamic and uncertain. There is no single pathway to progress in obtaining higher sustainability, different and multiple pathways are possible (Scoones et al., 2007).

Most policymakers only have the intention to look a couple of years ahead and see the long term as perhaps 20-30 years from now, jeopardizing the timeframe of sustainability. Given the global systems and resources, it is better to perceive sustainability in a very extended timeframe, say into an indefinite future. This time period should help in conducting actions not jeopardizing the needs of future generations and societies. Such an approach is also able to help in avoiding the trend towards discounting the long-term future. Moreover, it will urge for approaches considering fundamental and gradual steps of global balance and change (Dahl, 1995).

Sustainability is about the ability to manage this transition from stable state to stable state, which will take place sooner or later. It should however occur with an acceptable quality of life for everybody on the earth. In order to do so, there is a clear need for tools to identify the progress towards sustainability (or the further moving away from it) at a certain point of time (snapshot or monitoring). Given the above, tools might sometimes monitor unsustainability, but anyhow provide insights into the system quality at a certain period of time, necessary to reveal the followed path: progress towards or diverting away from it. The system quality hence needs to be monitored (assessed over time) in order to reveal the progress and adjust/intervene the situation for restoring the balance. This proves the need for assessment tools on one hand, but also the need for sustainability management tools which are helpful in constructing the right pathway towards sustainability.

Sustainability is about exploring pathways in trying to reach more sustainable states. Interactions of ecological and social/economic systems therefore take place. Taking such interactions into account plays a vital role for constructing and understanding pathways and intervention strategies towards sustainability, tackling economic, social and environmental issues (Scoones et al., 2007).

The economic view of sustainable development is presented by the concept of pursuing maximum income, while at least maintaining the assets on which this income is based. Ecologically sustainable development refers to preserving the ability of 'natural' systems to adapt, whereas socio-cultural sustainable development aims at maintaining the stability of social and cultural systems. To fully reach sustainable development all three dimensions of sustainability have to be included (Voß, 2005; Van der Vorst, 2010).

In 1992, this was acknowledged by the Rio Declaration (U.N. Report of the United Nations Conference on Environment and Development, 1992). Often mentioned are the "3 P’s" of sustainable development: People, Profit, Planet which was introduced by Elkington as the **Triple Bottom Line (TBL)** (Elkington, 1997). This is another way of expressing the necessity to integrate the social, economic and environmental components of sustainable development and can be presented as shown in figure 1.

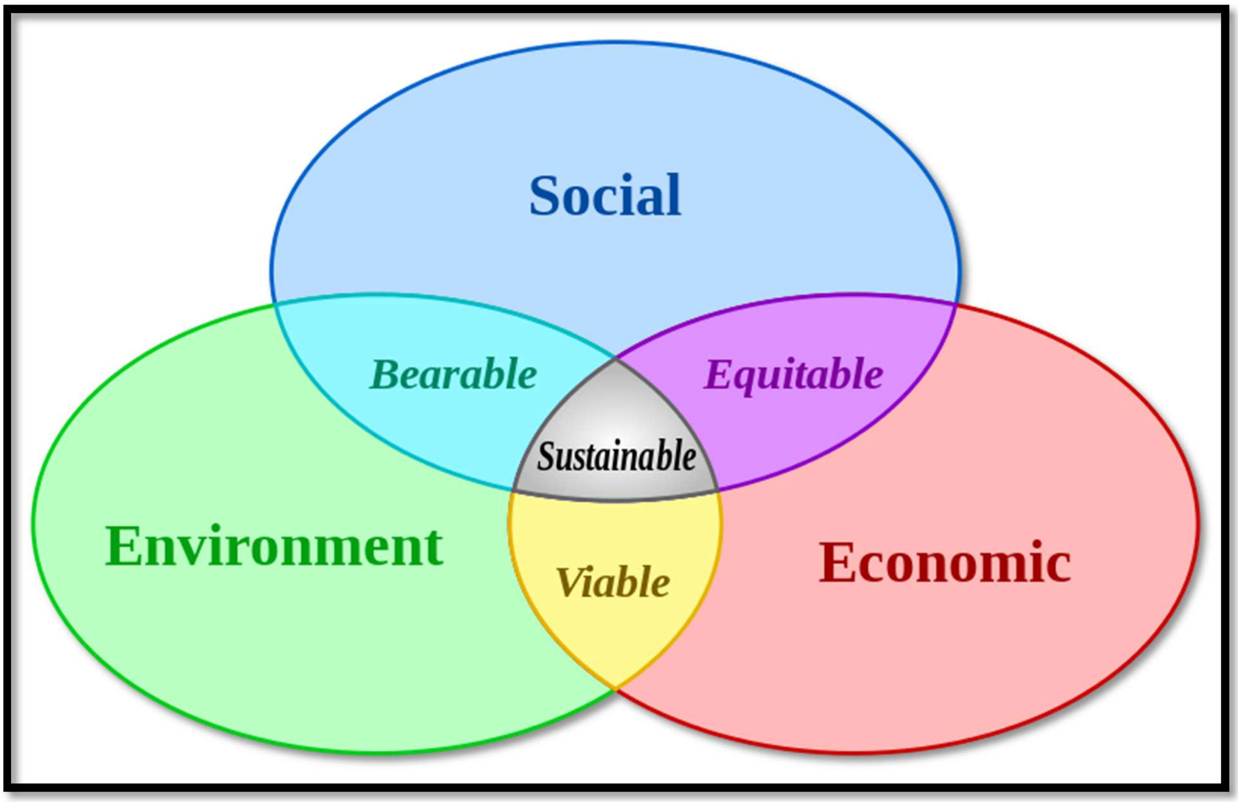

Figure 1. The three P (People, Planet and Profit) of sustainable development.

The triple bottom line (TBL) framework adds two main elements to the discussion. First, it helps in focusing on specific criteria for progress in each of the three domains. Second, the TBL framework highlights the relationships among the three main elements (see Figure 1). In the ideal situation, humankind operates at the intersection of this Venn diagram, where all three goals are met: sustainability.

The three P's or dimensions of sustainability have been introduced. It is important to clearly explain what they precisely entail. The next section further elaborates on this matter.

1.3 Sustainability dimensions

In the narrow sense, the **economic dimension** can be seen from a fiscal and financial sustainability point of view: deposits plus interest must balance with the withdrawals plus charges in accounts. Financial debts created in the present are hence a bottleneck on the ability to meet future needs. Moreover, the present advantage mankind possesses and not being able to pay back the debt, creates an imbalance and is unsustainable. Obviously, economic sustainability involves much broader issues than solely financial ones. It also includes maintenance of assets and capital value and productivity through investments which are at least equal to depreciation.

In economy, there are several types of capital: man-made, natural and social (World Bank, 2006). Goodland-Ledec (1987) define sustainable development as using renewable natural resources in such a way that does not exclude future generations to access them easily.

Given the definition, sustainability however does not require for each generation to get access to the same amount of renewable resources. In seeking a balance it might occur that new man-made capital turns a renewable resource into a resource with no use for future generations. Within the framework of defining the sustainability concept in more concrete terms, the need to reduce environmental impacts and climate change can definitely be underpinned. This is more difficult when confronted with the question of whether the use of finite energy resources is compatible with sustainable development, as oil and gas might not be available for use by future generations. From this perspective, only renewable resources would be possible under the concept of sustainability as Goodlands definition does not take into account non renewable resources.

This is however not possible for 2 reasons. At first, using renewable energy always goes hand in hand with the use of non-renewable resources (think of some scarce resources (oil and minerals) to construct PV solar panels). Given the scarcity of these resources, the price mechanism will do its work in finding alternatives. Second, it would actually mean that non-renewable resources cannot be used at all, not even by future generations. Due to the second law of thermodynamics (life needs the consumption of workable energy) it is inevitable to use non-renewable resources. Therefore, for development to be sustainable it is important to leave future generations a resource-base which is usable and allows to satisfy needs at the same level as today's generation is enjoying.

Neo-classical economists started talking about theories of substitutable capital in order to define **weak sustainability.** Elements of the natural capital can be replaced by man-made capital such as infrastructure, labor and knowledge. Natural capital covers the stock of environmental assets such as fossil fuels, biodiversity and other ecosystem services. In **very weak sustainability**, the overall stock of man-made capital and natural capital remains constant over time. It is important to note that any substitution between the various kinds of capital is allowed within weak sustainability. This means that natural resources may decline as long as human capital is increased. Examples include the ozone degradation, tropical forests and coral reefs but with benefits for the human capital (financial profits). If capital is left constant over time it is possible to obtain intergenerational equity, and thus achieve Sustainable Development. A debate emerged whether such a weak definition was appropriate and whether a stronger definition emphasizing the lack of substitutability of 'critical natural capital' was needed (Pearce and Atkinson, 1993, Turner, 1992; Goodland, 1995; Goodland and Daly, 1996). Sustainability concepts attributed to the school of ecological economics take into account a view of **strong sustainability**, preferring ecologically based limitations. Arguments are the limited level of natural resources and the non-substitutable functions of nature.

The two models are not very realistic. Regarding weak sustainability it might occur that if the environment has been degraded in favor of other capital, it no longer can be restored. There are limits to capital substitution (Seidler, 2009). The choice of society about how to balance the value of the environment against other services might be destructive in the long run. Strong sustainability on the other hand is not in line with the second law of thermodynamics, as each activity conducted by man produces entropy by consuming workable energy. As such, life constantly requires inputs, which is in contradiction with the non-substitutability of natural capital.

The definition on sustainable development by Markandya and Pearce (1988) does not require economic growth. It is sufficient to give future generations the same real incomes as the current generation has plus a capital stock allowing them to meet their needs.

Today there are a lot of concerns about the **ecological/ environmental sustainability** of human development on our planet. Population growth and increasing resource consumption are bringing down stocks of natural resources and their renewable productive potential. At the same time, increasing amounts of waste further harm natural systems and disturb vital support processes such as the carbon cycle and the maintenance of the stratospheric ozone layer. There is in fact an accumulation of resource and pollution debts which can only be removed by means of thorough future investments. Many ecological processes are so complex and thus poorly understood that it is difficult to deduce indicators of ecological sustainability. Given the current knowledge, the best possible approach is to use indicators of unsustainable pressures and impacts. These should be reduced and minimized in a precautionary way to the management of such complex systems (UNEP, 1995) in order to achieve higher sustainability.

Environmental sustainability is about maintaining nature's services at a suitable level (Moldan et al., 2012), protecting the resources for human needs without exceeding the sinks for human wastes.

There is also a dimension that can be referred to as **social sustainability.** Setting a definition for social sustainability is less clear-cut.

Black (2004) defines social sustainability as " the extent to which social values, identities, relationships and institutions can continue into the future." Torjman (2000) defines it as follows: "human wellbeing cannot be sustained without a healthy environment and is equally unlikely in the absence of a vibrant economy." Gilbert et al. (1996) see it as follows: "Social sustainability requires that the cohesion of society and its ability to work towards common goals be maintained. Individual needs, such as health and well-being, nutrition, shelter, education and cultural expression should be met." These definitions remain however statements of the general goal of social policy rather than clear definitions (Colantonio, 2007). Social sustainability might play a crucial role in the long-term survival of human civilizations (Diamond, 2005). Despite this acknowledgement, it is not yet clear what determines social unsustainability. The sustainability concept shows the ambition to meet the needs of a growing population in the present and in the future, and to sustain human living conditions for all and permanently without living on the account of future generations.

Sustainability is linked with ecological, economic, social, cultural and institutional aspects of development. Taking into account these different aspects, terms as "dimensions" and "pillars" have been acknowledged. Two generic categories can be distinguished: single-pillar and multi-pillar models. Single-pillar models want to preserve the ecological balance in a socially and economically compatible way (Voß, 2002). Nature needs to be preserved as basis for living and the economy. Multi-pillar models focus on equity among the different dimensions of sustainability. Most studies follow the three-pillar model, thus accounting issues of environmental, economic and social dimensions under equal ranking conditions. This PhD dissertation follows the three-pillar model as well. Some sources in literature want to include a cultural or institutional dimension too. I suggest not to include it as it has a qualitatively different function. The 3 main dimensions are more related to the content of issues of sustainable development, whereas the institutional dimension is more linked with the question **how** sustainable development might be implemented and **what skills** institutions should possess for carrying out the job (Voß, 2002).

The temporal dimension

Indicators for sustainable development should include new ways of accounting for sustainability over time. Including the sense of time requests a temporal dimension. Measures trying to map a balance cannot solely look at a static situation at several moments (monitoring), but need to look at measures integrated over time to document processes and trends. Such way of accounting should be able to incorporate past trends which led to the present. At the same time, projections into the future are needed to identify what is needed to reach or maintain sustainability. For these reasons, many present economic balance sheets are having shortcomings: many capital stocks and externalities are excluded, and the sheets are not well summed over time, in particular with respect to future implications of the present situation (UNEP, 1995).

Nothing is permanent on our planet. Wealth is created and destroyed again; energy is degraded; objects and materials have a useful life and then become waste with disposal or recycling costs. Ageing and death are as much part of life as reproduction and growth. Sustainability expects to account for all this over time, a way of monitoring given its dynamic nature. The ultimate sustainable balance should involve processes of maintenance, replacement and renewal and at the same time exceed those of depreciation, degradation and loss (Voß, 2002).

In order to be able to monitor, there is a clear need to assess sustainability in the first place. In this dissertation, the focus is on assessing sustainability from a case study point of view by means of tools. Indeed, part of sustainability assessment is context-specific (Gibson, 2006) and therefore requests cases. It is not just the dynamics of sustainability that matter, but the dynamics-in-context which are critical (Scoones et al., 2007). In order to do so, a reference point is needed. Once this is defined and in place, it is possible to start assessing the progress towards sustainability.

Research on sustainability assessment has so far highlighted 4 major shortcomings:

First, the multi-functionality of the concept is not addressed as it involves several dimensions. Many studies do not take into account this multi-functionality. There is a need for an integrated assessment, covering all sustainability dimensions (Rossing et al., 2007; Inyang, 2009; Moldan et al., 2012; Bausch et al., 2014).

Second, there is an imbalance across dimensions: most studies focus too much on 1 dimension (mostly environment) during assessments, hence creating an imbalance (Moldan et al., 2012, Bausch et al., 2014). Moreover, there is less good framing and validation of socio-economic dimensions.

Third, despite the demand from policymakers and scientists, very little information and experience is there with indicators linking sustainability dimensions to target levels taken from a sustainability point of view (Moldan et al., 2012).

Fourth, decision-making requests more transparency: Assessment results are mostly difficult to implement in decision-making (Morse et al., 2001). Transparency is high on the political agenda and is very prominent in sustainability (Wognum et al., 2011). It can be defined as disclosing information from producers, chains, certification bodies and failing state authorities for civil society and consumers (Mol, 2013). Especially Global value chains are more and more confronted with voluntary and mandatory demands for transparency and to disclose information on the sustainability qualities of products and processes along value chains. Consumers as well as the public domain in emerging countries (like Brazil) encounter difficulties in getting access and applying sustainability information on the production and products of international value chains (Mol, 2013).

A lot of countries have set nationwide strategies for obtaining sustainable development, including sustainability targets and indicators (FAO, 2012). Several attempts have been undertaken to make the agricultural sector in general and the food sector in particular more sustainable. However, there is no single standard on an international level defining what sustainable production should involve (van der Vorst et al., 2013). Moreover, there is no international agreement on which set of indicators to include when measuring integrated sustainability performance, covering all dimensions (including the social one). Many organizations, both public and private (e.g. The Sustainability Consortium, The ENVI protocol and the COSA consortium) have made attempts to build a scientific foundation driving innovation to improve consumer product sustainability. It is therefore not clear when a certain product or process can be seen as sustainable (FAO, 2012). Similar findings are reported by Hassini et al. (2012).

Intrinsic properties of food products and processes in combination with raising sustainability concerns lead to the need for decision support tools able to integrate economic considerations with quality preservation and environmental protection in food chains (Soysal, 2014). In order to perform sustainability assessments at an international scale, new assessment tools and methods are needed, covering the multi-dimensionality of sustainability (Bausch et al., 2014).

1.4 Measuring sustainability

A conceptual construct like sustainability has no standardized method for measuring (Heink and Kowarik, 2010).

Attempts to measure sustainability have been undertaken in the past. The ecological footprint for example has been used as a measure of sustainability, however literature suggests it has a lot of shortcomings (Fiala N., 2008). This footprint is a measure of the resources needed to produce the goods that an individual or population consumes (Fiala N., 2008). It offers a simple and intuitive estimate of production inputs needed for a certain consumption level, however it fails to address the sustainability of consumption where it was actually meant for. The criticisms towards the ecological footprint include the assumptions that need to be made: the arbitrariness of assuming zero greenhouse gas emissions and national boundaries, which makes extrapolating from the average ecological footprint problematic. If prices for food increase and the business movement in the market is free, there will be an increased search for producing more food (Fiala, 2008). This can be done by extensive or intensive production. In the extensive approach will the producers look for more land, whereas the intensive approach will increase production by applying better production technology to obtain higher yields. In this regard, the ecological footprint also cannot include intensive production, and so comparisons to biocapacity are wrong (Fiala, 2008). Biocapacity comparisons, such as the argument that it would take 5 Earths to sustain consumption if everyone consumed like

Americans, assume that the average consumption of an area can be extended to the entire world population, with all production at the current technology level. However, it is known that this kind of calculation is meaningless and partial (Fiala, 2008). The most important limitation is the fact that the footprint cannot address the issue of land degradation. The lack of correlation between land degradation and the ecological footprint hides effects of a larger sustainability problem. In fact, an ecological footprint measures inequality in resource use between regions and nations rather than measuring sustainability of a certain area. Better measures of sustainability should be able to address these issues more directly. There is hence a need for new tools to assess sustainability. It is more useful to look directly at measures of sustainability, such as land degradation and CO_2 aggregations, rather than using a footprint (i.c. ecological) that very poorly integrates these problems (Fiala N., 2008).

The combined empirical research focus on both the single sustainability dimension level as well as the integrated sustainability approach, request case studies. This PhD dissertation includes sustainability assessment and sustainability management tools from a case study point of view. The applied cases are described in the end of the introduction. The following sections of the introduction further elaborate on this matter. The rationale behind the relation between research locations and the topic is described in the penultimate section of the introduction. The subsequent sections present underlying theories and methods, followed by the conceptual framework. Moreover, the research questions are presented.

There is a reason why there is not yet a fixed method in place providing a single figure as a result of a sustainability assessment, related to the concept's complexity. It is understandable that decision makers of any kind urgently need such information to assist them in their decision making.

In light of the above, the aim of this PhD is twofold:

- To measure sustainability performance based upon 3 cases and to provide tools for conducting sustainability assessments. Herewith raises the question whether sustainability can be expressed as 1 indicator and whether this is necessary.

- In order to assess different potential pathways to sustainability, there is a need to have a closer look at how systems respond to internal and external changes. The aim is thus to assess sustainability for the cases in place in order to identify hot spots and those factors able to contribute to higher sustainability by restoring the balance for the dimension in place.

1.4.1 Sustainability indicators

In order to convert sustainability into an operational approach, for assessing the sustainability of systems, quantitative approximations of the systems performance are needed. At the same time, performance measurement allows to identify hot spots and bottlenecks in the system in order to restore the balance and obtain higher sustainability. Thus, sustainability indicators are needed to provide decision makers and the public with necessary information about the state of a system.

Sustainability moved from a rather qualitative nature to a quantitative one. A straightforward instrument for a multidimensional approach (i.c. sustainability) is a suitable set of indicators. They are used to identify and monitor a change over time towards a certain target. Indicators also play a huge role in communication with and among stakeholders.

One of the most important reasons why sustainability concepts were demanded by its early defenders like the Brundtland Commission and other people was the perception that the natural environment (air, water and soil) should be protected. The carrying limit of natural systems was considered and estimations were done to predict when they would be reached.

The above mentioned carrying capacities, like the cumulative emission of greenhouse gases to limit climate change to an acceptable level, might subsequently be used to assess sustainability of developments of the system under scrutiny. This fits into the second aim of the PhD: identifying factors within the system disturbing the balance in order to restore the equilibrium and increase sustainability.

1.4.2 Environmental indicators

Sustainability should contain at least interacting economic, social and environmental factors. Progress towards higher sustainability obviously requests to direct policy attention to all three factor types. There is however no agreement on the fact whether existing economic and social indicators (e.g. GDP, unemployment index, …) are useful in identifying the progress toward sustainable development. So far, no consensus is formed on indicators of sustainable development. As such, many highly aggregated economic and social indicators have been widely adopted. However, there are virtually no comparable national environmental indicators to help decision makers or the public evaluate environmental trends (Hammond et al., 1995).

Wherever human activities are treating the environment as a sink, there is a big need to measure emissions and wastes. Such activities might degrade the environment in several ways, creating local, regional or global impacts. The focus here is on phenomena changing the character or health of the earth's physical and biological systems. Climate change (related to emissions) and the disposal of solid waste very well fit into this.

Climate change

Greenhouse gas emissions change the composition of the atmosphere on earth. As a consequence extra heat is trapped by the earth, increasing the chance to global warming. The main greenhouse gases are carbon dioxide (CO_2), methane (CH_4), nitrous oxide (NO_x), chlorofluorocarbons (CFCs) and halons. Emissions of these gases, increase the warming potential of the atmosphere. How much greenhouse gas emissions contribute to the global warming potential depends upon the retention time in the atmosphere before being removed or broken down into other compounds and moreover how well the heat is radiated by the earth. This is covered by the Global Warming Potential (GWP) for each gas. The GWP is used as a weighting factor for those gas emissions. The weighted summation of the discharge of CO_2, CH_4, NO_x and N_2O expressed as CO_2 equivalents, forms the climate change indicator. CO_2 aggregations are also known as the carbon footprint (see also section 1.5). This is a first sustainability assessment tool, which will be elaborated in chapter 2: carbon footprint as a sustainability performance measure.

One major guiding principle of EU policy is the "20-20-20 target", being an example of environmental sustainability: 20 % reduction of greenhouse gas emissions, 20 % share of renewable energy and a 20 % rise in energy efficiency by the year 2020, compared to 1990. Actions are set in order to reach the targets, by means of a climate-energy package.

The research gap with regard to carbon footprint is a methodological one. Scientific literature on the subject of carbon footprint is scarce and most studies were carried out by private organizations and companies due to their sense of business rather than from a standpoint of environmental responsibility (Kleiner, 2007; Wiedmann and Minx, 2007; East, 2008). Moreover, there is very little uniformity in the definitions of carbon footprint within literature and available studies (Wiedmann and Minx, 2007). Finally, there is a lack of uniformity over the selection of direct and embodied emissions (Pandey et al., 2011).

Disposal of solid waste

Disposing of solid waste takes into account the collection, treatment, processing, recycling, reuse and incineration. The European policy is mainly there to prevent waste creation (according to the waste management hierarchy). If waste products do arise, the main goal is to create a shift from dumping and incineration to recycling and reuse.

Many policies address the waste problem. The problem of increasing waste generation has not been addressed a lot (Persson, 2007).Therefore it is of utmost importance to identify those factors having an impact on the generation of residual household waste, a certain waste fraction. Decreasing its generation is important to design waste management plans providing the possibility to reach a certain waste target: they are needed to define a strategy for improving the sustainability related to waste production: decreasing waste generation and increasing recycling.

Targets and indicators are helpful in improving sustainability. Objectives to prevent the quality of ecosystems from surpassing critical threshold levels, beyond which the benefits to humans fall to unacceptable levels, are often set as policy targets, for example, reducing deforestation, halting biodiversity loss, mitigating and adapting to climate change and improving sanitation. But to be effective, targets must be specific and linked to relevant sustainability indicators. Indicators provide current information about the health of the environment and allow the 'distance' or 'proximity' to the target to be measured, for example, percentage of forested area, proportion of species threatened with extinction, CO2 emissions and proportion of population with access to clean water.

The EU '20-20-20' policy is a successful example of applying sustainability targets and indicators. Its objective is to achieve a 20% reduction in greenhouse gas (GHG) emissions by 2020 (compared with 1990 levels) while achieving a 20% increase in energy efficiency and ensuring that a 20% share of energy consumption originates from renewable sources.

1.5 Conceptual framework

The main concepts of this PhD dissertation are summarized below into a conceptual framework. The literature review on one hand and the theoretical concepts described so far in the introduction chapter on the other hand, provide a strong basis for selecting few key concepts (see figure 2). The concepts form the core of the empirical work carried out in this PhD dissertation in order to comply with the objective: revealing whether sustainability can be expressed as 1 indicator and whether this is necessary, or that the concept remains a matter of monitoring given its dynamic nature.

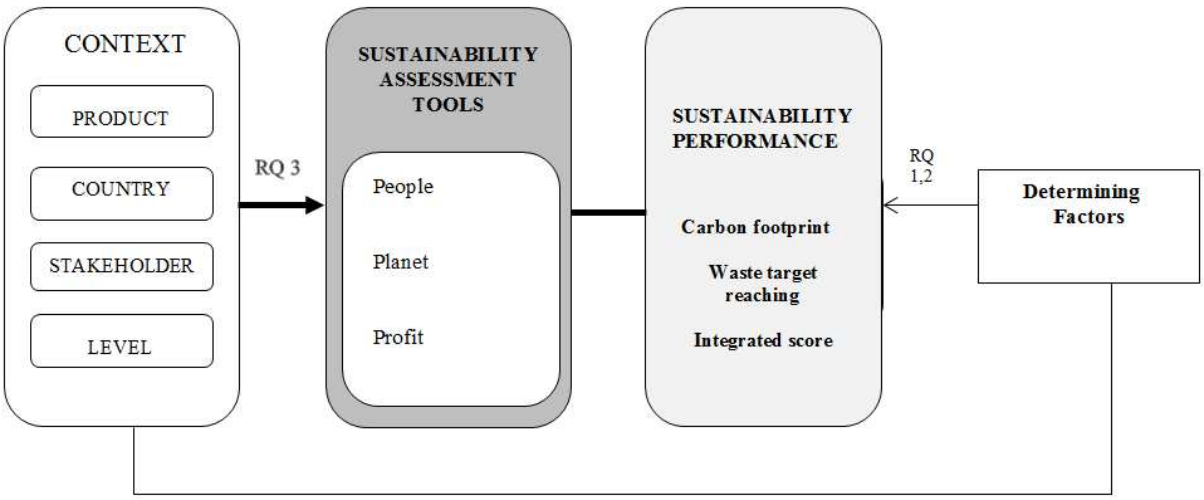

Figure 2. Conceptual framework

The **Context** determines which sustainability assessment tool needs to be followed: single dimension or integrated including all dimensions (3 P's).

A Product is anything that can be offered to a market to satisfy needs. In the agribusiness one can think of food products (pigmeat, beef, poultry, milk, …).

Country refers to a geographic location where the assessment takes place.

A **stakeholder** is a person, group or organization that has interest or concern in an organization. Stakeholders can be involved in the selection of the framework of sustainability indicators to be used during the assessment. Stakeholder involvement during the setup phase determines the type of assessment tool. Their preferences with regard to what should be included into an assessment is captured. Sustainability indicators are selected based on their involvement. Moreover, they can be involved during the weighting of indicators as well in order to reveal preferences.

Level refers to the level in the chain (e.g. farm level) for which the assessment is conducted.

A **sustainability assessment** concerns the guiding concepts, frameworks and information sets suitable for decision support as the scope is broadened from narrowly defined economic issues, to the planet's ecosystems and the long term (O' Connor, 2010). Sustainability assessment is being increasingly viewed as an important tool to help in the shift towards sustainability (Pope et al., 2004). From a policy point of view, it can be seen as a process by which the implications of that policy on sustainability are evaluated.

In literature, sustainability assessment is seen in the light of impact assessment processes, related to Environmental Impact Assessments (EIA) for projects and Strategic Environmental Assessment (SEA) for policies and other programmes (Devuyst, 2001). Assessments are either EIA, EIA-driven or objectives-led SEA (Partidario, 1999; Eggenberger and Partidario, 2000; Sheate et al., 2001, 2003). EIA-driven assessment is a reactive, ex-post process with the aim to evaluate environmental impacts of a policy, plan or programme in order to evaluate the acceptability of the impacts and to identify potential improvements in the environmental outcomes (Sheate et al., 2001, 2003; Sippe, 1999; Devuyst, 1999). Extension of environmental assessment processes with the aim to include all pillars of sustainability can occur in all 3 types of environmental assessments. Approaches to sustainability assessment are thus either EIA-driven integrated assessments or objectives-led integrated assessments (Pope et al., 2004). The integration of environmental, social and economic aspects is what Scrase and Sheate (2003) call 'integration among assessment tools' which will be applied in this PhD dissertation.

Sustainability Assessment Tools can be very helpful to assess what progress is being made in terms of sustainability, and what still needs to be done to reach a certain target.

Depending upon the context, the current **sustainability performance** is being measured with the result provided as either carbon footprint, reaching a certain waste target or an integrated sustainability performance score. As such, this PhD dissertation includes three different sustainability assessment and sustainability management tools. Moreover, the key factors (**determining factors)** in the environmental assessments bringing about unsustainability are determined in order to find a pathway to higher sustainability and to progress towards sustainable development. They also depend upon the defined context.

Appropriate sustainability indicators need to be selected in order to measure the system performance from any impact in the dimensions under assessment. The sustainability assessment tools within this PhD dissertation are applied to cases. Indicators must have a policy relevance, be analytically sound and measurable at a reasonable cost.

Given its dynamic nature, sustainability needs to be assessed over time (monitored) to detect the progress towards a sustainable state. Literature shows that sustainability concepts and evaluation tools are most of the time focusing on one of the three aspects at once (Van der Vorst, 2010). The ecological footprint has proven not to be an appropriate tool for sustainability measurement.

It cannot be emphasized enough that the different aspects of sustainability are not separate entities, but are interlinked. However, concepts and evaluation techniques integrating all three aspects at the same time are not available (Van der Vorst, 2010). In order to assess total sustainability, a combination of these concepts and measuring tools should be done to get the full sustainability picture of products, processes, sectors or even society as a whole (Van der Vorst, 2010).

An assessment is integrated when it presents a broader set of information than is normally derived from a standard research activity. Because integrated assessments bring together and summarize information from diverse fields of study, they are often used as tools to help decision makers understand very complex (environmental) problems.

The indicators of the **three Ps of sustainable development.** The dimensions (3 P's itself) have been elaborated in section 1.2 and 1.3 above.
Each of the dimensions consist **of indicators** to be used during sustainability assessments to indicate what is being measured. Several organizations and institutes defined indicators related to economic, environmental and social sustainability. The Global Reporting Initiative (GRI) for instance is a non-profit organization that developed several indicators which are today the world's most widely used sustainability criteria.

Indicators in the **economic dimension** are for instance economic performance, market presence, economic impacts and entry barriers. For the **environmental dimension** it relates to materials, energy, water, biodiversity, emissions, effluents and waste, transport, … the **social dimension** mainly covers labor practices, human rights, health and safety and product responsibility.

A **footprint** is one possible outcome of the sustainability performance assessment and is a quantitative measurement describing the appropriation of natural resources by humans (Hoekstra, 2008). It describes how human activities might impose several types of burdens and impacts on global sustainability (UNEP/SETAC, 2009). Footprints are mostly being measured in area units. This implies high variability and the chance for possible errors regarding the results (Cucek et al., 2012). Converting into a land area should be based on a large number of assumptions increasing the uncertainties related to a particular footprint (Wiedmann and Minx, 2008; Lenzen, 2006). For non-area based footprints it might be problematic to convert some processes to units of area. The ecological footprint is a composite indicator and emerged as the first indicator for measuring (environmental) sustainability. It can be seen as a measurement of the human demand for land and water areas, comparing the human consumption of resources and absorption of waste with the earth's ecological capacity to regenerate (GFN, 2010). It should be noted that the ecological footprint does not take into account all environmental concerns, and excludes social and economic indicators. The ecological footprint is measured in global area units per person. Converting the data to area units can however be problematic. Moreover, there is limited data availability, uncertainty about the data and geographic specificity. This provides another reason for the fact that the ecological footprint (expressed in units of area) is not a good indicator for (environmental) sustainability (see also section 1.2).

During the last years, the **carbon footprint** became one of the most important environmental protection indicators (Wiedmann and Minx, 2008; Lam et al., 2010; Galli et al., 2012). A carbon footprint takes into account the amount of CO_2 and other greenhouse gases (methane, nitrous oxide), which were emitted along the full life cycle of a process or product (BSI, 2008). The carbon footprint is quantified by using an indicator as the Global Warming Potential (GWP), representing the amount of greenhouse gases contributing to global warming and climate change. A time horizon of usually a 100 years is taken into account (IPCC, 2009). Wiedmann and Minx (2008) have proposed that the carbon footprint stands for a measurement of the direct (on-site, internal) and indirect (off-site, external, upstream and downstream) CO_2 emissions of an activity, or over the life cycle of a product, expressed in units of mass.

With growing awareness regarding climate change, a huge concern has grown in individuals over their responsibility of contributing to greenhouse gas emissions (Pandey et al., 2011). Achieving sustainable development in the agricultural sector can be established by limiting agricultural greenhouse gas emissions (non-ETS) in order to reach a stabilization of them (Dalgaard et al., 2011). It proves the need for evaluating the current situation, and assessing where the agricultural production system needs improvements (Eriksson et al., 2005). In a way, only things which are measurable are manageable. Mensuration of greenhouse gases of products and processes is ongoing, and expressed as the carbon footprint. It is an important tool for the management of greenhouse gases (Pandey et al., 2011). Carbon footprinting is a strong tool for identifying hotspots in the livestock production chain. Carbon footprint reporting to a third party or disclosing this information to the public is needed in return to legislative requirements, or carbon trading or moreover as a part of corporate social responsibility, or for improving the image of a brand (Carbon Trust, 2007; L.E.K. Consulting LLP, 2007). At the same time it should be noted that a carbon footprint is limited in its scope regarding environmental sustainability. It is a very partial environmental indicator only focusing on the air compartment of the environment and solely taking into account greenhouse gases.

1.6 Research questions

The final aim of this PhD dissertation is to identify whether it is possible and necessary to express sustainability as 1 integrated indicator. In order to reach this objective, based on the conceptual framework, a total of three main research questions are formulated based upon the sustainability assessment in place. Each of them is related to the sustainability topic of interest: climate change impact-carbon footprint, residual household waste and integrated sustainability performance assessment of the soy chain. The three key research questions are formulated in a broad way, and hence specific sub-questions are needed to investigate specific relationships, to analyze issues in terms of methodology and moreover to put the research questions in a much broader context. Some concepts might not be present in the framework, but will occur in the research questions.

RQ 1. Which determining factors making up the carbon footprint with regard to beef and pigmeat production in Flanders are – upon mitigation – able to contribute to progress towards sustainability ?

RQ1A. What is the level of carbon footprint in absolute terms for beef and pigmeat in Flanders ?

RQ1B. What are the hotspots (highest impact production stages) with regard to carbon footprint for beef and pigmeat in Flanders causing unsustainability ?

RQ1C. What is the impact of changes in feed and herd characteristics on the carbon footprint of pigmeat and beef in Flanders ?

RQ1D. What are the opportunities to reduce the carbon footprint as improvement options to initiate a development towards a more sustainable production, based upon the hotspots found for pigmeat and beef in Flanders ?

Footprints go hand in hand with Life Cycle Assessment (LCA) (Weidema et al., 2008). LCA is a structured, internationally-standardized tool (ISO 14040 and 14044) for quantifying those emissions, resource consumption and other environmental impacts associated with processes, products or activities over their complete life cycle (Cucek et al., 2012). Many studies are however not clear whether the carbon footprint number actually includes the full life cycle. Thus, one of the key issues in carbon footprinting calculations are the system boundaries. These different approaches in terms of methodology make it impossible to compare carbon footprints between products and sectors.

In order to initiate a development towards more sustainable production one needs to find out where along the production chain improvements can be made. As such, all related emissions need to be quantified. One such method is to calculate the Carbon Footprint (CF) of the livestock product (Espinoza-Orias et al. 2011). Within the CF, all GHG emissions (carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O)) are combined and expressed as CO_2 equivalents. Of all the major sources of world meat production, pigmeat represents the highest proportion (37%) compared with poultry (33%) and beef (23%) (FAO 2008). Flanders is a key player in pigmeat production, but should however at the same time comply with the EU 2020 target. Especially since the agricultural sector is a non-ETS sector. In view of the importance of pigmeat in terms of world consumption and livestock production, a study is needed to calculate the CF of Flemish livestock produce in general and pigmeat in particular.

The second case relates to household waste. Sustainable household waste management and recycling systems have the aim to reduce the amount of natural resources consumed, and at the same time to ensure that any resources already taken from nature are reused many times, and that the amount of waste produced is kept to a minimum. The processing of waste plays a key part in this. It is therefore important to identify factors contributing to waste generation and to keep this generation to the lowest level possible.

Waste management plans play a key role in achieving sustainable waste management. Their purpose is to give an outline of waste streams and treatment options. More specifically their aim is to provide a planning framework compliance with waste policy and target achievement. Waste management plans are important instruments contributing to the implementation and achievement of targets set up in the field of waste management at the national and the European Union level (Commission, 2003). The waste hierarchy is the major guiding principle for waste management policies (Lang et al., 2006; Powell et al., 2002). In the Flemish region of Belgium, the implementation plan household waste (OVAM, 2002) incorporates all European and regional requirements and describes strategy, goals, actions and instruments for the collection and treatment of household waste. The plan is binding to local authorities unless otherwise stated (Article 36, (Afvalstoffendecreet, 1981). The central mandatory goal and target for Flemish municipalities is to reduce the amount of residual waste to 150 kg/capita/year by 2007 (OVAM, 2002) and maintain it during the 2007-2015 period.

Municipalities are responsible for the collection of household waste and have some degrees of freedom in designing local waste management plans. Nevertheless, they have to adhere regional waste management plans and European directives, setting goals and limiting the use of policy instruments. The local authorities as a policy level stand closest to households, and therefore are sensitive to avoid societal failure. Societal failure arises when policy measures and infrastructure are inadequate to facilitate the behaviors they may wish to exhibit (Bisson, 2002). Local authorities want the waste management scheme to facilitate participation by their citizens, without putting too much of a burden on them and without losing sight on mandatory targets. Policy recommendations to local authorities include the provision of bespoke recycling services to suit the variety of residential conditions. A reasonable body of information has been published, but the information is complex, often contradictory and difficult to interpret, giving significant problems to those responsible for developing waste strategies and policies (Martin et al., 2006).

RQ2. What are the main characteristics of residual household waste generation able to define a strategy for improving the sustainability related to waste production in Flanders ?

RQ2A. What are the key/determining factors having a significant impact on reaching the mandatory goal of 150 kg residual household waste per capita per year ?

RQ2B. Is there a significant relationship between the generated amount of residual household waste and a) income per capita, b) the cost of residual household waste collection and c) the collection frequency ?

RQ2C. Is there a significant difference in the cost-efficiency between private or public collection of residual household waste ?

RQ2D. What factors determine the decision to opt for public versus private waste collection services ?

RQ 3. Can sustainability performance be expressed in an overall sustainability score, integrating all dimensions ?

RQ 3A. What are the different qualitative and quantitative indicators encompassing the sustainability performance measurement framework of Latin America-EU soy chains ?

RQ 3B. What is the current performance and stakeholder perception of Latin American EU soy chains with regard to sustainability, covering all dimensions ?

RQ 3C. Is there a difference in sustainability performance with regard to the dimensions of sustainability for different soy production systems ?

From the extraction of natural resources to the recycling and disposal of waste at the final stage, from cradle to grave, a product's environmental, social and economic impact can be monitored throughout its supply chain. By means of an LCA (Life Cycle Assessment), the environmental impact of a product or process is measured. The assessment of sustainability however covers more than 1 dimension. Assessing sustainability performance of a product involves the simultaneous implementation of three life cycle techniques. Previous efforts were done by combining 2 LCA techniques (Saling et al., 2002; Itsubo and Inaba, 2003; Poulsen and Jensen, 2004; Franze and Ciroth, 2004). Efforts to combine 3 LCA techniques have been done in the meantime as well (Traverso et al., 2012; Vinyes et al., 2011; Halog and Manik, 2011; Spoerri et al., 2011; Vinyes et al., 2011). The concept to combine 3 LCA techniques (one for each sustainability dimension) was introduced by Klöpffer (2008), followed by

Finkbeiner et al. (2010) and can be described as follows: LCSA = (environmental) LCA + LCC (Life Cycle Costing) + social LCA. Life Cycle Sustainability Assessment (LCSA) evaluates all environmental, economic and social negative impacts and positive benefits of a product along the whole life cycle. Results can help to back up processes in decision making (Finkbeiner et al., 2010).

It is therefore acknowledged that LCSA is a technique to assess sustainability performance of a product. The scope however is not to discuss the current state of the art regarding LCSA. Life cycle techniques are able to give information in order to manage the value chain and social responsibility of an organization, from cradle to grave, addressing the triple bottom line dimensions of sustainability.

LCA as a technique however, makes use of quantitative indicators. In general, environmental LCA and economic LCC use quantitative data. However, issues on social LCA (and also sensitive economic and ecological conditions are hard to capture in 1 or more figures) are somewhat more challenging to quantify. The nature of some indicators requests a more qualitative approach, hard to cover with an LCA. This is not tackled in current LCSA, which is for the most part quantitative or to some extent semi-quantitative.

Indicators are acknowledged as a very important tool in decision making processes and communication to a broad audience. The main characteristic of indicators is their possibility to wrap up, focus and underpin the huge complexity of the dynamic environment we operate in. They are able to narrow it down to a reasonable body of information (Kumar Singh et al., 2009). By visualizing phenomena and emphasizing trends, indicators quantify, analyze and communicate information which would be difficult to interpret (Warhurst, 2002).

Sustainability is very complex, so it enables a very large number of perspectives upon it, as it is harder to prove any of them wrong in simple terms. The fact that sustainability is not precisely defined and agreed upon does not mean it cannot be used and assessed as a concept. The extensive use of the concept and the variety of actors committed to it, motivate this thesis on the demystification of the discourses and perceptions of stakeholders on sustainability.

In the context of increasing concerns regarding the sustainable production of animal feed and food for humans, a debate has emerged with respect to the transatlantic export of soya from Latin America to Europe.

1.7 Research contribution

This PhD dissertation shows a major research contribution, referring to the following aspects:

- Adaptations and moreover improvements in Life Cycle Sustainability Assessment methodology: quantification of primary data into a single sustainability index for different production systems.

- The combined empirical research focus on both the single sustainability dimension level as well as the integrated sustainability approach, covering all dimensions and composed of multiple individual assessments. This approach is applied to several cases most suitable for the assessment in place. The household waste management, Flemish pigmeat and the Latin American EU export of soy can be considered as cases for an integrated and non-integrated sustainability assessment. No other study was found using such approach.
- High relevance to stakeholders, policy and decision-makers of any kind.
  e.g. waste management and environmental policy makers in Flanders
  stakeholders interested in sustainable sourcing of raw materials in food production

This thesis has two major scientific contributions, namely an empirical and a methodological one. The latter lies in the fact that sustainability performance and assessment tools are developed and tested based upon a number of cases.

As such, many tools at present do not address the multi-functionality of the sustainability concept (Rossing et al., 2007), whereas the integrated sustainability tool developed and presented in this dissertation takes into account all dimensions.

There are clear advantages of sector-specific research for a better understanding of the performance, accompanying contextual factors. Cases were chosen to best fit the appropriate sustainability assessment and to contribute to the bottom up approach in order to obtain an integrated holistic sustainability assessment. This doctoral research uses an integrated approach. This integrated approach provides opportunities for future inquiries on setting case-based targets.

For environmental sustainability, well established tools like LCA are available. For the economic and social dimension, there is a clear need for robust indicators and methods. Life Cycle Sustainability Assessment is at the early stage of implementation, results can contribute to further development of this methodology.

Next to the integrated and more theoretical perspective, the contributions of this PhD dissertation can be accommodated towards theoretical, empirical, methodological, managerial and policy relevance.

1.7.1 Theoretical contribution

This PhD dissertation mainly aims at contributing to the theory of the hot topic of sustainability and its measurement. More specifically, it contributes to ‘theory building’ by scrutinizing the possibility and necessity of expressing sustainability as 1 indicator. At the same time it contributes to ‘theory testing’ by applying the developed sustainability tools. Moreover, these tools are applied as managerial instruments in order to identify factors causing unsustainability with the aim to remove them and achieve higher sustainability.

1.7.2 Methodological and empirical contribution

The methodological contribution mainly relates to the development and application of three different sustainability performance and sustainability management tools. Tools are provided to help in future sustainability assessments, to identify whether the implications of certain initiatives bring about the necessary changes in the direction of sustainability.

Sustainability is a complex, multidimensional and contested concept. For environmental sustainability, well established tools like LCA are available. For the economic and social dimension however, there is a clear need for robust indicators and methods. Life Cycle Sustainability Assessment – as implemented in the final tool - is at the early stage of implementation, results of this dissertation can contribute to further development of this methodology.

This thesis is contributing to empirical research by, first of all, producing transparent sustainability assessments, which remains a challenge. Second, by contributing to scientific insights in the role of sustainability assessment with regard to sustainable development. It will be examined whether policy targets are able to help in contributing to reach sustainable development. Third, Sustainability indicators are a way to compile and structure knowledge and to express societal and political norms and preferences. While this dual role has been acknowledged by many people writing on this topic, it is less clear in how far this duality has been acknowledged in sustainability indicator development processes. Therefore, the socio-political dimension (normative and stakeholders' preferences frameworks) needs as much recognition and consideration as the knowledge dimension during the design and implementation of sustainability indicators. This gap is being tackled by including stakeholder perceptions into the integrated sustainability assessment.

In short, the objectives of the assessment are to: 1) address lacuna in literature on social and economic aspects of commercial soy production following a chain approach, 2) develop an assessment approach incorporating all sustainability dimensions, 3) deliver a subsequent tool useful to stakeholders for conducting future sustainability assessments.

A major source of added value to empirical research is the inclusion of information from different countries in general and subsectors in particular. This dissertation includes Latin America (Brazil/Argentina) and the Flemish region of Belgium, Flanders. The covered sectors are the agribusiness and waste collection/processing sectors. Selection of the included products occurred based upon their socio-economic importance (turnover and consumption rates). Section 1.6 elaborates on the scope and relevance of the topic including the different research locations.

1.7.3 Managerial and policy relevance

Too many sustainability indicators are just meaningful to scientists and people in the field, but not necessarily to policy makers and a broad audience. No adequate communication and no sustainability index is set. During the literature analysis, no other study neither came up with a one-metric sustainability score, nor used the same methodology for indicator selection.

The conducted research also tries to show practical relevance for policy makers on one hand and practitioners on the other hand.

The European Lisbon treaty has revealed a strategy to increase the competitiveness of the region. The transition to a knowledge-based economy has been acknowledged as the key factor for obtaining this goal. Looking at the agrifood sector, the Knowledge-Based Bio-Economy (KBBE) has been set as the core target. Bioeconomy is defined as an economy based on biomass, resulting in food (cfr. pigmeat and soy). This dissertation provides a tool for integrated sustainability performance assessment in selected agribusiness chains. It helps in defining sustainable production for these kinds of chains, thus on a case-based basis. The tool is hence applicable for these cases, but can set the tone for others and serve as a leverage.

Besides the integrated sustainability assessment tool, this PhD dissertation provides 2 sustainability management tools: carbon footprint and sustainable waste management. In order to design pathways to increase sustainability, it is necessary to reveal the factors for managing sustainable systems. This is especially important in the light of the Europe 2020 strategy for smart, sustainable and inclusive growth.

Sustainability is an attractive topic for problem analysis and is an area of study where the use of indicators can be useful. It is 1) comprehensive and inclusive, 2) simple in concept and flexible, 3) value-based and 4) an approach that makes consistency necessary across different areas of policy (Shields et al., 2002).

When sustainability indicators are in place, one wants to measure them, by both quantitative and qualitative techniques (Moldan et al., 2012). Albeit absolute values do not completely matter, one wants to have a notion of what is an acceptable level of sustainability. People need an assessment of how far they are from sustainable targets: “what is the distance to target ? ” (Stieglitz et al., 2009). This is much more straightforward on the individual dimension level. Environmental sustainability for instance (compared to social and economic sustainability) is more open to define and set targets which are incorporated in the biophysical properties of the system. Sustainability indicators should be linked to reference values and targets (Moldan et al., 2012). Indicators are meant to cover the 3 pillars of sustainable development. Once policymakers commit to and start planning for sustainable development, they make decisions in support of sustainability goals. Decisions should be informed by science that enhances understanding of systems, and identifies targets for desired levels of performance (NRC, 2000).

1.8 Scope and relevance of the topic

Exchanging and disclosing information is a crucial element for improving sustainability in complex value chains like the transatlantic soy chain (Wognum et al., 2011). Transparency should bring about a certain transformation towards higher sustainability. In order to succeed in transparency, 2 specific prerequisites should be met: users should get easy access to the disclosed information and should be able to use and understand it in a proper way (Mol, 2013). Moreover, chain actors should be responsive and vulnerable to complaints regarding poor sustainability performance (Mol, 2013). Both prerequisites are not always met. Consumers as well as the public domain in developing countries encounter difficulties in accessing and applying sustainability information about products and production in international value chains (Mol, 2013).

Disclosing the right environmental information does not just depend upon the receiver side of the story. It can sometimes be part of a conscious tactic to disclose information as part of a green washing strategy. Disclosed information is often made too complex and abstract (Mol, 2013). Creating global value chains encompasses sustainability challenges. According to bilateral EU-Latin American cooperation agreements, there is a mutual interest in developing strategies to tackle Latin American sustainability challenges.

Next to climate change, municipal waste is relevant to study because it is less researched from an instrument mix point of view than climate (see e.g. Sorrell and Sijm, 2003; Holm-Pedersen, 2003; Smith, 2004) and agri-environmental policies (see e.g. Daugbjerg, 1998ab, 1999). While municipal waste has been a topic on the environmental policy agenda for some time, the real problem of increasing (household) waste generation has not been tackled to a large extent. In many EU countries, it is estimated that per capita municipal waste generation almost doubled during the 1985-2000 period, from 300 to 540 kg (Commission of the European Communities, 2003). According to the OECD, this amount might further increase to 640 kg per capita per year by 2020 in OECD countries. As such, it suggests that policy instruments used so far have not been effective in limiting waste generation.

In many EU countries (including Belgium) waste management policies seemed not so successful, as waste generation kept on growing. Waste management policies at the EU level are implemented at the national level of each member state. Waste prevention and reduction are high priorities in the waste management hierarchy. Based upon this hierarchy, ambitious waste management targets emphasizing waste recycling have been defined (Lang et al., 2006). In Flanders, the OVAM (Waste Authority) is responsible for drawing up sectoral waste management plans. Chapter 3 provides a sustainability management tool with regard to household waste management. As such there is no impact assessment of waste but of the factors influencing the possibility to reach a certain waste target (i.c. 150 kg per capita per year). Identification of these factors is needed to define a strategy for improving the sustainability related to waste production.

Determining factors of household waste collection performance

Local authorities have a variety of instruments at their disposal to influence the total waste production of households. To what extent they are used by local authorities varies greatly. Even within Belgium there are great differences among municipalities. When working out a local waste management plan, municipalities have to take into account several variables. A group of external variables is not in control of local authorities. Those variables describe the socioeconomic background of the municipality and might influence the amount of waste generated and recycled.

Two groups of controllable variables can be discerned. A first group describes how the municipality uses pecuniary incentives (e.g. fees). A second group of variables describes the service level provided by the local authorities (Gellynck & Verhelst, 2007). Decisions on pecuniary incentives and scheme design can influence the amount of waste generated and recycled.

Authorities can use incentives and prompts to endorse pro-environmental behavior. Hopper and Nielsen (1991) found that incentives can influence recycling behavior. But De Young (1986, 1990) found they do not create long-lasting behavioral changes and do not last when the economic incentives are withdrawn.

Instruments can either be controllable or uncontrollable for municipality waste managers. Both types are elaborated below.

Socioeconomic background (Uncontrollable)

Results about the impact of socioeconomic factors are equivocal (Tucker et al., 1998). Findings on sociodemographic factors related with participation in recycling schemes are rather inconsistent. Even when significant relationships have emerged, the variance explained by these variables is rather small (Hornik et al., 1995). Common socioeconomic factors are among others income, population density, household size, single versus multiple family dwellings, age and ethnic background.

Scheme design (Controllable)

Curbside recycling schemes are complex systems with different parameters, each of which can influence overall performance (Harder et al., 2006). The design of any scheme for the separation and recovery of recyclables from the domestic waste stream must facilitate householders' use (Mattsson et al., 2003; Perrin and Barton, 2001; Price, 2001). The convenience of curbside recycling is one of the main factors influencing participation (Barr et al., 2003; Boldero, 1995; Coggins, 1994; De Young, 1990; Do Valle et al., 2004; Domina and Koch, 2002; Everett and Peirce, 1993; Ewing, 2001; Guagnano et al., 1995; Noehammer and Byer, 1997; Perrin and Barton, 2001; Thomas, 2001; Tucker and Speirs, 2003; Vining and

Ebreo, 1992; Woodard et al., 2005). However Martin et al. (2006) found that convenience of curbside recycling is not necessarily a major factor in recycling participation.

Convenience is a multi-facet concept. Several indicators have been distinguished. Recycling schemes should recognize convenience in terms of the householders' contributions of time and effort expended in cleaning and sorting materials (Bruvoll et al., 2002; Burnley and Parfitt, 2000; Perrin and Barton, 2001; Tucker and Speirs, 2003). Furthermore, required storage space is an issue (Barr et al., 2003; Boldero, 1995; Jakus et al., 1997; Knussen et al., 2004; Martin et al., 2006; McDonald and Oates, 2003; Mee et al., 2004; Perrin and Barton, 2001; Robinson and Read, 2005).

Pecuniary incentives (Controllable)

Most municipalities financed solid waste collection services either with local property taxes or with fixed monthly or quarterly fees. Charging residents a fixed fee for waste collection services provides no financial incentive to minimize the total amount of waste. The cost of contributing one additional bag of garbage to the household is zero, which suggests households will generate more waste than is socially desirable (Kinnaman, 2006). However, McFarland (1972) finds an inelastic price elasticity of demand for waste collection services of –0.455 based on differences in fixed fees between municipalities.

Economic literature devoted to designing waste management policies to achieve the efficient quantity of waste and recycling argues that municipalities should charge according to marginal costs to maximize economic efficiency (Porter, 2002).This can be achieved with a direct tax on household garbage. Households paying the well set unit-based charge internalize all social marginal costs of their garbage production. In response to this fee, households could reduce the amount of garbage they generate or divert some materials for recycling (Kinnaman, 2006). In practice, communities applying some form of unit-pricing usually turn to average cost pricing that sets the unit-price equal to the average total cost per unit (Miranda et al., 1996).

Two common implementation methods of unit-pricing are found. The first type, often called the bag/tag program, requires households either to purchase specific garbage bags, or purchase stickers or tags to affix on each of their own garbage containers. Only garbage identified with the bag, sticker, or tag is collected. The second type of program is a weight-based system. Garbage trucks are fitted with scales, and collectors weigh each household's garbage and bill that household accordingly. These weight-based systems eliminate the incentive for households to reduce garbage collection expenses by compacting waste into fewer containers. This is not particularly helpful since most garbage trucks compact household waste anyway (Kinnaman, 2006).

Studies on the change in disposal behavior by households facing unit-based pricing programs consistently estimate the demand for garbage collection services to be inelastic (Dijkgraaf, Gradus, 2004; Fullerton, Kinnaman, 1996; Hong, 1999; Jenkins, 1993; Kinnaman, Fullerton, 1997; McFarland, 1972; Miranda et al., 1994; Morris and Holthausen, 1994; Podolsky, Spiegel, 1998; Van Houtven, Morris, 1999; Wertz, 1976). However, Hong et al. (1993) found that a user fee does not appreciably affect the quantity of waste produced at the curb. Efaw and Lanen (1979) found a high inelastic price elasticity of demand for waste collection services, if not perhaps zero or even positive in sign.

Weight-based fees represent the cost of waste disposal better than do volume-based fees, such as unit-pricing by the bag. They also provide a clearer and continuous pricing signal to household producers of waste. Volume based fees provide no additional waste reduction incentive below the lowest level of service, i.e. one bag or bin per collection round (Miranda et al., 1996). Some studies have examined the effects of weight pricing, and estimated changes in disposal behavior are roughly equal to those of the bag/tag studies (Dijkgraaf and Gradus, 2004; Linderhof et al., 2001). So, weight-based systems induce households to reduce garbage by about the same magnitude as bag/tag programs (Kinnaman, 2006).

The research focus of this PhD dissertation regarding household waste is limited to the examination of municipal waste policy and is not addressing policy instruments for industrial, agricultural, commercial nor hazardous waste.

Research location

Figure 3 explains how each sustainability concept relates to the cases. Each case is carefully chosen to fit the appropriate sustainability concept.

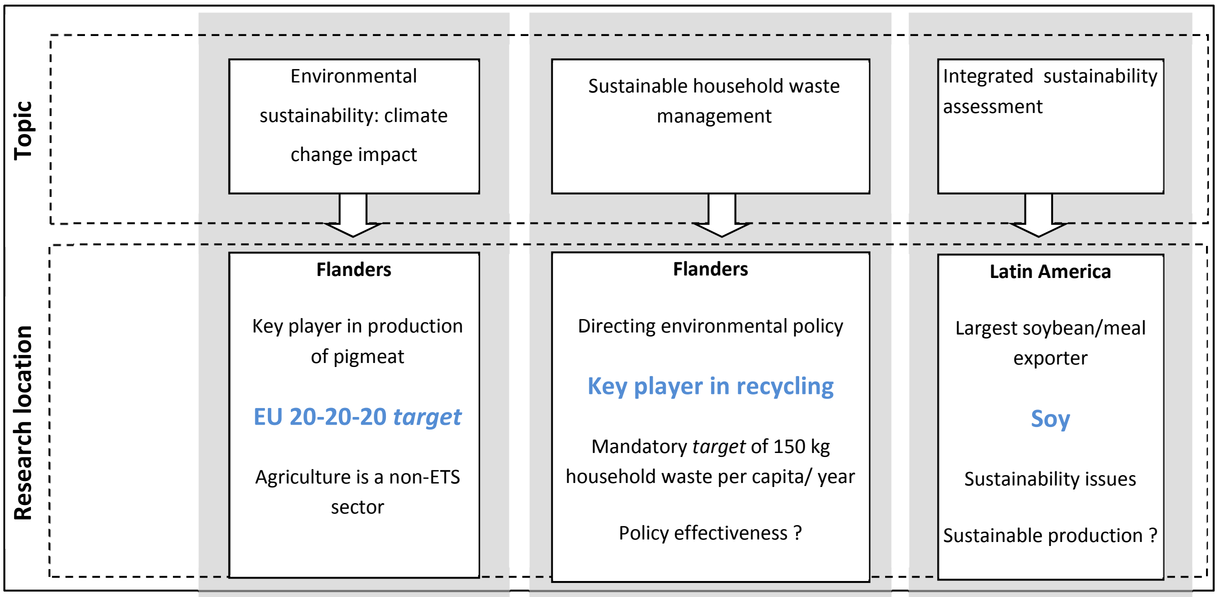

Figure 3.Focus and main lines of inquiry of the PhD manuscript.

EU 2020 target

Nowadays, one of the guiding EU principles is the "20-20-20 target": 20 % reduction of greenhouse gas emissions (compared to 1990), 20 % share of renewable energy and a 20 % rise in energy efficiency. Actions are set in order to reach the targets by 2020 in a climate-energy package.

Belgium is a federal country where every region is responsible for implementing EU environmental policies of any kind. There are two types of arrangements on the EU level: the EU Emissions Trading System (EU ETS) and policies covering the Non-ETS emissions. Sectors (energy-intensive industry) in the EU ETS are regulated at EU level, whereas Member States themselves are responsible to implement national policies and measures to limit emissions from the sectors covered by the non-ETS emissions. The EU 2020 target is converted into legally binding national targets for each individual Member State.

The national targets for emissions not being covered by the ETS (EU Emissions Trading System), are put under the Effort Sharing Decision (ESD). The ESD establishes binding annual greenhouse gas emission targets for Member States during the 2013–2020 period. These targets involve emissions from sectors not included in the EU Emissions Trading System (EU ETS), such as transport (except aviation and maritime shipping), buildings, agriculture and waste. The ESD forms part of the climate-energy package that will help move

Europe towards a low-carbon economy. Targets vary among countries and apply to each Member State individually. They can range from limiting the increase in GHG emissions to 20 %, to reducing them by 20 %. The national targets on greenhouse gas emissions use 2005 as reference year.

In 2013, Belgium performed 4 % better comparing 2012 emissions and 2013 targets under the ESD. However projections estimate that Belgium will fail to meet its national 15 % reduction target by 2020 in the sectors (e.g. agriculture) not covered by the EU emission trading system. In fact, there is a gap of 11 % between the 2020 projected non-ETS emissions versus 2020 ESD targets (European Commission, 2014). Negotiations on cooperation and burden allocation between the federal government and the regions in Belgium have not resulted in a clear distribution of efforts.

If Belgium needs to comply with EU2020 GHG targets, all regions of the country should gain insights into their climate change contribution in absolute terms. More specifically, the agricultural sector needs to monitor its activities for mapping the environmental impact in general and climate change impact in particular. There is a clear need for identifying hotspots in the production chain of Flemish livestock produce (beef and pigmeat) for installing ex ante mitigation measures. Carbon footprinting is a strong tool for doing so, and might be used as one of the indicators for sustainable development (Pandey et al., 2011).

Key player in Recycling

For societies moving towards sustainable development, new systems for the sustainable use of resources as well as for solid waste management should be developed. Efforts for reducing negative environmental effects caused by waste generation are amongst others: source separation, collecting, transportation, processing, and disposal of waste. Waste management is very much related to the needs of Maslow's pyramid. Fulfilling needs of people, is accompanied with a high(er) degree of consumption, leading to more household waste, which needs to be taken care of. Landfilling is restricted and minimization of waste generation is encouraged. Sustainable waste management should be able to save resources of any kind, to reduce and minimize the environmental impact and to establish a sound business model. Much of the household waste can be recycled. This is beneficial for the environment as waste is led away from landfills and can be used as secondary material. Moreover, recycling might be innovative and create jobs, which is beneficial for the economy.

Council Directive 75/442/EEC (Council, 1975) urges European Member States to take appropriate measures encouraging the reduction of waste production and ensuring waste is recovered or disposed of, without endangering human health and without using methods harming the environment or resources for future generations. For realizing these objectives, Member States need to make waste management plans.

The so-called waste hierarchy is laid down in Directive 75/442/EC (Council, 1975) and in the EU Strategy for waste management (Council, 1997). Waste prevention is given the highest priority, while optimum final disposal is placed at the bottom of this hierarchy (Figure 4).

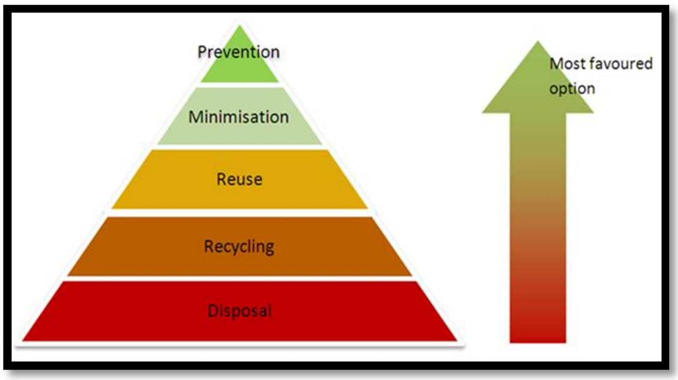

Figure 4. Waste management hierarchy

Source: (UNEP, 2013)

Nevertheless, there have been a number of challenges to the assumed ranking of management options within the waste hierarchy (Aadland & Caplan, 2006; Ackerman, 1997; Bruvoll et al., 2002; Kinnaman, 2006; McDougall et al., 2001; Porter, 2002; Rousakis & Weintraub, 1994). The European Council recognizes that for recovery operations, the choice of option in any particular case must have regard to environmental and economic effects, but considers that at present, reuse and material recovery should be preferred insofar they are the best environmental options (Council, 1997). Thus, there is a risk of government failure, implying that the chosen policy instruments do not adjust prices to their social marginal cost or are not cost-effective (Bisson, 2002).

Based on the waste management hierarchy, ambitious waste management targets have been defined, emphasizing the necessity of raising the amount of waste to be recycled (Lang et al., 2006). In the Flemish region of Belgium the government implemented Council Directive 75/442/EEC (Council, 1975) in the Waste Decree (Afvalstoffendecreet, 1981). The responsibility of prevention, recovery, collection and treatment is laid with the municipalities. Local authorities decide on the terms of waste collection (Article 6, § 1 and 2, (Afvalstoffendecreet, 1981). However, the regional government determines what types of waste need to be collected separately and can determine what kind of treatment options are open (Article 7, § 1 and 2, (Afvalstoffendecreet, 1981)). Specific household waste streams that have to be collected separately are stated in Vlarea, a decision made by the Flemish government (Vlarea, 2003). The OVAM (regional waste authority) is responsible for drawing up sectoral waste management plans. This PhD dissertation addresses the target effectiveness of the current household waste management policies for all municipalities in Flanders, the northern part of Belgium.

Transatlantic soy value chain

In the context of increasing concerns regarding the sustainable production of animal feed and food for humans, a debate has emerged with respect to the transatlantic export of soya from Latin-America. Soy is one of the main raw materials for the global feed and food industry. About 87 % of the global soybean production is crushed into roughly 80 % meal and 20 % oil (MVO, 2011). Soy meal is one of the main animal feeds in the world due to rich protein content and high performance diets (FAO, 2012). United States, Brazil and Argentina are the main soybean producers and the European Union is one of the foremost consumers of this feed. EU countries have to import large amounts of soybean meal to comply with feed sector demands. It is forecasted that the world meat consumption will experience a high rate of growth until 2020, due to demographic growth and increasing purchasing power (MVO, 2011), subsequently leading to a rising demand for soy meal as feed compound ingredient. In recent years sustainability concerns have been growing. The Latin American EU trade in soy, mainly for the EU feed and biofuel industries, has raised questions especially in the EU about its impacts. Criticism mainly involves the impact on global climate change and deforestation in Latin America. On the other hand the agribusiness is contributing to strong economic growth in a large number of Latin American economies. Soy export taxation in Argentina for instance provides financial resources to support public expenditures on social welfare (Severi and Zanasi, 2013). Evaluating a complex concept such as sustainability therefore requires a critical reflection of a wide range of social, economic and ecological issues. This is a condition for sustainability initiatives claiming legitimacy, authority and neutrality. In this regard, there is a common interest between Latin America and Europe to sustain the production and exportation rates of soy (SALSA, 2010).

1.9 Research design and structure of the thesis

1.9.1 Research design

Different research methods are applied to analyze the various parts defining the conceptual framework (figure 5). For answering the research questions, and to reach the objective of this dissertation, data were gathered through primary and secondary data sources. Analysis occurred by a combination of both statistical and systematic techniques like Life Cycle Assessment. The design of the research is shown in figure 5, and relates to the different studies conducted in the chapters. The published papers in each chapter provide a thorough overview of each methodology. Therefore this section solely provides a brief overview. As such three different sustainability assessments are conducted:

- Environmental sustainability: climate change impact (carbon footprint)

Both primary and secondary data have been used to model the meat production system through a chain approach. The data are representative for an average Flemish production system.
The emission factors are derived from approved Life Cycle Inventory (LCI) databases. The data on other steps in the chain (mainly manufacturing) were collected by means of interviews and primary data collection.

- Environmental + economic sustainability: household waste management

All 308 municipalities in the Flemish region were surveyed. For the data analysis, nine coastal municipalities were left out. Coastal tourism creates an extra amount of waste in a few months, which has a negative impact on the outcome of the model. The total amount of waste created does not reflect the amount of household waste originating from the native inhabitants. Furthermore, three other municipalities were also excluded as outliers. Their amount of residual household waste/capita was more than three standard deviations higher than the mean value for all municipalities. Finally, another municipality did not fully participate so the necessary information could not be gathered. Consequently this municipality
was left out. As a result 295 of the original 308 municipalities were included in the final analysis.
A structured questionnaire was compiled to obtain data on instituted pecuniary incentives and characteristics of waste management programs in place for the year 2003. Information was collected on the use of a flat rate fee, a pay-by-the-bag system or a weight-based fee, the type of residual waste and recyclable collection methods utilized both curbside and drop-off, the existence of composting programs and descriptive data on waste legislation. In a first step, data were gathered from municipalities' web sites and brochures on waste management. Then these data were verified and completed via telephone interviews with the competent public official of each municipality. To control for municipality characteristics that can possibly influence waste generation, data on this matter were gathered through the National Institute of Statistics (NIS). Included variables were average income/capita, the area of a municipality per inhabitant and the number of companies. The final analysis was done, using an empirical logit model.

- Integrated sustainability assessment: 2 parts need to be distinguished: performance assessment (objective) and the stakeholder perceptions (subjective)

In order to assess sustainability performance, one needs to define a sustainability framework, covering the most important indicators for the soy chain. To do so, a literature review was conducted, followed by stakeholder interviews in order to grasp the hotspots on sustainability. The literature review was conducted by selecting quantitative and qualitative scientific research, organization reports, etc. The selected studies covered a large share of the comprehensive indicators for classifying and analyzing food supply chains in detail. The review was performed in three steps. First, integrated triple bottom line (simultaneous consideration of sustainability indicators) studies were reviewed to identify key environmental, social and economic indicators of food chains. The outcome of this first step was used as the base for creating a list of indicators to be included in any sustainability assessment. In the second step, the literature review was expanded with any economic, environmental, and social indicators that were evaluated separately for soy chains.

The results of the stakeholder interviews (N=74) were used to extend the analytical framework with indicators aiming at measuring or exploring sustainability issues. These indicators are not quantifiable as such, they are manifest variables, which need to be validated. Moreover the comprehensiveness of the extended analytical framework was evaluated using the SAFA guidelines, developed by the FAO (2012) as a reference conceptual framework. With the aim of improving the transparency and comparability of sustainability performance of food and agricultural sectors, FAO (2012) developed guidelines on the Sustainability Assessment of Food and Agricultural Systems (SAFA guidelines). The SAFA guidelines were developed through an ongoing process of text elaboration, stakeholder consultation and participation. Continuous reviews take place and therefore SAFA can be considered as a reference extended analytical framework involving a broad range of sustainability dimensions and related indicators supporting the evaluation of the comprehensiveness of the selected sustainability indicators. The framework can be extended with additional indicators to increase the comprehensiveness in order to touch upon a broad set of ecological, social and economic themes.

Subsequently, the whole process of identifying important indicators was finalized by conducting expert interviews.

In order to reveal the perceptions of a wide range of stakeholders in Europe and Latin-America a web survey based upon the input from the defined framework was set up. In total, 36 business and 34 non-business stakeholders completed the survey, implying an EU/LA geographical spread of 32/38.

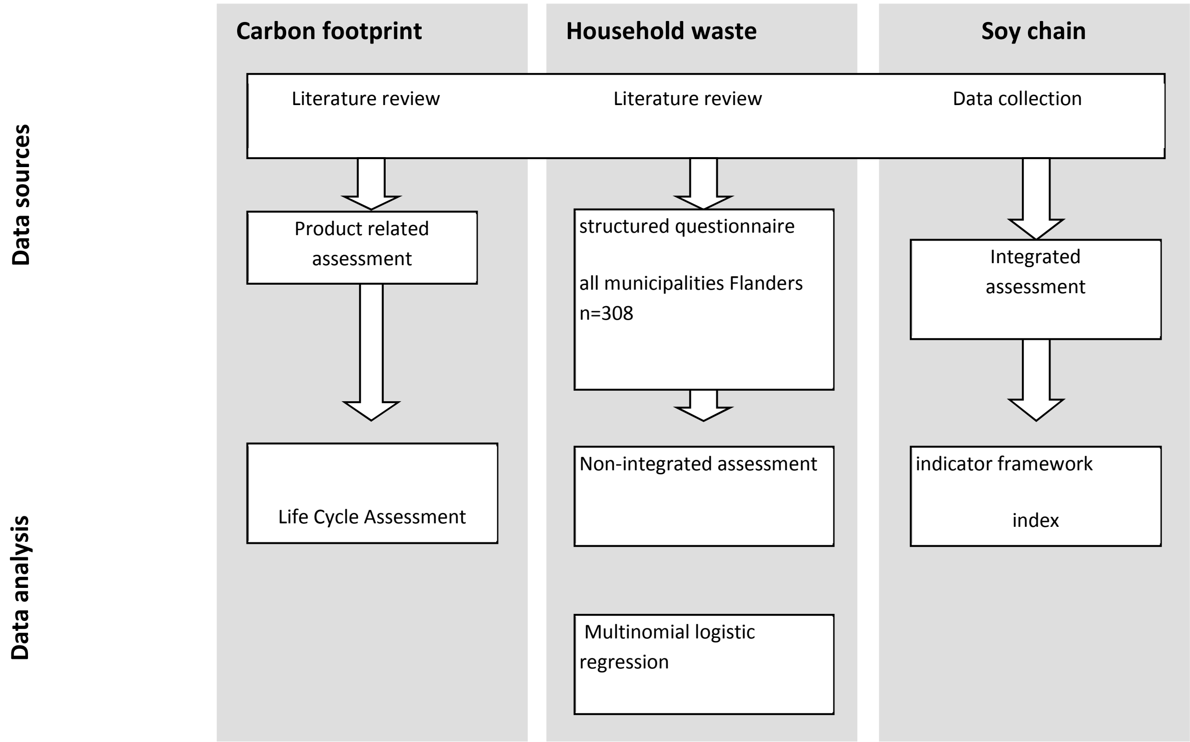

Figure 5. Research design, data sources and analyses, per scrutinized topic.

1.9.2 Thesis outline

This thesis contains 6 chapters, as shown in figure 6. Chapters 2 to 5 are research chapters answering the aforementioned research questions. The introduction chapter presents the rationale behind the research, the research questions, the conceptual framework, research contribution and research design. Chapter 2 describes environmental sustainability with regard to a case on carbon footprint. It is adapted from a book chapter and a paper discussing the carbon footprints of pigmeat and beef in Flanders. Chapter 2 provides an alternative sustainability performance measurement tool for the ecological footprint, namely carbon footprinting or CO_2 aggregations. This is a better suggested alternative as it directly looks at sustainability (Fiala N., 2008). Carbon footprint might be used as an environmental sustainability indicator. Chapter 3 tackles sustainable waste management. This chapter is based upon the results and discussion of 2 published papers on residual household waste in Flanders. The first part of this chapter covers environmental sustainability, whereas the last part focuses on the economic part of sustainable waste management.Chapter 4 goes one step beyond the previous assessments, by including the social dimension of sustainability. The chapter scrutinizes the perceptions of stakeholders towards sustainability, a more subjective approach. Whereas chapter 5 assesses sustainability in an integrated and objective way. The last chapter describes the general conclusions. It answers research questions, describes limitations of the research, describes future research and provides policy implications.

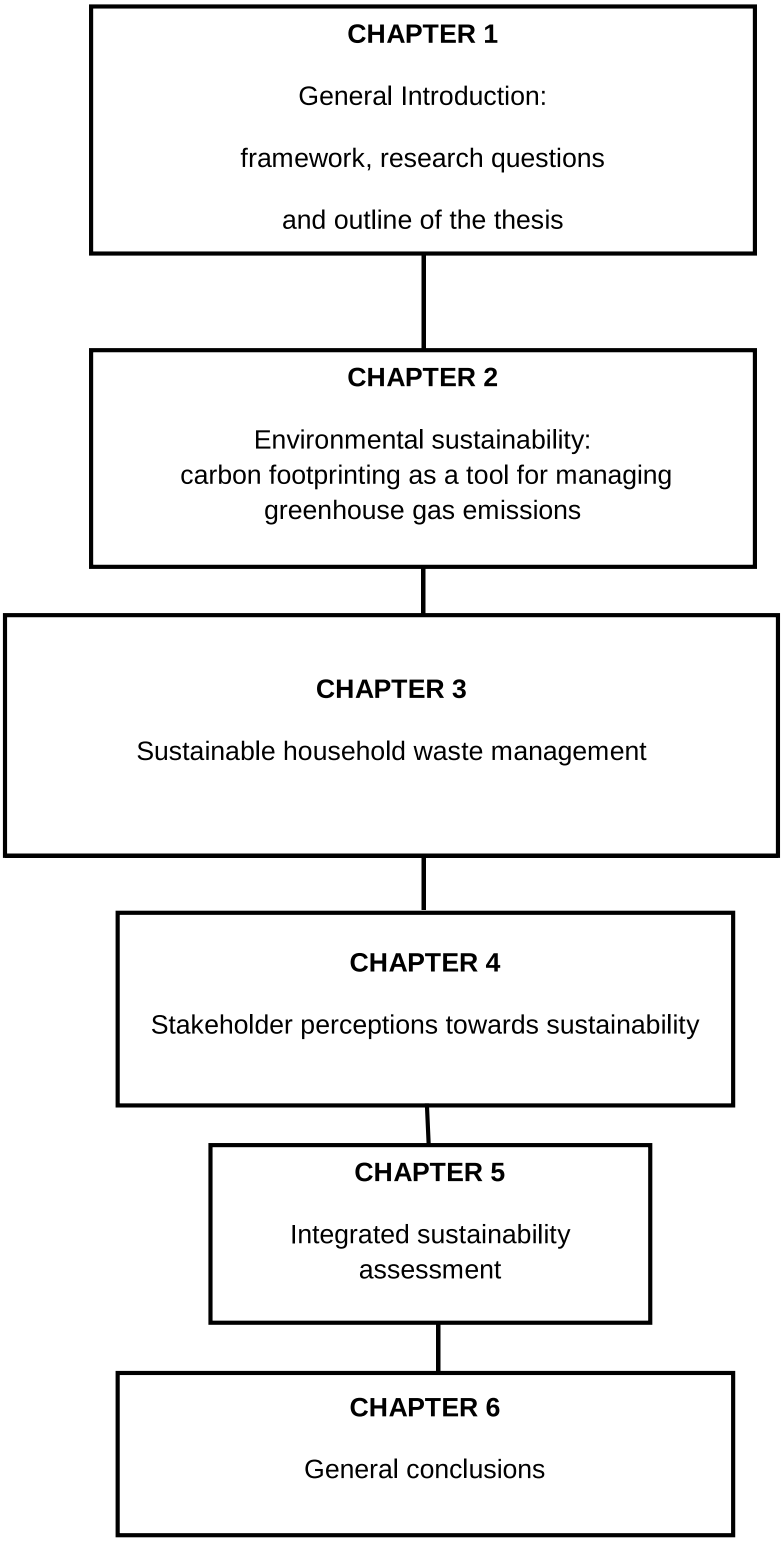

Figure 6. Outline of the thesis

We are living on this planet as if we had another one to go to.

(Terri Swearingen)

2 Environmental sustainability: carbon footprint as a tool for managing greenhouse gas emissions

Adapted from:

Peer-review web of science publication

R. Jacobsen, V. Vandermeulen, G. Van Huylenbroeck & X. Gellynck (2014). Carbon footprint of pigmeat in Flanders, *International Journal of Agricultural Sustainability*, 12:1, 54-70

Book chapter

R. Jacobsen, V. Vandermeulen, G. Van Huylenbroeck & X. Gellynck (Eds.) (2014). A Life Cycle Assessment Application: The Carbon Footprint of Beef in Flanders (Belgium). Assessment of carbon footprint in different industrial sectors, volume 2. In EcoProduction : *Environmental Issues in Logistics and Manufacturing* p.31-52.. Springer, Singapore.

2.1 Introduction

This chapter emphasizes environmental sustainability with regard to carbon footprinting as sustainable development indicator. Meat forms a major part of the human diet in many countries (Van Wezemael, 2011). The associated livestock production leads to substantial greenhouse gas (GHG) emissions causing climate change (Johnson et al., 2007). More specifically, livestock production accounts for half of all GHG emissions related to the human diet in Europe (Kramer et al., 1999; European Commission, 2009).

In order to initiate a development towards more sustainable production, there is a need to analyse the current situation, as well as the potential for identifying improvements in the production system (Eriksson et al., 2005). To find out where along the production chain improvements can be made, all related emissions need to be quantified. One such method is to calculate the carbon footprint (CF) of the livestock product (Espinoza-Orias et al., 2011). A CF quantifies the climate change impact of an activity, product or service. Within the CF, all GHG emissions (carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O)) are combined and expressed as CO_2 equivalents. As such, the carbon footprint tool is applied to two different livestock products in Flanders: beef and pigmeat. The latter will be considered most thoroughly. The methodology for beef will only be addressed where it is different from pigmeat.

2.2 Importance of pigmeat

Of all the major sources of world meat production, pigmeat represents the highest proportion (37%) compared to poultry (33%) and beef (23%) (FAO, 2008). Pigmeat provides an interesting case to study because, although many organisations and institutions are now determining the carbon footprints (CF) of their products and services, estimations for the CF of pigmeat are rather limited, especially compared to, for example, the carbon footprints of milk where examples in literature abound (Blonk et al., 2008b; Muller-Lindenlauf et al., 2010, Sonesson et al., 2009, Thoma et al., 2010, van der Werf et al., 2009). Moreover, not all studies on the CF of agricultural products are clear in terms of the methodology or standard used, the chosen system boundaries and definition. This leads to ambiguous results and hence wrongly informed stakeholders. These different approaches in terms of methodology also prohibit a fair comparison of carbon footprints between products and sectors. A CF relies on the Life Cycle Assessment methodology (LCA). LCA takes into account the life cycle principle (Glavic and Lukman, 2007). The fact that each Life Cycle Assessment (LCA) has to deal with many different issues (such as allocation, scope, system boundaries, data, land use etc. (Finkbeiner, M., 2009)) makes it necessary that each of these aspects is described properly in any LCA study. A literature review on existing CF studies on pig production revealed that in some cases information on several aspects was missing (e.g. Dalgaard et al. (2007) and Leip et al. (2010) did not report which allocation method they used). This problem was also mentioned by de Vries and De Boer (2010): they had to exclude some sources from their meta-analysis due to a lack of data.

This chapter of the PhD dissertation presents the results of an estimation of the CF of pigmeat and beef production in Flanders, according to the most up to date product category rules and standards. Furthermore, a step by step approach is followed in order to provide a clear overview of the applied methodology, tackling the gaps in literature.

2.3 Pigmeat and beef production

In Flanders, the northern part of Belgium, the concentration of pigs is one of the highest in Europe (European Commission, 2008). The pressure on the environment from livestock production in Flanders is substantial, with a major impact on GHG emissions.

Flemish pig farmers annually produce 10.5 million pigs, which is about 4% of the EU production (Statistics Belgium, 2010; Eurostat, 2012). The animals are distributed over 5.377 farms (Statistics Belgium, 2010). Half of these farms (42%) specialize in pig production, producing about 95% of all pigs (Van Liefferinge, 2011). The focus is hence on this type of farming.

Flemish farmers hold a total of 262,280 heads of beef cattle per year. The animals are distributed over 5,544 farms (Statistics Belgium 2010). Approximately 80 % of the farms specialize in beef production, being the focus. The remaining farms produce a combination of crops and beef.

2.4 Methodology

One major gap in literature is the fact that many CF studies do not include the used methodology. Therefore it is not clear how results were obtained. Below is described which methodologies and standards were used to obtain results for the carbon footprint of pigmeat and beef in Flanders.

2.4.1 Standard and used method

In the study, the Intergovernmental Panel on Climate Change (IPCC, 2006a) standard was used, in line with the National Inventory Report of Belgium (VMM, 2011). Although the IPCC (2006a) directive clarifies the calculation of total GHG emissions, it does not mention how it can be allocated to a particular product. Therefore, a specific methodology is needed, such as Publicly Available Specification (PAS2050) or ISO14067 (Espinoza-Orias et al., 2011). At present, the PAS(PAS2050, BSI 2011) is the most developed method and was therefore chosen. Moreover, the PAS2050 has been used in the past for estimating GHG emissions within the agricultural and horticultural sector (ERM, 2010). Based on the PAS2050, specific Product Category Rules (PCR) have been developed for dairy and horticultural products (ERM 2010). In 2012, specific PCR have been developed, according to the international EPD system (Environdec 2013), for mammal meat (including pigmeat) in which slaughter activities, packaging processes and storage are core processes (Studio LCE, 2012). The upstream activities within this PCR, namely animal production, feed cultivation, manure management, transportation and packaging, are seen as core activities in the current study. Based on the IPCC 2007 (IPCC, AR4 2007), the following global warming potentials (GWP) have been used for methane and nitrogen gas emissions: 1 kg of methane (CH_4) equals 25 kg of CO_2 and 1 kg of nitrogen gas (N_2O) equals 298 kg CO_2. In order to be able to compare and correctly understand the value of the estimated CF, it is necessary to describe the assumptions made, in relation to the scope and system boundaries, the functional unit used, the allocation method and whether land use change (LUC) has been accounted for.

2.4.2 Scope and system boundaries

The scope of the study is from cradle to gate. PAS2050 states that those emission factors contributing less than 1% of the total CF can be ignored (BSI 2011). The majority of the emissions occur at farm level, rather than during distribution, consumption, transport and waste processing (Blonk et al., 2008b; Campens et al., 2010). The majority of emissions occur within farm gate compared to post farm gate (Scollan et al., 2010). The relatively small proportion of post farm gate emissions is also indicated in a report on the carbon footprint for exported lamb from New Zealand (Ledgard et al., 2010). These last steps are not included in the CF calculations. The system boundaries were identified based on the scope of the study. The study includes all GHG emissions as shown in Figure 7, including the production of materials and energy, and also taking into account transport steps.

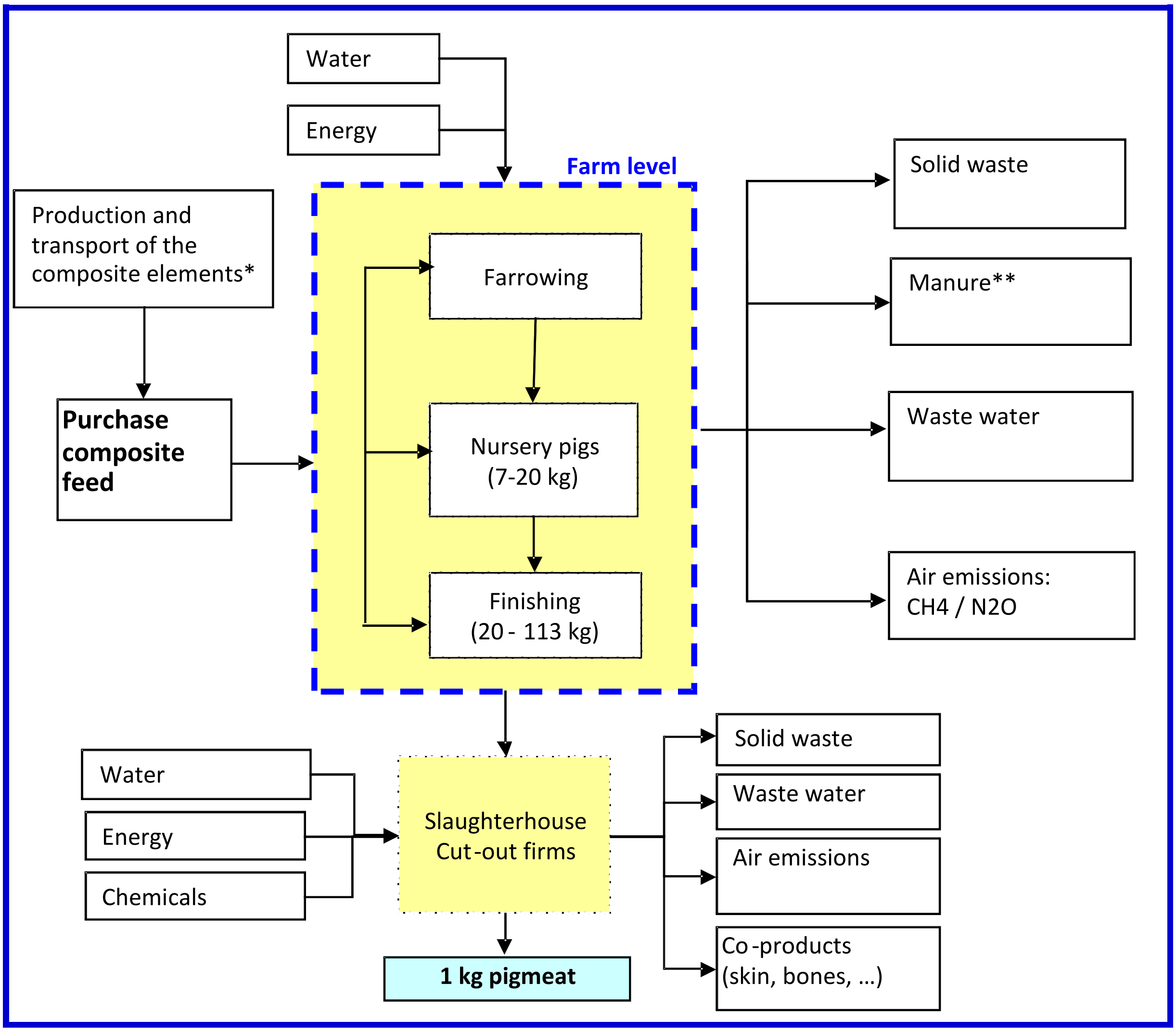

Figure 7. The system boundaries for pigmeat production. Boxes with dashed lines present a process, those with full lines a productflow. The coloured boxes are the foreground system, whereas the other parts were based on generic data.

*Including land conversion for feed stock grown outside Europe.

**Manure storage on the farm and manure usage outside the farm.

The focus is solely on the meat production chain. Hence the system boundary excludes the production of capital goods (machinery and equipment), adopting a similar approach as other international studies.

Figure 8 presents a similar approach for the beef production system.

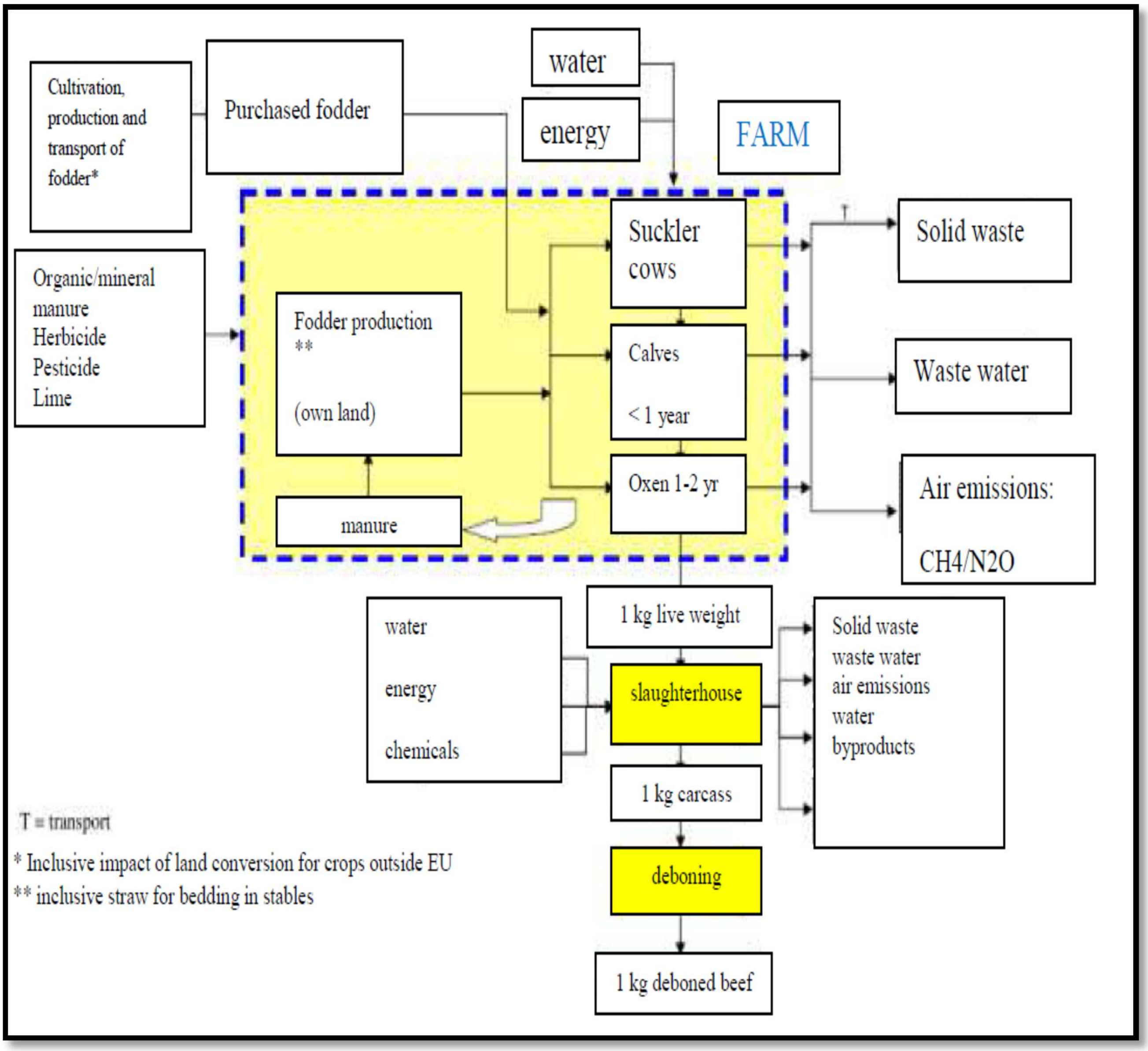

Figure 8. The system boundaries for beef production. The boxes with dashed lines present a process, those with full lines a product flow. The coloured boxes are the foreground system.

After setting the system boundaries, it is necessary to obtain an overview of all emission sources. Table 1 gives an overview of the included emission sources throughout the chain.

Table 1. Overview of emission sources within the covered system boundaries.

Emission source	GHG	Description
Feed mixtures (purchased)	CO_2 and N_2O	Farming, transport, processing and land conversion is taken into account in the covered emission factors.
Animal	CH_4	The IPCC method is applied (Tier 2 calculation)
Manure storage and disposal	CH_4 and N_2O	The IPCC method is applied (Tier 2 calculation)
Manure application (not used for own feed mixtures)	CH_4 and N_2O	Allocation between animal (40%) and vegetable production system (60%) based on nitrogen uptake by plants.
Energy/ water consumption	CO_2, CH_4 and N_2O	Energy consumption (electricity; red diesel; gas). Water consumption (tap and ground water)
Transport of goods	CO_2, CH_4 and N_2O	Assumptions made for the goods entering and leaving the farm.
Processing materials	CO_2, refrigerant	Cleansing products and refrigerants

2.4.3 Allocation method

Another critical assumption describes how to allocate GHG emissions between various co-products emerging from a single process. International standards were followed in determining the allocation method. These standards prescribe that whenever possible, allocation should be avoided. If not possible, a single allocation method should be used. Allocation was needed in the case of Flemish pigmeat and beef. In terms of the slaughtering and deboning process, economic allocation was used, in which the economic value of the by-products (bones, fat, skin, hide, heart, blood, etc.) represents the market prices multiplied by the mass fraction per incoming product. When the byproduct has a negative economic value (a cost), the share of these byproducts in the GHG allocation is considered to be 0. However, if one were to apply this to the manure production process, allocation to crops would be zero, because of the oversupply of manure in Flanders. However, manure does contribute to the production of crops and therefore physical allocation was used to allocate the GHG production of manure between crops and animal production.

2.4.4 Land use and land use change

This study also considers the impact of Land Use Change (LUC). CO_2 Emissions due to changes in land use are mainly the result of cutting down forests and using the land instead for agriculture or built-up areas, urbanization, roads, etc. When large areas of (rain) forest are cut down, land often turns into less productive grasslands with smaller carbon storage capacity. According to PAS2050, LUC should be taken into account if there has been land conversion in the last 20 years. For agricultural products in the European Union (EU) this is not the case, and thus LUC is zero. However, part of the feed compound (mainly soy and palm) is imported from countries outside the EU, and for those FAO statistics (2010) are used to define the LUC in the past 20 years. The total emissions originating from this LUC are calculated and 1/20th is attributed to each forthcoming year (Kloverpris et al., 2010). The information on the emissions themselves comes from Blonk (2008a), Ecoinvent (2011) and LCA Food (Nielsen et al. 2010).

Land use (or carbon sequestration in the soil) is not included since a lot of uncertainty remains regarding the net effect (absorption or emissions) of land use (Smith et al., 2008,; Gill et al., 2010; Scollan et al., 2010). It is necessary to develop more precise databases of land use and emission factors (Scollan et al., 2010). Carbon is lost from soils due to cultivation and other management activities. These latter variables are currently excluded from PAS 2050 due to the difficulty in accurately estimating this emission source (Plassmann et al., 2010).

2.5 Data sources

PAS2050 has specific rules favoring the use of primary over secondary data (BSI, 2011). Whenever possible, primary data have been used in the study – collected through interviews (e.g. on slaughtering and deboning). These primary data were complemented with data from existing databases (e.g. on farming) and other reports (e.g. on emission factors). The data were collected for the year 2009. For certain data it was necessary to obtain more recent values, for example, for the feed compound composition which changes on a daily basis.

2.5.1 Raw materials and farm level

Distinction is made between pigmeat and beef.

2.5.1.1 Pigmeat

The data on the representative conventional pig farm were collected from the data set of the Farmers Union (Boerenbond), keeping track of farm data from an economic point of view. The data set includes general farm accountancy data (such as farm type, region, type of crops or livestock, input, production, etc.) as well as financial data, technical–economic data, environmental indicators (such as water, nutrient or energy balance sheets) and a balance sheet for the farm (Boerenbond, 2011). This data set was used to select pig farms specialized in pig production breeding their own piglets (i.e. closed farms). The average of their data was identified, in order to obtain a representative pig farm. This involved a limited group of farms, whereby the data showed no outliers and therefore could represent a real farm in Flanders. Data on manure production was not available within the Farmers Union data set and was therefore collected from reports of the Flemish Centre for Manure Processing. The average manure production per animal was linked to the number of animals on the farm (VLM, 2011a). Other data were collected from the National Institute for the Accounts in Belgium – NIR, Belgium (VMM, 2011), IPCC (2006b, 2006c), Ecoinvent (2011), Statistics Belgium (2010), LCA Food (Nielsen et al., 2010) and IDF (2010). Furthermore, the most common pig stable system implemented in Flanders was included. During the farrowing phase, a semi-slotted floor was used, while for the nursery pigs and finishing phase a slotted floor was used. Since most pig farms in Belgium do not produce own fodder, feed production at the farm was not included, and only concentrated feed was used for the pigs. The average composition of the feed concentrate was given by BEMEFA (Belgian Feed Compound Association, personal communication, April–October 2011), for different animal categories, including the origin of the composite elements (see Table 4). The composition has been given for 4 October 2010 – randomly chosen. The composition of feed compound varies daily upon the availability of components on the market. By not taking the average for the whole year, one can be sure that the used feed has an appropriate composition for the animals. The emissions associated with the production of the feed mixture are calculated based on emission factors for the resources, which were derived from Blonk Milieuadvies (Blonk et al., 2008a), LCA Food (Nielsen et al., 2010) and Ecoinvent (2011).

2.5.1.2 Beef

Data on the representative conventional farm was collected through the Farmers Union dataset (Boerenbond 2011). This dataset was used to select the farms specializing in beef production. Flanders has 4,334 specialized beef cattle farms (NIR Belgium, 2011). The database allowed identification of the average to obtain a representative and existing farm. Outliers in the data were omitted. The average represents a real farm with the following main characteristics:

• 53 calves < 12 months,

• 47 heads of young cattle with an age between 1–2 years,

• 65 suckler cows.

Farms are confronted with the loss of animals. The mortality rate amounts 1.25 % for mature animals and 11 % for calves. The replacement rate amounts 34 %.

Data on manure production were collected from reports of the Flemish Centre for Manure processing and were linked to the number of animals on the farm (VLM, 2011).

The applied fodder partly originates from the farm's own production and is partly purchased. The average composition of the feed concentrate was given by the Belgian Feed Compound Union (BEMEFA; personal communications, April–October 2011). Emission factors were derived from BlonkMilieuadvies (Blonk et al., 2008a), Nielsen et al. (2010), and Ecoinvent (2011).

A suckler cow consumes the following raw feed components (homegrown) per year: 8.053 kg of grass, 7.697 kg of feed corn and 636 kg of other raw feed components (mainly fodder beets). This is extended with purchased fodder consisting of 1.120 kg wet sugar beet pulp, 1.072 kg composite feed concentrate, 422 kg wet byproducts, 140 kg single forage, 20 kg soy meal, and 14 kg sugar beet pulp per suckler cow per year, as well as 13 kg of milk powder.

The animals are kept in stables on a bed of homegrown straw. Suckler cows and female young cattle (1–2 years) stay outside for 24 h a day, during a period of 6 months/year. Female calves remain outside for 24 h a day, during a period of about 4 months. The male young cattle and the male calves stay inside. The animals produce 623 kg of manure per day. Approximately 32 % of this manure ends on the grassland during grazing, 60 % is preserved as stable, and 8 % as mixed manure.

The farmer possesses a total of 48 ha containing grass and cropland: 22.6 ha grassland, 10.3 ha of maize, 0.7 ha temporary grassland, 0.5 other roughage and 13.9 ha of wheat.

Fertilizer use amounts 650 kg per hectare of grassland (with 170 units of nitrogen) and 100 kg of starter fertilizer per hectare of maize (20 units of nitrogen and 2 units of phosphorus). The farm uses on average 2.65 kg of herbicides and 500 kg of lime, both per hectare.

The farm annually consumes 8.637 kWh of electricity and 8.054 L of oil fuel. In terms of water consumption, the farm consumes 773 m³ of ground water and 243 m³ of tap water.

Table 2. Emission factors for electricity and gasoline oil in Belgium and the emission value for animal exhaustion of pigs.

Category	Value	Unit	Source
Electricity	0.40	kg CO_2eq/kWh	Energy convenant
Gasoline oil	2.66	kg CO_2eq/kg	Energy convenant
Animal exhaustion	37.5	kg CO_2/pig/year	IPCC

2.5.2 Meat processing

After primary production, the live animals are transported to slaughterhouses for meat processing. This operation also includes deboning (cut-out) activities.

2.5.2.1 Pigmeat

Data were gathered through in-depth interviews (N=5) with the biggest slaughterhouses and cut-out firms in Flanders. The contacted slaughterhouses collectively represent 20% of the volume of processed pigmeat in Flanders and is therefore representative. Data with regard to the price and weight of meat, by-products and carcass, energy consumption (water, electricity, refrigerants and cleansing products) and transport characteristics (distance and type of truck) were collected. Furthermore, secondary data were used from the Flemish Meat Federation (Febev), Flemish Waste Authority (OVAM) and the National Institute of Statistics.

2.5.2.2 Beef

The contacted slaughterhouses (N = 4) represent 33 % (weight basis) of processed beef in Flanders and hence are representative for the whole sector. Data were collected on meat weight and prices, byproducts and carcass, energy consumption and transportation characteristics. Missing data, such as the price of cuts and amount of waste/meat generated through slaughtering, were given by the Flemish Meat Federation (Febev).

2.6 Data analysis

Emissions emerge throughout the meat production chain. Below is elaborated how the different chain stages contribute to climate change impact.

2.6.1 Fodder production emissions

2.6.1.1 Pigmeat

Fodder production begins with the production of resources and transport in order to sow, grow and harvest crops (i.e. seeds, fertilizers, pesticides and diesel). GHG emissions associated with the resource production itself are allocated to the crops, as well as transport to the processing plant. Emissions associated with processing (grinding, crushing, mixing, etc.) are also taken into account. Based on the information in Table 5, column four of Table 6 gives the emissions resulting from the feed consumption. The last column gives an indication of the importance of LUC, caused by the use of soy and palm, in terms of emissions.

Table 3 shows the percentage composition of the feed compound during several production phases. For instance, barley makes up 37 % of the feed composition of a 7-12 kg piglet.

Table 3. Feed compound ingredients during several production phases for pigmeat.

Component	**% LUC**	**Piglet 7-12 kg**	**Piglet 12-22 kg**	**Pig 22-35 kg**	**Meat Pig 35-75 kg**	**Meat Pig 75-115 kg**	**Sow, gestation**	**Sow lactating**
Barley	0,00%	37	35	21	21	20	22	16,5
Soy meal import	75,83%	5,3	14,6	13	12,5	6,4		10,8
Palm kernel import	8,15%				4	5	5	
Wheat	0,00%	8,7	18,1	30	35	35	34,2	35
Maize	0,00%	15	12,3	11,6	13,3	9,7		10
Barley	0,00%			10	7,5	15	20	6,7
Sugar beet pulp	0,00%						4,5	
Soy hull import	20,16%	15	10					
Blend (21% protein)	0,00%	15	7					
Sum		96,0 %	97,0 %	85,6 %	93,3 %	91,1 %	85,7 %	79,0 %

The relevant emissions were calculated on the basis of resource emission factors and are given in Table 4. The second column of table 4 shows the yearly average number of animals on the farm, divided according to their growth stage. Column three indicates the consumed amount of feed compound.

Table 4. Fodder consumption per animal category for the modeled pig farm (based upon the exact composition and multiplied by the retention time (in days) in each category) and accompanying emissions.

Animal category	**Number of animals**	**kg product**	**kg CO_2eq/kg product**	**% LUC**
Sows (gestation and lactating)	219	251 000	0.76	9.7 %
Piglets (<12 kg)	5 319	57 282	1.02	16.9 %
Piglets (12-22 kg)	5 319	92 652	1.10	19.3 %
Meat pigs (22-35 kg)	5 133	196 012	0.79	10.8 %
Meat pigs (35-75 kg)	5 133	603 116	0.79	9.9 %
Meat pigs (75-115 kg)	4 857	542 150	0.62	5.4 %

2.6.1.2 Beef

Regarding a beef cattle farm, distinction should be made between homegrown and purchased fodder. The production of fodder also comprises the production and transportation of the inputs used in the fodder production at the farm (seeds, fertilizers, pesticides, and diesel). The accompanying emissions are allocated to the crops. Land use during cultivation of the crops results in extra GHG emissions. Laughing gas is the most important greenhouse gas for land use.

For purchased fodder, crops are being transported to a processing plant. Emissions accompanying transport and processing are included. A 50/50 male/female ratio for the young cattle and calves population is assumed.

Purchased fodder for beef production

Concentrate feed is added to the other feed raw materials for obtaining compound feed. The Belgian Feed compound union BEMEFA and feed specialists were consulted to indicate the composition. Databases were consulted on August 4, 2011. Table 5 on the next page indicates the representative composition of approximately 80 % of feed concentrate for beef cattle.

The applied emission factors are derived from literature (Blonk Milieu Advies, Wageningen University). The available data were extended with other sources: the Ecoinvent database, LCA food database, and Carbon Trust. Table 6 presents the emission factors for purchased fodder.

Table 5. Composition of purchased feed concentrate for beef cattle (source: BEMEFA).

Feed compound resources	Beef cattle ALLMASH 16 Share in %
Barley	12,5
Soy meal	5
Maize yellow from France	5,9
Maize gluten feed	22,5
Sugarbeet pulp	20
Linseed flakes	12,5
Rapeseed flakes	8,3

Table 6. Emissions accompanying purchased fodder per kg of feed compound ingredient.

Resources	kg CO_2 eq/kg product	% LUC*
Soy meal	3,06	71 %
Sugar beet pulp (dry)	0,11	
Sugar beet pulp (wet)	0,03	
Wet byproducts	0,03	
Milk powder	7,9	
Single forage	0,30	
Composite forage	0,42	19,8 %

* Land Use Change

Homegrown roughage for beef cattle

Farm land is applicable both for grassland as well as for cultivating fodder crops. The yield of the own crops is used as roughage. Table 6 presents the calculated emissions. Energy and fuels used for machinery and transportation are included within the total energy consumption of the farm. They are not mentioned in Table 6. GHG emissions accompanying production and transportation of fertilizers, herbicides, insecticides, and fungicides are included too. Emission factors were calculated from the Eco-invent database. Emissions due to the application of these substances are mentioned in Table 7.

Table 7. Emissions accompanying the cultivation of own crops.

Resources	Area (ha)	kg CO_2 eq per year
Wheat (home-grown) for straw	13,9	17.659
Maize (home-grown)	10,3	32.150
Grass silage	23,3	23.9408
Fodder beets (home-grown)	0,5	4.989

Table 8. Emission factors for the production, transportation of fertilizers, herbicides and lime.

Name	**Value**	**Unit**
Fertilizer (Calcium Ammonium Nitrate)	8,81	kg CO_2 eq/kg N
Herbicide	10,730	kg CO_2 eq/kg
Lime (Calcium carbonate)	0,02	kg CO_2 eq/kg

Laughing gas emissions

N_2O emissions due to crop cultivation are calculated according to the IPCC 2006a, b, c method. Nitrogen sources applied to land are in this case (natural) fertilizers and crop residue. Direct laughing gas emissions are the result of denitrification.

Denitrification comprises the stepwise reduction of nitrate (NO_3^-) and nitrite (NO_2^-) to nitric oxide (NO), nitrous oxide (N_2O), and dinitrogen gas (N_2) (Zumft, 1997):

$$NO_3^- + 2e^- + 2H^+ \rightarrow NO_2^- + H_2O$$

$$NO_2^- + e^- + 2H^+ \rightarrow NO + H_2O$$

$$NO + e^- + H^+ \rightarrow \tfrac{1}{2} N_2O + \tfrac{1}{2} H_2O$$

$$\tfrac{1}{2} N_2O + e^- + H^+ \rightarrow \tfrac{1}{2} N_2 + \tfrac{1}{2} H_2O$$

It is assumed that 1 % of all nitrogen applied to land converts into laughing gas (uncertainty interval 0.3–3 %). The IPCC value (1.25 %) was still applied in the national inventories until 2013. Current research in Flanders points out that 3.16 % of all nitrogen converts into N_2O. Indirect laughing gas emissions due to nitrogen leaching are calculated with data recorded in the National Inventory Report for greenhouse gases (NIR) of Belgium (2009). The amount of leaching nitrogen was determined with the Systems for the Evaluation of Nutrient Transport to Water model. In Flanders, 9 % of all applied nitrogen is leaching (NIR 2010, H6, p. 120). Of this, 0.75 % is finally converted into N_2O (IPCC 2006a, b, c). Indirect N_2O emissions due to nitrogen evaporation as ammonia (NH_3) and NOx are calculated with the same data from the NIR Belgium (2009). The amount of evaporated nitrogen as NH_3 or NOx, depends upon the nitrogen source:

(1) Fertilizer in Flanders: average NH_3 evaporation amounts 3.3 % and the NOx evaporation amounts 1.5 % (NIR, 2010).

(2) Organic fertilizers in Flanders: average nitrogen evaporation as NH_3 or NOx

amounts 20 % (NIR, 2009).

According to the IPCC calculation method, 1 % (0.2–5 %) of evaporated nitrogen

(as NH_3 or NOx) is converted into N_2O.

Lime Application

Lime is applied on land to increase the soil pH, causing CO_2 emissions. For beef cattle farms, the total amount of lime applied per year is 1000 kg. The used emission factor is 0.48 kg CO_2 eq/kg lime (e.g. dolomite).

2.6.2 Emissions during animal production

The previous section discussed the emissions during fodder production. The section below elaborates on emissions emerging during animal production.

2.6.2.1 Pigmeat production

Two types of emissions are related to pig production itself: energy use and emissions from the animals. Since a specialized pig farm was being considered, all energy consumption at the farm can be attributed to pigmeat production. The energy consumption was not further divided into more farm sub-activities. Energy consumption per meat pig amounts 46.4 kWh (25% electricity and 75% gasoline oil). Table 10 shows the related emission factors. Pigs will have a low level of methane output compared with ruminants, because of the limited gastro-intestinal fermentation. The IPCC Tier 1 method (IPCC 2006a) has been used to calculate methane (CH_4) output. The IPCC Guide for National Inventories states that each pig annually emits 1.5 kg of CH_4 (IPCC 2006b). This amount is converted into kg CO_2 eq. taking into account the Global Warming Potential (GWP) of methane (25 kg CO_2 eq./kg CH4).

2.6.2.2 Beef cattle breeding

Yet again, two types of emissions emerge: energy consumption and emissions from animals.

Energy consumption at the farm

The energy consumption is included as a whole and not allocated. In Table 9, emission factors are presented. Each suckler cow annually consumes about 1.512 kWh of energy (10 % electricity and 90 % gasoline oil).

Table 9. Emission factors of electricity and gasoline oil.

Name	Value	Unit	Source
Electricity	0.40	kg CO_2eq/kWh	Energy convenant
Gasoline oil	2.66	kg CO_2eq/kg	Energy convenant

Animal emissions: rumen fermentation

Emissions due to rumen fermentation are calculated based upon the IPCC guidelines (Tier 2 method). For calculating the necessary gross energy uptake (GE) per animal aspects as the daily need, growth, and gestation are included. It is assumed that suckler cows produce a negligible amount of milk. The digestible energy (DE) is expressed as %GE. An adapted value is calculated based upon the fodder and the number of grazing days. It is calculated that approximately 169 MJ of GE is needed per suckler cow, 121 MJ for young cattle, and 82 MJ for calves. The digestible energy is calculated to be on average 74 %GE based upon the fodder. Table 10 represents the digestible energy per feed compound. The time spent on grassland is included in order to determine an adapted digestible energy content of the animals' diet.

Approximately 6.5 % (weight basis) of the ingested gross energy is converted into gas (methane) (IPCC 2006a). If the animals are fed more than 90 % with composite fodder, the above number can be lowered to 3 %. The ingested gross energy is calculated based upon the fodder composition and the digestible energy content of each feed component.

Table 10. DE-value for different types of fodder (source: FAO* and NIR Belgium).

Feed component	DE	Unit	Source
Wheat/Barley	86	%GE	FAO
Maize- roughage	72	%GE	NIR Belgium
Soy meal	80	%GE	FAO
Beet pulp/citrus pulp	81	%GE	FAO
Wet byproducts	78	%GE	FAO
Composite fodder	80	%GE	NIR Belgium
Protein, vitamines	80	%GE	Proxy: composite fodder
Feed concentrate	80	%GE	Proxy: composite fodder
Composite young feed	80	%GE	Proxy: composite fodder
Fodder for young cattle	80	%GE	Proxy: composite fodder
Full milk	90	%GE	NIR Belgium
Grass silage	72	%GE	NIR Belgium
Fresh grass (grazing)	79	%GE	NIR Belgium

* GHG emissions from the Dairy sector a Life Cycle Assessment, 2010, FAO.

2.6.3 Manure emissions

Emissions also emerge through the usage and storage of manure.

2.6.3.1 Pigmeat

Pigs manure is stored as manure slurry in a pit under the stables creating methane and nitrous oxide emissions. Manure can be used during crop production, and hence emissions need to be allocated towards the crop and pigmeat production, through physical allocation (see the 'Allocation method' section).

Methane

Methane emissions due to manure production depend upon the amount of excreted volatile solids, the maximum methane production capacity of manure and the way in which manure is stored. The amount of excreted volatile solids is calculated according to the IPCC (2006a) formula, whereby the IPCC 2006 reference value for the urine fraction (2%) and dry matter content (2%) is used (IPCC 2006b). The methane conversion factor for manure slurry stored under the stables amounts to 19% (IPCC 2006b).

Table 11. N_{ex} per type of animal in Flanders (source: NIR Belgium/manure database).

Animal category	Nex (kg/head.yr)
Piglets (<20 kg)	2.24
Meat pigs (20-110 kg)	11.24
Meat pigs (> 100 kg)	21.18
Sows (inclusive piglets< 7kg)	21.66

Nitrous oxide

Stored manure gives rise to nitrous oxide emissions through a combination of nitrification and denitrification (see also section 2.6.1.2). The amount produced of nitrous oxide emissions depends upon the total nitrogen emissions by the animals (Nex). This can be calculated by identifying the difference between the nitrogen uptake (in fodder) and the amount retained in the body or within the products. The amount of excreted nitrogen per animal category is taken from the Belgian NIR report (VMM, 2011).

The amount of nitrous oxide released from manure depends upon the way of storage. It is assumed that 0.1% of total nitrogen is converted into nitrous oxide for manure slurry stored under the stables (this is lower than the IPCC (2006b) value (0.2%), but in line with the value for Flanders in the National Inventory (VMM et al. 2011)). This concerns *direct* nitrous oxide emissions.

There are also *indirect* nitrous oxide emissions from manure. These are formed through volatilized ammonia (NH_3) and NOx. The amount of NH_3 and NOx formed from the pile of manure depends upon the storage method. Based on the IPCC guidelines, it is assumed that 25% of the total nitrogen emissions originating from pig manure storage underneath stables is converted into ammonia (NH_3) or NOx (IPCC 2006b). Furthermore, it is assumed that 1% of the total nitrogen losses are converted into nitrous oxide through indirect nitrous oxide emissions (IPCC 2006c). Leaching is assumed to be 0%.

2.6.3.2 Beef

Manure production takes place on the meadow and in the barn. Suckler cows and female young cattle (between 1 and 2 years) stay approximately 183 days/year on the meadow, and female calves (<1 year) approximately 122 days. Male young cattle and male calves are not put on the meadow. Manure produced in the stable is temporarily stored. It is assumed that 80 % of the manure production in the stable is being stored as stable manure. Manure disposal on grassland and manure storage are accompanied with methane and N_2O emissions. The calculations per greenhouse gas are explained below, based upon the IPCC 2006a, b, c guidelines (Tier 2).

Methane

Methane emissions related to manure production depend upon the excreted volatile solids, the maximum methane production capacity of the manure, and the storage. The excreted volatile solids are calculated by using the IPCC (2006a) formula. Moreover, the IPCC 2006a, b, c reference value for the urine fraction (4 %) and dry matter content (8 %) were used (IPCC 2006b). Allocation of manure production between meadow and stable is presented in Table 12 on the next page.

Table 12. Methane conversion factors and manure storage systems (source: IPCC, NIR Belgium, Farmer's union).

Category	Stable manure	Mixed manure	Manure disposal on grassland
Methane conversion factors	2 %	19 %	1 %
% manure suckler cows	40 %	10 %	50 %
% manure young cattle (1-2 years)	60 %	15 %	25 %
% manure calves (<1year)	83 %		17 %

Laughing gas

Through a combination of nitrification and denitrification, N_2O is released from stored manure or is disposed on land. The amount of produced laughing gas emissions depends upon the nitrogen excretion of the animals (Nex). The excreted nitrogen per type of animal is taken from the Belgian NIR report. The amount of N_2O from the total amount of nitrogen depends upon the manure storage. It is assumed that 0.5 % of total nitrogen is converted into N_2O during manure storage. For mixed manure stored underneath the slatted floor, it is assumed that 0.1 % of the total nitrogen is converted into N_2O during storage. For manure disposed on the meadow by the animals, a 2 % conversion into N_2O is assumed (direct emissions) (Table 13).

Table 13. N_{ex} per type of animal (source: NIR Belgium/manure database).

Animal category	Nex (kg/head.yr)
Calves (<1 year)	33
Young cattle (1-2 year)	58
Suckler cows	65

Indirectly, there are N2O emissions formed through volatilized NH3 and NOx.

The amount of NH3 and NOx formed from the manure depends upon storage. Table 14 presents how much of the total nitrogen converts into NH3 and NOx. It is assumed that 1 % of indirect nitrogen losses converts into laughing gas (indirect laughing gas emissions).

Table 14. NH3 or NOx losses from manure as a function of manure storage systems (IPCC 2006).

Manure type	N volatilization (NH_3/ NO_x)
Beef cattle – stable manure (fixed manure)	45 %
Beef cattle – mixed manure storage	40 %
Beef cattle – manure disposal on grassland	20 %

Manure usage for crop production

Pigmeat

When manure surpluses are exported and used on agricultural land for growing crops, emissions have to be divided between the crops (manure usage) and livestock (production of manure surpluses). On the studied pig farm, all produced manure is transported to another farm. Emissions related to the use of manure surpluses are divided, using physical allocation, based upon the nitrogen efficiency (0.60) of the crops (Kramer et al., 1999). This implies that 40% of the emissions related to manure usage are allocated to pig livestock and 60% to the crops. It is assumed that all pig manure is distributed on agricultural land without any prior treatment, representing the most common practice in Flanders. Nitrous oxide emissions accompanying manure usage are calculated according to the IPCC methodology (IPCC 2006a).

Beef

When manure is used on agricultural land for growing crops, emissions are allocated among crops and livestock. All produced manure is disposed of on the farm's own land. Accompanying emissions are described in Section 2.6.1.2.

2.6.4 Transportation emissions

Transport takes place on several levels throughout the production system in question (see Figure 6-7). First, feed components are transported to the feed compound processing plant. For those components grown in Europe, the transportation distances are limited (less than 1,000 km). The soy component might originate from Brazil, Argentina and the USA; and for this maritime transport is needed. Emissions related to both types of components are taken into account in the applied emission factors, and are covered by the production of purchased fodder (see section 2.6.1.2).

Second, the feed compound needs to be transported from the fodder processing plant to the farm. An average distance of 30 km for the purchased fodder is assumed (source: BEMEFA) and the related emissions are covered within the farm data.

Third, animals need to be transported from the farm to the slaughterhouse. The average distance for this in Flanders amounts 25 km (based on the interviews with slaughter houses). Half of the animals are transported with a 23-ton truck (e.g. about 200 pigs), the rest with a 12-ton truck (e.g. about 110 pigs). The associated emissions are taken into account.

2.6.5 Meat processing emissions

Emissions from slaughtering relate to electricity and fuel consumption, the use of cleansing products, water usage and waste processing. Transport of the necessary resources (pigs and other processing materials) is also taken into account. Emission factors related to electricity and gasoline are derived from Table 9. More emission factors were derived from the Ecoinvent database (2011). As a final step, emissions were allocated to meat and other useful by-products. For this, economic allocation was used (see the 'Allocation method' section) in two steps: once at the slaughtering and once upon deboning of the carcass. After the two steps, 63% of the live weight of the pig was retained as pigmeat (see Table 15 below). Using the economic value of pigmeat, compared with the economic value of the by-products, 92% of the emissions were attributed to pigmeat production.

Table 15. Allocation between pigmeat and byproducts.

Step		**Total**	**Carcass – meat**	**Byproducts which can be sold at a positive price***	**Byproducts which cannot be sold at a positive price****
Slaughtering	Mass (kg)	115	90.5	13.6	10.9
	Unit price (€/kg)		1.5	0.4	0.0
	Economic value (€)	141	136 (96.5%)	5	0
Deboning	Mass (kg)	90.5	72.4	16.7	1.8
	Unit price (€/kg)		2.0	0.4	0.0
	Economic value (€)	152	145 (95.5%)	7	0

* Such as blood, head, intestines, liver, hart, tung, bones, fat

** Such as waste and losses

For beef, economic allocation is used at the slaughtering and deboning phase. The carcass yield is 67 % and the meat yield on carcass is 81 % for the Belgian White-Blue race.

2.7 Carbon footprint results

In order to answer the research questions, results are presented as the carbon footprint on the one hand and a sensitivity analysis on the other hand.

2.7.1 The carbon footprint of pigmeat

The results are presented in Figure 9 and can be summarized as follows: 1 kg of pigmeat (after slaughtering and deboning) creates a CF of 5.7 kg CO_2 eq. Production and transport of purchased fodder, as well as manure production and usage, have the greatest share in the overall CF. The slaughtering and deboning process only contributes approximately 4% to the total CF. It is noteworthy to mention that transport of purchased fodder contributes less than 1 % to the total carbon footprint, and thus indicating the less relevant weight of transport on the overall environmental sustainability performance.

Fodder production contributes approximately 63.4% to the total carbon footprint. This is a high proportion and is strongly influenced by the used emission factors. The total impact of Land Use Change (LUC) amounts about 7%. Manure storage accounts for 25.2% of the emissions, whereby 88% relates to methane and 12% to nitrogen gas emissions. The use of manure surpluses accounts for 3.1% and animal methane production for 4.9% of the life weight CF. Energy and water contribute 3.4%, whereby electricity has a share of 33%, gasoline oil 66.9% and water 0.1%.At the slaughterhouse/deboning firm level, an extra 0.15 kg CO_2 eq./kg of liveweight is added. The biggest contribution (79.3%) relates to energy consumption. Waste management of the by-products (about 20 kg per pig) accounts for 16.6%, whereby 61% is related to energy consumption and the rest (39%) originates from fossil fuel combustion. Furthermore, animal transport between farm and slaughterhouse accounts for 4%. Production of process materials (0.1%) is negligible.

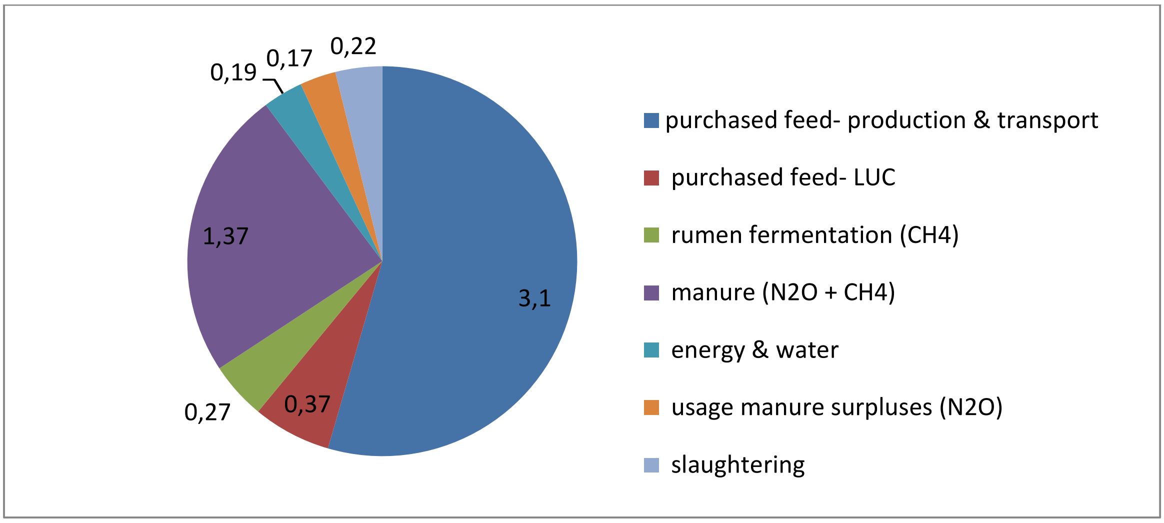

Figure 9. The CF of 1 kg of pigmeat (after slaughtering and deboning) in Flanders, in kg CO_2 eq.

2.7.2 The carbon footprint of beef

Results are presented in Fig. 10 and summarized as follows: a kg of deboned beef meat creates a CF of 22.2 kg CO_2 eq. Rumen fermentation, home-grown crops and manure production and usage, have the lion's share in the overall CF. The slaughtering process contributes 0,01 % to the total CF.

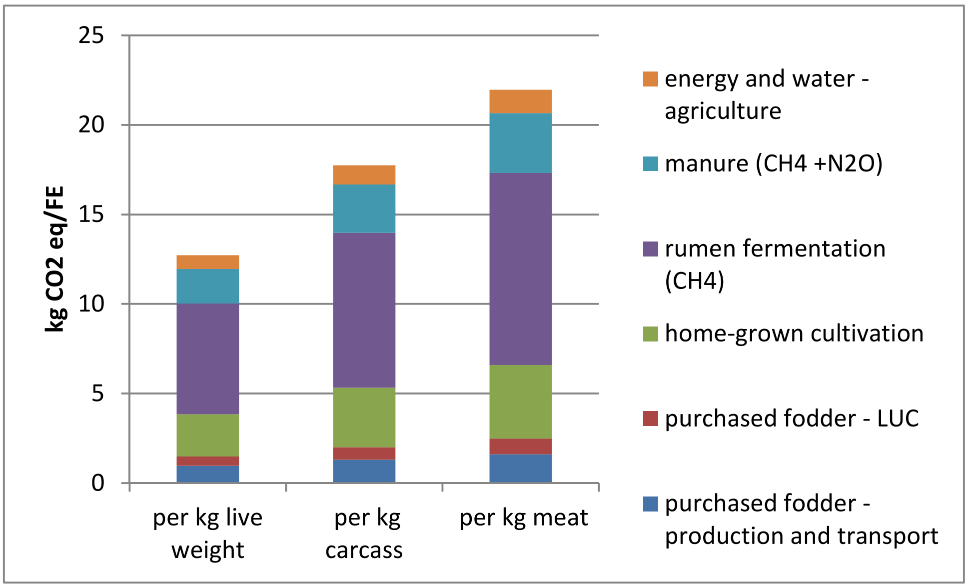

Figure 10. The CF of 1 kg of beef in Flanders (FE = Functional Entity).

The carbon footprint of liveweight amounts 12,7 kg CO_2 equivalents. At farm level rumen fermentation represents 48,8 % of the carbon footprint, strongly determined by the feed uptake and digestibility. Feed consumption is estimated with available data, however the reference values for digestibility are accompanied with a high uncertainty.

Fodder production is responsible for 29.5 % of the CF. Albeit only 13 % of the fodder is purchased, the impact is only 50 % compared to home-grown crop cultivation. The total impact of LUC amounts 4 %. Manure storage and application on grassland contribute for 15,3 % to the emissions (19 % is due to methane and 81 % due to N_2O emissions).

Energy represents 5,9 % (electricity consumption represents 13,8 % of this impact, gasoline oil 86 % and water 0,2 %). The slaughtering process only contributes 0.01% to the total CF. The largest contribution (52,7 %) relates to the waste management of byproducts. Energy consumption contributes for 37,5 % (75 % electricity consumption, 25 % combustion of fossil fuels). Furthermore, animal transport between farm and slaughterhouse amounts 9,7 %. Production of process materials is negligible.

2.7.3 Carbon footprint sensitivity

The single figure CF of meat production should be used with caution, since it is based on specific input data. Giving a range in which the CF is expected to fall, will give a much more robust result (Flysjo et al., 2011b). Therefore, a sensitivity analysis was carried out to define how the single figure can be expected to fluctuate.

2.7.3.1 Pigmeat

First, the effects of some changes in feed and herd characteristics on the CF were estimated (table 16). When the mortality rate for piglets in the first stadium decreases from 12 to 8%, the CF of pigmeat decreases by 0.08 kg CO_2 eq./kg of deboned meat. Greater effects can be identified for an increase in the number of piglets per sow per year from 27.6 to 30, leading to a decrease in CF of 0.14 kg CO_2 eq./kg of deboned meat; and for an increase in the final weight of the meat pigs from 115 to 125 kg, leading to a decrease in the CF of 0.16 kg CO_2 eq./kg of deboned meat.

Table 16. Applied sensitivity analysis: the impact of changes in herd and feed concentrate parameters on the CF of 1 kg deboned pigmeat.

Parameter	**Initial value**	**Min**	**CF shift**	**Max**	**CF shift**
Mortality rate of piglets in stadium 1	12 %	8 %	-0.08	16 %	+0.08
Piglets per sow per year	27.6	25	+0.18	30	-0.14
Final weight meat pigs	115	105	+0.18	125	-0.16
Digestible energy content of fodder (%GE)	85 %	75 %	+0.13	95 %	-0.14
Use of soy beans and meal in the piglet feed	*	-10%	-0.02	+10%	+0.01
Use of soy beans and meal in the meat pig feed	*	-10%	-0.13	+10%	+0.12
Use of soy beans and meal in the sow feed	*	-10%	-0.03	+10%	+0.03
Manure management (solid versus pit storage)	6/94	0/100	-0.01	100/0	+0.18

* See table 4

Second, the impact on the CF of the use of soy beans and meal in the fodder was analysed. It was expected that the impact would be substantial due to the impact of using soy on LUC. Limiting the use of soy in the piglets' feed by 10% can lead to a decrease in the CF of 0.02 kg CO_2 eq./kg of deboned meat. Limiting the use of soy in the feed of the sows by 10% can lead to a decrease of 0.03 kg CO_2 eq./kg of deboned meat, while limiting the use of soy in the feed of the meat pigs by 10% can even lead to a decrease in CF by 0.13 kg CO_2 eq./kg of deboned meat. However, limiting the use of soy with a high emission factor implies that a substitute has to be found (e.g. soy produced without LUC, locally produced protein-rich products, animal by-products, etc.), and an estimated emission factor for those alternatives has to be added to come to a final conclusion on the impact of soy banning policies on the CF of pigmeat (see section 2.8.2).

Third, because manure production is a major contributor to the estimated CF, the impact, on the CF, of changes in manure management were estimated. In the original setup, 6% of the manure was stored as solid manure and the rest as pit storage. A shift towards complete pit storage leads to a decrease in CF of 0.01 kg CO_2 eq./kg of deboned meat. A shift towards complete solid storage leads to an increase in CF of 0.18 kg CO_2 eq./kg of deboned meat. Based on this sensitivity analysis, the estimated CF of pigmeat production in Flanders is expected to lie between 5.5 and 5.9 kg CO_2 eq./kg of deboned meat.

2.7.3.2 Beef

A similar approach as pigmeat is used.

Feed and herd characteristics

Table 17 presents trends in feed and herd characteristics and their possible impact on the CF.

A shift in the digestible energy content of fodder also has an impact. Changing the percentage from 85 to 95% leads to a decrease in the CF of 0.14 kg CO2 eq./kg of deboned meat. Table 17 also shows the shift in CF when an increase in mortality rate, a decrease in piglets per sow, a decrease in final meat pig weight and a decrease in digestible energy content is assumed. Keeping all other parameters in the model constant, and shifting only one feed and herd parameter at a time, leads to changes in life weight CF from minus 0.16 kg CO2 eq./kg to plus 0.18 kg CO2 eq./kg.

Table 17. Applied sensitivity analysis: impact of changes in herd and feed concentrate parameters on the beef's CF.

Parameter	**Initial value**	**Min**	**Shift in CF**	**Max**	**Shift in CF**
Mortality rate animals < 1year	11 %	5 %	-0,6	15 %	+0,50
Mortality rate animals > 1year	1,25 %	0,5 %	-0,05	3 %	+0,1
Calving interval	365	-*	-	420	+2
Final weight bull/cow	680/690	660/670	+0,25	700/720	-0,3
Digestible energy content of fodder (% GE)	76 %	66 %	+1,9	86 %	-1,2

* The initial calving interval is not lowered because not possible.

When the mortality rate for animals <1 year is decreased from 11% to 5%, the CF decreases with 0.6 kg CO_2 eq. per kg of deboned meat. Greater effects are identified for the rise in value for the GE from 76 to 86 %, leading to a CF fall of 1,2 kg CO_2 eq. per kg of deboned meat and a decrease from 76 to 66 % GE, leading to CF rise of 1,9 kg CO_2 eq. Finally, a switch in the final weight also has an impact. Changing it from 680 to 700 decreases the CF with 0.3 kg CO_2 eq. per kg of deboned meat, whereas a decrease in weight to 660 kg increases CF with 1,9 kg CO_2 eq.

Manure storage/disposal

An allocation between manure disposal on grassland (59 %) and in the barn is defined. For the latter, part of the manure is stored as barn manure (8 %) and the other as mixed manure (33 %). In total, three scenarios are considered. It is assumed that 100 % of the manure production is disposed on grassland (scenario 1), as barn manure (scenario 2), or stored as mixed manure (scenario 3) (Table 18). The results are little influenced by these parameters. Storage as stable manure provokes the least emissions. With extended time on grassland, animals consume more energy. The calculated gross energy is higher—hence, the rumen fermentation associated methane emission.

Table 18. Variation on parameters regarding manure storage.

Parameter	Initial value	Scenario 1	Scenario 2	Scenario 3
Disposal grassland	59 %	100 %	0 %	0 %
Stable manure	8 %	0 %	100 %	0 %
Mixed manure	33 %	0 %	0 %	100 %
Result (relative towards initial)	1	1,01	0,97	1,02

Allocation method influence

Byproducts are produced at slaughterhouses, and deboning facilities. Emissions from these byproducts need to be taken into account as well as they have made their way through the production process. It is a matter of allocating the emissions to the right product. Economic allocation was opted for since this method takes into account the products' value. An alternative allocation method is based upon mass basis.

Fig. 11 presents the impact of using either mass or economic allocation methods on the overall CF, applied to three different functional units (live weight, carcass and deboned meat).

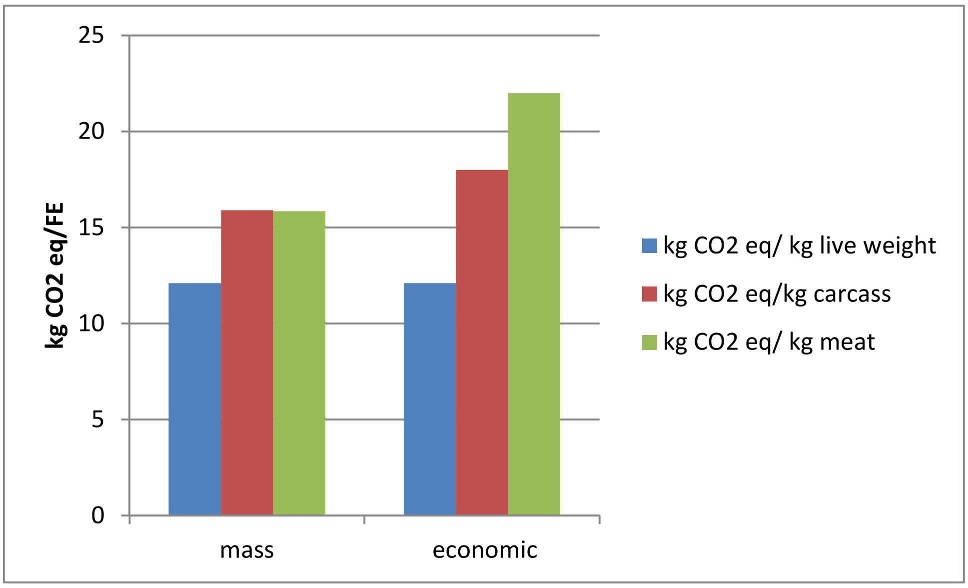

Figure 11. Sensitivity on allocation method for beef meat's CF (FE = Functional Entity).

2.8 Discussion and conclusions

2.8.1 Relative importance

In order to fully understand what this CF of Flemish pigmeat and beef means, it is helpful to compare the results with those of non-Flemish pigmeat/beef and the CF of other animal products. Comparing the results with other international studies is difficult due to different choices made in relation to allocation, functional unit and/or system boundaries (see section 2.1) on one hand and the lack of transparency on the other hand.

Some authors report a similar CF for pigmeat, while others report a lower as well as a higher CF. For example, Leip et al. (2010) found a CF of 7.5 kg CO_2 eq./kg of meat for European pork meat, using a physical allocation method and taking into account LUC. Williams et al. (2006) reported a CF of 6.4 kg CO_2 eq./kg of deboned pigmeat. Carlsson-Kanyama (1998) reported a CF of 6.1 kg CO_2 eq./kg for Swedish pigmeat, including transport to the consumer. Leip et al. (2010) reported a CF in the UK of 5.7 kg CO_2 eq./kg for pigmeat. In 2007, Blonk Consulting reported 5.0 kg of CO_2 eq./kg of pigmeat CF in the Netherlands (Blonk, 2007). However, 1 year later, Blonk Milieuadvies (Blonk, 2008b) found a lower CF of 3.6 kg of CO_2 eq./kg of pigmeat in the Netherlands. Although they used a similar methodology to the one in this report, they did not include soy in the feed, while this component is a very important contributor to the CF in the current study. A similar CF was found by Dalgaard et al. (2007) using the ISO14044 methodology to calculate CF of 1 kg Danish pork, delivered to the port of Harwich (UK). Without using an allocation method and not taking into account LUC, they found a CF of 3.6 kg CO_2 eq./kg of pigmeat.

Next, the CF of pigmeat can be compared with the CF for beef and milk production. Within the same study it is shown that the CF of Flemish beef lies between 22.2 and 25.4 CO_2 eq./kg of deboned meat and of milk between 1.03 and 1.36 kg CO_2 eq./kg of milk consumed (1.5% fat).

Looking at consumption, it is known that in Flanders people consume more pigmeat than beef per annum (namely 6.8 kg of pigmeat and 5.6 kg of beef in 2010 (GfK 2013)). Owing to the much higher CF per kg of beef compared with pigmeat, the contribution of pigmeat to the total CF for consumption is much lower: eating pigmeat leads to an average annual production of 38.0 kg CO_2 eq., while eating beef leads to an average annual production of 132.3 kg CO_2 eq. The consumption of milk also makes a higher contribution to a person's CF than the consumption of pigmeat. On average, Flemish people drink about 52.7 L (which equals 54.4 kg) of milk per annum (GfK 2013), leading to an average annual production of 65.0 kg CO_2 eq. Therefore, the importance of pigmeat consumption in the Flemish diet is limited compared with milk and beef.

When production is considered, recent reports for Flanders (Platteau et al., 2012) show that the CF for milk production amounts to 2.1 billion kg CO_2 eq. (for 2 billion litres of milk); and for beef production amounts to 5.0 billion kg CO_2 eq. (for 0.3 million tons of beef) and up to 6.3 billion CO_2 eq. for pigmeat (for 1.1 million tons of pigmeat). This shows that the importance of pigmeat production in Flanders, in terms of GHG emissions and the region's CF, is quite important and much higher than the production of milk.

Until such time as a similar study using the same method is undertaken for all the other components of the Flemish diet it is impossible to precisely determine the relative importance of pigmeat consumption.

When looking on a global scale, it can be stated that only food production already might exceed total global targets for greenhouse gas emissions by 2050 (as in, emissions targets for everything). The food system however is a massive semi-hidden variable in the global warming progression. As such, some studies make the argument that changing the global food system is crucial to slowing climate change and for ensuring food supplies in general (Bajzelj et al., 2014).

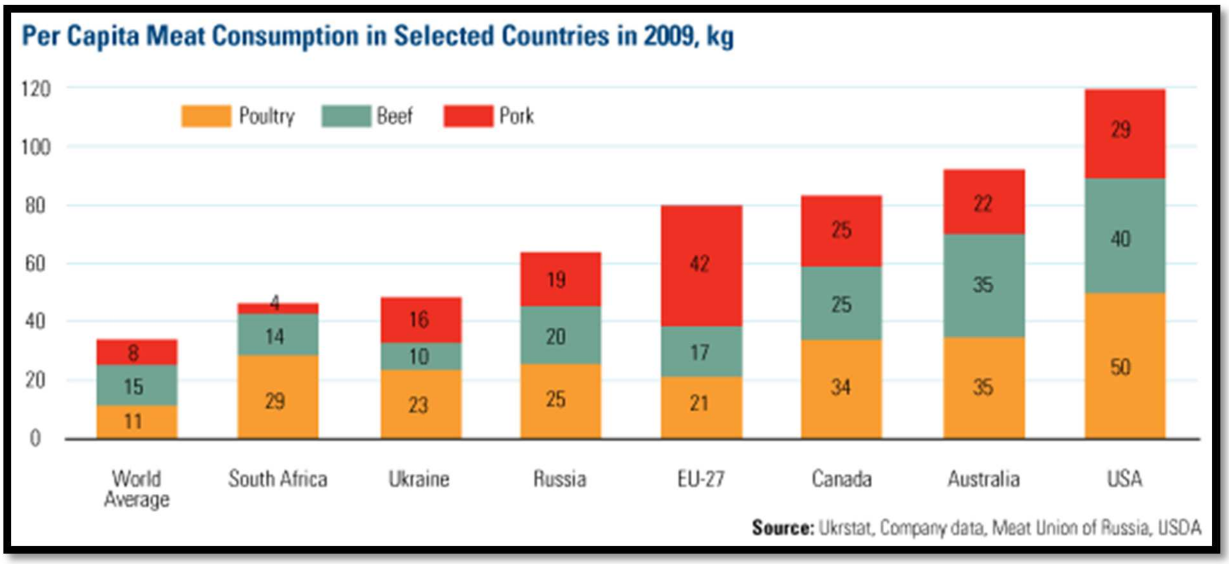

Figure 12. Worldwide per capita meat consumption (Source: Ukrstat).

Research has shown that future population growth coupled with current demands for food might imply almost a doubling of global crop production by 2050. Farmers are slowly increasing crop yields, but only at about half of the rate needed to meet future demand (Bajzelj et al., 2014).

Agriculture is a major contributor to climate change and pollution, and so further expansion is questionable. It is hence almost imperative to achieve global food security without expanding crop or pastureland and without increasing greenhouse gas emissions (Bajzelj et al., 2014).

Fortunately, according to the studies, this food system shift does not force everyone to switch to soylent-based diets so much as jumping to generally healthier diets in the conventional food pyramid sense. Healthier diets are ought to be more efficient diets at the production end, which in turn means tearing up less land and hacking apart fewer GHG-absorbing trees (Bajzelj et al., 2014). It should not be the aim to all become vegetarians, it should be an argument to eat meat in sensible amounts as part of a more healthy and balanced diet.

2.8.2 Mitigation measures

This penultimate section will address issues on how to lower the CF and the accompanying climate change impact for increasing sustainability. This can be seen as a Sustainability Impact Assessment (SIA) with the aim being to identify mitigation measures for minimizing and even avoiding negative and adverse impacts.

2.8.2.1 Pigmeat

Results revealed two major hotspots in the life cycle of pigmeat production for which the highest contribution to GHG emissions can be identified: fodder production, manure production and usage. Based upon these hotspots, it is possible to define opportunities to reduce the pigmeat's CF. In particular, the composition of feed has a major impact on the CF.Within Europe, the use of, for example, soy bean in feed concentrates has grown dramatically. However, the use of soy has a substantial impact on the CF because of the negative LUC impact and the need to transport the feed over long distances (Hortenhuber et al., 2011). Therefore, using regional products instead of overseas products can reduce the CF by limiting the consumption of energy needed for transport.

When overseas products are necessary, preference should be given to products which are sustainably produced and having a limited LUC impact. Hortenhuber et al. (2011) show that limiting the use of soy in feed compounds will have a positive impact on the CF. In their study on milk, replacing soybean meal by 50% with alternative regionally produced, protein-rich feed leads to a decrease of about 26% in GHG emissions by dairy cattle.

However, the composition of the feed does not depend on its contribution to GHG emissions, but more on availability, price and characteristics of the feed compound components. Price and availability are two important economic factors influencing the final price of the feed and the possible usage of it by farmers. Hortenhuber et al. (2011) comment that regionally produced alternatives are not always available within Europe. Shifting production in Europe towards these alternatives might lead to LUC effects in Europe (Steinfeld et al., 2006). The increase in demand for the alternatives and limitations in the supply of traditional products might put pressure on the prices of both products. Therefore, one cannot simply suggest that all carbon-negative components should be banned, because this would limit the economic sustainability of farming practice.

The characteristics of the components will define whether they can be combined to provide a digestible feed for pigs. A pig, like any other animal, needs a balanced diet meeting all nutritional requirements. The inclusion of certain feed components is therefore necessary, while the inclusion of others might have upper limits. For example, excluding soy products from the feed compound would increase the need for protein-rich alternatives, which might not all be as digestible for pigs as soy. Moreover, these alternatives can be very expensive, thereby limiting the usability of the feed (as described above). This economic aspect is often neglected in other literature, as described by Verspecht et al. (2012).

Therefore, parameters as price, availability and characteristics of the feed components need to be considered alongside the CF to ensure that the production of pigmeat is not compromised in an effort to reduce the GHG emissions (Espinoza-Orias et al., 2011).

Manure production, storage, management and usage represent the second important contributor to the CF. Especially in Flanders (and by extension also in the Netherlands) manure production and usage creates a serious problem, not only in relation to GHG emissions, but also in terms of nutrient leakage and water pollution (Verspecht et al. 2011). By improving the nutrient efficiency, for example, by processing the manure (Masse et al. 2010), several problems relating to sustainability could be dealt with at the same time.

In Flanders, one type of manure management, which is the most popular, involves the separation of liquid and solid manure components. The liquid element is then cleaned so that it can be disposed of as water. The solid component contains all nutrients from the manure and can be used as fertilizer. This solid part will eventually create a similar quantity of nitrate emissions as would do the storage and use of untreated animal manure. Some sources point out that the emissions of nitrate might even be higher (Hansen et al. 2006). In terms of methane emissions, a decrease can be expected, although the exact decrease is hard to estimate and will depend on the treatment conditions (Sommer et al., 2000). In the study at hand, not enough data were available to estimate the exact impact of manure management on the CF.

As was the case with changes in the feed, described above, parameters such as price and availability will also have an important influence on the mitigation possibilities in relation to manure.

Veillette et al. (2012) for example describe how the system of biofiltration can substantially reduce methane emissions. This system is however very expensive: investment costs amount € 870 per m³ of filter material, whereas operational costs amount € 0,25-1,25 per 1000 m³ of treated air (Energie en milieu-informatiesysteem van het Vlaamse Gewest, 2012).

Both issues exemplify the potential trade-offs between dealing with GHG emissions and other aspects of sustainability. Sustainability consists of three pillars: environmental protection, economic growth and social equity, and it is only when a mitigation measure positively affects all aspects that it leads to better sustainability. For example, a reduction in the CF at farm level (e.g. by adapting the feed diet) needs to be combined with food safety and public health, product quality, genetic diversity, efficiency, environment, animal health and welfare in an economically viable way (Hoffmann, 2011).

Moreover, it is important to note that the CF is a good indicator for GHG emissions in relation to climate change impact, but is not an indicator for environmental impact in general.

2.8.2.2 Beef

Three huge hotspots in the production chain were revealed: rumen fermentation, fodder production, manure production and the usage of it. Some opportunities to reduce the CF of beef are defined.

In particular, the composition of feed has a very big impact on the overall CF. Within Europe, the use of soy meal in feed concentrates has increased rapidly. However, the use of soy has a substantial impact on the CF (negative LUC impact and transportation of feed components over long distances (Hortenhuber et al., 2011)). Therefore, replacement with regional products, can reduce the CF. When overseas products are to some extent indispensable, priority should be given to products produced in a sustainable way with a restricted impact on LUC. However, the composition of the feed depends more on availability, price and the characteristics of the components. Price and availability are major economic factors influencing the final price of the feed and the possible usage by farmers. Hortenhuber et al. (2011) clearly indicate that regional and local products are not always at people's disposal. Shifting production in Europe towards these alternatives might lead to LUC effects in Europe (Steinfeld et al., 2006). However, one cannot remove all carbon negative components, since this limits the economic sustainability of farming practices.

Therefore, parameters as price, availability and feed component characteristics need to be taken into account alongside the CF to ensure that meat production is not compromised in an effort to reduce the GHG emissions (Espinoza-Orias et al., 2011). These economic aspects are often neglected in literature (Verspecht et al. 2012). Manure production, storage, management and usage is the second largest contributor to the overall CF. In Flanders, one type of manure management, which is the most popular method, involves separation of liquid and solid components of manure. The solid part gives rise to a similar quantity of nitrate emissions as the storage and use of untreated animal manure would do.

Both things exemplify the possible trade-offs between dealing with GHG emissions and other aspects of sustainability, put in a larger perspective. A mitigation measure only has a positive affect when all aspects lead to better or higher sustainability. Moreover, it is important to stress that the CF is a good indicator for GHG emissions as 1 environmental indicator, but it is not an indicator for environmental impact in general. It only reports on one single environmental impact and when considered alone and not placed alongside other potential environmental impacts, it could misdirect resources away from actions which are more important (Ridoutt et al., 2011). This proves once again the need for an integrated sustainability approach.

2.9 CF Limitations

Accounting greenhouse gases, referred to as carbon footprints, is a common used method for assessing climate change impacts. It is the main focus of many sustainability policies in governments and companies (see section 1.6.4 about the EU Gothenburg Sustainable Development strategy). In fact, carbon footprint is often used as an indicator for sustainability in general, and environmental sustainability in particular. However, environmental issues concern more than climate change (e.g. chemical pollution, depleting natural resources, …). Solely focusing on carbon footprints might bring about the event of problem shifting or even pollution swapping. Reducing the carbon footprint hence might cause an increase in other environmental issues. It is however clear that in many situations the carbon footprint is a poor representative of the total environmental impact of products. The toxic submissions for instance do not covary with climate change impacts (Laurent et al., 2012). As such, pollution swapping might occur as products are managed to become more green due to problem shifting towards other environmental impacts.

With the aim to fully quantify the environmental impact of products, a Life Cycle Assessment (LCA) takes into account all emissions and resource consumptions from a life cycle point of view. LCA is covering a broad range of impact categories and has the ability to come up with a straightforward assessment of the environmental impact. Many studies have tried to aggregate several environmental impact categories in order to create simplified indicators, serving as proxies for the environmental performance (e.g. ecological footprint). Nevertheless, no study emphasizes how these aggregated indicators can be used in a decision-making context, accompanied with a risk of problem-shifting.

As such, carbon footprints seem more easy to perform compared with a full Life Cycle Impact Assessment. Climate change issues have come on top of many political agendas. This gave rise to the development of stand-alone greenhouse gas methods. Many recent initiatives for standardizing carbon footprint calculations like PAS2050, PAS2060, ISO14067 were developed and thus indicate the main focus of environmental policies nowadays.

Carbon footprint has the ability to raise awareness about the environment among stakeholders of any kind. However, the limitations of the carbon footprint as representing the whole environmental impact should be handled with care. In very few cases, carbon footprint might serve as a good proxy for environmental sustainability. This occurs when a correlation can be found between carbon footprint and other environmental impact indicators, originating from covarying processes. However it is outside the scope of this PhD dissertation to scrutinize the correlation between carbon footprint and other environmental impact indicators. Carbon footprint as environmental sustainability proxy needs to be used and documented on a case-by-case basis (Laurent et al., 2010).

Carbon footprint relies on Life Cycle Assessment, which has limitations. A potential weakness of LCA is the huge amount of data involved, the availability of those data, and the resource and time intensities of LCA (Cucek et al., 2012). The first limitation is the high degree of uncertainty arising from the Life Cycle Inventory causing the results to show high variability (Guinée et al., 2002; Curran, 2006; Finnveden et al., 2009). A next limitation is the lack of a method for generating and revealing sustainable solutions (Grossman and Guillén-Gosalbez, 2010; Gerber et al., 2011).Moreover, conventional LCAs are mostly for new technologies, based upon average technology at labscale and extrapolated to large scale (Gerber et al., 2011).

Apart from the few cases where carbon footprint is a good proxy, it should be seen as a transition indicator (Laurent et al., 2012). This implies that it should be used as a leverage in order to progress towards more integrated approaches. It hence proves the need for a complete environmental sustainability on one hand, but moreover an overall and integrated sustainability assessment on the other hand.

Recycling is great, but the ultimate goal is to get people to prevent waste.

(McKenzie Jones, Arizona Daily Sun, 11/2/2016)

3 Sustainable household waste management

Adapted from:

Peer-review web of science journal papers:

Gellynck, X., Jacobsen, R., Verhelst, P. (2011). Identifying the key factors in increasing recycling and reducing residual household waste: A case study of the Flemish region of Belgium. *Journal of Environmental Management* 92, 2683-2690.

Jacobsen, R., Buysse, J., Gellynck, X. (2013). Cost comparison between private and public collection of residual household waste: Multiple case studies in the Flemish region of Belgium. *Journal of waste management* 33, 3-11.

3.1 Introduction

Modern societies generate far more solid waste than ever before. Solid waste management handles all of this garbage. Municipal waste collection is a part of solid waste management. Garbage collection entities remove tons of garbage yearly and make it ready for recycling or final disposal. The end goal is a reduction of the amount of garbage clogging the streets and polluting the environment, whether that garbage is disposed of or recycled into something useful. Solid waste management also focuses on developing environmentally sound methods of handling garbage.

Waste has been a huge topic on the environmental agenda, however the problem of increasing waste generation has not been covered to a large extent.

Waste generation has positive income elasticity, its generation grows as income grows. The cities of Western Europe, as well as those of North America, generate more solid waste than they did a century ago (Porter, 2002). In the UK however, waste growth appears to have started to diverge from GDP growth, though the reasons are not yet fully understood (Parfitt and Robb, 2009). According to OECD statistics, municipal waste generation increased by 14% between 1990 and 2000, from 530 to 605 million tonnes. Measured per capita, municipal waste generation increased from 509 to 540 kg, a rise of 6% (de Tilly, 2004). In Western Europe, municipal waste generation measured per capita increased from 476 kg in 1995 to 580 kg in 2003, a rise of 22% (European Environment Agency, 2005).

This chapter covers sustainable waste management. Environmental sustainability should be seen in the light of waste reduction, with the aim of identifying factors impacting waste generation. Economic sustainability refers to the efficient use of scarce resources, and thus implies that waste management services preferably occur in an economic efficient way. From this perspective, the cost of waste collection services is scrutinized.

3.2 Sustainable waste management

As indicated before, the central mandatory goal for the Flemish competent waste authority is obtaining an amount of 150 kg residual household waste per capita per year for all municipalities. This amount can be seen as a policy target for sustainable household waste management. It aims to reduce the environmental impact of waste through the recovery, recycling, and reuse of resources, and the minimization of waste streams. This includes the management of resources in an environmentally sound and economically effective way.

Many policies are installed in order to reach this objective (150 kg). The question remains how effective these policies are on the one hand, and on the other hand which factors are able to contribute in reaching this target. Compliance with these factors can help in defining strategies for improving sustainability related to waste production.

3.2.1 Waste generation

For its producer, the marginal private cost of dumping waste in nature is zero. This is however not the case when there is naming and shaming (higher private cost of dumping). As some wastes have negative effects on human health and the environment, the damaging effect on nature or society as a whole may be considerable, although the direct effect on any individual will be rather small. There are however examples of pollution with a high cost for the individual: noise pollution, particulate matter emissions and heavy metals. The marginal social cost of waste disposal can thus be considerable. The difference between the marginal private cost and the marginal social cost is the marginal external cost. When it comes to waste, the market fails such that the price of creating and dumping an extra unit of waste does not match the marginal social cost (Porter, 2002). Hence, waste policies are necessary. When production and consumption cause negative externalities such as pollution and waste, governments should intervene in private markets and attempt to attain the level of waste or pollution where the marginal social benefit of reducing it by one more unit is just equal to the marginal cost of reducing it. Overall social welfare is maximized and economic efficiency is reached (Walls, 2004).

3.2.2 Waste policies

Within the European Union, policies to address waste issues are increasingly formulated by the European Commission and subsequently enacted by Member States. Council Directive 75/442/EEC (European Council, 1975) urges the European Member States to take appropriate measures to encourage the prevention and the reduction of waste production and to ensure that waste is recovered or disposed of without endangering human health and without using processes or methods harming the environment or the resources of future generations. In order to realize these objectives, the obligation for Member States to draw up a waste management plan is mandated.

In the Flemish region of Belgium, the government implemented Council Directive 75/442/EEC (European Council, 1975) in the Waste Decree (Afvalstoffendecreet, 1981). The responsibility for prevention, recovery, collection and treatment rests with the municipalities. Local authorities decide on the terms of waste collection (Article 6, Afvalstoffendecreet, 1981). However, the regional government determined what type of waste has to be collected separately in Vlarea (2003). The OVAM, the Flemish competent waste authority, is responsible for drawing up sectoral waste management plans. The ‘Implementation plan household waste 2003-2007’ (OVAM, 2002) and the ‘Implementation plan sustainable management’ (OVAM, 2008) incorporate all European and regional requirements and both plans describe strategies, goals, actions and instruments for the collection and treatment of household waste. The plan is binding to local authorities unless otherwise stated (Article 36, Afvalstoffendecreet, 1981). The central mandatory goal for Flemish municipalities was to reduce the amount of residual household waste to 150 kg per capita per year by 2007

(OVAM, 2002). The new Implementation plan wants to maintain the amount of residual household waste to 150 kg per capita per year between 2010 and 2015 (OVAM, 2008). This amount can be seen as a policy target. This figure is a reasonable target to strive for, based upon all data from Flemish municipalities over the last years. The implementation plan includes the provision of unique recycling services to suit the variety of residential conditions. In literature, a reasonable body of information has been published on the effectiveness and efficiency of a variety of policy instruments, but the information is complex, often contradictory and difficult to interpret, giving significant problems to those responsible for developing waste strategies and policies (Martin et al., 2006).

According to Noehammer and Byer (1997) each local authority has to adapt to its own socio-economic conditions, so it is not possible to develop one waste collection system that could be adopted by all. There is no single, ideal design and the characteristics and needs of the community should dictate the scheme's design (Williams and Kelly, 2003). Also Tucker et al.

(2000) state that the local context is important and that any successful change in the design of one scheme may not necessarily be replicable elsewhere. Shaw et al. (2006) on the contrary argue that regional differentiation may be less important than previously perceived. Their data show a similar pattern of recyclable material occurrences in household waste across broad geographical and socio-demographic ranges. They conclude that given the apparent consistencies in material occurrence and recovery, the adoption and implementation of waste collection schemes sharing common objectives (with respect to the range and quantities of materials that are recovered) are feasible. These broad similarities also offer considerable scope for performance enhancement through exchange of best practice, based on the cumulative experience of the waste management community.

The objective of this chapter is to identify key factors in residual household waste minimization for the Flemish region of Belgium, using binary logistic regression. Reaching the mandatory goal of 150 kg residual household waste per capita per year is used as dependent variable. Independent variables that could explain a municipality's chance of reaching this goal are derived from literature. Findings from this case-study add to the knowledge of waste management policy instruments.

3.2.3 The case of Flanders

Flanders is the northern region of Belgium, situated between the Netherlands and France, and bounded by the North Sea. It has 6.0 million inhabitants, with a population density of 444 inhabitants/km². It is subdivided in 308 municipalities (NIS, 2004). Through the 'Implementation plans household waste 2003-2007 and sustainable management 2010-2015', the Flemish Government directs environmental policy. However, the municipalities remain responsible for prevention, collection and treatment of household waste. This has resulted in a wide variety of policy approaches. The municipalities decide to some degree on fees, receptacles and type and frequency of collection. However, municipalities are encouraged to adhere to the objectives through instruments put forward in the implementation plan. The Flemish authorities subsidize municipalities who implement suggested instruments. Early adopters receive a significant subsidy. The level of subsidization then diminishes through the years.

The main objective of the 'Implementation plans household waste 2003-2007 and sustainable management 2010-2015' is to reduce and maintain the level of residual household waste to 150 kg per capita per year. Residual household waste consists of mixed household waste collected through curbside collection, bulky household waste and municipal waste such as street-cleaning residues, waste from markets and illegal dumping (OVAM, 2002). To reach this objective, a variety of instruments are put forward. The 'polluter pays principle' is selected as the adequate economic instrument to reach this goal. A price of € 1.50 per 60 L waste bag is recommended in order to encourage citizens to reduce the amount of residual waste and encourage recycling efforts. The option of recycling is made available through a curbside recycling program for several recyclables such as paper and cardboard, PMD (plastic bottles and flasks, metal packaging and drink cartons) and in some municipalities even organic waste, glass and other recyclables. Furthermore, each municipality has to have a drop-off facility. All these activities are conducted to make it convenient for citizens to reduce waste through recycling.

The policy mix instituted by the Flemish authorities is implemented in varying degrees by local authorities. This gives variability that makes it possible to assess which instruments have a significant impact on reaching the goal of reducing the amount of residual household waste to 150 kg per capita per year. Binary logistic regression will be used to determine the relevant variables.

3.2.3.1 Data collection

The model contains 10 variables and 3 dummy variables. For each variable, data were gathered from the municipalities to set up the model in order to get insights in what factors contribute significantly to decrease the amount of residual waste to 150 kg per capita per year. In this way, the study differs from others, since the amount is a target set by the Flemish waste authority and the model will reveal those factors that will help to reach that target for the Flemish situation. All 308 municipalities in the Flemish region were surveyed. For the data analysis, nine coastal municipalities were left out. Coastal tourism creates an extra amount of waste in a few months, which has a negative impact on the outcome of the model. The total amount of waste created does not reflect the amount of household waste originating from the native inhabitants. Specific actions such as a higher frequency of recycling collection during high season or specific receptacles for recyclables on camping sites are set up to tackle this problem (OVAM, 1998). Furthermore, three other municipalities were also excluded as outliers. Their amount of residual household waste/capita was more than three standard deviations higher than the mean value for all municipalities. Finally, another municipality did not fully participate so the necessary information could not be gathered. Consequently this municipality was left out. As a result 295 of the original 308 municipalities were included in the final analysis.

In a first step, data were gathered from municipalities' web sites and brochures on waste management. Then these data were verified and completed via telephone interviews with the competent public official of each municipality. To control for municipality characteristics possibly influencing waste generation, data on this matter were gathered through the National Institute of Statistics (NIS). Included variables were average income/capita, the area of a municipality per inhabitant and the number of companies. In order to divide municipalities into a group that already reached the goal of 150 kg/capita residual household waste in 2003 and a group that did not, the amount of residual household waste/capita for 2003 for each municipality was obtained from the regional competent authority, namely OVAM.

A structured questionnaire was compiled to obtain data on instituted pecuniary incentives and characteristics of waste management programs in place for the year 2003. Information was collected on the use of a flat rate fee, a pay-by-the-bag system or a weight-based fee, the type of residual waste and recyclable collection methods utilized both curbside and drop-off, the existence of composting programs and descriptive data on waste legislation.

3.2.3.2 Model specification

Following Pampel (2000) it was found that each municipality has a probability of reaching the goal of 150 kg/capita residual household waste, defined as P_i. Several factors will influence the municipality's probability of reaching the goal, gathered in a vector X, so that

$$P_i = F(X, ß)$$

The set of parameters ß reflects the impact of changes in X on the municipality's probability of reaching the goal of 150 kg/capita residual household waste. Using linear regression with a dichotomous dependent variable is inappropriate. The binary logistic or logit transformation is used because of its desirable properties and relative simplicity. The logit transformation begins by transforming probabilities P_i into odds, O_i. Probabilities vary between 0 and 1, and express the likelihood of an event as a proportion of both occurrences and non-occurrences. Odds express the likelihood of an occurrence relative to the likelihood of a non-occurrence, so that

$$O_i = P_i / (1 - P_i) \text{ implying that } P_i = O_i / (1 + O_i)$$

Second, taking the natural logarithm of the odds, the logit equals

$L_i = \ln [P_i / (1-P_i)] = b_0 + b_1X_i$

Expressing the probabilities rather than the logit as a function of X gives

$P_i / (1 – P_i) = e^{b0+b1Xi}$

Solving for P_i gives

$P_i = (e^{b0+b1Xi}) / (1 + e^{b0+b1Xi}) = O_i / (1+ O_i)$

The details on the variables in the vector X are given in Table 21 and justified in the next section.

Only 5.8% of the investigated municipalities use a system of weight-based pricing for residual household waste collection. Most of the municipalities have a once-a-week collection of household waste and a separate collection of organic waste at the curbside (56.6% and 60% respectively).

3.2.3.3 Variables in the empirical logit model

Most of the variables included in the empirical model were chosen based upon theory and/or evidence from previous studies. Some variables were, however, included based on a hypothesized relationship with the dependent variable. Weight-based pricing, frequency of collection and separate organic curbside collection are chosen as dummy variables due to the fact that these variables can only take on 2 values, whereas the others can have a range of values.

AVGINC refers to the average income per capita in € 1000. On a country level, increasing per capita income results in the generation of more solid waste (Porter, 2002). The same holds on a municipality level. In the UK however, waste growth appears to have started to diverge from GDP growth, though the reasons are not yet fully understood (Parfitt and Robb, 2009). According to Eurostat statistics the average income in Belgium increased with 23% between 1998 and 2005, on yearly basis from € 29 600 to € 36 700. Measured per capita, municipal waste generation increased during the same period from 530 to 547 kg, a rise of 3.1%. McFarland (1972) determined a small, positive income elasticity of demand for waste collection services of .178. Podolsky and Spiegel (1998) found the strongest relationship between waste quantities and income with an income elasticity of demand for waste collection services of .55. Other studies also found a positive but weaker relationship between income and waste. Throughout the literature, estimated income elasticity of demand for waste collection services include .279 and .272 (Wertz, 1976), .22 (Reschovsky and Stone, 1994), .242 (Richardson and Havlicek, 1978), .262 (Kinnaman and Fullerton, 1997), between .2 and .4 (Efaw and Lanen, 1979) and .41 (Jenkins, 1993). Results of Gellynck and Verhelst (2007) indicate that as the annual average income of people in a municipality increases by € 1 000, the residual household waste collected increases with 1.822 kg/capita. This gives an income elasticity of demand for waste collection services at the mean of the data for residual household waste collection services of .326. While the estimates for income elasticity of demand for waste collection services vary by a factor of almost four, all show waste collection services to be a normal good (Morris and Holthausen, 1994). However, Hong et al. (1993) estimated a positive but statistically insignificant relationship between waste and the wage rate, which can be considered as a proxy for income. Only Cargo (1976) found a negative correlation between waste generation and income.

Table 19. Summary statistics of the regression variables.
Units are explained in body of the text below the table.

Variable	Mean	Standard deviation
MSW (1≤150 kg/capita/year, 0≥150 kg/capita/year)	146.81	35.60
AVGINC: average income	26.30	3.31
AREA: area per capita	3258.44	2592.79
COMP: number of companies	473.68	956.43
FEE: flat user fee	65.68	26.38
COST: yearly cost of curbside collection	51.37	19.59
PERDIR: % of direct costs in total costs of waste management	0.68	0.23
CREC: number of waste fractions collected at the curbside	21.68	6.21
DOREC: number of waste fractions via drop-off recycling	7.19	1.12
CMASTER: number of compost masters per 1000 inhabitants	0.43	0.38

(continued)

Dummy variable	0	1
WEIGHT: weight-based pricing	94.2%	5.8%
FREQ: frequency of collection	56.6%	43.4%
ORGANIC: separate curbside collection	40.0%	60.0%

Furthermore, the more affluent are more likely to be recyclers, be it through a curbside collection scheme or recycling centers (Belton et al., 1994; Domina and Koch, 2002; Gamba and Oskamp, 1994; Jakus et al., 1997; Martin et al., 2006; Oskamp et al., 1991; Perrin and Barton, 2001; Tucker et al., 1998; Vencatasawmy et al., 2000; Vining and Ebreo, 1990). As the more affluent recycle more and thus divert waste to the collection of recyclables, less residual household waste will be collected. However, results concerning household income and recycling are ambiguous as other studies have not found a significant relationship between income and recycling (Derksen and Gartrell, 1993; Do Valle et al., 2004; McDonald and Ball, 1998; Scott, 1999).

AREA is the area per capita in km^2. Cargo (1976) found that waste generation is positively correlated with population and density. Also Johnstone and Labonne (2004) found that population density and (more ambiguously) the degree of urbanization have positive effects on household waste generation. Dijkgraaf and Gradus (2004) on the contrary found that area per inhabitant increases the waste stream.

COMP gives an indication of the number of companies in a municipality. Residual household waste comprises residual waste from small commercial and industrial activities that cannot be separated from residual household waste because of its small amount. It is hypothesized that a municipality with a high number of companies, in particular SMEs, is likely to have a larger amount of residual waste because of this, although assimilated waste (residual waste from SMEs) is considered in a specific implementation plan and is collected and measured separately (OVAM, 2000).

FEE refers to a flat user fee to be paid by households for the collection and treatment of waste. If no flat user fee is due, the value will be 0. Traditional waste management systems charge residents a fixed annual fee for waste collection services. However, this system provides residents no financial incentive to minimize the total amount of waste they produce. The cost of contributing one additional bag of residual waste to the household is zero, which suggests households will generate more waste than is socially desirable (Kinnaman, 2006). However, this fixed annual fee differs between municipalities and McFarland (1972) found an inelastic price elasticity in the demand for waste collection services based on differences in fixed fees of -0.455.

COST measures the yearly cost (direct + indirect) of curbside residual household waste collection for a representative household in €. Economic literature devoted to designing waste management policies to achieve the efficient quantity of waste and recycling, argues that municipalities should charge according to marginal costs to maximize economic efficiency instead of charging a fixed annual fee (Porter, 2002). The most direct approach is to tax or charge each bag of waste presented by the household. In response to this fee, households could reduce the amount of waste they generate or divert some materials for recycling (Kinnaman, 2006). In practice, communities adopting some form of unit-pricing usually turn to average cost pricing that sets the unit-price equal to the average total cost per unit (Miranda et al., 1996). This is commonly implemented through a bag/tag program. It requires households either to purchase specific garbage bags, or purchase stickers or tags to affix on each of their own garbage containers or bags. Only garbage identified with the bag, sticker, or tag is collected. Results of studies estimating the change in disposal behavior by households facing unit-based pricing programs consistently estimated the demand for garbage collection services to be inelastic (Dijkgraaf and Gradus, 2004; Fullerton and Kinnaman, 1996; Hong, 1999; Jenkins, 1993; Kinnaman and Fullerton, 1997; McFarland, 1972; Miranda et al., 1994; Morris and Holthausen, 1994; Podolsky and Spiegel, 1998; Van Houtven and Morris, 1999; Wertz, 1976). However, Hong et al. (1993) found that a user fee does not appreciably affect the quantity of waste produced at the curb. Efaw and Lanen (1979) found a high inelastic price elasticity of demand for waste collection services, if not perhaps zero or even positive in sign. Pay-by-the-bag systems are likely to perform best when there is a comprehensive system in place for the collection of segregated materials for recycling (OECD, 2004). Several studies attribute part of the effect of the introduction of a pay-by-the-bag system to accompanying recycling programs. Kinnaman and Fullerton (2000) developed a model of household behavior with empirical implications.

Households are predicted to respond to an increase in the value of the user fee by decreasing the quantity of waste presented at the curb. They state that albeit the implementation of a municipal recycling program diverts some material from waste to recycling, it also frees up additional household resources for consumption, which may result in more waste. Gellynck and Verhelst (2007) found that as the yearly cost of curbside residual household waste collection increases by € 1, the waste collected decreases by .396 kg/capita. This gives a price elasticity of demand at the means of the data for waste collection services of -0.139.

WEIGHT is a dummy variable that takes the value of 1 if weight based pricing is used for residual household waste collection, 0 otherwise. In a weight-based system, garbage trucks are fitted with scales, and collectors weigh each household's garbage and bill that household accordingly. Weight-based fees represent more closely the cost of waste disposal than do volume-based fees, such as unit-pricing by the bag. They also provide a clearer and continuous pricing signal to household producers of waste. Volume based fees provide no additional waste reduction incentive below the lowest level of service, i.e. one bag or bin per collection round (Miranda et al., 1996). Fullerton and Kinnaman (1996) found that the implementation of a price-per-bag program leads to a slight decrease in the weight of waste, but the volume of waste, i.e. number of bags or cans, is characterized by a higher decrease. Efaw and Lanen (1979) called the observation 'stomping', the changes in user fees can be moderated by the household through volume reduction. Weight-based systems eliminate the incentive for households to reduce garbage collection expenses by compacting waste into fewer containers. This is not particularly helpful since most garbage trucks compact household waste anyway. The effects of weight pricing on disposal behavior are roughly equal to those of the bag/tag studies. Linderhof et al. (2001) found a price elasticity of demand for waste collection services of -1.10. Dijkgraaf and Gradus (2004) estimated the price elasticity of demand forwaste collection services at -0.47.

PERDIR indicates the percentage of direct costs in total costs of the waste management program for a representative household. Direct costs can be directly attributed to the waste services provided, i.e. a fixed annual fee, if any, the costs associated with curbside collection of waste or recyclables and costs of dropping of recyclables at a drop-off center. Indirect costs are other general municipal taxes paid by households with no direct link to waste collection. Indirect taxes are used to cover municipal expenses not covered by direct taxes. Gellynck and Verhelst (2007) found that as the percentage of direct costs increases, the waste collected decreases by .159 kg/capita.

CREC is the number of waste fractions collected through the curbside recycling program, whereas DOREC gives the number of fractions collected through drop-off recycling. These collected fractions differ across municipalities. For instance glass is not collected through a curbside recycling program in every municipality. Households may recycle more materials that are included in local collection programs. Any increase in recycling presumes that this option is available and that residents find it more convenient than disposing waste through various illegal or undesirable means (Fullerton and Kinnaman, 1996). Noehammer and Byer (1997) found that the range of materials collected is an important design variable of a curbside recycling-scheme. The amount of effort, time and storage space demanded from the householder for recycling will increase with the degree of sorting and preparation of materials prescribed by the scheme. Participation is higher with binary sorting or commingled collection, i.e. separation of recyclables from non-recyclables, than multi-sorting or segregated collection, i.e. separating different recyclables (Bruvoll et al., 2002; Chung and Poon, 1994; Noehammer and Byer, 1997; Oskamp et al., 1996; Thomas, 2001). However, Shaw et al. (2006) found that the success of curbside recycling is not necessarily directly related to the range of materials collected. Furthermore, Harder et al. (2006) and Woodard et al. (2006) found that the participation rate is higher in schemes that collected more types of materials. When only one material is included in the curbside scheme, the emphasis of the whole collection system is still on waste collection. When more materials are incorporated into the collection of recyclables, the population may shift their perception of the process from one of waste collection with a limited recycling service to a system dominated by recycling with minimal actual 'residual waste' (Woodard et al., 2006).

FREQ is the frequency of collection service. It has the value of 1 if mixed household waste collection is once-a-week, 0 if waste collection is every other week. Early work focused on the effect of collection frequency on the overall amount of waste collected. Wertz (1976) found that the frequency of service could influence the amount of waste collected. On the contrary, Kemper and Quigley (1976) found no significant relationship between the number of collection visits per year and the annual quantity of waste discarded. More recent research focuses on the effect of collection frequency towards the recovery of recyclables. Findings point to a gain in recovery when the collection frequency of recyclables is increased (Everett and Peirce, 1993; Noehammer and Byer, 1997; Platt et al., 1991). However, reducing the collection frequency does not necessarily have a huge impact upon recovery (Tucker et al., 2000; Woodard et al., 2001). Nevertheless, some authorities are moving toward a kind of recycling dominated system by reducing the frequency of collection of residual waste to every other week, while also increasing the frequency and range of recyclable materials collected, so that the public perceives collection of the recyclable fraction as being the main element of the system (Woodard et al., 2006). Gellynck and Verhelst (2007) found a significant impact from differences in the number of collection visits. A weekly collection of residual household waste yields higher amounts of waste than an every other week collection round.

ORGANIC is another dummy variable that will take the value of 1 if separate curbside collection of organic waste is in place, 0 otherwise. What would induce a household to generate or throw away less waste (source reduction) hinges on at least two elements: the incentive built into the unit-pricing structure for waste collection and disposal, and the availability of convenient (and legal) alternatives such as recycling and yard waste collection or composting programs (Folz and Giles, 2002). One of the major components of household waste is organic material such as kitchen and garden waste, typically comprising 43% by weight of an average household's waste in Flanders (OVAM, 2002) and may include vegetables, fruit, cooked and processed foods, weeds, grass, leafs and other garden waste. Gellynck and Verhelst (2007) show that the implementation of a curbside collection program for organic waste has a significant negative impact on the average amount of residual waste generated.

CMASTER measures the number of compost masters per 1000 inhabitants. Compost masters are volunteers who completed a course on composting and are willing to train other people in proper composting methods. Reschovsky and Stone (1994) state that an incentive to participate in composting yard and food waste is likely to generate substantial savings for a locality. One approach that can be adopted by local authorities is to minimize the kitchen and garden waste components of household waste entering the collection stream, through the provision of subsidized waste digesters or compost bins to residents. Home composting has the potential to make a significant contribution to household waste minimization (Bench et al., 2005).

3.2.4 Results

About 53% of the municipalities had reached the goal of 150 kg/capita residual household waste in 2003. The amount of residual household waste collected ranged from 67.9 to 259.7 kg per capita. The results of the empirical model characterizing a municipality's probability of reaching the goal are given in Table 20. The overall model is statistically significant with χ^2 being 65.558, significant at the .001 level. Correct classification of municipalities as either reaching or not reaching the goal of 150 kg/capita residual household waste based on the explanatory variable information is an important measure of goodness-of-fit. The model correctly classified 219 of 295, or 74% of the municipalities. Four out of the 12 initial independent variables in the model contributed significantly to the classification.

An increase of AVGINC by € 1000 per household per annum lowers the logged odds of reaching the goal by .155. This means that odds are reduced by a multiple of .856 or by 14.4%. Consequently the probability of reaching the goal of 150 kg/capita residual household waste is reduced by 3.9%.

An increase in COST by € 1 raises the logged odds of reaching the goal by .028. Odds then increase by a multiple of 1.028 or by 2.8%. As a consequence the probability of reaching the goal increases by .7%.

Table 20. Factors affecting a municipality's probability of reaching 150 kg per capita per year. Average income, yearly cost of curbside collection, frequency of collection and separate curbside collection of organic waste contribute significantly to the classification (Sign. < 0.05).

Variables	Parameter estimate bi	Standard error	Sign.	O_i
Area	0.000	0.000	0.389	1.000
Average income	-0.155	0.051	0.002	0.856
Number of companies	0.000	0.000	0.250	1.000
Flat user fee	0.005	0.006	0.419	1.005
Cost of curbside collection*	0.028	0.010	0.006	1.028
% of direct costs in total costs of waste management	-0.93	0.994	0.925	0.911
Drop-off recycling	-0.005	0.024	0.828	0.995
Fractions collected at curbside	-0.102	0.138	0.459	0.903
Frequency	-1.132	0.331	0.001	0.322
Weight	0.877	1.084	0.419	2.402

(continued)				
Organic separate collection	1.916	0.398	0.000	6.791
Compost master	-0.095	0.280	0.733	0.909
Constant	2.824	1.858	0.129	16.839
χ^2 = 65.588			0.000	

* COST variable and amount of waste were tested for endogeneity bias. The direct cost in the logit model consists of taxes and retributions. The latter are dependent on the amount of waste produced. As such, in some cases it might occur that endogeneity potentially takes place during analysis.

In Fig.13, the probabilities are shown as a function of range for the variables COST and AVGINC. The odds of reaching the goal are 67.8% lower for municipalities with a weekly collection of residual household waste than for municipalities with an every other week collection. At the sample mean, municipalities with a weekly collection of residual household waste have a 26% less probability of reaching the goal than municipalities with an every other week collection. Hence, a weekly collection of waste yields higher amounts of waste than an every other week collection. Odds are 579% higher for municipalities with a separate curbside collection of organic waste than municipalities that do not collect organic waste separately at the curb. At the sample mean, municipalities with a separate curbside organic waste collection have a 37% higher probability of reaching the goal than municipalities that do not collect organic waste separately at the curb.

Calculation of standardized coefficients following Menard (1995) and Long (1997) indicates that the implementation of separate curbside collection of organic waste has the strongest influence on reaching the goal of 150 kg/capita residual household waste. The frequency of collection of residual household waste, the yearly cost of curbside residual household waste collection and the average income per capita have a less strong but fairly equal impact on whether the goal of 150 kg/capita residual household waste is reached (see Table 21).

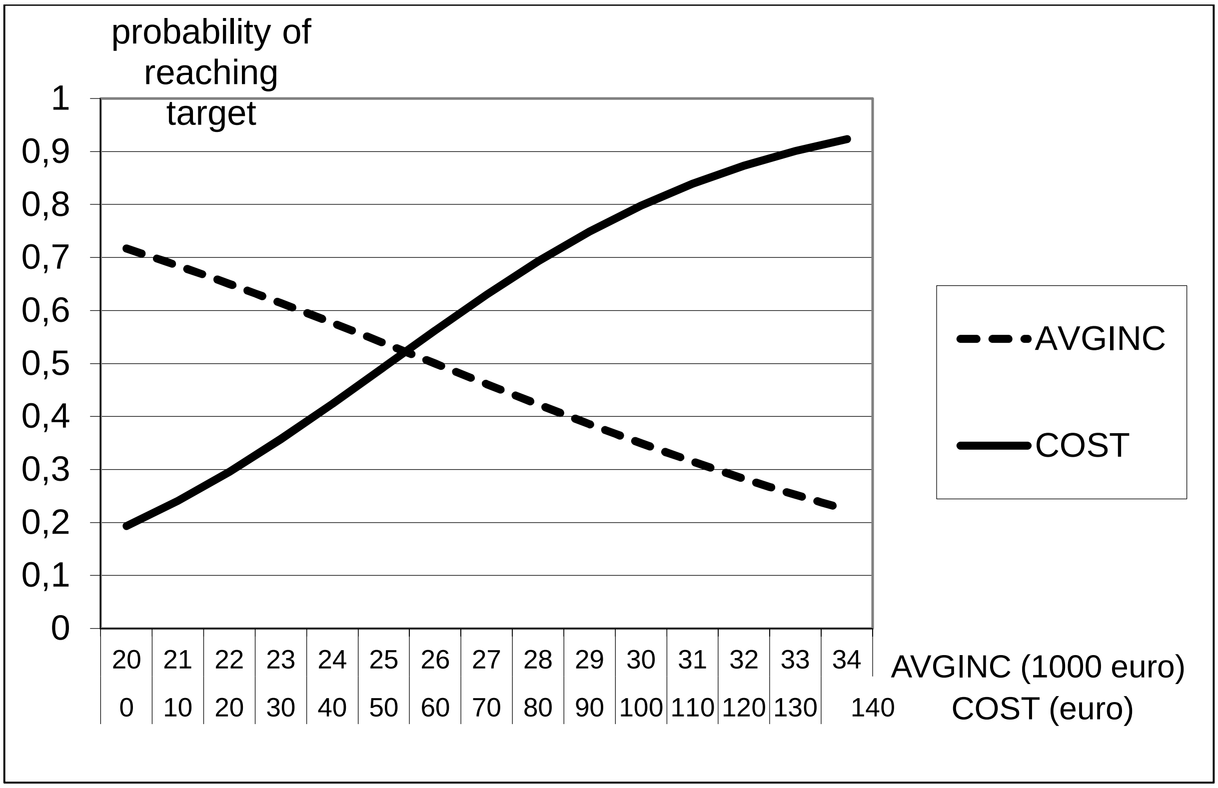

Figure 13. Probabilities as a function of range for the variables average income and yearly cost of household waste collection. The range for the 2 parameters is the one from the database and the probability is set on 0.5 for the average value (for the data, see annex).

Table 21. Standardized coefficients for the significant independent variables. The implementation of separate curbside collection of organic waste has the strongest influence. The other 3 variables have a fairly equal impact on reaching the goal.

	b	B* Menard	B* Long
AVGINC	-.155	-0.391	-0.521
COST	.028	0.418	0.557
FREQ	-1.132	-0.428	-0.570
ORGANIC	1.916	0.718	0.957

3.2.5 Discussion

Increasing per capita income results in the generation of more solid waste. This result supports with earlier findings on country level (Porter, 2002) or on municipal level (Efaw and

Lanen, 1979; Gellynck and Verhelst, 2007; Jenkins, 1993; Kinnaman and Fullerton, 1997; McFarland, 1972; Podolsky and Spiegel, 1998; Reschovsky and Stone, 1994; Richardson and Havlicek, 1978; Wertz, 1976).

Economic literature on waste management illustrates the effectiveness in reducing the amount of residual household waste of ‘pecuniary incentives’, evolving from a fixed annual fee (McFarland, 1972) over unit-pricing by the bag (Fullerton and Kinnaman, 1996) to weight-based fees (Miranda et al., 1996). Gellynck and Verhelst (2007) found that there is no significant influence from a fixed annual fee, nor from the implementation of a weight-based fee.

Analysis of this case using binary logistic regression supports the conclusion that the implementation of a pay-by-the-bag system has a significant influence on the amount of residual household waste collected and is a useful instrument in helping municipalities to reach the goal of reducing the amount of residual household waste to 150 kg per capita per year. However it is worth noting that the model applies data of the year 2003. When using a pay-by-the-bag system, citizens have more incentives to produce less waste. However Sterner and Bartelings ’ report (1999) on a weight-based billing system in Sweden, states that average waste per household had declined by 35%.

Municipalities with a weekly collection of residual household waste have a lower probability of reaching the goal of 150 kg per capita per year. This is in line with findings of Wertz (1976), Platt et al. (1991), Everett and Peirce (1993) and Gellynck and Verhelst (2007). The result is explained intuitively because compared to recycling, throwing away waste as residual household waste is less time and space consuming. Furthermore, a higher frequency of collection prevents other negative side effects, as there are foul odors, problems with rodents and other pests as well as a lack of space.

Municipalities with a separate curbside organic waste collection have a higher probability of reaching the goal of 150 kg per capita per year. As the organic waste fraction, both yard waste and waste from fruit and vegetables, makes up nearly 40% of the total municipal solid waste weight in Flanders (OVAM, 2002), this finding is reasonable.

For this case using multiple linear regression, there is no significant influence from the area per capita, the number of companies in a municipality, the number of compost masters per 1000 inhabitants, the number of waste fractions collected through the curbside recycling program nor through drop-off recycling.

However, Gellynck and Verhelst (2007) found that the percentage of direct costs to total costs of the waste management program for a representative household has a significant influence on the amount of residual waste collected, concluding that the ‘polluter pays’ principle is an effective instrument in reducing the amount of residual household waste. However its effect does not significantly impact a municipality’s probability of reaching the goal of 150 kg/capita residual household waste. This may indicate that the implementation of the ‘polluter pays’ principle beyond residual household waste can have some influence on the amount of residual household waste collected but is not sufficiently strong to substantially reduce the amount of waste. It is an instrument better suited to finetune the waste management policy.

The implementation of a pay-by-the-bag system and separate curbside collection of organic waste, together with bringing back the frequency of mixed household waste collection to an every other week collection proofs to be a good policy mix to gain substantial reductions in the amount of mixed household solid waste collected and consequently raising a municipality's probability of reaching the goal of 150 kg/capita residual household waste.

Furthermore the model reveals that residual waste amounts will further decrease when municipalities install a high recycling program: the more fractions are collected at the curbside, the less waste will end up in the residual waste bags. In revealing the influence of source prevention the model could contain another dummy variable: a certain amount per year could be set as a standard for prevention campaigns in municipalities (e.g. € 10 000). Municipalities with a lower budget take on the value of 0, those with a higher budget the value of 1.

In considering further developments, the model could contain the following independent variables: communications and public engagement activities by municipalities and behavioral variables e.g. the level of participation in recycling collections or using a home compost bin. Furthermore, it is worth considering in future research to adapt the model in order to accommodate different types of dependent variables (e.g. a different mandatory level of residual waste).

3.3 Economic sustainability

Sustainable waste management not only covers the environmental aspect from a source-sink point of view. It also addresses the fact that it should be done in an economically effective way. Economic sustainability can thus in this regard be seen as using assets of a company or institution in an efficient way to allow it to keep it going in the long run.

Against this backdrop, the cost of household waste collection in Flanders is further examined. It is also one of the determining variables contributing to household waste generation (see section above). From this perspective it needs further examination. A more explicit analysis of costs might be a critical step in understanding of how costs are generated for solving the issue of efficiency (Porter, 2006; Kaplan, 2011). Dealing with economic sustainability would imply to find better ways to assess what are priorities in the allocation of resources and simply getting the most out of waste collection systems.

Economic efficiency can always be linked with the efficient use of resources. The general economic principle corresponds with the efficient use of resource principles deducted from the sustainability concept. The principle not only refers to scarce and finite energy sources, but takes into account all other scarce resources like capital, work and environment (Klaassen and Opschoor, 1991).

In the economy, costs and prices are measures for the use of resources which are scarce. Lower costs for providing the same service imply an economically more efficient execution which is less resource demanding. With regard to household waste collection, it is therefore necessary to scrutinize the cost structure.

In the previous section, it was found that the cost of waste collection has a huge impact on the probability of municipalities of reaching the mandatory target of 150 kg residual waste per capita per year. This target is seen as an environmental sustainability target. To some extent it might be seen as an economic sustainability one too. However, it does not fully cover the economic perspectives as such. Extension with economic principles is thus needed.

Striving for more efficiency can thus be seen in the light of the New Public Management (NPM) paradigm which tries to focus on performance and policy (Schiavo-Campo, 1999). The paradigm believes in higher private efficiency compared with public institutions (Hezri and Dovers, 2006). Household waste collection can be executed by both private and public operating companies and it is the aim to reveal differences in terms of cost structure.

Currently, municipal solid waste management emerged due to the complexity of the service, caused by the growth of cities and higher service costs (Bel and Mur, 2009; Benito-Lopez et al., 2011). The current economic and financial crisis limits the development of public services, including waste management services. Therefore the cost of waste management in general and residual waste collection in particular is scrutinized in the following sections.

3.3.1 Municipal solid waste management

Managing municipal household waste requires specific cost-related assets; market entry barriers remain high and a monopoly structure of services remains to be the usual approach upon waste collection services (Warner and Bel, 2008). However, this is linked with high costs of service provision and given the limited budgetary freedom, local authorities seek ways for reducing costs.

The main question is one of ownership (Simões and Marques, 2012) and more specifically whether the management of household waste should be done by the private sector or remain a public activity conducted by governments and/or local authorities. From this perspective, all factors delivering public services should be taken into account. This dissertation examines whether cost efficiency and other issues are able to influence the decisions made by local authorities willing to change their service level. Differentiation should be made between many forms of outsourcing and supralocal joint ventures (intermunicipal cooperation). Due to its high cost and complexity, this topic has been addressed by many other studies (Bosch et al., 2006; Termes-Rifé and Alerm-Domènech, 2007; Bel et al., 2010; Benito-López et al., 2011; Rogge and De Jaeger, 2012).

3.3.2 Household waste collection in Flanders

In Flanders, the collection of household waste is done by both private and public organisations. Due to rising pressure in terms of cost efficiency and service expectations on public services, governments are pushed to transfer part of those services to the private sector. The collection of municipal household waste is also characterized by a trend towards more privatizing. Local authorities like municipalities and supralocal joint ventures are in charge of organizing waste policy on a local scale. They have the control over drop-off facilities and take care of the collection of household waste.

In recent years, a lot of municipalities were forced to evaluate the solid waste management program due to cost, health and sustainability related concerns. This evaluation comprises cost effectiveness in terms of collection, transportation and processing (Rubenstein and Zandi, 2000; Lombrano, 2009; Passarini et al., 2011; Porter, 2002). Literature review regarding the cost of municipal solid waste management shows that the question of the cost of solid waste management is very complex. Some examples of factors having a thorough effect on the cost of solid waste management are municipality characteristics (population, density, area), the quantity/quality of the solid waste and its composition, the collection and transportation, distances and labor expenses (Gellynck and Verhelst, 2007; Karam et al., 1988). Recently, Warner and Hefetz (2003) and Bel and Costas (2006) suggested that intermunicipal cooperation might be a good alternative to local privatization, especially in smaller rural municipalities with a small number of potential outside contractors. On the contrary, intermunicipal cooperation proved to be incompatible with private production in the Netherlands (Dijkgraaf and Gradus, 2008), but this is not the case in Spain (Bel and Warner, 2008).

There are plenty of numerical examples in terms of variability of the cost of municipal solid waste components. The cost of curbside collection and transport is approximately $3.5 per ton mile in the US (Pollock, 1987), and between $2.9 and $10.4 per ton in Thailand (Danteravanich and Siriwong, 1998). The total cost of municipal solid waste management in the New York City region is $143 per ton (Clark, 1993), while it has been estimated that the state of Florida spends on average $16.6 per ton for the collection (Young, 1991).

In literature there is a reasonable body of information regarding privatization of solid waste services (Hefetz and Warner, 2004). One reason commonly forwarded for contracting out solid waste services is to decrease service costs; however, the evidence regarding such savings are somewhat ambiguous (Bel et al., 2007; Bel and Warner, 2008). Plenty of literature studies found no difference in cost due to public or private contract arrangements (Hirsch, 1965; Pier et al., 1974; Collins and Downes, 1977; Stevens, 1978; Domberger et al., 1986; Dijkgraaf and Gradus, 2003, 2008; Ohlsson, 2003; Bel and Costas, 2006; Bel and Mur, 2009). Upon a larger number of bidders, there are more cost savings (Gomez-Lobo and Szymanski, 2001). Senternovem (The Netherlands) did some research regarding the waste market in 2006 and concluded that public companies provide for a large part the residual household waste collection. In almost 70% of the cases the waste is being collected and processed by a public company. There is no indication that government-dominated companies are more expensive than private operating companies or vice versa.

On the other hand there are plenty of other studies stating that costs differ. Competition encouraged public managers to keep costs down. Szymanski and Wilkins (1993) found similar results in the 1984–1988 period. They found a 20% savings in the first year, but these savings disappeared in 2 years. A study by Szymanski (1996) on 365 English municipalities provided some evidence that although savings eroded over time, private production costs were lower than public production. Three other studies have found lower costs with private production (Kitchen, 1976; Tickner and McDavid, 1986; Reeves and Barrow, 2000).

More recent studies on waste collection show no differences in costs (Dijkgraaf and Gradus, 2003). Only cities that recently privatized show cost savings. Cost savings from privatization appear to erode over time, since there were no cost differences between cities that had privatized earlier and those retaining public production (Dijkgraaf and Gradus, 2008).

Literature gives plenty examples of cost comparisons between private and public production, however it does not reflect the deeper causes why municipalities not always opt for the most economic solution in terms of waste collection.

The objective of this PhD chapter is twofold: first, the aim is to reveal if there are cost differences between private and public production for the selected cases in Flanders, without generalizing the results. The latter are then used for the second objective where the aim is to identify the deeper causes why local governments opt for a certain type of waste collection program (public or private). Qualitative data from in-depth interviews were used to tackle this problem for case studies (Eisenhardt, 1989).

One should pay attention with the word privatizing. In a narrower sense, this would mean that each citizen could decide upon the way of collection and processing.

This dissertation focuses on outsourcing a certain service to a private collection firm. It should be emphasized that only the collection and transportation of residual household waste is considered, not the processing operations. These are all the same all over Flanders, differences occur in collection and transportation.

In order to obtain economies of scale, municipalities transfer their responsibility regarding solid waste management towards a supralocal joint venture of municipalities (Fig. 14 on the next page). These joint ventures arrange the collection and processing of all household waste fractions for the participating municipalities. A total of 308 municipalities in Flanders form 27 different supralocal joint ventures.

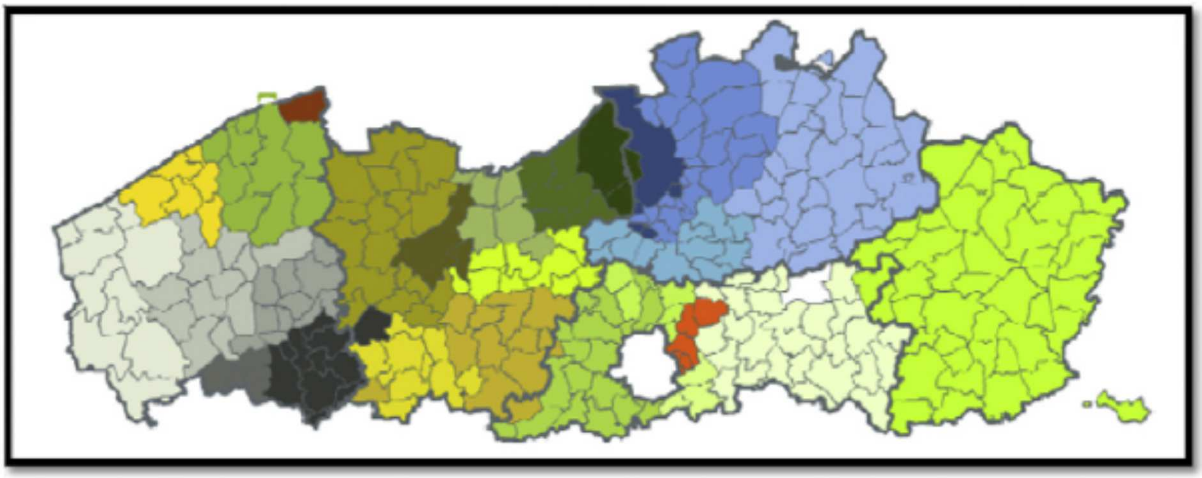

Figure 14. Map of Flanders with all supralocal joint ventures (same color is 1 joint venture).

Household waste consists of five relevant fractions: residual household waste, PMD (plastic bottles and metal packaging), paper and cardboard, organic waste and glass. The Flemish waste authority OVAM is responsible for defining the Municipal Solid Waste (MSW) management policy.

The idea is to compare the cost of waste collection services for mutual comparable municipalities, one having a private service and one having public service. Fig. 15 provides a framework to guide the investigation. As mentioned before, OVAM is responsible for defining the policy regarding MSW in Flanders. Municipalities are responsible for the collection of mixed/separate collection of MSW. Citizens, commercial services and industries generate MSW. Waste collection can be done in three ways: municipalities use own waste collection trucks, supralocal joint ventures collecting the waste or private firms commissioned by the joint ventures to do so. Furthermore the waste is being processed in an incineration plant with energy recovery. The plants being owned either by the supralocal joint ventures or private companies.

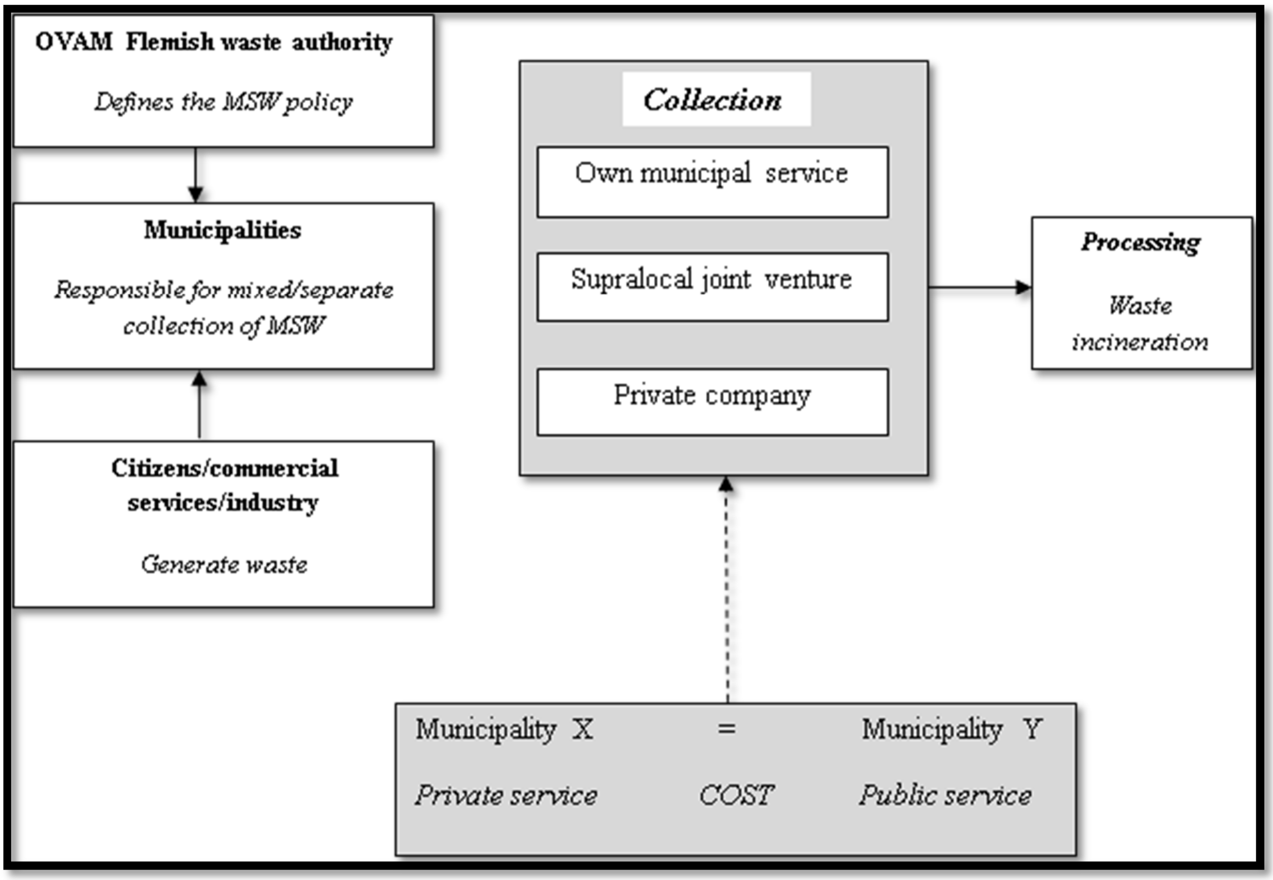

Figure 15. Flow diagram of residual household waste collection. The shaded boxes and the dotted arrow indicate where the analysis takes place.

3.3.3 Methodology

Case study

The case study method is considered to be the most suitable methodology with regard to the exploratory nature of the study which combining both qualitative and quantitative data (Voss, 2002). The case study methodology has been selected because of the need for studying the phenomenon in its natural setting, and the focus on contemporary instead of historical events (Yin, 2009). A multiple case study design was employed in order to get the comparative insights of the study. A case study approach was opted for because of the ‘‘how’’ and ‘‘why’’ nature of the questions to be answered.

Approach and data collection

For this case study the OVAM database regarding municipal waste was used. Information about the name of the municipalities, members of the supralocal joint venture and the amount of residual household waste collected in each municipality is systematically stored.

In order to identify mutual comparable municipalities one needs a set of parameters. Derived from literature (Cargo, 1976; Karam et al., 1988; Dijkgraaf and Gradus, 2004) there are some characteristics which can help to find mutual comparable municipalities. Those parameters are population, population density and the yearly amount of residual household waste per capita. The mutual comparable municipalities should nearly have the same value for the above mentioned parameters. To control for municipality characteristics like population and population density, data on this matter were gathered through the National Institute of Statistics (NIS).

The original database was manually split up into two subdivisions, following collection frequency. Each municipality or supralocal joint venture decides to some extent between weekly or biweekly collection. Consequently the database for municipalities was divided in two more groups: one collecting residual household waste on a weekly basis, the other one on a biweekly. Subsequently, within both groups, two more groups are made for homogeneous clusters by using the above mentioned parameters (population, population density and amount of residual household waste per capita): One cluster containing municipalities where the standard deviations of the selected parameters move within a narrow range around the cluster average, another similar working cluster for the municipalities with parameters differing significantly from the cluster average. As a result of the hierarchical cluster analysis it was opted for obtaining two groups within each collection frequency (weekly or biweekly), meaning four groups in total. The use of more clusters is not beneficial due to the outliers, clusters with just one municipality for instance. Cluster analysis is conducted via the use of the software ‘‘Statistical Package for Social Sciences’’ (SPSSs). Since it is the aim to obtain 2 x 2 clusters, it was opted to take k-means in SPSS. The latter is a very good method when one does not want to determine the number of clusters. Fig. 16 shows the step by step approach of cluster analysis executed on the municipalities in Flanders.
It should be noted that it is assumed that municipalities just have 1 waste collection system in place for the whole area. In reality it might however occur – mostly in cities – that municipalities apply two or more waste collection systems (city centre versus the outskirts have different collection rounds). Municipalities with a high student population and a lot of coastal tourism have been omitted in the analysis. Therefore the analysis starts with 289 instead of 308 municipalities.

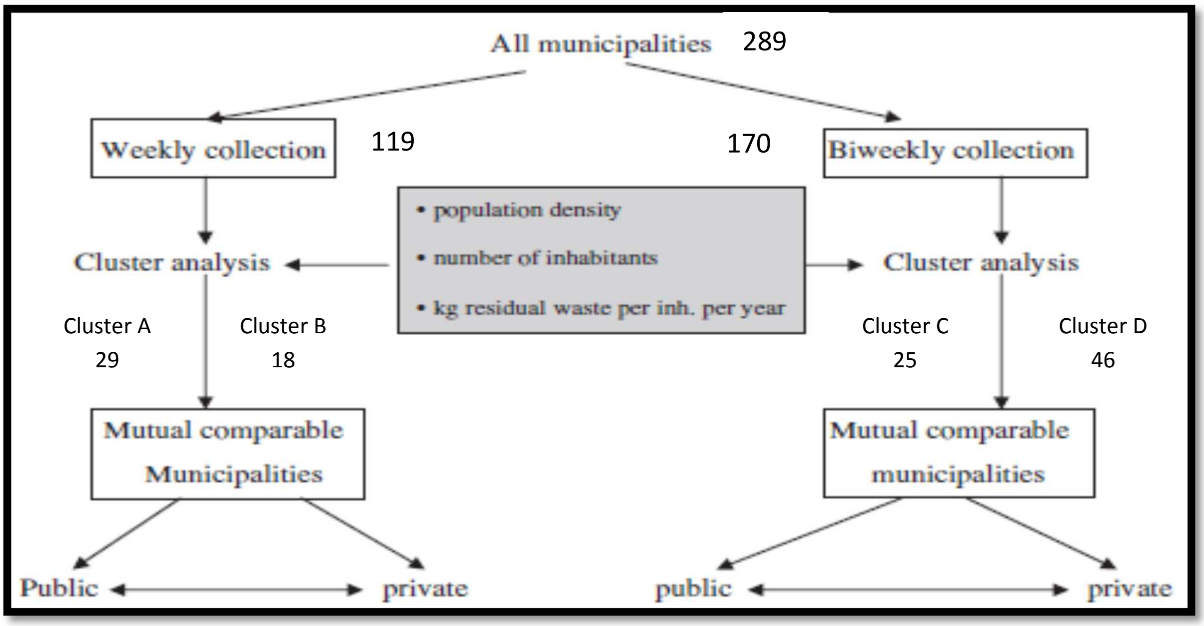

Figure 16. Step by step approach in order to select mutual comparable municipalities. The first database is split up in two databases. On each database a cluster analysis was conducted to form four homogeneous groups in total.

Consequently there are two groups of municipalities with a weekly collection and as many with biweekly collection. Within each group it is the aim to purposively find mutual comparable municipalities according to the three selected parameters, bearing in mind that one has public and the other a private service level.

In the multiple case study, municipalities collecting residual household waste through receptacles are withdrawn, waste collection through bags is considered in order to refrain more municipalities. This decision was made in collaboration with the stakeholders taking part in the project. A total of 218 out of 308 municipalities collects residual waste through bags. In total 12 municipalities, six mutual comparable pairs, were identified purposively. Each pair comprises municipalities with almost the same value for the cluster parameters, whereby one has public service and the other one private service in terms of waste collection.

The withheld municipalities were then contacted to give further details regarding the cost of private or public service. A structured questionnaire was compiled to obtain data to address the first part of the objective. The questionnaires addressed following topics: the number of municipalities taking part in the joint venture, identifying the company(ies) collecting residual household waste and the other fractions, the number of weeks needed per year in order to collect the different waste fractions, the provided service level, percentage of time spending for administrative tasks, identify if there is a solidary price among all participating municipalities in the joint venture, the distance covered by the collection trucks, the number of staff needed, the salaries, the number of collection points, the time needed for waste collection, the hourly price charged from the collection firm, the number of collection rounds, the presence of a transshipment point. Furthermore data were gathered from municipalities websites and brochures on waste management. Then these data were verified and completed via telephone interviews with the competent public official of each municipality. Finally in depth-interviews were taken from the directors of the different waste collection companies and supralocal joint ventures in order to address both parts of the objective. The structured questionnaire as mentioned above has been used as well.

Descriptive statistics

The method of cluster analysis is described above. Table 22 describes the clusters according to the selected parameters. The average value for the parameters in each cluster is given. Cluster 2 contains the least municipalities, cluster 4 the most. The group of municipalities with a biweekly collection of household waste (clusters C and D) is larger than the group with a weekly collection (clusters A and B). From these clusters 12 cases (six mutual comparable pairs) are chosen purposively in order to collect qualitative and quantitative data at municipality level. It should be noted that each pair of selected municipalities nearly had the same value for the selected parameters (see Table 23).

Table 22. Description of the clusters executed on 289 Flemish municipalities. Municipalities in clusters A and B have a weekly collection, clusters C and D a biweekly collection.

Cluster	Average inhabitants	Average population density (inw/km²)	Average amount of household waste per capita per year (kg/cap.yr.)	Number of municipalities in cluster
A	12 192	338	118	54
B	42 427	479	131	43
C	24 776	676	94	50
D	13 879	285	81	71

Table 23. Description of the selected municipalities with a weekly or biweekly collection of household waste. Each pair is presented two by two. W = weekly collection, B = biweekly collection.

Municipality	Cluster	Service	Population	Area (km^2)	Density (inh./km^2)	Residual waste per capita (kg/yr.)
A	2	Public, w	22 180	60.34	368	134.52
B	2	Private, w	24 113	63.24	381	113.52
C	1	Private, w	20 043	30.05	667	147.83
D	1	Public, w	21 165	32.92	643	119.56
E	1	Public, w	10 298	20.43	504	125.69
F	1	Private, w	10 641	16.16	658	124.08
G	3	Public, b	24 882	56.65	439	91.62
H	3	Private, b	16 448	42.78	384	89.02
I	4	Public, b	9 711	26.67	364	78.73
J	4	Private, b	10 235	32.69	313	84.94
K	4	Public, b	17 989	26.92	668	97.21
L	4	Private, b	14 907	22.49	663	109.02

Several stakeholders (representatives from private and public organizations) involved in the project agreed upon the ad random selection of the cases (Table 23). Due to confidentiality reasons, municipalities are given letters of the alphabet. It can be clearly seen that biweekly collection lowers the residual household waste generation. These are findings confirmed by literature: frequency of service influences the amount of waste collected (Platt et al., 1991; Everett and Peirce, 1993). A higher collection frequency makes it easier to throw waste away than to recycle. The collection of household solid waste is performed on a fixed day for each municipality and is performed between 06:00 am and 01:00 pm. Every household is being served by the collection firm, which is possible due to the fact that Flanders is a dense region.

3.3.4 Cost comparison

3.3.4.1 Public versus private settings

In Flanders there are several forms of managing the municipal solid waste. Simões and Marques (2012) studied earlier research on the economic management of waste collection services. They came to the conclusion that a major area of study implies the form in which the service is delivered. More specifically, the question of ownership was benchmarking the service level provided by private service management and public administration (Bel and Warner, 2008; Bel and Fageda, 2007; Bel and Warner, 2010; Simões et al., 2012). Three theories are seen as being in favour of private management of services: agency theory, public choice theory and organization theory (Simões et al., 2012).

In most developed countries, private service management is common. It has however lost importance in a process called privatization in reverse (Hefetz and Warner, 2004). Despite the increasing trends of privatization, collecting residual household waste is still perceived as a task for municipalities. However, both public and private service management have the property of showing a monopolistic nature with regard to the market structure (Stiglitz, 2000; Bel, 2006). Private firms have a clear overview of their cost structure. So collection and transportation is the responsibility of the private firms, however other aspects like composting actions or prevention campaigns applicable for the municipalities is done by the supralocal joint ventures themselves. On the contrary, some public operating collection firms (or joint ventures) offer a complete service level (collection of all waste fractions, transportation and prevention campaigns). Moreover the cost of this total service package is being transferred to the municipalities. Therefore, single costs involving transportation and collection of residual household waste are not available. In order to obtain these costs, a filter was used for some joint ventures.

This filter allocates the right costs to the right waste collection services, but is based on assumptions: collection and transportation cost of each curbside collected waste fraction is hypothesized to be equivalent. Assumptions were agreed upon during several stakeholder meetings. The total yearly budget for collecting and transporting all waste fractions was allocated to each waste fraction separately based on their respective yearly number of collection weeks outweighed to the total amount of collection weeks. This gives a good but rough indication and should therefore not be seen as the exact cost for collecting residual household waste. Table 24 presents the yearly cost per inhabitant for the collection and transportation of residual household waste. The last column shows how the data were obtained, through interviews and questionnaires only (non-filter) or in combination with the filter assumption described above.

Table 24. Yearly cost per inhabitant for collection and transportation of residual waste (in EUR). Municipalities are presented pairwise (cases two by two). W= weekly collection, B = biweekly collection.

Municipality	Service level	Price (€/inh.year)	Origin of data obtained	
			Non filter	Filter
A	Public, w	12.2		X
B	Private, w	8.9	X	
C	Private, w	11.4	X	
D	Public, w	11.8	X	
E	Public, w	11.8		X
F	Private, w	8.9	X	
G	Public, b	8.54	X	
H	Private, b	4.98	X	
I	Public, b	8.0	X	
J	Private, b	4.8	X	
K	Public, b	4.8	X	
L	Private, b	4.47	X	

From the above table, it can be seen that in all cases the price per inhabitant in municipalities having a private service level is lower than the price in municipalities having a public service. At least this is the case for the examined municipalities.

In order to compare the cost of private and public service level, assessment should go beyond simple comparison of costs as such. Therefore it is feasible to have a closer look at the efficiency and effectiveness of the collection firms. Efficiency indicators show how efficiently the firm uses the contracted budget for the collection and transportation of municipal waste (Koushki et al., 2004).

On the other hand, effectiveness indicators demonstrate how effective the service is in terms of population being served and areas being covered (Koushki et al., 2004). Based on the collected data, an attempt was made to calculate efficiency and effectiveness for the collection and transportation of residual solid household waste. The calculated efficiency indicators are cost per tonne, cost per collection point and cost per tonne and covered area. Effectiveness on the other hand was measured by including the population being served (persons per total price) and the area being covered (square kilometer per total price). Table 25 presents the results of the calculations.

Table 25. Efficiency and effectiveness of collection and transportation of household solid waste. W= weekly collection, B= biweekly collection (concept derived from Koushki et al., 2004).

Municipality	**Service level**	**Efficiency indicators**			**Effectiveness indicators**	
		EUR/ton	EUR/stop	EUR/ton.km²	km²/cost	Person/cost
A	Public, w	83.5	28.2	1.3	0.22	0.08
B	Private, w	79.1	20.8	1.2	0.29	0.11
C	Private, w	71.98	26.1	2.3	0.13	0.08
D	Public, w	92.8	27.1	2.8	0.13	0.08
E	Public, w	78.2	26.1	3.8	0.16	0.08
F	Private, w	65.5	20.4	4.0	0.17	0.11
G	Public, b	92.9	18.6	1.6	0.26	0.11
H	Private, b	55.3	12.8	1.2	0.52	0.20
I	Public, b	98.0	18.5	3.6	0.34	0.12
J	Private, b	55.2	12.3	1.6	0.66	0.20
K	Public, b	43.7	12.7	1.6	0.31	0.20
L	Private, b	40.7	10.9	1.8	0.33	0.22

The lower the efficiency indicators are, the more cost-effective firms work.

From Table 25 it can be seen that more or less the same trends are obtained as in Table 24 (private service at a lower price), however less distinct. Effectiveness indicators on the other hand should be high. Subsequently from the above table there is no general distinction between public and private service concerning effectiveness.

Although the results are more distinct for municipalities with biweekly collection. It is important to note that factors such as the population size and density, area size, the quantity of the generated solid household waste (function of socio-economic characteristics), the geometric design of streets and the level of area traffic congestion have a pronounced impact on the cost of collection and transportation.

In-depth interviews also showed that three main factors have great influence on the cost for the collection of residual household waste: level of depreciations of the equipment, transportation route during collection and labor costs. The latter are somewhat high in Belgium (Eurostat, 2012). However the budget for collecting and transporting solid waste is estimated 1 year before the real execution.

3.3.4.2 Identification of factors influencing the decision to opt for public versus private waste collection services

It is of utmost importance to reveal those factors triggering managers to prefer one or the other form of management in collecting household waste. Two major economic drivers are considered for opting private management: inefficiency and fiscal stress (Bel and Fageda, 2007). As such, it should be possible for the public sector to incorporate ideas from the private sector for increasing efficiency (Plata-Diaz et al., 2014). At the same time private managers have a large amount of incentives at their disposal to conduct cost cutting activities and to introduce competition providing public services (the “Public Choice” argument) (Niskanen, 1971; Savas, 1987; Plata-Diaz et al., 2014). With regard to private operating companies, there is even an advantage when it comes to economies of scale and the accompanying efficiency: same service can be provided in several municipalities (Christensen and Laegreid, 2011). Economies of scale and cost reduction in providing public services, play thus a central role in deciding which form of waste management is opted.

Differences in costs under public and private production can be attributed primarily to competition. However, the importance of management, service characteristics and the industrial organization of the sector itself should not be underestimated. Local governments are interested in more than just costs (Belton et al., 1994; Carver, 1989; Bel and Fageda, 2007; Domina and Koch, 2002; Hefetz and Warner, 2007). Communities may prefer private/ public delivery even it is more costly, if that reflects their view on the role of government in service delivery (Dubin and Navarro, 1988; Tucker et al., 1998).

In Flanders competitive tendering for private solid waste collection is compulsory. Furthermore there is the recognition that cost competition is not enough, but that tendering should also include aspects of service quality, stability, innovation and citizen engagement

(Bel and Warner, 2008; Gamba and Oskamp, 1994; Jakus et al., 1994; Martin et al., 2006; Oskamp et al., 1991; Vencatasawmy et al., 2000; Vining and Ebreo, 1990).

In Flanders, several waste management options are available: 1) municipalities collecting waste themselves, 2) municipalities contracting out to a single entity, 3) supra-local joint ventures forming a public company, 4) joint public production and 5) private management but with cooperation

Therefore through qualitative data collection it was the aim to find deeper causes why municipalities and local governments opt a certain way of collection (private versus public).

The in-depth interviews with the different public officials revealed that municipalities with a public service waste collection opt to focus on a broad service level rather than cost efficiency and hence a lower cost for municipalities and their citizens. They are offered a total and complete service level in terms of waste collection, and therefore these municipalities are willing to pay extra. However, it should be noted that citizens do not decide on the way of collection, services are offered by the municipalities. It is due to political reasons and economies of scale that municipalities opt to join a supralocal joint venture. Municipalities pay a yearly fixed amount to the joint venture they are part of. This fee includes waste collection services of any kind: curbside collection of each waste fraction (yard waste, paper, cardboard, plastics,. . .), street-refuse removal, prevention and composting campaigns, glass spheres are emptied every week, mistakes by the collection services are corrected, agreements are made during road constructions, citizens are notified when they wrongly offer waste at the curbside, several negotiations between waste collection managers and citizens take place in order to clarify certain issues, planning of the waste collection day is negotiated with the municipality, solidarity price among all participating municipalities in the supralocal joint venture, etc.

Municipalities appreciate this way of working since a lower service level would give rise to additional work and hence higher costs for the civil servants: coordination, corrections, assessments, etc.

Some quotes of public servants and staff from public joint ventures strengthen these insights:

‘‘It is impossible for private operating firms to deliver services to the same extent as public operating firms. We are closer connected to municipalities and their citizens than private service firms ever can be.’’

‘‘We collect all waste fractions and offer a complete service level, whereas private firms simply focus on the cheapest way of collecting each waste fraction separately. They can even temporarily work below the economic break-even point in order to obtain the waste collection contract and gain market share. Biennial tendering helps to reach the cheapest operating firm, however it remains a snapshot and the cost advantage can erode over time.’’

‘‘The citizens are used to the broad service level and do not want to narrow it down.’’

Municipalities counting upon private service waste collection firms on the other hand, focus more on cost efficiency driven by the technique of tendering. Therefore, competition is more likely.

Once a year different firms can submit a proposal for the separate collection of each household waste fraction. When collecting five different waste fractions at the curbside, it can be possible that five different firms execute the operation. The local governments set the framework in terms of price and basic service that the firms need to provide. All contacted municipalities with private operating collection firms, declared that the given service by the firms is strictly limited to the collection and transportation of the waste itself. Extra service level dimensions, as present in the public operating firms, are being taken care of by the local governments.

Although these services are not to the same extent, as compared with the public bodies. Quotes of some staff members working for joint ventures outsourcing the collection to private operating companies:

‘‘We opt for a biannual tendering system in order to get the company with the best offer for each waste fraction separately.

Sometimes the companies have the best bid when submitting a tender for collecting two waste fractions simultaneously e.g. residual and yard waste or residual and bulky waste.’’

‘‘The specific service level included in the cost comprises the waste collection itself and the follow up of complaints. Other service aspects are handled by the joint ventures.’’

‘‘There is a fixed cost per inhabitant and a variable cost per tonne of waste collected. We keep track of the costs for each waste fraction separately. We know that public service operates differently: costs are related to the whole service level and not to each curbside collected waste fraction separately.’’

The above outcomes can be linked with political factors too. From literature, it has been suggested that politicians whose ideology is close to that of neoliberalism are more in favor of policies based upon private management (Pallesen, 2011). All these factors might favor a change towards new types of management. Contracting out is hence a possibility, but might not at all be attractive for smaller municipalities (Bel and Fageda, 2006; Warner, 2006).

Understanding why municipalities are interested in contracting out or to join a supralocal joint venture is the core of this chapter. Therefore the economic factors are considered first. Next the relationship between the costs and the chosen management option is scrutinized.

3.3.5 Public or private ?

Given the limited amount of examined cases and the fact that assumptions were made in order to obtain the cost for some municipalities, it is not designated to generalize the results for the whole region of Flanders. It cannot be stated that private service level operates cheaper and hence more efficiently than public operating companies. However, the results in Table 27 give an indication into that direction. In general, even with these extended calculations, it is not clear whether private operating firms are more cost-effective in collecting and transporting solid waste than public operating firms. Albeit there is an interest in establishing whether there are differences in the costs and service levels between public and private waste collection services, there are clear difficulties in establishing comparisons that can be made without having to rely on a large number of assumptions and corrections and there are also difficulties in generating interpretable results.

An obvious problem arises because private firms are motivated to undercut public service provision until such time as public services are withdrawn. Private firms are able to determine the costs of public provision from data that are to some extent in the public domain or can be estimated with reasonable accuracy or can be established through contact with employees in the public service. They will only win contracts to supply services if these are below the costs of public provision, but there is the issue of private sector strategy to take into account, which could include the possibility of cross-subsidies between and among activities in the private companies' portfolio of services.

Many of these issues presented are actually internal affairs of the companies, and were not at disposal. There is also the issue of cost structure to consider and how the structure of fixed and variable costs is likely to differ between public and private sector suppliers. Moreover public operating firms do not have a complete overview of all costs themselves.

It is noteworthy that the determination of the cost for collecting household waste is a snapshot. When the study would be conducted in 2 years time, different results could be obtained. Indeed costs like depreciations of the rolling stock, maintenance, gas prices all have some influence on the final price. Furthermore they can outweigh more from 1 year to another. This holds too for the comparison among supralocal joint ventures. One might just have purchased new waste collection trucks or have an older and hence more expensive staff. This implies that these factors could outweigh more during a short period of time in comparison with another joint venture which already has depreciated for example all its equipment and machinery. This counts both for private and public. Previous aspects play a huge role in the cost, however their influence on it could not be revealed since the addressed joint ventures did not give insight in their analytical bookkeeping.

Another aspect influencing the cost are salaries. The public sector uses pay scales of the public administration, whereas workers at private companies follow the rules of a joint committee per sector. Working in the public sector has consequences regarding seniority and extra holidays. It is hard to reveal in which sector the salary and working conditions are the best and hence outweigh more in the cost. Since salaries make up 75% of the total cost in some joint ventures, it is a non-negligible factor.

A last aspect influencing the cost is the followed route by the waste collecting trucks. Furthermore there are costs inherent to waste collection which are not being taken into account in the costs for private waste collection since they are borne by either the municipality or the joint ventures. It comprises costs related to administrative tasks (non-exhaustive): communication, setting up of the waste calendar, budget discussions, complaints treatments,…These costs are often partly or completely incorporated in the cost of public service. Calculations with the focus on residual household waste prove that these costs regarding administrative services per inhabitant are really low and hence do not form a profound explanation for a potential higher cost.

During the interviews it became clear that in fact three major parameters influence the cost. More specifically it are the followed route, the level and degree of depreciations of the investments made and the staff salaries. In order to conduct a fair comparison between the cost of private and public service, one should keep track of the evolution of the costs during a longer period of time, by means of panel data.

Most probably the cost ratios between private and public will remain, however the cost level will be different.

The discussion about privatization should move from competition and instead look more closely at the costs of contracting and the organization of the service sector itself (Bel and Warner, 2008). It lies indeed with the municipalities themselves to decide upon which degree of service level they want to offer their citizens and what price they are willing to pay for it regardless the efficiency of the collection firms. Many local government budgets are facing extraordinary challenges as decreasing revenues and rising expenses lead to major shortfalls. One of the most logical solutions is to reduce the cost and hence size of government by concentrating on providing critical municipal services. To achieve this, many cities are increasingly privatizing other activities for which the private sector is best prepared to provide improved service at a lower cost, increased efficiency and other benefits. Waste collection, recycling and disposal are among the most prominent candidates for privatization. More and more municipalities/communities are privatizing these vital services now.

3.3.6 Conclusions

3.3.6.1 Environmental sustainability

Using binary logistic regression, key factors in residual household waste minimization for the Flemish region of Belgium were identified. Reaching the mandatory goal of 150 kg residual household waste per year per capita was used as dependent variable. Independent variables that could explain a municipality's chance of reaching this goal were derived from the literature.

The model correctly classified 74% of the municipalities. Four out of 12 variables in the model contributed significantly. Higher per capita income lowers a municipality's chance of reaching the goal.

An increase in the cost of residual waste collection, changing from weekly to an every other week residual waste collection and introducing separate curbside collection of organic waste increase a municipality's chance of reaching the goal. It is concluded that binary logistic regression is an appropriate tool for identifying key factors in a waste management plan. The implementation of separate curbside collection of organic waste has the strongest influence. This is not surprising as the organic waste fraction, both yard waste and waste from fruit and vegetables, makes up about 40% of total municipal waste weight in Flanders (OVAM, 2002).

Frequency of residual household waste collection, yearly cost of curbside residual household waste collection and the average income per capita have a fairly equal impact.

Using multiple regression analysis, Gellynck and Verhelst (2007) found that the percentage of direct costs to total costs of the waste management program has a significant impact. This finding was not supported using binary logistic regression. This variable captures also pay-as-you-throwpolicies for recyclables. Future research could focus on this particular aspect of the waste management policy as the effect of introducing the 'polluter pays' principle for recyclables is not well understood and the results using the percentage of direct costing as a proxy are ambiguous. Furthermore, a specific policy mix for areas with tourist visitors should be assessed.

3.3.6.2 Economic sustainability

Multiple case studies regarding the collection of residual household waste in the Flemish region of Belgium were executed. In total 12 cases, divided into three mutual comparable pairs with a weekly and three mutual comparable pairs with a biweekly residual waste collection, were investigated. For the considered cases the results give a rough indication of the fact that the cost of private service is lower than public service in the collection of residual household waste. However, it should be noted that in some cases assumptions were made in order to calculate the cost since neither municipalities nor joint ventures keep track of all costs related to each waste fraction separately. Given the lower cost, municipalities and their citizens will benefit in economic terms from private service. However if the expectations regarding the service are high, one will benefit more from public service. Furthermore, municipalities do not always opt the most economic efficient service and they are free to organize the policy regarding household waste. They can manage it on their own, they can cooperate with a private partner or they can be a part of a bigger entity, namely a joint venture of municipalities which has a private or public service level. With this joint venture they try to diminish the price by creating economies of scale. Such cooperations collecting and processing the household waste offer a total service level, which is attractive to some municipalities.

Based on just 12 cases, it is impossible to draw conclusions whether private service in general is cheaper than public service in Flanders. This case study however gives rough indications regarding the cost for collecting and transporting solid household waste as such. In order to avoid assumptions during calculations, joint ventures and its municipalities should have a better and more clear overview of their cost structures, related to the different waste fractions. Therefore if one wants to know the exact cost, tools and assumptions should be excluded. A method like Activity Based Costing can help to allocate the costs to the right services. From the interviews emerged that each joint venture uses different ways to calculate or to keep track of their costs. As such, there is no harmonized way to calculate the cost for waste collection. Future research could tackle this problem by setting up a model using full cost analysis. In agreement with the different stakeholders all factors that contributing to the collection cost, should then be listed. The outcomes could help to detect differences in cost or efficiency between companies regarding waste collection. Moreover, these results can help policymakers, governments, politicians, municipalities and waste collection companies to aim for higher efficiency.

A snapshot was used in order to calculate the costs. Future research should gather data over a longer period of time, using panel data. However, the challenge will be to guarantee comparative cost data over time. Furthermore, it would be interesting to take into account the value of explanatory variables at the moment of structuring the waste management service rather than when the data are recorded.

In order to conduct a fair comparison between the cost of private and public service, one should keep track of the evolution of the costs during a longer period of time. Most probably the cost ratios between private and public will remain, however the cost level will be different. Further research is needed to clarify this issue.

Cost savings crucially rely upon the characteristics of public service markets, the nature of the service itself, geographical dimensions of the market in which municipalities are located, and the overall structure of the sector. It has been proven that there exist plenty of cost calculation schemes for waste collection out there.

In order to avoid controversy regarding the price municipalities and their citizens are paying, waste collection firms should be more transparent in terms of what waste collection comprises and what are the accompanying costs.

Sustainability is the mother of all collaboration.

(Miriam Turner)

4 Stakeholder perceptions towards sustainability

Adapted from:

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4.1 Introduction

Issues on sustainability and sustainable development attract more and more attention from decision makers of any kind. Sustainability is however a contested concept. Sustainability is very complex, so it enables a very large number of perspectives upon it, as it is harder to prove any of them wrong in simple terms.

Many have contested the common definitions and many have come up with different ones. different perceptions of sustainability exist and that these arose from a better understanding and hence more precise conceptualization by analysts, as from different actors with different interests and concerns trying to proliferate and legitimize their claims in the sustainability debate in terms favorable to them.

Peasant organizations, for instance, might associate sustainable development with e.g. global inequity, thereby stressing on the rights, needs and aspirations of the poor. Environmental NGOs may stress the need for nature conservation; and for business actors sustainable development may come down to economic growth and profit maximization. The fact that sustainability is not precisely defined and agreed upon does not mean it cannot be used and assessed as a concept. The extensive use of the concept and the variety of actors committed to it, motivate this chapter on the demystification of the discourses and perceptions of (business) stakeholders towards sustainability.

A straightforward instrument for a multidimensional concept like sustainability is a suitable set of indicators. The latter are used to identify and monitor a change over time towards a certain target. Indicators however also play a huge role in communication with and among stakeholders. Different types of stakeholders can interpret indicators differently due to different values, cultures, backgrounds and academic contexts (Mascarenhas et al., 2014). Gaps between indicator data and the perceptions of stakeholders might originate from wrong communication. Most literature on stakeholders focuses on participation in indicator selection schemes and in data collection for sustainability assessment, e.g. with regards to Life Cycle Assessment.

The sustainability concept, also found its way into the business world. The triple bottom line philosophy emerged, emphasizing that a single economic business objective is not sufficient and that a threefold approach is more sustainable in the long run for all stakeholders (Elkington, 1997). In light of the above it is noteworthy to identify stakeholder preferences with regard to what should be included in a sustainability assessment (stakeholder involvement).

Sustainability indicator development occurs in a very static way and focuses on scientific/objective assessment work only while it should also include “social” norms (Binder et al., 2010). Regarding the normative dimension, a challenge exists in the application of the concept of sustainable development to the different aspects of the system (three sustainability dimensions of the agricultural sector (Binder and Wiek, 2007)). More specifically, targets need to be deducted from the general concept of sustainable development for matching the specific problems of the agricultural sector. Moreover, as targets are preferably consistent they leave decision-makers flexibility for taking measures in a specific context (Morse et al., 2001 and Park, 1996; Binder et al., 2010). The normative dimension might play an essential role if the indicator-based decision-making system wants to be useful for sustainability assessment and its subsequent application (Pope et al., 2004). Normative concepts might show cultural and social variation (Empacher, 2002) and thus, the question of the extent to which a certain assessment is applicable to other countries or must be modified given the cultural conditions, needs to be critically considered before extrapolating results or methodologies to other contexts (Binder and Wiek, 2007).

The sustainability concept can be either completely theory-based (i.e., Bossel, 1999 and Niemeijer, 2002), or developed in a way stakeholders' perspectives can be included (Hacking and Guthrie, 2008, Woodhouse et al., 2000; Lee, 2006), creating a shift from research for society towards research with society, whereby mutual learning between science and society is aimed for (Scholz, 2000; Scholz et al., 2000). Sustainability moves actually on the edge between science and society. Sustainability became what Thomas Gieryn (1999) calls a 'boundary term': one where science meets politics, and politics meet science. Boundary work takes place at this intersection between science and politics. This work often has ambivalent and contested meanings but at the same time plays a thorough political role in policy making and development (Scoones, 2010). Democratic politics should decide upon problems and solutions regarding sustainability. Once these issues have been decided upon, technical considerations might be appropriate. (Lundqvist, 2004).

Deploying sustainability science is not possible without strong political commitment. Otherwise, neither the traditional scientific community nor businesses will use sustainability science processes. From an anthropocentric point of view, sustainable development is about the progress of humankind and thus should reflect social consensus about what is unsustainable and what improvement is needed. Therefore it cannot be translated into a blueprint or a defined end state outlining specific criteria and calling for unambiguous decisions (Voss & Kemp, 2006).

Principle 10 of the Rio Declaration indicates that "environmental issues are best handled with the participation of all concerned citizens, at the relevant level" (UNCED 1992). The Johannesburg Declaration again confirms the need for "broad-based participation in policy formulation, decision-making and implementation at all levels" as well as the "need for more effective, democratic and accountable international and multilateral institutions" (UN 2002 (principles 26, 31), Bell, 2002). Most political liberals stick to a more or less egalitarian theory of intergenerational justice, which urges each generation to make the possible effort to secure for future generations conditions in which it is (at least) possible "for all to have a decent standard of life" through "social cooperation" (Rawls 2001, p. 84). However, political liberals might acknowledge that the structure of representative democracy threatens to undermine the protection of the interests of future generations (Bell, 2002). If only current generations are represented and if political parties have the main interest in seeking re-election every couple of years, representative democracy is not well designed to protect the interests of future generations in terms of sustainable development (Bell, 2002). For political liberals, society is “a fair system of social cooperation (among free and equal citizens) over time from one generation to the next” (Rawls 2001, p. 5). Each generation has the task to hand on “a fair system of cooperation” to future generations and in that sense all generations share in the historical project of maintaining just institutions in perpetuity. As such, all are equal members of a transgenerational society. However, a transgenerational society is not one in which all generations (or, at least, current and future generations) control policy-making at all times. Each generation leads its own life – its own contribution to the historical project – always in the knowledge being part of a transgenerational community with obligations to future generations. In that way, conflicting interests might occur as many people might act out of self interest instead of focusing on the future.

Therefore, integrated assessments of sustainability science may help to identify directions requiring change. Transition management will lead towards a sustainability transition even when no one knows what a sustainable society would actually look like and the very idea of achieving sustainability may be illusory (O’Riordan, 1996). Greater involvement of society in the transition- management process is needed and more attention should be given to issues of societal embedding (Kemp and Martens, 2007). It is thus not only science but also society’s opinion that matters what should be taken into account upon evaluating the soy chain sustainability. A prerequisite however is that society’s opinion is well reflected upon the involved stakeholders during the assessment.

Given this perspective, the objective of this chapter is to identify the preferences of involved (business) stakeholders along the transatlantic soy chain. While there has been a strong focus on impact and performance in the chain, the mechanisms leading to these impacts may become black-boxed. Therefore it is interesting to study the driving forces behind the sustainability management and the inclusion of sustainability values and concepts into supply chains. The aim is to grasp the perception of stakeholders in the supply chain, i.e. business community, non-governmental and governmental actors, in order to better understand the different actors’ reactions to, and outcomes of sustainability initiatives and to understand conflicts and opportunities in sustainability management of supply chains.

This chapter reports part of the findings of an EU-FP 7 project on sustainable transatlantic soy chains. Given its economic importance, it was opted to focus on the soy chain. Elkington’s business philosophy on triple bottom line wants to integrate a sustainable economic, social and environmental value for all stakeholders with each dimension given equal emphasis in the business (Elkington, 1997). The question arises whether (business) stakeholders want to address an equal weight to each sustainability dimension and its indicators. Posed differently, what is the weight (and hence preferences) stakeholders want to address to certain issues. Sustainability preferences towards the soy chain can help several businesses (and other) stakeholders to comply with claims or demands of other stakeholders (e.g. supplier- customer relationships), being revealed or for producers to switch marketing strategy. Furthermore the outcomes can help policymakers to take into account stakeholder preferences when installing sustainability related policy measures. Given the changing soy international trade flows structure, where Chinese import is becoming more and more relevant, the EU wants to sustain business with Latin-America. For them as producers, it also important to understand the

European market. Results can potentially lead to mutual understanding, beneficial for all parties involved. To wrap up, the aim is to increase understanding about sustainability perceptions and indicators perceived as most important by soy chain stakeholders.

4.2 Soybean production

In this regard, there is a common interest among the compound feed industry in Latin America and Europe to sustain the production and exportation rates of soy (SALSA, 2010). Table 26 presents some data on the world's leading soy exporters.

Table 26. Data on the world's leading soy exporters for the period 2010-2011 (USDA, 2012).

Country		Million tons	% global export
USA			
	Bean	42	44 %
	Meal	10	14 %
Brazil			
	Bean	32	34 %
	Meal	14	24 %
Argentina			
	Bean	10	10 %
	Meal	29	49 %

4.3 Methodology

4.3.1 The need for indicators

Indicators are seen as a mean of communication. They are considered as tools in policy making and public communication (Kumar Singh et al., 2009). Meadows (1998) applies the following definition:

"Indicators arise from values (we measure what we care about), and they create values (we care about what we measure)".

Indicators are able to package a dynamic environment into a reasonable amount of meaningful information (Godfrey and Todd, 2001). By means of visualizing phenomena, indicators are able to quantify, analyze and communicate complicated information (Warhurst, 2002). No single indicator is able to monitor the human impact. Indicators need to be used and interpreted jointly (Galli et al., 2012).

With regard to perceiving sustainability and its indicators, one needs to obtain an overall framework of indicators. In order to obtain a sustainability framework, the following approach was followed:

1) stakeholder interviews

2) SAFA guidelines review (see also section 4.3.2): With the aim of improving the transparency and comparability of sustainability performance of food and agricultural sectors, FAO (2012) developed guidelines on the Sustainability Assessment of Food and Agricultural Systems (SAFA guidelines). and

3) expert interviews.

The stakeholder interviews give insights into broad sustainability areas and indicator fields. The latter each fit into one of the sustainability dimensions and represent aggregated themes derived by grouping the specific indicators. The SAFA review and the expert interviews deliver a set of specific indicators related to the soy chain. The combination of the former and the latter, leads to an overall sustainability framework, able to grasp all important soy chain indicators. Figure 17 on the next page provides the flow of operations. A framework is a suitable and straightforward set of indicators for evaluating and assessing sustainability.

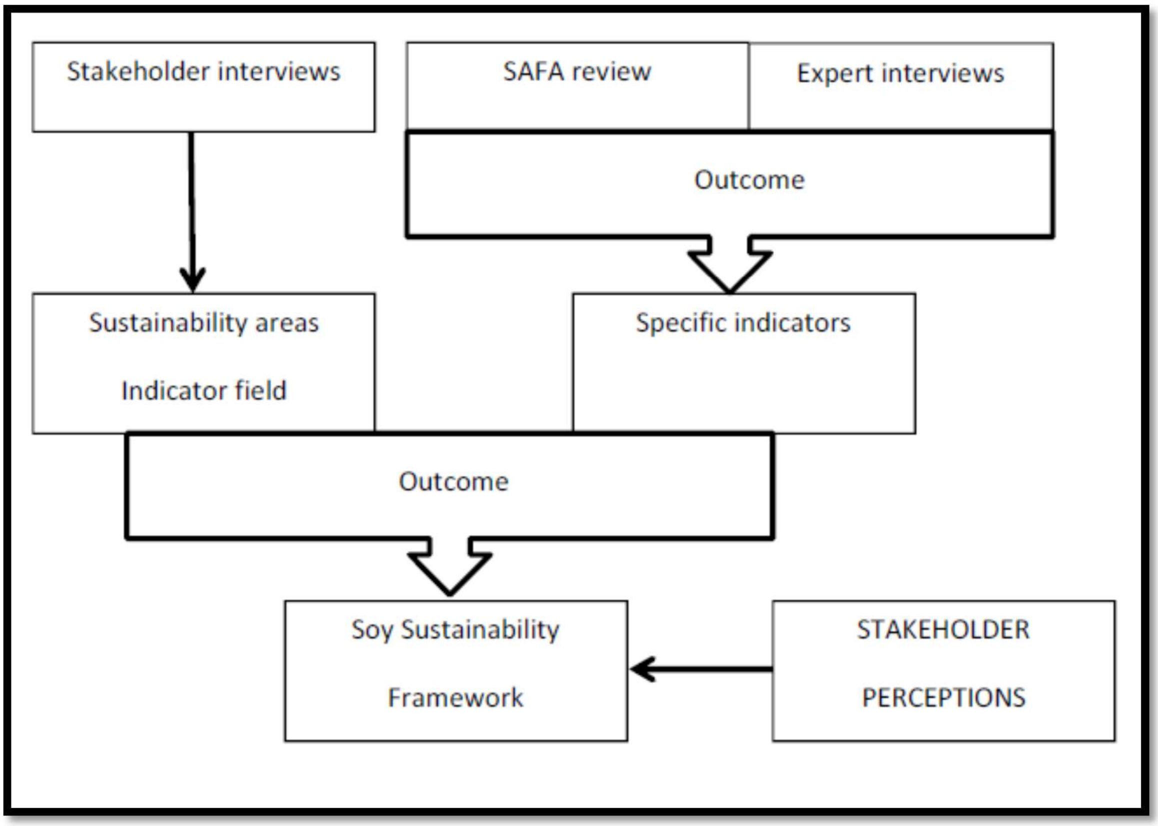


Figure 17. Flow of operations to follow for obtaining a specific soy sustainability framework of indicators.

Stakeholder interviews were conducted to touch upon a variety of topics regarding sustainability, standards and certification schemes from a more broad perspective.

In addition to the literature review, the opinion of experts from Latin America and the European Union is used to define a 'core set' of quantitative indicators.

Experts were business stakeholders in the beef and soy chain in Brazil, Mexico, Belgium, Italy, Hungary and Switzerland. This core set of indicators will serve as the basis of the analytical framework. The aim of the expert interviews was to explore bottlenecks regarding different sustainability issues among and between representative soy chains.

4.3.2 Indicator selection

Dale and Beyeler (2001) have indicated that there is a lack of robust procedures to select indicators. This makes it difficult to validate the information provided by those indicators. Hence, there is a lot of room for improving the selection process. A more transparent indicator selection process increases the value and scientific credibility of environmental assessment reports in order to come accross management concerns (Dale and Beyeler, 2001). In current practice, indicators are mostly chosen based upon historical practices or on intuitive assessment of experts (Bossel, 2001). Indicators are mostly selected based upon individually chosen criteria. Schomaker (1997) believes indicators should be specific, measurable, achievable, relevant and time-bound. OECD (2001) takes into account policy relevance, analytical soundness and measurability as criteria to include indicators. Talking about agriculture, Pannell and Glenn (2000) suggest that it is of importance to have a high uncertainty about the indicator level to be monitored, a low uncertainty about links between the indicators, management practices and production. Indicators should be measured reliably and in an accurate way, and the cost for monitoring should remain low.

4.3.2.1 SAFA review

The SAFA (Sustainability Assessment for Food and Agriculture) guidelines are developed through an ongoing process of text elaboration, stakeholder consultation and participation and continuous reviewing and may therefore be considered as a strong generic template for evaluating the comprehensiveness of selected sustainability indicators. Where necessary the framework can be extended with additional indicators or alternative methods to ensure a comprehensive and balanced framework, touching upon a broad set of ecological, social, economic and even governance (institutional) themes.

Core performance indicators help in providing standardized metrics to help assessing sustainability. A SAFA is giving indicators for users who do not necessarily have the knowledge to develop indicators themselves, avoiding the risk of reducing the assessment level. The SAFA default indicators are applicable at the macro level (to all firms, regardless size or type), and in all contexts. Therefore the SAFA guidelines are well applicable in a food context in general and for soy chains in particular. The 118 SAFA default indicators were developed through expert analysis. As not all indicators are applicable for soy chains, one needs to narrow down the amount. For this chapter, stakeholders and experts along soy chains were involved in order to reveal and identify soy-specific issues and topics.

4.3.2.2 Soy chain indicators

The SAFA framework was used as background information when exploratory interviews were carried out with a wide range of stakeholders (table 27), people who are actively involved in soy chains. By means of these interviews the most important and relevant sustainability issues were identified in order to narrow down the research (sustainability areas). Next, experts were involved to identify soy-specific indicators after stakeholders selected the major issues. Experts are not actively involved but are familiar with the social, economic and environmental performance of the soy chains. The expert interviews allowed to come up with a selection of sustainability indicators relevant to their specific industry.

Each project partner interviewed at least 4 experts, including representatives of academic, NGO or governmental institutions, familiar with social, environmental and economic issues regarding the soy chain. During the interviews, experts were asked to give their opinion on a selection of indicators, according to their respective expertise. The exploratory nature of the interviews allowed interviewees to come up with themes and sustainability issues they deemed relevant. The aim of the expert interviews was to explore bottlenecks regarding different sustainability issues among and between case studies representative of the most widespread soy farming systems in Argentina and Brazil. The justification for bringing down the broad SAFA framework to a more general list of issues is: validation, relevance to stakeholders and comprehensiveness. In this way a balance was found between feasibility of data collection on the one hand, and comprehensiveness and discriminating power of the sustainability framework on the other hand. The origin of the stakeholders goes hand in hand with the location of the project partners (UNIBO (Italy), UGent (Belgium), Wageningen univ (Netherlands), Fibl (Switzerland), ProQ (Germany), RTRS and Solidaridad (Argentina), Embrapa (Brazil)).

Table 27. Number of interviewees (stakeholders).

Origin of stakeholder	Number	Type of organization
Italy	10	Business (5) Academic (3) Governmental (2)
Belgium, Netherlands, Germany, Switzerland	22	Business (17) Academic (3) Governmental (2)
Argentina	12	Business (6) Non-governmental (5) Governmental (1)
Brazil	30	Business (8) Academic (17) Governmental (1) Non-governmental (4)

4.3.2.3 Sustainability framework

The resulting framework after consulting experts and stakeholders can be seen in table 28.

Table 28. Sustainability framework of indicators for transatlantic soy chains.

Indicator/dimension	Environment	Social	Economic
	Global warming	Employability	Operating profit
	Primary energy use	Working conditions	Chain entry barriers
	Land use efficiency	Food safety	Local economic development
	Biodiversity	Global food security	Economic vulnerability
	Water consumption	Land distribution	
		Human health and safety	

4.3.3 Perceptions measuring

In order to measure sustainability perceptions, two complementary approaches were followed: one survey as a tool for ranking by stakeholders, followed by a multi-criteria analysis for prioritizing dimensions and indicators (rating). The first open question however, was to reveal how stakeholders define the concept of sustainability. Short online surveys were created and used to collect responses over a three week period in July-August 2013. The survey was spread among all participating countries in the project: Belgium, the Netherlands, Germany, Switzerland, Hungary, Italy, Brazil and Argentina.

4.3.3.1 Web survey

In order to reveal the preferences of a wide range of stakeholders in Europe and Latin-America a web survey based upon the input from the above defined framework was set up. Respondents were asked several sustainability related questions, following a top down approach. The following three main topics were covered:

1) general definition on sustainability (open question),

2) absolute scoring of sustainability dimensions and indicators to make a ranking,

3) weighting of dimensions and indicators (rating).

Rating forces respondents to make trade-offs, rather than giving them an individual score one by one time after time.

In total, 36 business and 34 non-business stakeholders were interviewed. Out of a total of 70 people interviewed 32 were from the EU and 38 from Latin America.

4.3.3.2 Stakeholders description

Stakeholders were first asked to rank dimensions and second to rank indicators within the dimensions. Descriptive statistics give a nice way of describing the group of stakeholders. A total of 15 indicators needed to be given a score, one by one:

Environment: (1) Impact on water use and quality, (2) Impact on the atmosphere, (3) Impact on land degradation and soil quality, (4) Impact on biodiversity, (5) Impact on non-renewable energy and material use.

Social: (6) Impact on food/product safety, (7) Impact on human health and safety of the local community, (8) Impact on working conditions and labor rights of employees, (9) Impact on global food security, (10) Impact on land distribution and land acquisition for smallholders, (11) Impact on food sovereignty

Economy: (12) Impact on local economic development, (13) Impact on profitability, (14) Impact on national economic development, (15) Vulnerability: chain entry barriers

Table 29 gives an overview of the top 3 in ranking of dimensions and indicators. Ranking involves assigning each decision element a rank reflecting its perceived degree of importance. Since data are not normally distributed, a non-parametric test (Kruskal Wallis one way analysis of variance) was used for detecting significant differences in means between Latin-American and European stakeholders. The significance level was set on 5 %. Environment and economy show a significant difference in ranking. On indicator level, significant differences were found for: water use and quality, land use efficiency, energy use, local economic development, profitability and national economic development.

Table 29. Top 3 ranking of sustainability dimensions and its indicators, by stakeholders

Dimension/indicator	EU	LA
Rank 1 dimension Within dimension rank 1 indicator	Environment Water and land use	Environment Water and land use
Rank 2 dimension Within dimension rank 1 indicator	Economy vulnerability	Social Working conditions
Rank 3 dimension Within dimension rank 1 indicator	Social Food/product safety	Economy National economic development

The web survey based upon a relative small group of stakeholders reveals a difference in ranking between Latin American and European people regarding sustainability dimensions and its indicators: Latin Americans rank higher on an individual basis. The absolute scores for all indicators are given in annex. It seems that environmental and economic indicators also show significant differences in scoring. Social indicators are to some extent granted similar. Within stakeholder group differences could not be detected by means of post hoc tests, as the sample size was too small. Differences could rather occur through sectoral affiliations.

4.4 Results and discussion

4.4.1 Integrated sustainability perception

At first stakeholders were requested to give an overall (soy) sustainability definition. Most definitions of sustainability were referencing Brundtland. Others referred to the triple bottom line approach and several respondents also mentioned cultural issues. In light of the above, there were many definitions (especially from EU side) focusing on environmental issues, as there are enhancing natural assets, natural resources management and human impact reduction.

Although there seems to be cohesive common ground among soy business stakeholders towards sustainability, a closer look into the narratives and storylines reveals a more differentiated perception of and perspective on sustainability in the soy chain.

Latin American producers, while in general recognizing the need or possibilities for reducing negative impacts of soy production, often stated to feel targeted in the sustainability debate in the soy chain. While demands for sustainable products and processes generally originate in Europe, Latin American producers often stated that commitments of traders and importers for more sustainable chains mostly entail relatively high efforts from producers without clear and sufficient benefits. Soybean producers strongly relate sustainability with sustainable land management with a very prominent role of no-tillage practices. Assurance of satisfactory market premiums for sustainable products and binding assurance of market access, are crucial for many Latin American producers in order to create the necessary market incentives for sustainability initiatives to be successful.

European and multinational traders and importers often refer to sustainability as the business ' license to operate and a commitment to corporate social responsibility. Traders, processors and importers often refer to the global food crisis in the debate on sustainable soy chains. Evidence of the growing importance of the food security agenda in companies CSR-strategies (Corporate Social Responsibility) can be found on these companies' web pages. In contrast with the market pull suggested by many interviewed Latin American producers, business actors at the demand side of the soy chain, not surprisingly, advocate a market push. Scaling-up the production of sustainable soy and mainstreaming the market for sustainable products, would create the necessary incentives (low premiums and high supply) to tip the market in favor of more sustainability. According to European and multinational companies, especially European interviewed retailers, to be successful any initiative should be reassured to the consumer. Any sustainabilty oriented initiative should provide the consumer a credible guarantee system. Table 30 below is a result of the general discourses on the sustainability of the soy value chain given by respondents in the web survey. This table allows to better identify matches on sustainability issues between EU and LA.

Table 30. General (matching) discourses of EU and LA (business) stakeholders towards a sustainable soy trade.

Discourse	EU	LA
Publishing information about economic, social and environmental performance of companies in the chain	X	X
Facilitating trade of sustainable soy products by eliminating import barriers in the EU	X	X
Encouraging multi-stakeholder dialogues	X	X
Harmonizing sustainability criteria to simplify the choice of buyers	X	X

(continued)		
More strict national or international environmental regulation and law enforcement	X	X
Increase the price premium for certified sustainable soy		X
Subsidies to support more sustainable production systems		X
Creating alternative trade networks	X	X
Using local or alternative protein sources for the EU feed industry	X	
Soy originating from deforested land should be banned from the market	X	X
Implementation of a strict traceability system in the entire chain	X	
Pursuing a strict, ethical business conduct		X

European stakeholders want to be less dependent on the import of proteins. Furthermore they really indicate to care about environmental (deforestation) and social issues in the soy exporting countries. Moreover the existing sustainability schemes and standards should be harmonized in order to obtain a good quality management system and to facilitate trade with the European Union by eliminating trade barriers. Subsequently European stakeholders are not in favor of increasing the price premium for purchasing sustainable soy from producers. This is an issue found back in the expert interviews as well: European (business) stakeholders are not willing to pay extra for sustainable soy. On the other hand, creating more labels is not an option, the counterpart of harmonizing all sustainability schemes.

Also Latin American stakeholders are in favor of harmonizing the existing standards. Obviously, for economic reasons they are not in favor of stimulating the European feed market to find local proteins and hence to encourage the European consumer of reducing its meat consumption. Furthermore, the environmental and social issues are very important and these indicators should be taken into account when producing sustainable soy. However, when producing this, Latin American stakeholders expect to receive a higher premium from Europe, which the latter is not very keen on.

4.4.2 Stakeholder perceptions

Within the ranking section, they indicated absolute scores for sustainability dimensions and indicators, on a one-by-one basis. A shortcoming of this methodology is that scores can be given over and over again starting from scratch, implying that almost every issue might become important as such. It is however more interesting to reveal their preferences, by making them choose among several options: weighting (rating) by making trade-offs. Stakeholders were requested to allocate a 100 points among dimensions and indicators within a certain dimension. This allows to come up with a prioritizing list of sustainability dimensions and indicators, revealing interesting information for decision makers of any kind. Table 33 presents the assigned weights on both continents. The framework was simplified in order to conduct straightforward data collection and present a user-friendly framework for weights allocation.

Rating is somewhat similar to ranking, except that decision elements are given ‘weights’ between 0 and 100. The weights for the elements being compared must add up to 100. Thus, to score one element high means that a different element must be scored lower. For Latin American stakeholders economy (36 %) is the most important dimension, followed by social issues (33 %) and environment (31 %). The latter is most important for European stakeholders (42%), followed by social issues (31 %) and economy (28 %).

Table 31. Top 3 of stakeholder perceptions (rating) towards sustainability dimensions and its indicators.

Dimension/indicator	EU	LA
Rate 1 dimension **Within dimension rate 1 indicator**	Environment (42 %) Global warming	Economy (36 %) Operating profit
Rate 2 dimension **Within dimension rate 1 indicator**	Social (31 %) Working conditions	Social (33 %) Working conditions
Rate 3 dimension **Within dimension rate 1 indicator**	Economy (28 %) Operating profit	Environment (31 %) Land use efficiency

The rating delivers different results compared to the ranking method. The latter allows respondents to rate each dimension and indicator on an individual basis, implying that almost every issue as such might be important to them. It is however not the aim to compare methods. By revealing perceptions, stakeholders have to make trade-offs among dimensions and indicators.

The outcomes of the web survey should be seen as subjective. The findings prove that Latin Americans perceive sustainability differently in the sense that higher importance is addressed to sustainability issues on an individual basis. This is however to some extent contradictory and intuitive in against the European expectations, where the debate on sustainability and initiatives in this regard are expected to surpass the ones in Latin America. The great interest in sustainability issues from the Latin Americans does hence not match with reality in terms of initiatives, which originate from Europe.

Understanding each other's perceptions towards sustainability is important. On a global level efforts have been done to reach a common understanding (UN conferences, global civil society and business community approaches like GRI, ISO 26000; ISEAL codes of conduct etc.), however no paper focused on the mutual understanding of stakeholders on both sides of the Atlantic.

The main differences between EU and LA might very well be explained with the fact that the overall improvement on (environmental) indicators in the EU countries studied has been quite good in the last 2 decades, while in Latin America people are still clearly experiencing e.g. the environmental degradation, and thus give higher relevance scores.

Overall, the match between EU and LA stakeholders seems good: social and economic issues were rated to the same extent. The discrepancy is situated at the level of the environmental dimension. For Europeans environmental problems are most important to address, whereas Latin Americans place environment in third. Moreover, within environment LA seems to care about land use and its efficiency, whereas European stakeholders are more keen on controlling water consumption and mitigating climate change. The former is not at all important to LA stakeholders. Land use covers both the environmental (primarily) as well as the economic dimension (agricultural importance) of sustainability.

4.4.3 Limitations

The used methodology however shows some limitations. The survey is applied to a small selection of various stakeholder groups at LA and EU side. Furthermore the selection of stakeholders is very close to the inner circle around RTRS (Round Table on Responsible Soy), which might omit interesting stakeholders at any chain level. Future research could focus on a larger sample size in order to conduct post hoc tests and reveal within group differences to identify the source. Moreover, now it remains a question whether these differences are attributable to sectoral affiliation (business, academic, …) or nationality. This would be an interesting finding and might suggest the relevance of an interests-based approach to policy as a theoretical framing.

Moreover the environmental indicator weights are likely to be underestimated due to their higher number in the questionnaire, when compared to the economic and social ones. This is however fully comprehensible since environmental indicators abound, when compared to other sustainability indicators on economic and social aspects in sustainability frameworks of any kind. The results however reveal the most important issues and can help stakeholders of any kind to understand each other in terms of sustainability. As a consequence, agents can collaborate for creating a more sustainable chain, fulfilling the needs and expectations of everybody involved.

4.5 Conclusions

The limited sample size has shown that in a sustainability assessment European stakeholders find the incorporation of issues like global warming, water consumption, operating profit and working conditions important. In the same assessment Latin American stakeholders share the same economic and social interests but prefer to focus on land use efficiency instead of climate change regarding the environmental impact. Both continents want to facilitate trade with one another by harmonizing sustainability criteria. The EU however wants to be less dependent upon the import of proteins. Latin America on the other hand wants to keep on trading with Europe and hence needs to comply with sustainability claims. Aiming for harmonization thus indicates that sustainability perception towards the same crops (i.e. soy) should be scrutinized on both sides of the ocean. Therefore this chapter clearly identified the need to fully understand other stakeholder perceptions in a world where the concept of sustainability is gaining more and more interest. Learning from each other preferences and perceptions can hence help to further develop a market for sustainability, especially in the case for overseas exported crops.

Because of its pioneering role in bringing together stakeholders and striving for tangible results, due attention is given to the RoundTable on Responsible Soy (RTRS) and its participants. Many large food companies engage in sourcing sustainable raw materials. McDonald’s in Europe announced its intention to have 100% sustainably certified soy in its chicken meat supply chain by 2020. Also Unilever, a founding member of RTRS, finds sustainable sourcing very important. Unilever Brazil bought certificates covering 5,000 tonnes of sustainable soy oil. Unilever wants to sustainably source all soy beans by 2014 and all soy oils by 2020. Sustainable sourcing is an important part of the Unilever Sustainable Living Plan. Many companies would consider engaging in multi-stakeholder initiatives and cooperating together with NGOs, as successful partnerships and proof of their commitment to sustainability. Nonetheless, for a large group of civil society groups, cooperation with multinational companies would be considered as a major risk. These civil society groups often argue that solutions to increase sustainability in the soy chain should be sought outside the mainstream soy market. The assumptions, analyses and perspectives, related to these more radical discourses on sustainability are often incompatible with the reformist discourse of participants in the RTRS and other market based sustainability initiatives. In this way, a gap gradually emerged between participants, companies and organizations outside multi stakeholder initiatives. This might jeopardize the ambition of RTRS to claim legitimacy, neutrality and authority of its global voluntary initiative in the soy chain. If RTRS wants to live up to its goals, RTRS and its opponents should be open to other discourses. In doing so, the focus of further research should not be solely on EU consumers or LA producers. In Latin America, civil society is shaping its own discourse on sustainability. The growing awareness of the potential impacts of soy production especially related to pesticides and GM-soy use, will influence the debate on sustainable soy. European business stakeholders are facing an uncomfortable position, in which they are confronted with civil society organizations, criticizing the mainstream soy complex for a wide variety of reasons and taking a more radical stance in the sustainability debate. On the other hand, retailers and food companies only face a niche market demand for sustainable soy products. Soy is a hidden ingredient in a wide variety of food products; informing consumers is an indispensable condition in order to encourage consumer behavior and demand. In a challenging global market, European business stakeholders are experiencing decreasing deliberative power to put forward demands in the global soy chains. Growing Asian market power might soften the sustainability demands, which have strong roots in Europe, in favor of more traditional quality aspects.

Another possible constraint to the increase of export to Asian markets is the orientation towards importing unprocessed soybeans from Latin America. The share of added value related to the processing stage is hence transferred to the importing country (e.g. China). The opposite happens when exporting to EU, mostly oriented towards importing processed soy (soymeal). Finally, the European retailers, food and feed industries, which rely upon soy import (a high quality and relatively cheap protein source) are working hard to get public acceptance for sustainable soy. The acceptance of sustainable soy of any kind would be important for European business stakeholders, who are facing rising costs for assuring segregated streams of soy differing from GM (non-GM, organic, RTRS or other certified soy).

Sustainable development is widely debated. It covers and integrates numerous practices and related undesired impacts. It relates to the widest diversity of related stakeholders, who from their specific positions contribute to and feel the effects of these practices. Perceptions of the diverse stakeholder groups are closely related to these positions. The same goes for views on what sustainable development and corporate sustainability should be about. The main differences between EU and LA might be explained with the fact that Europe has done improvements in terms of environmental degradation, whereas people in Latin America are still experiencing these issues in their daily lives, and therefore have given higher relevance scores. The same holds true for other sustainability issues. Moreover, future research could try to reveal whether the within group differences are attributable to sectoral affiliation and not to continent or nationality. This would be an interesting finding and might suggest the relevance of an interests-based approach to policy as a theoretical framing.

The above chapter covered a more or less subjective assessment of sustainability, in terms of perceptions and preferences. There is however also a need for assessing sustainability in an objective way. Sustainability should incorporate both knowledge and social norms in terms of perceptions (Binder et al., 2010). The objective assessment will be elaborated in the next chapter. Stakeholders were involved to designing an indicator framework. This will be the basis for the integrated sustainability assessment reported in the following chapter.

The future belongs to those who understand that doing more with less is compassionate, prosperous and enduring, and thus more intelligent, even competitive. (Paul Hawken)

5 Integrated sustainability assessment

Adopted from:

Jacobsen, R., Driesen, T., Galvão, A., Van Damme, P., Gellynck, X. (2014). Beyond Life Cycle Sustainability Assessment in soy chains: the usefulness of adding inferred manifest variables and aggregation methods. *Journal of ecological indicators* (submitted and under review)

5.1 Introduction

Increasing population, changes in the global nutritional status and higher incomes contributed to higher food consumption all over the globe (Tilman et al., 2002). More consumption impacts both demand and supply of food, thus bringing about stringent economic, social and environmental challenges on a global scale (Tilman et al., 2002). The food sector needs to produce more and at the same time decrease the accompanying negative impacts (SAFA, 2012). As such, sustainability came forward as an important agenda for society as a whole. A lot of countries have set nationwide strategies for obtaining sustainable development, including sustainability targets and indicators (FAO, 2012). Several attempts were undertaken to make the agricultural sector in general and the food sector in particular more sustainable. However, there is no single standard on an international level defining what sustainable production should entail (van der Vorst et al., 2013). Moreover, there is no international agreement on which set of indicators to involve upon measuring sustainability performance.

It is not clear when a certain product or process can be seen as sustainable (FAO, 2012). Hassini et al. (2012) report similar findings.

In order to guide decision-makers in selecting among sustainable products , a triple bottom-line (i.e. people, planet and profit) comprehensive assessment is needed (van der Vorst et al., 2013). This assessment allows to create more sustainable food production chains. Chains which are able to reduce environmental degradation, economic instability and social insecurity (van der Vorst et al., 2013). Such integrated assessments will ideally cover the entire lifecycle of a product ensuring the inclusion of all relevant impacts (UNEP, 2009).

Albeit the lack of decent recipes for sustainability (Wals and Jickling, 2002), producing a transparent assessment is still a challenge (Binder et al., 2010; Pischke and Cashmore, 2006).

Consumers and the public domain in developing countries (like Brazil) encounter serious limitations in accessing and understanding sustainability information on the production and products of both domestic and international value chains (Mol, 2013). There is thus a clear need for more transparency, especially with regard to global value chains as they are more and more confronted with voluntary and mandatory demands on transparency and at the same time to disclose information on the sustainability qualities of products and processes along value chains (Mol, 2013).

In light of the above, there is a clear need for designing an integrated sustainability assessment tool, covering all dimensions. Too many sustainability indicators are just meaningful to scientists and people in the field, but not necessarily to policy makers and a broad audience. (Moldan et al., 2012).

Section 5.2 covers the objective assessment of transatlantic soy chains. The final result of this section will be an integrated sustainability assessment tool, covering all dimensions. Several soy production systems are benchmarked. Moreover, this integrated performance assessment tool might serve as a baseline in defining sustainable production for these chains.

The objectives of the assessment are: 1) to address a gap in literature on commercial soy production, 2) to develop an overall sustainability performance assessment tool, 3) and to be able to deliver an assessment useful for stakeholders.

5.2 Objective assessment

5.2.1 Life Cycle Sustainability Assessment

The concept to combine 3 LCA techniques (one for each sustainability dimension) was introduced by Klöpffer (2008), followed by Finkbeiner et al. (2010) and can be described as follows:

LCSA (Life Cycle Sustainability Assessment) = (environmental) LCA + LCC (Life Cycle Costing) + social LCA. Life Cycle Sustainability Assessment (LCSA) evaluates all environmental, economic and social negative impacts and positive benefits of a product along the whole life cycle. Results can help to back up processes in decision making (Finkbeiner et al., 2010).

It is acknowledged that LCSA is a technique to assess sustainability performance of a product. The scope of this PhD dissertation is not to discuss the current state of the art regarding LCSA. Life cycle techniques are able to give information in order to manage the value chain and social responsibility of an organization, from cradle to grave, addressing the triple bottom line dimensions of sustainability.

5.2.2 Qualitative indicators

LCA makes use of quantitative indicators. In general, environmental LCA and economic LCC use quantitative data. However, issues on social LCA (and also sensitive economic and ecological conditions are hard to capture in 1 or more figures) are somewhat more challenging to quantify. The nature of some indicators requests a more qualitative approach, hard to tackle with an LCA. This is not tackled in current LCSA, which is for the most part quantitative or to some extent semi-quantitative (Finkbeiner et al., 2010).

An indicator is a quantitative or qualitative factor providing a simple and reliable way to reveal changes linked to an intervention. Indicators comprise of information, able to help indicating changes. They allow us to identify differences, improvements or developments linked to a change. " Indicators are approximations. They are not the same as the desired change, but only an indicator of that change. They are imperfect and vary in validity and reliability." (Michael Patton, 1996). Indicators are acknowledged as a very important tool in decision making processes and communication to a broad audience. “Indicators arise from values (we measure what we care about), and they create values (we care about what we measure)” (Meadows, 1998). The main characteristic of indicators is their possibility to wrap up, focus and underpin the huge complexity of the dynamic environment one is operating in. They are able to narrow it down to a reasonable body of information (Kumar Singh et al., 2009). By visualizing phenomena and emphasizing trends, indicators quantify, analyze and communicate information which would be difficult to interpret (Warhurst, 2002).

Qualitative indicators are seen as subjective, unreliable and difficult to verify. They are more difficult to ascertain because they probe the whys of situations and the contexts of people’s decisions, actions and perceptions. However, qualitative indicators are valuable to the evaluation process because projects and initiatives are involved with studying changes in people’s lives and in communities. They seek to measure the impact and evaluate the long-term effects and benefits of a project or an initiative.

Some indicators are of course not necessarily quantitative or qualitative per se. It is not a matter of finding “which one is better”, it is about “which one is more suited for which purpose”. Due to the nature of some indicators, a qualitative approach is more appropriate. It is not $Q_{quantiative}$ OR $Q_{qualitative}$, it is $Q_{quantiative}$ AND $Q_{qualitative}$. Qualitative aspects will be elaborated more in the next paragraphs.

5.2.3 Soy production sustainability

In recent years, concerns about the sustainability of soy supply chains have been growing. The Latin American EU trade in soy, mainly for the EU feed and biofuel industries, has raised questions -especially in the EU- about its impacts. Criticism involves the impact on global climate change and deforestation in Latin America. At the same time the agribusiness is contributing to strong economic growth in a large number of Latin American economies. Evaluating a complex concept like sustainability therefore requires a critical reflection of a number of social, economic and ecological impacts, which is a condition for sustainability initiatives claiming legitimacy, authority and neutrality. Regarding the sustainable production of animal feed and food for humans, a debate has emerged with respect to the transatlantic import of soya from Latin-America to Europe, being an ideal case for conducting an integrated assessment.

Soy production relates to all 3 aspects of sustainability: environmental and social problems accompanying the production and transportation on one hand and economic issues on the other hand. It involves several stakeholders, each of them having a different perception on sustainability (see chapter 4).

Information on sustainability assessments solely based upon Life Cycle Assessment (LCA) indicators somewhat abounds in literature, as the focus is mostly on 1 sustainability dimension. Indicators with a more qualitative nature however lack in these studies. Therefore, it is an interesting field to study. At one point, indicators with a more qualitative nature need to be validated, in order to obtain an overall aggregated score (complex indicator or an index). The latter has not been done before or is even not suggested to do so (Valdivia et al., 2013; Traverso et al., 2012; Capitano et al., 2011). It is moreover a challenge to extend the LCA framework with indirect quantifiable indicators, and to come up with an overall aggregated sustainability score. Indirect quantifiable indicators are not measurable by means of an LCA, and are identified by means of qualitative research/interview methods. As more issues regarding sustainability occur to be important, filtered from qualitative interviews, these aspects might need inclusion in any sustainability assessment framework too. Qualitative nature indicators are manifest variables, in order to help measuring the concept of sustainability (which is latent).

The aim of this chapter is twofold:

1) Delivering a sustainability performance assessment tool for defining a baseline for sustainable production. The aim is here to obtain a one-metric sustainability score, implying aggregation among and between dimensions.

2) To assess overall sustainability performance (both quantitative and qualitative) of LA-EU selected soy chains, covering different production systems as cases.

The focus is hence not on conducting a full Life Cycle Sustainability Assessment as such, but more on the translation of results to obtain overall sustainability scores and accompanying tools. Moreover, it is scrutinized how manifest (inferred from qualitative interviews) indicators deliver an extra value to a sustainability performance assessment. The sustainability assessment tool is delivered as a means for knowledge transfer and decision-making.

5.3 Background

The previous chapter already indicated the importance of soybean export (table 28). This section elaborates on the specific case of the Brazilian soy chain.

During the last three decades, soybean production has grown rapidly and has become one of the most important Brazilian agricultural products (Cavalett et al., 2009). The EU is only able to produce a small fraction of its consumption of oilseed meal and other protein-rich feedstuffs (required to feed livestock) due to climatic and agronomic reasons (Salin and Ladd, 2012).

In Brazil, soy is produced all over the country. The states of Mato Grosso and Paraná, however, are the most important producers, accounting for 27% and 20% of the national production, respectively. According to Aprosoja (union), the state of Mato Grosso alone is responsible for 8% of the world's soy production. In the harvest time during the period 2010/2011 the state produced more than 20 million tons of soybeans. In cultivated areas, the order of the ranking is virtually the same, with Mato Grosso in the first place, Paraná in second and Rio Grande do Sul in third (ICONE, 2011). USDA reported recently that more than 90 % of Brazil's soy crop is GM. GM production is increasing in Brazil and producers shift one year to another from non-GM to GM and viceversa. Decisions are taken due to yields, pest control and costs (more than the ethics dilemma about GM soy). If producers do not have an interest or market for certified non-GM soy, then GM soy production will increase every year. According to estimations the non-GM production was estimated to be 25 % in 2011 with a tendency to decrease in the upcoming years (Solidaridad, 2013). Regarding organic soy, the world production is estimated to be around less than 0.10% of the market. In Brazil the estimation is 30 000 tonnes on a total production of 73.8 Mtonnes, implying a market share of about 0.04 %. Hence this is a niche market, especially runned by smallholders.

Soybean production in Brazil can generally be classified into 2 types of farming systems:

(1) conventional farming including genetically modified (GM) and non-genetically modified (non-GM) farming systems, and (2) organic farming systems (Dalgaard et al., 2008). Non-GM and organic soybeans are typically cultivated in Rio Grande Do Sul, Paraná and Santa Catarina in the South of Brazil (Dalgaard et al., 2008), however organic and non-GM represent the minority of soybean production systems. These states are the regions in Brazil in which soybean production is grown traditionally. The region Paraná itself produced about 20 % of total Brazilian soybean production in 2009/2010 (Dalgaard et al., 2008). In Paraná and the southern states, soybeans are mainly produced on small to medium-sized farms. About 90 % of the soybean area in Rio Grande do Sul and Paraná is cultivated on farms smaller than 1000 hectares (Dalgaard et al., 2008). An increase in the farm size in these states is often not feasible, and consequently, peasants focus on niche markets such as organic and non-GM soybeans (Dalgaard et al., 2008).

However, in the central-west part of Brazil, farms are mainly medium to large-sized. In the states Mato Grosso and Mato Grosso do Sul, respectively 78 % and 51 % of the soybean region is cultivated by farms larger than 1000 hectares.

Organic soybeans are genetically the same plant variety as the non-GM soybeans, but are grown according to a different set of rules related to the organic standards. GM soy is a type of modified soy in which the DNA of soy is adjusted to allow high doses of glyphosate herbicides (mainly, there are other events) (Charles, 2002; Soares et al., 2005). All soy production systems have both strengths and weaknesses in terms of sustainability. Several studies stated that GM and organic products can positively affect a number of environmental, economic and social sustainability aspects (Azadi et al., 2011; Dalgaard et al., 2008; Speiser et al., 2013). Examples are increasing food production and cost reduction for GM soy; less soil pollution and human health effects for organic soy. However, other studies claimed that the shift in soy production systems from non-GM to

GM and organic not only affects the environment, but also affects other sustainability issues such as profitability and working conditions.

The United States, Brazil and Argentina are at the top when it comes to soy exports (Table 28). The three countries are responsible for 90% of the world’s soybean exports. Nevertheless, Argentina is bigger in the export of soy byproducts, especially oil and meal, due to its policy on export taxes. Therefore crushing is being done in Argentinean harbors, whereas Brazilian exported soybeans are mostly crushed, upon the port of arrival in Europe. In the season of 2009/2010 harvest time, Argentina exported 50% more soybean meal than Brazil. Similarly, Argentina exported approximately 66% more of soy oil than Brazil (ICONE, 2011). Brazil exported US$ 2,88 billion to EU-27 and US$11,88 billion to China (UN Comtrade, 2013).

The main destinations of exported soybeans are China and the European Union (Table 32). During the last ten years, Chinese imports of soybeans overtook the European, and it keeps on growing. European imports decreased about 30% during the last ten years, while Chinese imports showed an increase of 280% at the same time. The European Union, however, is still the main global importer of soybean meal, in order to comply with demands of the feed compound industry. Nevertheless, similar to the bean imports scenario, the European share in soybean meal imports has also been decreasing.

Table 32. Leading soy importers in 2009/2010 (ICONE, 2011).

Country	Million tons	% imports
China		
Bean	55	59 %
Meal	0,3	1 %
European Union		
Bean	14	15 %
Meal	23	40 %

Hence, a sustainability performance assessment of current soy chains related to GM, non-GM and organic systems in Brazil is necessary to provide insights into the contribution on the debate towards the environmental, economic and social consequences of these three production systems. The analysis comprises small-scale and large-scale farms in the region of Paraná and Mato Grosso do Sul in Brazil.

5.4 Methodology

5.4.1 Approach

In order to assess sustainability, there is a need to define a sustainability framework, covering the most important indicators for the soy chain. To do so, a literature review was conducted, followed by stakeholder interviews (table 33) in order to grasp the hotspots on sustainability. The literature review was conducted by selecting quantitative and qualitative scientific research, organization reports, etc. The selected studies covered a large share of the comprehensive indicators for classifying and analyzing food supply chains in detail. The review was performed in three steps.

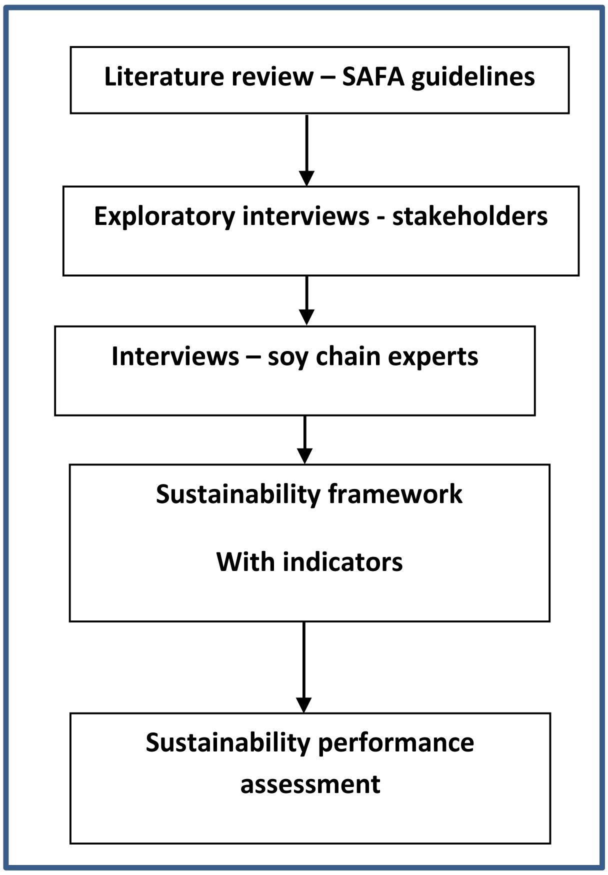


Figure 18. Conceptual flow of operations for conducting sustainability assessment

First, integrated triple bottom line (simultaneous consideration of sustainability dimensions) studies were reviewed to identify key environmental, social and economic indicators of food chains. The outcome of this first step was used as the base for creating a list of indicators to be included in any sustainability assessment. In the second step, the literature review was expanded with any economic, environmental, and social indicators that were evaluated separately for soy chains.

The results of the stakeholder interviews were used to extend the analytical framework with indicators aiming at measuring or exploring sustainability issues. These indicators are not quantifiable as such, they are manifest variables, which need to be validated.

Table 33. Overview of selection of interviewees (stakeholders).

	N° of interviewees
Soy chain	29
Geographic distribution:	
Latin American business stakeholders	14
European and multinational bus. stakeholders	15
Nature of the business:	
Production	9
Processing	7
Feed industry	4
International trade	3
Retail	3
service	3

Moreover the comprehensiveness of the extended analytical framework was evaluated using the SAFA guidelines, developed by the FAO (2012) as a reference conceptual framework. With the aim of improving the transparency and comparability of sustainability performance of food and agricultural sectors, FAO (2012) developed guidelines on the Sustainability Assessment of Food and Agricultural Systems (SAFA guidelines). The SAFA guidelines are developed through an ongoing process of text elaboration, stakeholder consultation and participation. Moreover continuous reviews take place and therefore SAFA can be considered as a strong generic template for evaluating the comprehensiveness of the selected sustainability indicators. The framework can be extended with additional indicators to increase the comprehensiveness in order to touch upon a broad set of ecological, social and economic themes.

Subsequently, the whole process of identifying important indicators was finalized by conducting expert interviews. Each partner in the (EU funded) research project interviewed at least 4 experts, including representatives of academic, NGO or governmental institutions, familiar with social, environmental and economic issues in the soy chain. The template of the interviews is included in annex 1. During the interviews, experts were asked to give their opinion on a selection of indicators, according to their respective expertise. The exploratory nature of the interviews allowed interviewees to come up with themes and sustainability issues they deemed relevant. The aim of the expert interviews is to explore the bottlenecks regarding different sustainability issues among and between representative soy chains originating from Brazil. Based on the results of the expert interviews the framework was finetuned to fit the specific character of soy chains. The number of indicators could therefore be reduced to a limited number of relevant, measurable and discriminative indicators.

GM, non-GM and organic soy chains in Brazil were considered. Processing and transportation in soy chains is very much bulk-oriented and hence comparable among production systems. Differences in sustainability impact are mainly detected at the production side (Tomei et al., 2011). Based upon the chosen framework, it was opted to select 3 production systems significantly differing in the following aspects: land use, pesticides use and chain relationships. These issues were derived from the expert interviews. In this way, a balance was found between feasibility of data collection on one hand, and comprehensiveness and discriminating power of the sustainability framework on the other hand. The interviews with experts allow to reveal deeper insights on sustainability issues, on what aspects are considered as hot spots. This approach works more deeply and thoroughly compared to quantitative secondary data collection.

5.4.2 Sustainability framework

The resulting analytical framework consists of quantifiable 'core' indicators and 'extra' qualitative indicators in order to guarantee the comprehensiveness of the assessment. Table 4 provides an overview of the screening process of the core indicators using the SAFA guidelines from the FAO. This provides a generic overview, to be narrowed down towards the extra indicators (see table 34).

Atmosphere: is referring to the change in composition of the atmosphere, related to greenhouse gases.

Freshwater: Agriculture needs a lot of water for its activities. The largest sources of water use during soy production are the plantation phase, product processing (crushing or milling phase) and for transportation.

Land (use): refers to the use of land, including land transformation

Biodiversity: is the degree of variation in life.

Materials and energy: direct and indirect energy needed along the chain in order to produce a certain amount of soy

Investment: Return on investment can be verified by means of the profitability. The latter can be either profit or loss depending on the size of costs and the revenues.

Vulnerability: the exposure to risk due to the nature of the economic activities. This can be verified by measuring the volatility.

Labour rights, human health and safety: can be verified by means of the working conditions, referring to the working environment, terms and conditions employees are working in.

Table 34. Screening of indicators using the SAFA guidelines.

	Themes	Soy framework
Environmental integrity	Atmosphere	Global warming
	Freshwater	Water consumption
	Land	Acidification, land use (change)
	Biodiversity	Biodiversity
	Materials and energy	Energy use
Economic resilience	Investment	profitability
	Vulnerability	Volatility
	Product safety and quality	
	Local economy	
Social well-being	Decent livelihood	Working conditions
	Labour rights	Working conditions
	Equity	
	Human health and safety	Working conditions
	Cultural diversity	

LCA indicators are limited in the narrow sense to the environmental dimension of sustainability, given the nature of the concept of a Life Cycle Assessment (LCA). As such, the concept can be extended to social LCA and economic LCC. Again, this proves the need for extending the framework with extra indicators to cover the whole range of sustainability dimensions, in a more qualitative way. The last column of table 34 therefore provides the withheld (selected) indicators in this analysis. Core indicators were selected from the SAFA guidelines themes, tackling the most important issues from the expert interviews.

5.4.3 Expert interviews

When dealing with the qualitative assessment of the sustainability performance in soy chains, the relevant constraints on its feasibility, related to data availability, the sensitiveness of data, the time and budget available, must be considered. As a consequence, expert interviews and literature review were chosen to collect data. Given the sensitivity of some questions, mostly those related to social aspects, the respondents were carefully selected to minimize the risk of getting biased opinions. The experts' semi-structured and open interviews were carried out by academics of the University Federal de Viçosa (UFV, for the Brazilian soy supply chain), RTRS (Round Table on Responsible Soy) and Solidaridad (NGO) for the soy supply chain in Brazil. Ghent university took care of the soy supply chain in the EU with a focus on Belgium and Netherlands, as soy meal is delivered to the Antwerp and Rotterdam ports. The template of the interviews, is included in annex 1. During the interviews, experts were asked to give their opinion on a selection of indicators, according to their respective expertise (e.g. social scientists, agricultural experts, representatives of governmental agencies, etc.). The exploratory nature of the interviews allowed interviewees to come up with themes and sustainability issues they deemed relevant. In total, 20 experts were interviewed (5 from Europe, 15 from Argentina and Brazil). The aim of the expert interviews was to explore the bottlenecks regarding different sustainability issues among and between relevant chains of soy originating from Brazil.

Due to the explorative character of the interviews, the list of indicators is not exhaustive, but reflects the issues that the interviewed experts deemed relevant. Albeit the expert interviews did reveal general bottlenecks regarding soy chain performance on a number of sustainability indicators, it was not possible to clearly distinguish between the selected chains on all selected sustainability indicators.

Throughout the data collection it became clear that from the perspective of Latin America – EU agrifood trade, specific issues might need further in-depth analysis, beyond the level of expert interviews. The expert interviews on the qualitative indicators have therefore been supplemented by the stakeholders' interviews (with business stakeholders and representatives of NGOs) carried out between April 2012 and September 2012. The results of these interviews provide a basis for further in-depth analysis on the specific sustainability indicators through a literature review to better contextualize the results of the interviews.

5.4.4 Sustainability assessment indicators

The SAFA guidelines provide the most important topics regarding sustainability, after screening. In combination with the expert interviews, the most important issues, topics and sustainability bottlenecks regarding the soy chain were revealed. Moreover, the following sustainability indicators were withheld in order to do the sustainability assessment (table 35). A top-down approach was followed: experts and researchers defined the framework and the set of indicators. As can be seen from the table, a total number of 9 indicators were taken into account for conducting the sustainability performance assessment. Six out of 9 indicators were measured by means of an LCA, whereas the remaining 3 manifest ones were validated on a Lickert scale.

Table 35. Soy indicator framework for conducting sustainability assessment.

Dimension	Indicator*	Input	Verifier
Environment	Global warming	LCA	Carbon footprint
	Energy use	LCA	Amount of energy
	Biodiversity	Quantitative verifiers	
	Water consumption	LCA	Water footprint
	Land use efficiency	LCA	m²/year
Economy	Operating profit	LCA	$/ton
	Barriers for chain entry	Quantitative verifiers	
Social	employability	Quantitative verifiers	
	Working conditions	Quantitative verifiers	

* LCA data collection and execution conducted in the frame of EU-FP7 SALSA project by the business economics subdivision within the social sciences unit of Wageningen University.

Working conditions is a manifest variable in the sense that the topic is measured by means of other observations. Working conditions is measured e.g. by means of other measurable parameters in order to quantify the working conditions. For example: the amount of subcontracted work, access to protective gear, health and safety training, …

Therefore the indicators with a more qualitative nature are inferred and validated by means of parameters covering the bigger concept.

5.4.5 Results translation

As indicated before, it is not the aim to conduct a complete Life Cycle Sustainability Assessment. Such a transparent assessment (LCA) was already conducted in chapter 2 of this PhD dissertation. The aim is to translate these results into a usable format for a wide range of interested stakeholders.

This section elaborates about the aggregation of individual results, the basis for developing the sustainability performance assessment tool. The tool is meant for farmers/producers on one hand and agricultural organizations on the other hand. It allows to assess impacts of production processes for key indicators and compare the results against alternative production systems. It can serve as a self-reflecting tool but also as one in order to comply with sustainability demands from any customer of the product. Analysis is being conducted by means of a Life Cycle Integrated (or Sustainability) Assessment with regard to soymeal, exported from Brazil to a European harbor. This tool can serve as a baseline for extension to other products.

At first, one needs to set the boundaries of the impact assessment. The procedure for the tool is explained below:

The user can pick among four different variables to structure the analysis:

• Step 1: COUNTRY: the country where analysis can occur;

• Step 2: SYSTEM BOUNDARIES: the system boundaries to be analyzed: the system offers two different options, namely the *farm level* as such or the *whole chain*

• Step 3: IMPACT CATEGORIES: The type of impact category. There are in fact 2 types of categories: sub-categories and upper-level categories. The latter are able to synthesize, with regard to the sustainability dimensions in the FAO SAFA guidelines, the preceding sub-categories. All upper-level categories are qualitative, albeit derived from quantitative values.

• The type of indicator, either qualitative or quantitative. Outputs in a quantitative way are only possible for sub-categories. All qualitative categories derive inputs from quantitative inputs and output values of sub-categories (SALSA, 2014). The system allocates an equal weight to every category.

Step 4: TYPE OF OUTPUT: single quantitative value, single dimensionless value, quantitative value compared to results of alternative production methods. Dimensionless values derive from quantitative values benchmarked to conventional type of soymeal production.

5.4.6 Overall sustainability score

The illustration below shows the score calculation after selecting the option of OVERALL SUSTAINABILITY SCORE at farm level for an LCIA of soy.

The sustainability performance assessment score is in fact composed of multiple unique assessments. A couple of soy production systems are assessed across nine categories. The individual scores for each soy production system are then aggregated to calculate a global sustainability score.

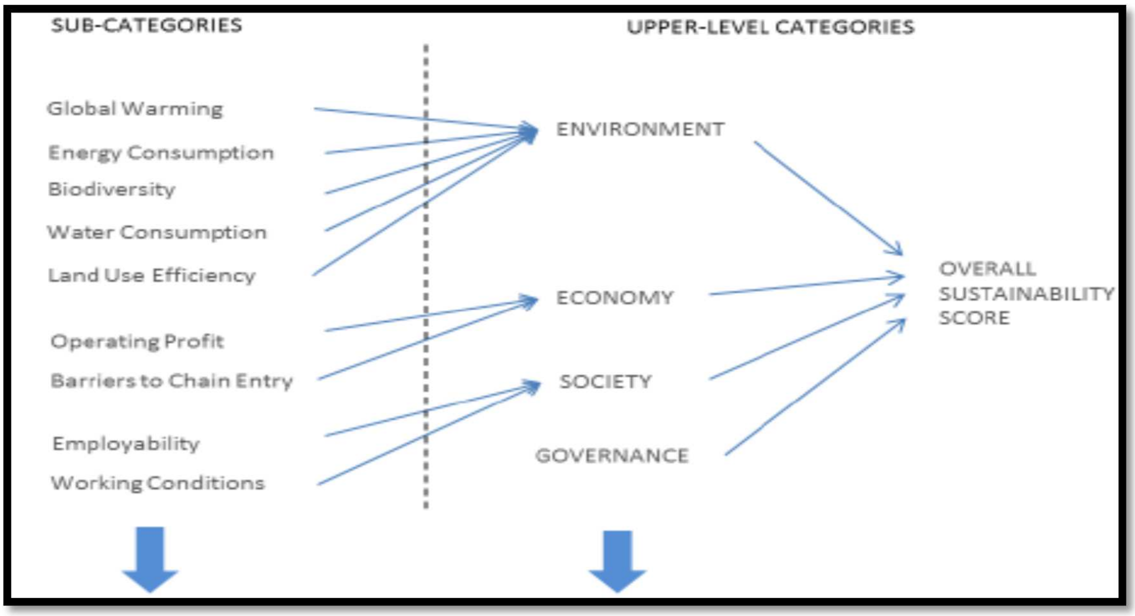


Figure 19. Sustainability score calculation structure.

(source: SALSA (2010-2014), with collaboration of ProQ)

Results of impact categories, by means of a weighted sum, is able to generate the score of the upper level category. A final qualitative score of the upper level category, involves a mathematical procedure to convert quantitative values into qualitative ones.

Each indicator has an objective measured outcome. As such, only the same indicators among different soy production systems can be benchmarked. Many authors, working on this topic, prevented the aggregation of results (Valdivia et al., 2013; Capitano et al., 2011; Traverso et al., 2012). However they leave the possibility open to aggregate indicator results within the same sustainability dimension, but not among sustainability dimensions.

Here lays the opportunity to come up with an overall one-metric sustainability score, aggregating objective scores within and across sustainability dimensions. Aggregation however, always implies a certain form of subjectivity depending upon the person in charge of the research. In constructing sustainability scores, one always encounters the problem of incommensurability, i.e. the lack of rules for weighting and aggregating data based upon the underlying scientific relationships. A common procedure implies the conversion of data from their original units to normalized units and subsequent aggregation (Ebert and Welsch, 2004).

It was opted not to follow the classic way of normalization. Most normalization involves ranging and standardization. This PhD dissertation applies more ad hoc rules with some degrees of freedom, as the selection of weights can occur depending upon the data and the analyst (Kumar Singh et al., 2009). In order to be able to sum up all scores, it is necessary to make them adimensional as they have a different unit of measurement. This is possible by converting each case outcome within each indicator, from an absolute score to a relative one. The latter can be done by applying relative 0-7 Likert interval scales. The higher the score on this scale, the better a production system chain is performing on this particular indicator. Zero presents the lower limit, and 7 the upper limit, which are in fact undefined ones. However, there is a risk of obtaining outliers (below 0 or beyond 7), when one is using fixed benchmarks. This can be solved by the following approach: the average of all measurements for a specific indicator is set equal to the neutral value of the Likert scale (namely 4). Each single observation is then weighted against this neutral Lickert scale value. If one has a sample size with more than 50 observations for this indicator, it can be seen as normally distributed. The mean of a normally distributed sample is hence representative. As such, a large sample size is necessary in order to avoid obtaining outliers.

An alternative methodology for identifying the ranking order is the Borda count. For the Borda Count Method, each alternative gets 1 point for each last place in the indicator received, 2 points for each next-to-last point place, etc., all the way up to N points for each first place vote (where N is the number of alternatives). The alternative with the largest point total wins, or in this case is most sustainable. The Borda count method has not been used in this PhD dissertation.

5.4.7 Tool analysis settings

In line with the first three steps in order to set up the analysis, below is illustrated how the final output is generated, by means of exemplary cases.

Case 1, LCIA soy, ANALYSIS SETTINGS:

Step 1: country: Brazil

Step 2: SYSTEM BOUNDARY: Farm

Step 3: ASSESSMENT CATEGORY: Global Warming (sub-category)

Step 4: OUTPUT: Quantitative value compared to results of alternative production methods

Figure 20 shows which specific input values (w.r.t. global warming) are requested in order for the system to conduct the analysis.

Figure 21 shows the output window.

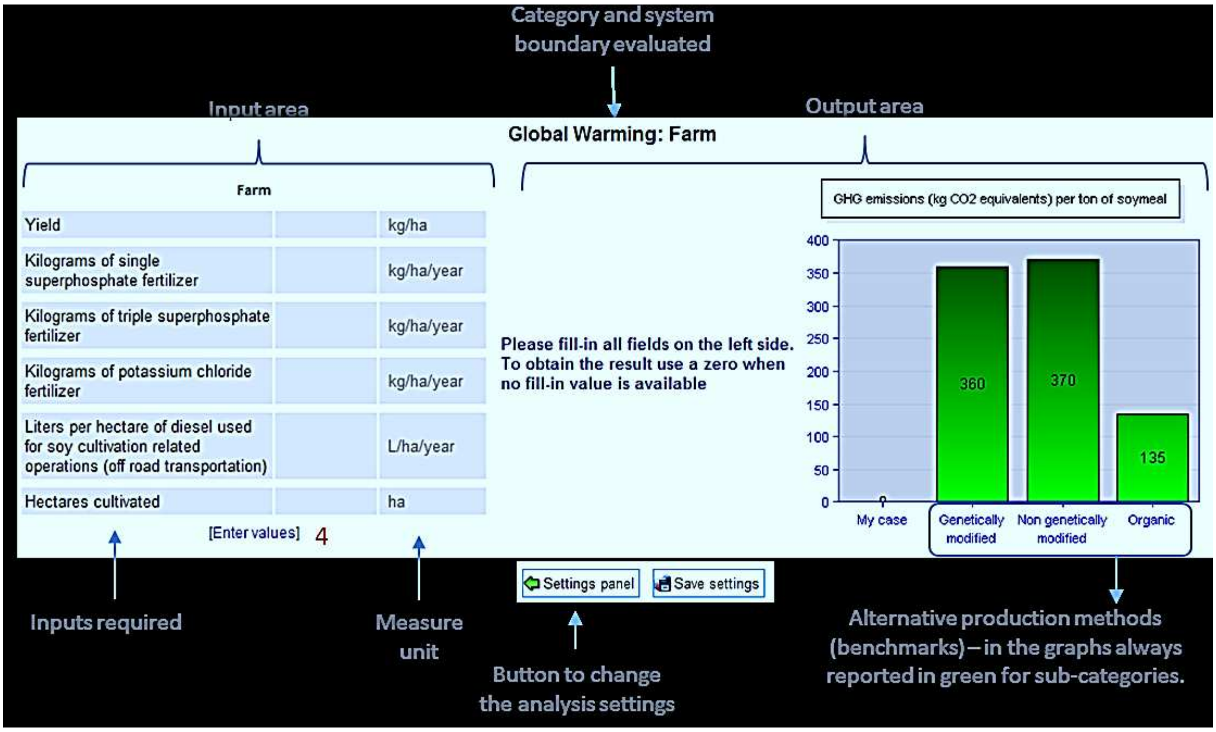


Figure 20. Input area window. (source: in collaboration with ProQ during SALSA project).

Step 5: Entering input values for the system to do the analysis (left side of figure 20).

Step 6:The output results appear on the right side of the input area and in the zone above my case (user of the tool).

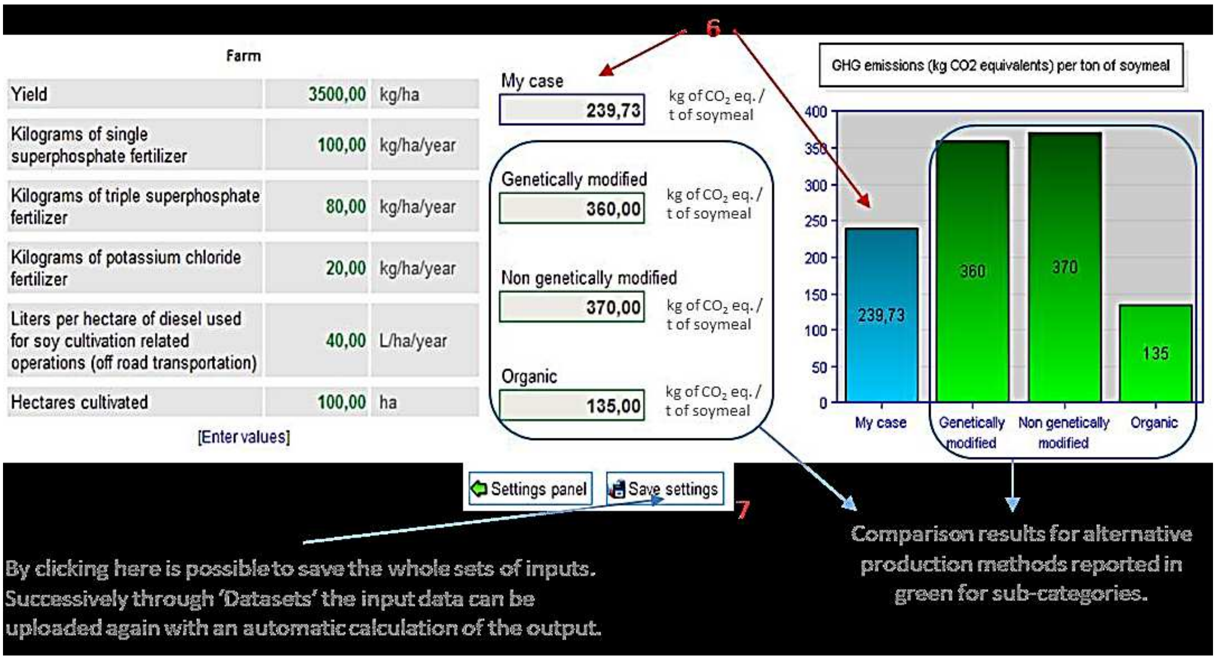


Figure 21. Output window and comparison results for global warming (source: in collaboration with ProQ during SALSA project)

5.5 Results and discussion

5.5.1 LCA results

In the course of the project, LCA data were collected on case studies of Brazilian farms. Table 39 gives an overview of the results on the selected LCA for the functional unit of 1 kg of soymeal, ending up in the Port of Rotterdam (EU). Hence, the whole chain is covered. The results of GM, non-GM and organic were measured with regards to the project. The "my case" observation was conducted by a farmer for testing the tool, with regards to the project. The application tool was designed in order to evaluate sustainability performance. Some parameters covered by the indicators, were requested to be inserted in the system. Only those parameters which have a significant impact on the indicator performance were selected. The my case outcomes are included in the results too, in order to supplement the number of observations.

Table 36. Objective LCA outcomes of sustainability indicators for soy chains. Results are given for the functional unit of 1 ton soymeal, unless stated differently. (The agricultural Economics Unit of the Bologna University, which was coordinating the project, collaborated with Wageningen, Gent University and the other partners involved in the data collection by contributing to the data collection and assessment quality and integration.).

Indicator	Production system			
	GM	**Non-GM**	**Organic**	**My case**
Global warming (kg CO_2 equivalents)	380	370	135	239
Energy consumption (MJ/kg)	6,775	6,854	3,677	7,039
Water consumption (L H_2O equivalents)	12021	11261	6889	10044
Land use efficiency (m^2/year)	1911	1911	2367	1964
Operating profit ($/kg)	15,167	9,295	9,227	11,791
Employability (hours/ton)	2,22	2,16	4,95	2,22

5.5.2 Qualitative assessments

The previous section highlighted results for the core quantitative indicators of the framework. The section below elaborates about the 'extra' qualitative indicators.

5.5.2.1 Biodiversity

The relation between soy production and the loss of natural fauna and flora is a very much debated issue in Brazil. There exists much controversy on the role of soy production in recent deforestation (and decreasing deforestation rates), especially in Brazil. The Soy Moratorium (Aguiar et al., 2011) is seen as a successful initiative and proof of the due diligence of the soy industry to tackle the negative reputation of the Latin American soy business and its impact on deforestation in the Amazon. Many interviewees regard the requirements of the Round Table on Responsible Soy (RTRS) standard as promising, especially the mapping project, in which the biodiversity of areas will be analyzed using satellite-verified maps with geographical indications for soy cultivation in order to identify areas eligible for land conversion and no-go areas. Nevertheless, the cut-off date of May 2009 is still a point of discussion for some interviewees, as it is seen as a weakening of the Soy Moratorium (cut-off date July 2006). According to the interviewed experts, the debate on soy production and deforestation is often limited to a discussion on the Brazilian Amazon.

Given the limited time and budget, Biodiversity was measured in an alternative way. It is seen as a 50/50 % share of two approaches. The first approach was that interviewees had to indicate the dosage of pesticides applied. A total of 12 different pesticides were presented (Opera, Talcord, Roundup, Engeo Pleno, Select 240CE, Flex, Boral, Aramo, Oberon, Nimbus, Aproach Prima and Priori xtra). According to literature, each of these pesticides has an active ingredient with a certain ecological impact (on biodiversity). Given the applied dosage in L/ha and the weighted ecology score, it is possible to obtain an ecological output-impact score. It needs to be emphasized that the higher the output-impact score, the worse the contribution to the environment. As such, organic had an output score of 0 (due to the organic's nature), GM 26 and non- GM 51. The my case observation was 27.

The second part comprised of 3 questions, with possible answers in a range linked to a Likert scale as follows:

1) Percentage of total arable area (i.e. excluding legal reserve) where natural or near-natural ecosystems and habitats are protected from human interventions? (depending on the region)

(1) 0-14% (2) 15-29% (3) 30-44% (4) 44-60% (5) 60-74% (6) 75-89% (7) 90-100%

2) Percentage of utilised area where a single plant species is grown, without rotation?

(1) 90-100% (GM) (2) 75-89% (3) 60-74% (non-GM) (4) 44-60% (5) 30-44% (6) 15-29% (7) 0-14% (organic)

3) Percentage of utilised area where no-tillage is performed?

(1) 0-14% (2) 15-29% (3) 30-44% (4) 44-60% (5) 60-74% (6) 75-89% (7) 90-100%

Table 37 provides the outcomes of part 1 and 2 of the biodiversity measurement. Part 2 delivers a Likert scale score on each of the questions. The average of the three questions was taken in order to obtain an overall score for part 2, and furthermore the average of part 1 and 2 for an overall score.

Table 37. Outcomes of the biodiversity sustainability assessment

Production system	Score part 1	Score part 2	Overall biodiversity score (0-7)
GM	4,13	3,33	2,91
Non-GM	0	3,33	1,67
Organic	7	6,33	6,67
My case	2,49	3,33	2,91

5.5.2.2 Barriers for chain entry

This indicator provides an insight on the extent to which new participants are able to enter the chain, whether there are thresholds for entering the market. It has been perceived through the interviews, that mostly smallholders suffer from problems related to these barriers. Despite the growing crushing volumes and capacity in Brazil, transportation costs remain relatively high in Brazil compared to Argentina, where soy production areas are very well connected to the major exporting ports. In Brazil, the high costs and environmental impact of transportation are still very much debated, as is illustrated by the statements of a Brazilian respondent: “The difficulties in transport mean that producers are looking for land closer to the ports.” When talking about the port in Paranagua, Para state, Brazil he states: "*This port was a magnet for producers and hence deforestation*."

Salin and Ladd (2012) argue that the competitiveness of the Brazilean soybean industry in particular, largely depends on its transportation costs. Hence, without proper transportation infrastructure, distant areas far from ports remain less economically interesting for soybean cultivation. Similarly, Altieri and Penque (2006) refer to the massive infrastructure projects required for transporting the soy from the production areas to the exporting harbors. This infrastructure, in its turn, attracts environmentally damaging activities by opening up large areas of natural land. Hence, the strategic location of ports may influence the rate and extent of land use change. All should be seen in the light of Industrial Organization (a field that builds on the theory of the firm by examining the structure of and the boundaries between firms and markets). This is used as starting point to say something about market relations in an industrial environment. For this reason, Chain entry barriers is measured by means of 3 questions in relation to Industrial Organization::

1) Cost for compliance to extra-legal chain requirements (auditing, monitoring, infrastructure, logistics, traceability): Estimate the period for total recuperation of investments (break-even point)?

(7) less than 1 year (6) 1-3 years (5) 3-6 years (4) 6-9 years (GM large) (3)9-12years (2) 12-15 years (1) more than 15 years

2)Please indicate the number of actual buyers for your soybeans? (the higher the better; if there is only one or a few buyers, then it might enlarge the chances for wrong market relationships (exploitation of market power through monopoly, pricing, etc.)

3) Please indicate the number of (potential) buyers for your soybeans? (the higher the better)

Table 38 provides the outcomes of the chain entry barriers assessment. The individual scores were obtained by summing up the three Lickert scale answers, and dividing them by three. The higher the individual score, the better the system is performing for this specific indicator.

Table 38. Overall score for the indicator chain entry barrier

Production system	Score
GM	6,8
Non-GM	4,1
organic	1,7
My case	3,1

5.5.2.3 Working conditions

While none of the interviewed experts was aware of violations of human rights in the soy sector in Brazil, many of them argued that there is a huge absence of clear data and statistics regarding working conditions (overtime, working hours, safety measures, etc.) and the implementation of labor rights in Brazil. According to European experts the soy sector in Latin America is characterized by increasing concentration of land, capital and power and by the commodification of rural labor (Severi and Zanasi, 2013). Men dominate in the primary sector and the interviewed Latin American experts estimate the wage gap between temporary and permanent employees to be significant.

Working conditions were analysed similarly:

1) Percentage of work being sub-contracted ?

(1) 90-100 % (2) 75-89 % (3) 60-74% (4) 44-60% (5) 30-44% (6) 15-29% (7) 0-14%

The following questions (from 2 to 7) get a response as follows: +1 if it is yes, 0 if respondent does not know and a minus 1 if the answer is no. The latter as a sort of correction in the answers.

Yes (+1), I don’t know (0), No (-1)

2)All workers are able to join a trade or labour union when wanted?

3) All workers have access to protective gear ?

4) All workers get training on health and safety issues?

5)All workers receive, at minimum, wages and benefits that meet national legal or industry benchmark standards?

6)Working hours are documented and overtime is compensated?

7)All workers have a legal contract?

Table 39 provides the outcomes of the working conditions assessment. The individual scores were obtained by 2 parts: first part was the simple answer on a Likert scale to the question. The scores on the second part were obtained by summing up the answers and add 7 (range Likert scale) to the result, divided by 2. The average of part 1 and 2, delivered the final score. The higher the individual score, the better the system is performing for this specific indicator.

Table 39. Relative scores on the working conditions indicator.

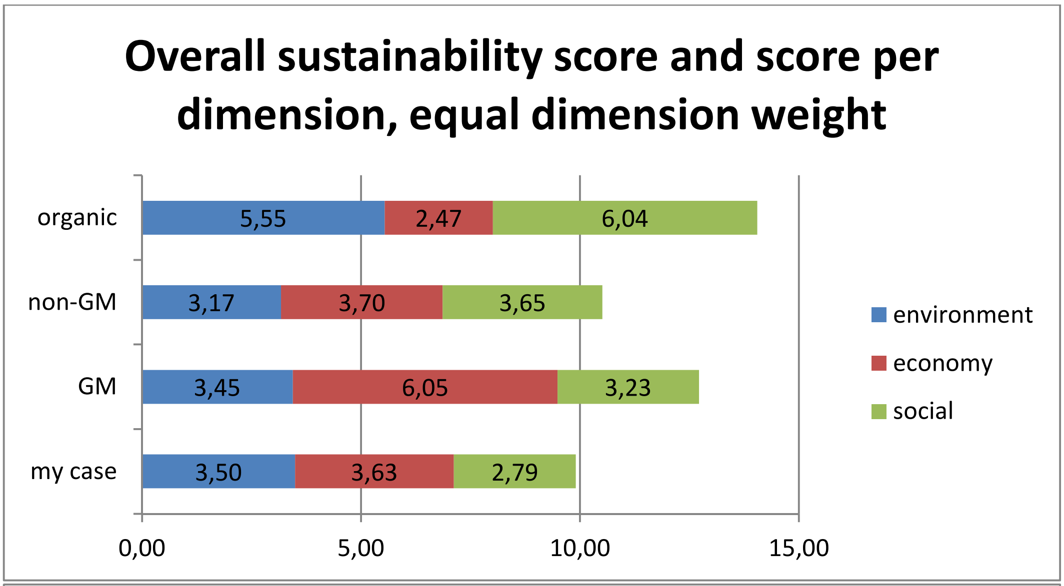
Production system	Score part 1	Score part 2	Final score
GM	2	4,8	3,4
Non-GM	4	4,6	4,3
Organic	6	4,5	5,2
My case	1	4	2,5

5.5.3 Overall sustainability score

In order to get an overall sustainability score, individual scores need to be aggregated within and among sustainability dimensions.

It is important that all three sustainability dimensions have equal weights in the analysis, namely 1/3. Weights can differ among stakeholders (see chapter 4). However, in order to obtain objective and scientific sound results, it is opted to allocate 1/3 of the weight at the dimension level. The latter implies that environment, social issues and economy are considered equally important. Moreover, results were obtained by adding up all individual indicator scores per dimension, and divide this score by the number of applied indicators within that dimension (5 in environment, and 2 each for the other).

Figure 22 provides 2 different outputs in terms of overall sustainability scores. The upper one allows to compare within and among sustainability dimensions. The lower one allows to compare the overall sustainability scores between dimensions.



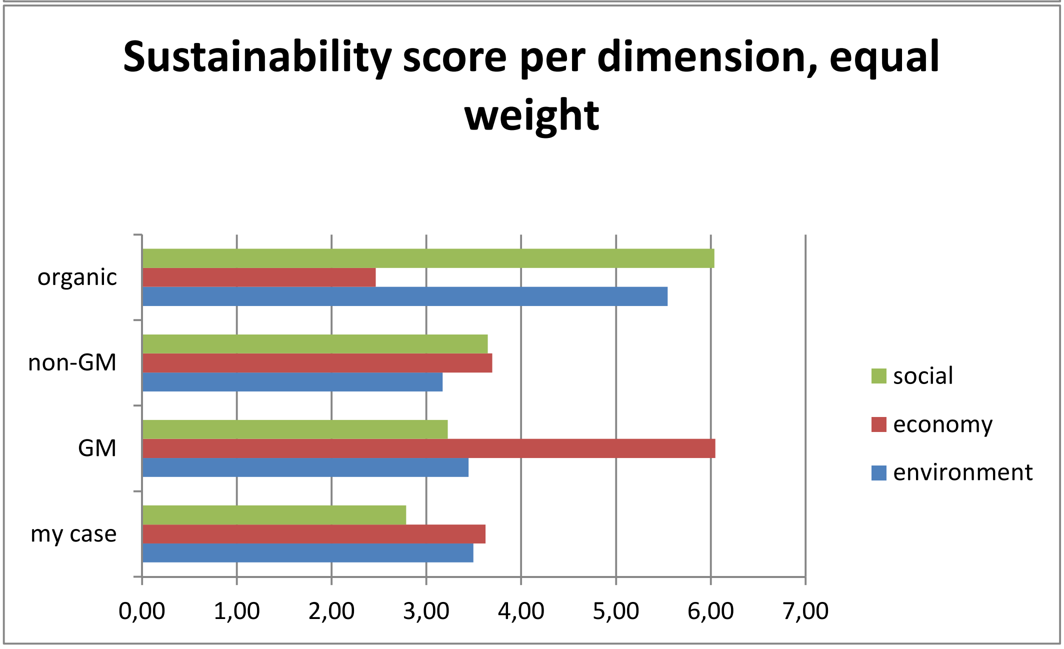


Figure 22. Overall sustainability scores for different soy production systems.

Scores are relative (not absolute) and should hence be interpreted this way.

It can be seen from the graphs that for the examined cases in Brazil, organic is obtaining the highest sustainability score for the withheld indicators, followed by GM and non-GM. Looking at the individual dimensions, GM is most profitable, and organic most environmental friendly. To some extent this can be expected beforehand, however now a tool provides insights across dimensions.

5.5.4 Inclusion of qualitative indicators ?

It is interesting to identify whether, the addition of indicators with a qualitative nature is necessary in the whole sustainability assessment. How necessary are they in fact, based on these case studies ? Therefore it is important to reveal the difference and identify the impact on the scores of adding these type of indicators to the sustainability assessment, before and after. The scores including both type of indicators were mentioned above. Subtracting the scores of the 3 manifest indicators, delivers the sustainability score without the manifest indicators. It is therefore interesting to see what the impact on each individual sustainability dimension implies. In fact, it is the aim to identify the difference with and without the manifest indicators: $x2 - x1 = \Delta x$. If Δx is > 0, the removal of the manifest indicator has a positive impact on the score and the opposite when Δx is <0. Figure 23 shows the impact on each sustainability dimension after removal of the manifest indicators.

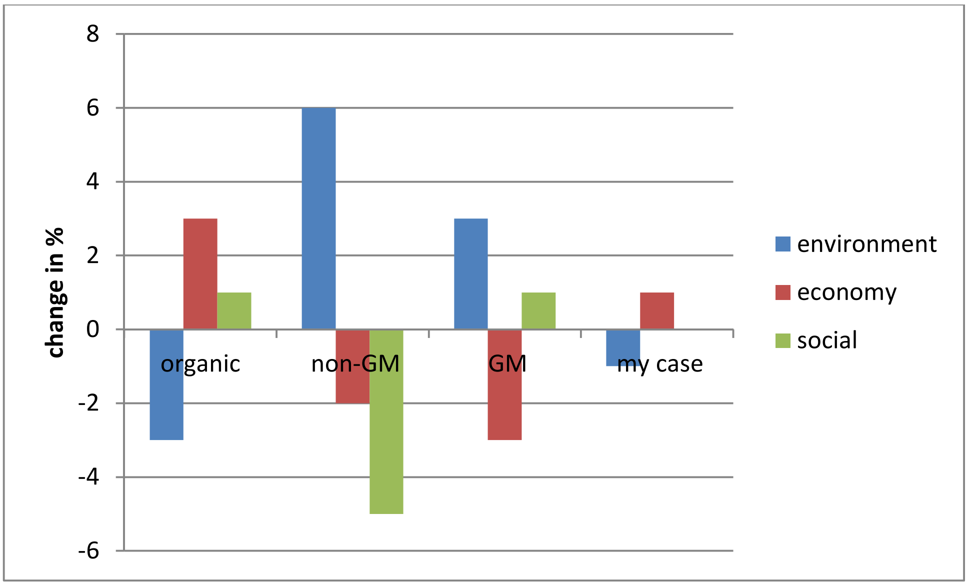


Figure 23. Sustainability assessment without the non-quantifiable indicators: percentage change in the dimensions.

The main results remain: organic keeps the highest sustainability score in the observations, followed by GM soy production system. Non-GM becomes last, after swapping with my case. Removing 1 indicator in economy appears to have an impact on the overall sustainability performance. Removing the biodiversity indicator, has the biggest environmental impact on the non-GM production system in a positive sense. At the same time, the social dimension of NON-GM loses about 5 % of share, and the economic one 2 % in the overall sustainability score. GM improves for the environmental and social dimension, but loses share in the economic dimension. Organic increases in economic and social dimension, but decreases in environmental dimension. One can see that mitigation measures for instance, might have a positive impact on a certain dimension, but at the same time harm another one. Therefore it is always a matter of trade-offs when applying mitigation measures.

As Binder et al. (2010) indicate, agricultural sustainability assessments have the intention to mainly focus on environmental issues of a system, putting aside economic and social aspects. Hence these assessments fail to take into account the multidimensionality of the system. It raises the question of what issues of the system are prioritized.

At first, it is important to note that case studies provide useful information in terms of contributing to knowledge building. However, at the same time, more cases are needed in order to underpin the necessity of adding manifest indicators to a sustainability assessment. The overall results only showed a change in scoring for 2 production systems. Adding these manifest indicators hence does have an impact, however the number of cases is too limited to draw general conclusions. More demonstration cases are needed in order to identify a statistical significant difference between adding manifest indicators or not.

Literature indicates the use of Life Cycle Sustainability Assessment (LCSA) for assessing the sustainability performance. LCSA is a good basis for integrating all dimensions, however excludes manifest indicators.

In literature results are always presented separately for the different sustainability dimensions. This however does not allow a comparison among sustainability dimensions. There are however 2 levels of aggregation: intra- and interdimensionsal. Regarding the first level, authors do not elaborate nor recommend on how to proceed on the individual techniques and pretty much endorse the way the practitioners have been doing: aggregation of E-LCA and LCC indicators and presentation of individual indicators concerning the S-LCA possibly except for few indicators (Valdivia et al., 2013). Authors agreed that if it is already controversial to aggregate on S-LCA indicators, and considering that S-LCA and LCSA are at the early stage of implementation, they discouraged the 2nd level of aggregation. However, one is confronted with possible trade-offs between concerns regarding sustainability (as proven by figure 22). This is important for decision makers who want to overcome bottlenecks regarding a process or product. Suppose a product is very environmental friendly, but at the same time socially unacceptable (Franze and Ciroth, 2011), then decisions need to be made according to trade-offs between sustainability dimensions.

The literature approach does not allow cross comparison and hence confronts the user with trade-offs between sustainability dimensions. Trade-offs can be underpinned by involving stakeholders of any kind along the chain. Each of them has a different perception or even preference towards sustainability. A technique like Multi Criteria Analysis (MCA) could provide useful information on how to allocate weights to the different sustainability dimensions in general, or the specific indicators in particular. This can help decision makers to address certain weights to the dimensions and indicators. Moreover it could help them in the decision making processes regarding certain existing artefacts (e.g. positive environmental score, negative social score). Perceptions and preferences however, remain subjective and are not an objective view on the facts. Perceptions also might differ from stakeholder group to stakeholder group. Value chain actors might perceive sustainability in a different way than workers or local communities. Therefore it depends upon the decision makers which group they tend to follow. To some extent this deviates from the objectiveness and tends to lead to a bias.

According to Finkbeiner et al. (2010) it is suggested neither to weight indicators, nor to aggregate results in a more intuitive way. The usual decision making process however makes use of both approaches to tackle the difficulty of the gathered results. More cases and experiences would allow to reveal the needs, bottlenecks and utmost ways to communicate sustainability assessment results. Traverso et al. (2012) indicate that the use of the LCSD (LifeCycle Sustainability Dashboard) approach is very useful in communicating sustainability performances of similar products. However, this technique still needs to deal with trade-offs, which is tackled in this chapter by aggregating the scores. Moreover aggregation of scores, with a clear view on each separate sustainability dimension score, allows to clearly see the impact of mitigation measures on the individual dimension on one hand and the overall sustainability score on the other.

Having to deal with trade-offs among sustainability dimensions however, can be excluded. This can be overcome by aggregating all sustainability dimensions in an overall score. It allows to cross benchmark between products and processes and hence draw conclusions regarding the overall sustainability score. Perceptions are then excluded and an overall objective score supports policy makers in their decision making processes, without having to deal with subjective perceptions. However, it should be noted that each dimension is equally contributing to sustainability: a weight of 1/3 each, being in line with the general definition of sustainability where each dimension is equal in weight (triple bottom line). At the same time, in order to evaluate the sustainability performance increase of certain underlying activities, it is still necessary to go back to the original list of dimensions: so two scenarios or products with the same sustainability index have different meaning for the stakeholders, so they can choose between those solutions based upon their preferences.

Literature studies focus more on aggregating results within sustainability dimensions, but not across. This thesis goes beyond this, firstly by adding manifest indicators to the assessment besides the classic LCA and secondly by aggregating results within and among all sustainability dimensions (intra- and interdimensional). As such, aggregation remains subjective as the practitioner addresses weights to dimensions and indicators. Full economic aggregation has been suggested in literature as one of the approaches. It implies expressing everything in monetary terms. However, one needs to make too many assumptions in order to convert environmental indicators to economic units for example.

So far, an overall and globally accepted set of indicators of the LCSA on one hand and for the sustainability framework on the other, has not yet been identified. In particular the social issues need more attention, as there is no agreement among scientists regarding an accepted set of indicators. The SAFA guidelines provide useful information in terms of agricultural products and processes, however are not applicable for other sectors. Therefore a general accepted set of social indicators would help to approach sustainability assessment in a uniform way across products, processes and countries. Moreover, using the same set of indicators allows fairer benchmarking as well.

The use of semi-quantitative and semi-qualitative information is a challenge when one wants to aggregate different type of data along the life cycle in LCSA (Valdivia et al., 2013). This chapter however has proved that it is possible to come up with an extended sustainability framework on one hand, and aggregate results in a clear way to obtain an overall score on the other. At the same time this has an impact on the communication of results, having to deal with different sustainability dimensions and stakeholders.

At first sight, adding manifest indicators seems necessary in conducting a sustainability performance assessment. The impact on the overall score is clear. However, given the small sample size of cases, it is hard to draw general conclusions. Further research and more case studies might be able to help assessing an integrated sustainability performance in a better way. The first steps in providing an overall sustainability assessment score have been made. Further research needs to reveal links between indicators across dimensions, which have to be included when combining the three dimensions of sustainability.

It could also investigate the impact of using quantitative and qualitative indicators in a fifty fifty approach, in order to reveal the real impact of applying one or the other indicator. More cases allow a statistical approach in order to reveal significant differences between those cases in terms of indicator performance. Moreover, one can clearly identify the impact of sustainability mitigation measures or improvement scenarios on the overall sustainability performance.

The downside of using indicators can be approached as follows: regarding agricultural sustainability assessments, data are scarce and sometimes lack quality. By using variables over indicators, there is a more dynamic approach regarding data inputs. Assessments of agricultural sustainability typically do not include interactions among variables or indicators. A lot of them describe sustainability as the linear outcome of multiple, semi-aggregated, weighted indicators (e.g., B. Hansen et al. 2001; Rigby et al. 2001; Taylor et al. 1993).

5.6 Conclusions

Sustainable development requests scientific sound techniques to better understand the product systems and their lifecycle. The findings from the Brazilian case studies, allowed to improve the existing LCA frameworks, in combining three dimensions of sustainability and adding manifest indicators. Organic soy production in our case obtained the highest sustainability score compared to other examined production systems. In spite of resulting in the most sustainable farming system, organic soy is still absolutely a niche segment of the overall soy export market in Brazil and, mostly, in Argentina. Concerns on the high risk related to its cultivation, complexity of obtaining a certification and, mostly, to the still non remunerative price for exported organic soy, suggest that the economic sustainability should be better investigated, also adding the institutional dimension (transaction costs) to the integrated sustainability assessment. More cases are however needed to draw conclusions on the sustainability scoring on the one hand, and to identify whether including manifest indicators has a significant impact on the other hand. The preliminary results suggest they do, however hard to draw general conclusions. Moreover, measurements were done at some point of time in some region, implying a snapshot. In order to draw thorough conclusions, one needs panel data to see the trends and evolutions over a longer period of time (monitoring tools).

The combination of three sustainability dimensions and comparison across them, allows to identify the impact of mitigation measures on both the individual dimensions and the overall sustainability score. Improvement options in one dimension, might harm another one and might cause a worse overall sustainability score. Sustainability performance assessment hence plays a crucial role in a lot of decision-making processes: at company, policy or consumer level. The overall score can help in making decisions which is facilitated by aggregating the scores and avoid having to make trade-offs.

A common set of social indicators cannot be provided at this point of time. There lays an asset for future research. In literature it is suggested that aggregation and weighting steps are not the best strategy to derive an overall sustainability score. This paper proves however, that it is feasible. It should be noted in this perspective that an aggregated stand-alone result does not make sense. Lancker and Nijkamp (2000) clearly state that " a given indicator does not say anything about sustainability, unless a reference value such as tresholds is given to it." It proves that benchmarks are needed to help identifying gaps, trends and improvement options. Aggregated scores can hence help in disseminating and communicating the results in a straightforward way.

This work was conducted with regards to an EU-FP7 project called SALSA, with the aim to assess the sustainability performance of soy and beef chains from Latin-America to Europe. The first steps related to an extended sustainability assessment have been done, contributing to the knowledge. The application seems feasible as a starting point, but needs extension with more cases. Sustainability assessment is an iterative and evolving process and our input can contribute to that.

6 General conclusions

6.1 Discussion

6.1.1 Sustainability and economy

The aim of the discussion section would be to first discuss sustainability with regard to the economy, and second to critically evaluate multi-pillar models regarding sustainability.

Economy is the total product of a society's activity for producing services and goods in order to meet the demand of people. Markets with given clearing rules determine the behavior of actors during the economic process, including rules for how to use the scarce resources in an efficient way.

Costs and prices in the economy contribute as a measuring instrument for using scarce resources. Providing identical service at a lower cost, means a more efficient and economically solution, using less resources.

Things are different with regard to the environment. Humankind can to some extent freely use natural resources, which results in ecological damages, so called external costs for the environment. These costs are not charged to the causer but to third parties (general public and also future generations). In a market system it is the aim to fully account for resource use and to utilize efficient market clearing rules. This shows the need to internalize the external environmental costs as much as possible (Voß, 2001). However, a quantification problem arises upon the internalization of external cost elements.

Cost-benefit analyses are a common approach to assess the pros and cons of policy measures for environmental issues. Many cost-benefit simulations for climate policy were conducted the last decades. These simulations have the aim to study for instance the relationship between global warming and economic growth. A well-known model is DICE (Dynamic Integrated Model of Climate and the Economy) developed by Nordhaus(1994). This model indicates the economic impact of global warming as a percentage of the Gross Domestic Product (GDP) (Nordhaus and Boyer, 2000). It is however hard to verify and judge the validity of the external cost calculation in the DICE model, as it is based upon rough estimates of the economic impact on several economic sectors (Nordhaus, 1994; Tol, 2002; Fankhauser, 1995).Simulation results show that global warming will account for 10 % to 40 % of all external costs in the 21st century. Land use and its changes will be responsible for the remaining cost. The internalization of the external cost will cause a decline in economic growth by approximately 5% (Kosugi et al., 2009).

Defining sustainability indicators is a challenge with several difficulties which need to be precisely clarified in order to create practical and mathematical sound indicators.

A first challenge relates to the unit of measurement. Problem is that the sustainability dimensions do not have a common unit of measurement and sometimes it is not clear at all which unit is being used. Profits are measured in a currency. What is social capital measured in? What about environmental or ecological health? Finding a common unit of measurement is one challenge.

Some authors plead for monetizing all the dimensions of the TBL, including social welfare or environmental damage, which has the benefit of having a common unit – a currency. At the same time many authors object to put a monetary value on wetlands or endangered species on strictly philosophical grounds. Others question the method of finding the right price for lost wetlands or endangered species. Monetization is challenging for non-economic parameters, otherwise the market can determine price and costs for economic parameters. A similar problem emerges with regard to the sustainability index, as the dimensions have different units of measurement.

Another solution for calculating the TBL is to calculate it in terms of an index, a composite indicator. This eliminates the incompatible units issue. A universally accepted accounting method would allow comparisons between entities, e.g., comparing performance between companies, cities, development projects or some other benchmark.

There are plenty of examples of indices: the Human Development Index, the national intellectual capital Index, Gini Index, …

There remains some subjectivity even when using an index however. It remains a question how the index components are weighted. Would each "P" get equal weighting? What about the sub-components within each "P"? Do they each get equal weighting? Is the people category more important than the planet? Who decides? That is when the social norms of society and politics become important (see chapter 4).

In order to avoid the monetizing problem and the index problem, it is better to use just each individual sustainability measure from a monitoring perspective to evaluate the progress over time (decay, status quo or positive progress). This brings the assessment on the level of individual dimensions and its indicators (e.g. carbon footprint).

An accepted universal standard method for calculating the TBL is still missing. Neither is there a universally accepted standard for the measures that comprise each of the three TBL categories. This can be seen as a strength because it allows a user to adapt the general framework to the needs of different entities (businesses or nonprofits), different projects, policies or different geographic locations.

Albeit that a common accepted set of indicators is not defined in literature, assessing sustainability by indicators is well established (Dahl, 2012), acknowledging operational bottlenecks (Gomez-Limon and Sanchez-Fernandez, 2010). Most problems occur upon the interpretation of a whole set of indicators and thus not providing the best policy information. Such approach is not able to assess the trend of changes and revealing causes. However, life cycle thinking is very reliable in supporting decision making for the assessment of the sustainability performance of products (Valdivia et al., 2013). Here, graphical representations of a Life Cycle Sustainability Dashboard (Traverso et al., 2012) are ways to communicate with experts and others for better decision-making processes. Policy-makers have no time to bother with in-depth studies. There is a request for compiled and packaged information to exhibit decision-making. Composite indicators are very suitable in summarizing multidimensional issues in order to support policy-makers. Moreover, they also facilitate communication to the public, media and other parties with the aim to strive for accountability. They are an aggregated index, composed of the individual indicator based on an underlying model. Composite indicators are more and more seen as useful tools for assessing sustainability (Esty et al., 2005).

Many tools, mostly monetary ones were used for evaluating sustainability in several fields and all approaches have both advantages as well as disadvantages. A lot of tools might be complementary but are unable to grasp the whole picture (Gasparatos and Scolobig, 2012). Moreover, during the aggregation step information gets lost in composite indicators and cost benefit analysis. Composite indicators allow trade-offs between sustainability issues (Böhringer et al., 2007). Such methodological assumptions are no major limitations, but prove that prudent consideration should be taken upon the construction of indices.

Composite indicators are popular tools for sustainability assessments at various scales (Krajnc and Glavic, 2005). They can follow a participatory approach by involving stakeholders during the steps of an index construction (indicator selection and weighting measurement), are flexible to quantify a huge range of issues on economics, environment and are able to conduct an integrated assessment. A composite indicator is able to summarize multidimensional aspects in providing a precise picture (Saisana et al., 2005) and is able to evaluate sustainability performance (Singh et al., 2007), to help in setting policy priorities and monitoring performance (OECD, 2008) and provides simple communication and interpretation to the public (Kondyli, 2010). On the other hand, if composite indicators are constructed in a bad way, they might show misleading policy messages or may be misused to underpin a desired policy. Constructing a composite indicator is difficult and requests a framework with a representative set of indicators. Compensation between dimensions can be seen as a form of weak sustainability. It indicates that a certain proposal (or production system in this case) can overall be considered positive as long as the net assets are not degraded (Neumayer, 2003). A project may thus have a positive outcome in 1 dimension, and negative outcomes in the other two. As long as the overall (net) outcome is positive, then negative impacts in other pillars are acceptable.

Integrated assessments are actually derived from Environmental Impact Assessment (EIA) and Strategic Environmental Assessment (SEA), but have been extended to include economic and social considerations, showing a Triple Bottom Line approach towards sustainability (Pope et al., 2004). The aim of integrated assessments can either be to reduce and even minimize unsustainable practices or to reach Triple Bottom Line objectives (Pope et al., 2004).

Most theoretical literature on sustainability assessment emerged from work done by practitioners in the field of EIA and more recently SEA. (Sheate et al., 2001, 2003). Sustainability assessment is considered to be the next generation in environmental assessment

(Sadler, 1999). EIA and SEA provide a decent basis which can be extended to include wider sustainability concerns (Gibson, 2001; Verheem, 2002).

In fact, distinction can be made between EIA-driven SEA and objectives-led SEA. An EIA-driven SEA is typically a reactive, ex-post process aiming to evaluate impacts of a plan or programme versus a baseline. Moreover, it has the aim to evaluate the acceptability of impacts and at the same time reveal possible adjustments for improving environmental results (Sheate et al., 2001, 2003; Sippe, 1999, Devuyst, 1999).

Literature also mentions the objectives-led SEA. This has the aim to test whether a certain action or programme is able to reach environmental objectives, rather than against a baseline (Sheate et al., 2001, 2003; Smith and Sheate, 2001; Twigger-Ross, 2003).

Broadening environmental processes with the inclusion of the three pillars of the triple bottom line can occur in both, the EIA-driven SEA and the objectives-led SEA for obtaining an integrated assessment. Both approaches are examples of sustainability appraisal (Sheate et al., 2001), integrated sustainability appraisal (Eggenberger and Partidario, 2000) or integrated impact assessment (Sheate et al., 2003). At the same time Lee (2002) applies sustainability assessment which equally refers to EIA-driven and objectives-led integrated assessment. Integrating environmental, social and economic aspects is what Scrase and Sheate (2003) refer to as ‘integration among assessment tools’. Integration in that sense entails that an integrated assessment should be more than a summation of individual and separate environmental, social and economic assessments. Eggenberger and Partidario (2000) state that it is widely acknowledged that the sum of parts does not equal the whole (Pope et al., 2004). Furthermore, it is suggested that integration in fact implies that a new entity should be created and new relationships established.

Against this backdrop, in this PhD dissertation it was opted for to start with a sustainability assessment tool related to the environmental dimension of sustainability in general and the climate change indicator (Carbon footprint) in particular. The prime aim was to identify mitigation measures by which negative effects can be reduced or eliminated. This is confirmed in literature by George (2001). At the same time the carbon footprint tool can be used in future sustainability assessments. A baseline value is formed and future impacts can be monitored and eventually benchmarked with the baseline. The carbon footprint approach can be seen of as ‘direction to target’ where the exact position of a sustainable state is unknown. It is however known that the target of 20 % greenhouse gas emission reductions compared to 1990 should be achieved by the year 2020. The carbon footprint tool thus can aid in monitoring the progress towards this state, by showing direction to the target and minimizing adverse effects.

This PhD dissertation has provided three different tools (carbon footprint, household waste and integrated sustainability assessment) to aid in future sustainability assessments for three different cases. From this perspective, the outcome of the tools provide a baseline. Sustainability assessment thus implies that the assessment can start after the baseline is set in order to assess what the implications of certain initiatives bring about on sustainability.

A second tool concerns household waste. This tool indicates the probability or chance for reaching a certain waste target in a specific municipality. It will help decision makers in gaining insights on the chances for reaching certain waste targets, Compliance with certain factors enabling target reaching (see chapter 3), can aid decision makers in gaining insights on the chances for reaching certain waste targets. Monitoring over time can help municipalities and supralocal joint ventures to evaluate the progress towards the waste target. The amount of waste can be monitored, but also the probabilities of reaching a certain waste target. The logit model used has an added value in the sense that it calculates a chance of reaching a certain policy target. So, one can see whether an event occurs (how much waste) but at the same time can see the probability of occurrence. This is much more interesting on the policy level as decision-makers can use this tool to see what the chances are of reaching a certain waste target, upon compliance with certain determining factors potentially decreasing waste generation. Reaching the target can thus easily be communicated by means of the Logit tool.

Sustainability assessment should however not only have the aim to minimize negative impacts (CO_2 and waste), but should also try to encourage positive steps (Gibson, 2001). The latter means a more proactive approach with a ‘direction to target’ characteristic, however yet again the position of the sustainable state is unknown (Pope et al;, 2004).

EIA-driven assessment approaches help decision-makers in deciding whether Triple Bottom Line Impacts are acceptable with the focus on minimizing negative impacts. Objectives-led assessment move further and help in assessing whether a certain project or proposal is able to positively contribute to goals of the Triple Bottom Line. Both methods can be perceived as ‘direction to target’ approaches.

The third tool in this PhD dissertation with regard to commercial soy production must be seen in the light of an objectives-led assessment. There is however no unanimity what sustainable soy production precisely entails. Therefore there is a need for a baseline, to identify potential objectives for the Triple Bottom Line pillars.

Against this backdrop, it is clear that measuring sustainability can either be done from a monitoring point of view (EIA-driven assessment) or an integration of the three pillars of the Triple Bottom Line.

6.1.2 Evaluation of three-pillar model

The constituting aspects of sustainability, being its dimensions economy, ecology and society, are used to conduct a quantitative analysis of the Brundtland Commission’s concept and to reduce its arbitrariness.

However, the three-pillar model has shown to be helpful for quantitative assessments within certain limits. The limited applicability is due to the fact that independent sustainability goals are set for each dimension separately. This is contradictory against the original integrated approach of the concept. The three dimensions approach has the benefit of being able to exhibit the problem area. At the same time it will not be possible to deduct concrete sustainability goals for one dimension independent from the other two.

Goals of sustainable development incorporate a decent economic growth in order to meet basic needs and aspirations for a better life with regard to a growing world population. In many EU countries, economic growth is a prerequisite to accomplish important social aspects. One can think of financing the social security system. Economy might therefore be the best means to meet the goals of society (efficient economic activities).

Additionally, the TBL is able to be case specific or allow a broad scope. The level of the entity, type of project and the geographic scope will drive many of the decisions about what measures to include. The set of measures will ultimately be determined by stakeholders and experts and the ability to collect the necessary data. While there is significant literature on the appropriate measures to use for sustainability at the state or national levels, in the end, data availability drives the TBL calculations.

6.2 Answering the research questions

Regarding the several addressed sustainability assessments, this PhD dissertation covers three major research questions. They were formulated in a broad way, implying that specific sub-questions were needed. Hereunder, answers to each research question and their sub-questions is provided, based on the results presented in the different research chapters.

RQ 1. Which determining factors making up the carbon footprint with regard to beef and pigmeat production in Flanders are – upon mitigation – able to contribute to progress towards sustainability ?

RQ1A. What is the level of carbon footprint in absolute terms for beef and pigmeat in Flanders ?

RQ1B. What are the hotspots (highest impact production stages) with regard to carbon footprint for beef and pigmeat in Flanders ?

RQ1C. What is the impact of changes in feed and herd characteristics on the carbon footprint of pigmeat and beef in Flanders ?

RQ1D. What are the opportunities to reduce the carbon footprint, based upon the hotspots found for pigmeat and beef in Flanders ?

Several functional units along the lifecycle of the meat production chain of pigmeat and beef were defined. It was found that 1 kg of deboned pigmeat creates a carbon footprint of 5.7 kg

CO_2 equivalents, whereas 1 kg of deboned beef has a CF of 22.2 kg CO_2 equivalents. The single CF figure should however be used with caution, since it is based upon specific input data. Presenting a range in which the CF is expected to fall, provides a much more robust result (Flysjo et al., 2011b). This can be seen in light of a sensitivity analysis in which several parameters are fluctuated. Based on this analysis, the estimated CF of pigmeat production in Flanders is expected to lie between 5.5 and 5.9 kg CO_2 equivalents per kg of deboned meat. For beef the CF would lie between 22.2 and 25.4 kg CO_2 equivalents.

With regard to pigmeat, fodder production contributes about 64 % to the CF. Manure storage accounts for 25 % of the emissions. With regard to beef, fodder production is responsible for almost 30 % of the CF. Albeit only 13 % of the fodder is purchased, the impact is only 50 % compared to home-grown crop cultivation. The total impact of Land Use Change contributes for 4 %. Manure storage and application on grassland contribute for 15.3 % to the emissions.

There are in fact three major parameters contributing to a shift in CF upon modification. First, the effects of changing some herd and feed characteristics like mortality rate, liveweight and digestible energy content of fodder are substantial. Second, the impact of using soy meal as feed compound was analysed. Limiting the use of soy can lead to a CF decrease, however implies that a substitute needs to be found. Third, changes in manure management also impact the CF results.

Results revealed two major hotspots in the lifecycle of pigmeat production for which the highest contribution to Greenhouse gas emissions can be found: fodder production and manure production and usage. Based upon these hotspots, there are possibilities to define opportunities for reducing the carbon footprint and contribute to sustainability. In particular, fodder composition has a huge impact on the CF. Within Europe, the use of soy beans in feed concentrates has increased a lot. Using soy meal has a substantial impact due to negative Land Use Change impact and the need to transport feed compound over long distances (Hortenhuber et al., 2011). Using regional products might reduce the CF by limiting the energy consumption needed for transportation. If however, overseas products are necessary to use, preference should be given to sustainably produced products with a limited impact on Land Use Change. Feed composition does not solely depend upon its climate change contribution, but more on aspects like availability, price and characteristics of components within the feed compound. Hence, one cannot simply suggest to ban all carbon-negative components, because this would limit the economic sustainability of farming practice.

Several issues hence exemplify the possible trade-offs which need to be made between dealing with GHG emissions on one hand and other sustainability issues on the other hand. A mitigation measure only has a positive effect when all other aspects lead to higher sustainability. It is important to stress again that the CF is a good indicator for GHG emissions, focusing on 1 environmental indicator, but that it is not an indicator for environmental impact in general.

RQ2. What are the main characteristics of residual household waste generation able to define a strategy for improving the sustainability related to waste production in Flanders ?

RQ2A. What are the key/determining factors having a significant impact on reaching the mandatory goal of 150 kg residual household waste per capita per year ?

RQ2B. Is there a significant relationship between the amount of residual household waste and a) income per capita, b) the cost of residual household waste collection and c) the collection frequency ?

RQ2C. Is there a significant difference in the cost-efficiency between private or public collection of residual household waste ?

RQ2D. What factors determine the decision to opt for public versus private waste collection services ?

Using binary logistic regression, key factors in residual household waste minimization for the Flemish region of Belgium were identified. Reaching the mandatory goal of 150 kg residual household waste per year per capita was used as dependent variable.

Independent variables that could explain a municipality's chance of reaching this goal were derived from literature. The model correctly classified 74% of the municipalities. Four out of 12 variables in the model contributed significantly. Higher per capita income lowers a municipality's chance of reaching the goal.

An increase in the cost of residual waste collection, changing from weekly to an every other week residual waste collection and introducing separate curbside collection of organic waste increase a municipality's chance of reaching the goal and thus might help in defining strategies for obtaining higher sustainability. It is concluded that binary logistic regression is an appropriate tool for identifying key factors in a waste management plan. The implementation of separate curbside collection of organic waste has the strongest influence. This is not surprising as the organic waste fraction, both yard waste and waste from fruit and vegetables, makes up about 40% of total municipal waste weight in Flanders (OVAM, 2002).

Frequency of residual household waste collection, yearly cost of curbside residual household waste collection and the average income per capita have a fairly equal impact.

With regard to the economic sustainability aspect of household waste (cost efficiency), multiple case studies of municipalities in the Flemish region of Belgium were conducted. In total 12 municipalities were investigated, divided into three mutual comparable pairs with a weekly and three mutual comparable pairs with a fortnightly residual waste collection. The results give a rough indication that in all cases the cost of private service is lower than public service in the collection of household waste. Albeit that there is an interest in establishing whether there are differences in the costs and service levels between public and private waste collection services, there are clear difficulties in establishing comparisons that can be made without having to rely on a large number of assumptions and corrections. However, given the cost difference, it remains the responsibility of the municipalities to decide upon the service they offer their citizens, regardless the cost efficiency: public or private.

The in-depth interviews with the different public officials revealed that municipalities with a public service waste collection opt to focus on a broad service level rather than cost efficiency and hence a lower cost for municipalities and their citizens. They are offered a total and complete service level in terms of waste collection, and therefore these municipalities are willing to pay extra. Municipalities counting upon private service waste collection firms on the other hand, focus more on cost efficiency driven by tendering.

RQ 3. Can sustainability performance be expressed in an overall sustainability score, integrating all dimensions ?

RQ 3A. What are the different qualitative and quantitative indicators encompassing the sustainability performance measurement framework of Latin America-EU soy chains ?

RQ 3B. What is the current performance and stakeholder perception of Latin American EU soy chains with regard to sustainability, covering all dimensions ?

RQ 3C. Is there a difference in sustainability performance with regard to the dimensions of sustainability for different soy production systems ?

In order to assess sustainability, one needs to define a sustainability framework, covering the most important indicators for the soy chain. To do so, a literature review was conducted, followed by stakeholder interviews (table 36) in order to grasp bottlenecks and hot issues on sustainability. The literature review was conducted by selecting quantitative and qualitative scientific research, organization reports, etc. The selected studies covered a large share of the comprehensive indicators for classifying and analyzing food supply chains in detail. The review was performed in three steps.

First, integrated triple bottom line (simultaneous consideration of sustainability indicators) studies were reviewed to identify key environmental, social and economic indicators of food chains. The outcome of this first step was used as the base for creating a list of indicators to be included in any sustainability assessment. In the second step, the literature review was expanded with any economic, environmental, and social indicators that were evaluated separately for soy chains.

The results of the stakeholder interviews were used to extend the analytical framework with indicators aiming at measuring or exploring sustainability issues. These indicators are not quantifiable as such, they are manifest variables, which need to be validated.

Moreover the comprehensiveness of the extended analytical framework was evaluated using the SAFA guidelines, developed by the FAO (2012) as a reference conceptual framework. With the aim of improving the transparency and comparability of sustainability performance of food and agricultural sectors, FAO (2012) developed guidelines on the Sustainability Assessment of Food and Agricultural Systems (SAFA guidelines). The SAFA guidelines are developed through an ongoing process of text elaboration, stakeholder consultation and participation. Moreover continuous reviews take place and therefore SAFA can be considered as a strong generic template for evaluating the comprehensiveness of the selected sustainability indicators. The framework can be extended with additional indicators to increase the comprehensiveness in order to touch upon a broad set of ecological, social and economic themes.

Subsequently, the whole process of identifying important indicators was finalized by conducting expert interviews. Each partner in the (EU funded) research project interviewed at least 4 experts, including representatives of academic, NGO or governmental institutions, familiar with social, environmental and economic issues in the soy chain. The template of the interviews is included in annex 1. During the interviews, experts were asked to give their opinion on a selection of indicators, according to their respective expertise. The exploratory nature of the interviews allowed interviewees to come up with themes and sustainability issues they deemed relevant. The expert interviews explore bottlenecks regarding different sustainability issues among and between representative soy chains originating from Brazil. Based on the results of the expert interviews the framework was finetuned to fit the specific character of soy chains. The number of indicators could therefore be reduced to a limited number of relevant, measurable and discriminative indicators. The soy sustainability performance indicator framework can be found in table 38.

The tool developed within the SALSA project is a way of giving expression in a practical way to the concept of sustainable soy production farms and by extension soy production chains. Based upon 9 sustainability indicators covering the three sustainability dimensions, this tool is developed as a self-assessment tool for both farmers and other decision-makers to support sustainability in agriculture in general and soy production in particular. The overall mono-metric sustainability score comprises of aggregated dimensional scores, where each dimension has an equal weight.

Given the examined cases in Brazil, it can be seen from the graphs that organic is obtaining the highest sustainability score for the withheld indicators, followed by GM and non-GM soy production systems. Looking at the individual dimensions, GM is most profitable, and organic most environmental friendly.

The limited sample size with regard to stakeholder perceptions has shown that European stakeholders prefer sustainability issues like global warming, water consumption, operating profit and working conditions. Latin American stakeholders share the same economic and social interests but prefer to focus on land use efficiency instead of climate change regarding the environmental impact. Both continents want to facilitate trade with one another by harmonizing sustainability criteria. The EU however wants to be less dependent upon the import of proteins. Latin America on the other hand wants to keep on trading with Europe and hence needs to comply with sustainability claims. Aiming for harmonization hence indicates that sustainability perception towards the same crops (i.e. soy) should be scrutinized on both sides of the ocean.

6.3 Policy implications

The several sustainability assessment tools and their applications are primarily targeting governments and their policy makers. Below is explained per topic what the potential implications of the conducted research might be with regard to sustainability (assessments). The targeted audiences are not limited to governments and its policymakers. The research is also of importance to the food industry in general and the meat processing companies in particular. Moreover, it is of importance to consumer organizations, NGOs and scientific researchers conducting sustainability assessments. In the multistakeholder debate characterizing the definition of sustainability strategies and policy each stakeholder is influencing the other. The results can, consequently contribute to a more informed debate and hopefully more effective sustainability policies.

Carbon footprint

If Belgium needs to comply with EU2020 GHG targets, all regions of the country should gain insights into their climate change contribution in absolute terms. More specifically, the agricultural sector needs to monitor its activities for mapping the environmental impact in general and climate change impact in particular. The results of the carbon footprint sustainability assessment might be useful in two ways.

First, monitoring the climate change impact of meat production systems emphasizes the need to make animal production more efficient. It helps to identify hot spots in the production chain and to come up with mitigation measures for reducing the environmental impact in the search for sustainability. Second, results might raise awareness about the climate change problem and relate it to achieve more healthy diets, bringing about a shift in the current food system.

Research has shown that future population growth coupled with current demands for food might imply almost a doubling of global crop production by 2050. Farmers are slowly increasing crop yields, but only at about half of the rate needed to meet future demand (Bajzelj et al., 2014).

Agriculture is a major contributor to climate change and pollution, and so further expansion is questionable. It is hence almost imperative to achieve global food security without expanding crop or pastureland and without increasing greenhouse gas emissions (Bajzelj et al., 2014).

This food system shift does not force everyone to switch to soylent-based diets so much as jumping to generally healthier diets in the conventional food pyramid sense. Healthier diets are ought to be more efficient diets at the production end, which in turn means tearing up less land and hacking apart fewer GHG-absorbing trees (Bajzelj et al., 2014). It should not be the aim to all become vegetarians, it should be an argument to eat meat in sensible amounts as part of a healthy, balanced diet.

Results hence help governments and stakeholders from the industry to monitor climate change impact of meat production systems and how to reduce its impact. Moreover current health policy with regard to healthy diets should be able to take into account this environmental impact as well. This first attempt to monitor climate change impact of meat production systems, should trigger governments and its policymakers to further improve and update results for detecting climate change impact over a longer period of time. At first the aim should be to achieve the EU 2020 target from an environmental point of view. At the same time a win-win situation can be created if these targets can be reached with more balanced, healthy consumer diets.

Household waste – determining factors

The chapter about household waste covers both environmental and economic sustainability. The results of the environmental sustainability part are of interest to local waste authorities and public managers organizing the residual waste collection rounds. It was found that at least 4 parameters have a significant impact on a municipality's chance of reaching the mandatory goal of 150 kg per capita per year. Policy makers of municipalities who already reached the mandatory goal, can further improve the result of decreasing waste generation, whereas decision makers of municipalities who did not reach the target yet, can take into account these factors when implementing measures for reaching the goal and searching for sustainability with regard to waste production. This thesis offers them a wide range of parameters to include in any scheme for reducing household waste generation.

Next, the economic sustainability part comprising the cost of residual household waste collection can be of interest to both waste authorities and private waste collection firms. With this information policymakers can decide which waste collection service to offer their citizens. Given the lower cost, municipalities and their citizens will benefit economically from private service. However if the expectations regarding the service are high, one will benefit more from public service. Furthermore, municipalities do not always opt the most economic efficient service and they are free to organize the policy regarding household waste. They can manage it on their own, they can cooperate with a private partner or they can be a part of a bigger entity, namely a joint venture of municipalities which has a private or public service level. With this joint venture they try to diminish the price by creating economies of scale. Such cooperations collecting and processing the household waste offer a total service level, which is attractive to some municipalities. Next to policy makers, private operating collection firms can gain insights into the waste collection market, e.g. where market share can be gained.

Soy chain integrated sustainability assessment

Based upon 9 sustainability indicators covering the three sustainability dimensions, this tool is developed as a self-assessment tool for both farmers and other decision-makers to support sustainability in agriculture in general and soy production in particular. The application of the sustainability tool is illustrated using Brazilian case studies. The final result is a measure of the current state of each variable indicator with regard to sustainability, as well as a measure of the current sustainability state of the system in an integrated way, covering all dimensions.

Moreover, a transparent assessment is delivered. By transparent, it is meant that in order to understand and use the results, one should be able to also understand the quality of the data, contributions of several participants to the process, the assumptions and uncertainties, as well as how and why each decision was taken (Bausch et al., 2014).

Next to the assessment, learning about the perceptions of stakeholders on both sides of the atlantic might help business stakeholders to gain insights into each others preferences toward sustainability in order to sustain business for example.

6.4 Limitations

The main limitations of this PhD dissertation are touched upon in this section. The focus will mainly be on broad, general cross-chapter limitations. Chapter specific constraints are covered within the specific chapters as such.

At first, the main limitation is related to trade-offs between the Triple Bottom Line Categories. Gibson (2001) indicates that some trade-offs might be possible in EIA-driven integrated assessment, and the risk of environmental standards outweighing socio-economic factors has been touched upon a lot in literature (Sheate et al., 2003; Jenkins et al., 2003, Gibson, 2001; Lee, 2002). An overall sustainability score expressed as a composite indicator allows for compensation among dimensions during aggregation. Such methodological assumptions are no major limitations, but prove that prudent consideration should be taken upon the construction of indices.

Second, the aspects that were used to give the best picture about environmental and economic sustainability in chapters 2 (carbon footprint) and 3 (household waste), might very well be replaced with others. They are hence no proxies for the respective sustainability concept in place.

Third, regarding the cost aspect of the economic sustainability on household waste collection in chapter 3, it was assumed that that there is just 1 waste collection system within each municipality. In practice it might however occur that the city centre and the outskirts have different collection systems due to logistical reasons. This obviously has an impact on the costs and on the overall cost comparison.

Fourth, the scope of the integrated assessment is limited regarding the product used for analysis, i.e. soy meal. The used indicators and results for the integrated sustainability tool cannot be generalized or applied for other food products. There is hence no such thing as one generic (farm) sustainability model. Indicators show the need to be adapted to local contexts and farming conditions. It is not very realistic to believe that one single tool could cover all types of production systems. Moreover, the engagement of stakeholders is unique in every context. This is one of the important decisions to be taken by an assessment team.

Fifth, the idea behind customizing the SAFA method to assess the sustainability of a specific product is very interesting. But while the practice of gathering expert ideas on the indicators through interviews might be the only way to consider such indicators so far, however, the selection of interviewees in terms of their expertise covering all the relevant dimensions and indicators might not be transparent enough.

Sixth, another issue is that the indicators' interlinkages in different dimensions has not been considered in the integrated sustainability assessment. For instance, with regards to the costs for society created by environmental impacts, the monetization of external costs tries to add up such impacts, included in the economic dimension. This means that a change in an environmental impact may also lead to changes in the results in the economic dimension. An economic impact (value added) quite often has a social dimension (income), and a social impact (health effect) has an ecological dimension (damage of ecological balance), and so on. Given that there is insufficient in-depth investigation into the actual range of interconnections, a blanket allocation of 1/3 weight to each dimension does not necessarily seem all that correct. However, environment, society and economy are considered being independent, but interrelated subsystems. The functionality and resistance to disturbance of those need to be preserved for future generations. The goal of sustainable development is the long-term preservation of the system and avoiding damage in all three dimensions (Voß, 2005). Literature provides 2 argumentations for the equity postulates among the dimensions. First, it is concluded that the heritage for future generations should not be limited to ecological goals. Sustainability must include basic needs for human living conditions for present and future generations. Second, the action area of sustainable development is limited by the carrying capacity of natural and social systems. The equity postulate is backed here by the fact that civilizing developments are not only threatened by ecological, but equally by economic and social risks.

Seventh, linked with the previous point is the methodology regarding the aggregation of scores. An all-inclusive mono-metric score based upon the combination of the three dimensions allows compensation among the dimensions, and hence the score might imply no real meaning. On the other hand, compensation can be allowed within the same dimension as favourable practices might offset practices harming another component (Baush et al., 2014).Compensation between dimensions can be seen as a form of weak sustainability. It indicates that a certain proposal (or production system in this case) can overall be considered positive as long as the net assets are not degraded (Neumayer, 2003). A project may thus have a positive outcome in 1 dimension, and negative outcomes in the other two. As long as the overall (net) outcome is positive, then negative impacts in other pillars are acceptable. Sadler (1999) points out that the chance of win-lose scenarios can be reduced by involving minimum acceptability tresholds into the Triple Bottom Line model and furthermore require that each project or initiative should meet this minimum tresholds (Pope et al., 2004).

Eight, the optimal or sustainable state of indicators can be identified by benchmarking them to other production systems. The comparison might allow to reveal target values. As the integrated assessment was meant to be practical and relevant to stakeholders, the inclusion of ideal and anti-ideal states for each indicator in general and overall sustainability in particular could have delivered an added value. Stakeholders could have engaged within the development of the tool by defining the ideal and anti-ideal states for all indicators. This could have helped to define overall sustainability targets for soy production systems.

Nineth, all conducted assessments are in fact snapshots. In the case of overall sustainability this is necessary to have in order to gain insights into the system property on a certain moment. Sustainability is however an iterative and dynamic process (Leach et al., 2010). The integrated assessment within this PhD dissertation can hence be seen as a point of reference or a baseline against which future assessments can be weighted. Thus, the conducted integrated assessment serves as a baseline and is as such too little to draw hard conclusions regarding sustainability. A follow-up study is needed to see the progress towards higher sustainability.

Tenth, both the carbon footprint assessment as well as the integrated sustainability assessment are very much dependent upon secondary data. Especially with regard to agricultural sustainability assessments, primary data are scarce, show a minor quality and were collected on a disperse basis. This might have influenced the results. A much more flexible approach is possible by using system variables instead of indicators. These variables describe the system in a proper way and indicate how they influence one another. This approach requires more details on system variables and on their mutual influence. As such, indicators sometimes might have little meaning regarding sustainability if they are not linked with a concept and context (Bausch et al., 2014). Second, comparison seems more difficult when specific indicators are applied without referring to a more broad variable covering a wide range of contexts.

Eleventh and ultimate, within stakeholder group perception differences could not be detected by means of post hoc tests, as the sample size was too small. Differences could rather occur through sectoral affiliations and not through continental differences.Results might hence be biased.

6.5 Suggestions for future research

Wiek et al. (2012) showed that assessments contributing to a transformational change toward higher sustainability might be limited. It was out of the scope of this PhD dissertation to identify how transformational the results of the integrated assessment were, but the tools provide a baseline to start an assessment with. As such, the integrated sustainability assessment conducted for the transatlantic soy chains should be seen as a baseline or reference point study with which future assessments can be compared with. The general approach in this PhD dissertation is transparent and systematic, providing a basis for assessing a future state of the system by monitoring.

Therefore, a follow-up study regarding transatlantic soy chains is needed in order to assess the sustainability situation after decision making in that area.

Second, future applications of the defined integrated sustainability tool should include targets for ideal and anti-ideal states. Therefore, an open discussion with stakeholders could be an appropriate way to increase awareness and understanding among scientists and stakeholders regarding the relationship between sustainability indicators. This thesis used a relative Lickert scale to convert absolute scores into adimensional ones. The Lickert scale ranges from 0 to 7. It could be an idea to let stakeholders define absolute indicator values accompanying the ideal and anti-ideal states.

Third, the integrated assessment approach might be extended towards other products and contexts. This creates opportunities to benchmark systems and to see how several strategies vary across contexts. Moreover there would be the possibility to identify how certain assessment decision options influence the assessment results.

Fourth, all data were collected in a certain temporary timeframe. This is especially necessary with regard to the sustainability assessment as a baseline is needed. However, in order to reveal the same trends in the results of carbon footprinting and household waste, one should clearly use panel data over a longer period of time.

6.6 The road ahead for sustainability

The transition towards sustainability requests to redesign production, consumption and waste management. In order to accomplish these goals, there is a need for reliable definitions and measurements (Cucek et al., 2012). Many tools to measure sustainability were developed to mostly evaluate the unsustainability of humans, processes and activities. However, defining a sustainability metric to support sustainability assessments remains an open issue in literature. This PhD dissertation tackled this gap.

At the same time it provided three different tools for conducting future sustainability assessments. The first two tools were about minimizing the negative impact on the environment, the last one was about extending the environmental aspect with social and economic ones to obtain an integrated assessment.

Which tool is being used and how sustainability is being measured depends upon the purpose. If communication towards a wide audience is the primary goal by policymakers, then a 1 indicator (or index) approach should be used. Policymakers do not have much time available to study all information and thus are interested in an easy and straightforward message to bring to their audience. If sustainability is more about minimizing negative effects in order to progress to a more sustainable state, it is better to apply the environmental sustainability tools (carbon footprint and household waste). They are able to help in identifying the direction to target (albeit unknown exact position): progress towards or diverting away from a more sustainable state.

Given its dynamic nature, measuring sustainability is a matter of monitoring progress towards a more sustainable state (transformational change). At the same time, following an integrated assessment, it is possible to express sustainability as 1 indicator, but more for communication purposes of policymakers. This approach has the disadvantage that a lot of information is lost during the aggregation process, indispensable with the other tools.

Therefore, all in all to have a thorough overview of sustainability and its components, it is better to split up the 1-metric score into the separate dimensions in order to clearly see where bottlenecks occur and improvements can be made to achieve higher sustainability.

The same occurs when 2 products have the same sustainability performance score. In order to evaluate the sustainability performance increase of certain activities on the dimension or indicator level, it is necessary to go back to the original list of dimensions. This implies that two scenarios with the same sustainability index have a different meaning for stakeholders in the sense that they can choose between those solutions based upon their preferences.

Measuring sustainability is thus a mix of monitoring aspects on one hand and integrating all dimensions into 1 index on the other hand. All depends upon the purpose of the measurements. There is no single optimum way of measuring the concept. Each approach has its advantages and disadvantages. A breakdown of the overall score into individual dimensions and indicator performance seems however necessary for identifying hotspots on one hand and to see the impact of improvement scenarios on the sustainability performance on the other hand.

Indicators which measure sustainability might be useful in decision-making. There is however still a long way to go to accomplish standardization of footprint definitions and units of measurement. Perspectives on footprints, sustainable development and extended LCA show that more work is needed to clearly integrate economic, environmental and social aspects during decision-making.

Summary

Sustainable development is described as using resources in a way that it meets the needs of the present generation without compromising the possibility for future generations to meet their needs. Nowadays sustainability is a hot topic, even in society as a whole. This societal impact created an extra leverage in attracting extra attention and funding for research on sustainability in the scientific world. Sustainability is the ability of a system to overcome shocks and stresses in seeking a balance upon the interactions of ecological, social and economic aspects. In the quest for sustainability, a pathway should be followed in order to move from one stable state to another and remove the factors causing unsustainability.

There is however a multitude of definitions out there. The fact that sustainability is not precisely defined, does not exclude it from being assessed as a concept.

Many countries have set national strategies for pursuing higher sustainability, including sustainability targets and indicators. Several attempts were undertaken to make the agricultural sector in general and the food sector in particular more sustainable. However, there is no harmonized standard defining what sustainable production should involve. Moreover, there is no agreement on which set of indicators to include when measuring sustainability performance; there is a need for a sustainability indicator framework.

The separation of the sustainability concept into three pillars tends to stress potentially competing interests instead of emphasizing the linkages and interdependencies between them. This makes integration very difficult and challenging and to some extent it is creating trade-offs among dimensions, mostly degrading the environment. The overall objective of this thesis is to examine the possibility and necessity to measure/ quantify sustainability as 1 composite indicator (index). There is a clear need in more transparent assessments on one hand and in integrated assessments (involving all dimensions) on the other hand.

This PhD dissertation is a compilation of research papers conducting sustainability assessments with regard to a certain dimension (in the case of 1 indicator assessments like climate change impact and household waste) and an integrated assessment with regard to a global value chain (i.c. transatlantic soy value chain), including sustainability perceptions. Soy is one of the main feed compound ingredients and Brazil is one of the main producers and exporters of soymeal. This trade between Latin America and the EU has raised questions about its impact, especially in Europe. Criticism mainly involves the impact on climate change and mostly deforestation in Latin America. At the same time, the agribusiness contributes to strong economic growth in Latin American countries. Given Brazil's position in the soy world market on one hand and sustainability issues on the other hand, assessing the sustainability performance of this global value chain seems of utmost importance.

The sustainability assessments are investigated following three lines of inquiry in a bottom-up approach in order to obtain an integrated assessment, covering all sustainability dimensions. As most sustainability assessments tend to only focus on environmental sustainability, it was opted to start assessing this dimension with regard to climate change impact, given its current

importance. The second line included the economic dimension into the assessment, on top of the environmental one. An interesting case for this was found in household waste, more specifically household waste generation. The third line integrates all sustainability dimensions to come up with an overall sustainability performance assessment tool, applied to a global value chain.

In line with these three lines of inquiry, several research questions are formulated and tackled, building upon theory, empirical evidence and existing knowledge gaps.

Data is collected based upon both primary and secondary sources. Several exploratory and conclusive research methods are applied. A structured questionnaire on waste collection issues with all Flemish municipalities (N = 308)was conducted. The study on carbon footprint was built upon literature review, structured interviews and life cycle assessments. And to assess integrated sustainability, an indicator framework was applied, using primary and secondary data.

Results showed that 1 kg of deboned pigmeat creates a carbon footprint of 5.7 kg CO_2 equivalents, whereas 1 kg of deboned beef has a Carbon footprint of 22.2 kg CO_2 equivalents. The single CF figure should however be used with caution, since it is based upon specific input data. Presenting a range in which the CF is expected to fall, provides a much more robust result. This can be seen in light of a sensitivity analysis in which several parameters are fluctuated. Based on this analysis, the estimated CF of pigmeat production in Flanders is expected to lie between 5.5 and 5.9 kg CO_2 equivalents per kg of deboned meat. For beef the CF would lie between 22.2 and 25.4 kg CO_2 equivalents.

Results revealed two major hotspots in the lifecycle of meat production for which the highest contribution to Greenhouse gas emissions can be found: fodder production and manure production and usage. Based upon these hotspots, there are possibilities to define opportunities for reducing the carbon footprint in the search for higher sustainability. In particular, fodder composition has a huge impact on the CF. Using regional products might reduce the CF by limiting the energy consumption needed for transportation. Feed composition does not solely depend upon its climate change contribution, but more on aspects like availability, price and characteristics of components within the feed compound. Hence, it cannot simply be suggested to ban all carbon-negative components, because this would limit the economic sustainability of the farming practice.

Several issues hence exemplify the possible trade-offs which need to be made between dealing with GHG emissions on one hand and other sustainability issues on the other hand. A mitigation measure only has a positive effect when all other aspects lead to higher sustainability. It is important to stress that the carbon footprint is a good indicator for GHG emissions, focusing on 1 environmental indicator, but that it is not an indicator for environmental impact in general.

The second line of inquiry focused on household waste generation and its determining factors. Using binary logistic regression, key factors in residual household waste minimization for the Flemish region of Belgium were identified. Reaching the mandatory goal of 150 kg residual household waste per year per capita was used as dependent variable.

Independent variables able to explain a municipality's chance of reaching this goal were derived from literature. The model correctly classified 74% of the municipalities. Four out of 12 variables in the model contributed significantly. Higher per capita income lowers a municipality's chance of reaching the goal.

An increase in the cost of residual waste collection, changing from weekly to an every other week residual waste collection and introducing separate curbside collection of organic waste increase a municipality's chance of reaching the goal. It is concluded that binary logistic regression is an appropriate tool for identifying key factors in a waste management plan able to contribute to higher sustainability. The implementation of separate curbside collection of organic waste has the strongest influence. This is not surprising as the organic waste fraction, both yard waste and waste from fruit and vegetables, makes up about 40% of total municipal waste weight in Flanders. Frequency of residual household waste collection, yearly cost of curbside residual household waste collection and the average income per capita have a fairly equal impact.

With regard to the economic sustainability aspect of household waste, multiple case studies of municipalities in the Flemish region of Belgium were conducted. In total 12 municipalities were investigated, divided into three mutual comparable pairs with a weekly and three mutual comparable pairs with a fortnightly residual waste collection. The results give a rough indication that in all cases the cost of private service is lower than public service in the collection of household waste. Albeit that there is an interest in establishing whether there are differences in the costs and service levels between public and private waste collection services, there are clear difficulties in establishing comparisons that can be made without having to rely on a large number of assumptions and corrections. However, given the cost difference, it remains the responsibility of the municipalities to decide upon the service they offer their citizens, regardless the cost efficiency: public or private.

The integrated sustainability assessment applied to the soy chain, learnt that there is a need to define an indicator framework. A cascade of operations was conducted for obtaining such a framework. The framework was then used as a starting point for developing an integrated sustainability performance tool. The tool developed within the project is a way of giving expression in a practical way to the concept of sustainable soy production farms and by extension soy production chains. Based upon 9 sustainability indicators covering the three sustainability dimensions, this tool is developed as a self-assessment tool for both farmers and other decision-makers to support sustainability in agriculture in general and soy production in particular. The overall mono-metric sustainability score comprises of aggregated dimensional scores, where each dimension has an equal weight.

Given the examined cases in Brazil, results showed that organic is obtaining the highest sustainability score for the withheld indicators, followed by GM and non-GM soy production systems. Looking at the individual dimensions, GM is most profitable, and organic most environmental friendly.

The limited sample size with regard to stakeholder perceptions has shown that European stakeholders prefer sustainability issues like global warming, water consumption, operating profit and working conditions. Latin American stakeholders share the same economic and social interests but prefer to focus on land use efficiency instead of climate change regarding the environmental impact. Both continents want to facilitate trade with one another by harmonizing sustainability criteria. The EU however wants to be less dependent upon the import of proteins. Latin America on the other hand wants to keep on trading with Europe and hence needs to comply with sustainability claims. Aiming for harmonization hence indicates that sustainability perception towards the same crops (i.e. soy) should be scrutinized on both sides of the ocean.

This PhD thesis provides an important baseline for stakeholders in the fields of sustainability assessments, environmental impact prevention and subsequent health prevention. It might stimulate the academic debate and raise awareness among the broad public with regard to sustainability for pursuing a higher quality level of it.

Samenvatting

Dit manuscript is een compilatie van research papers over geïntegreerde (alle dimensies samen) en niet geïntegreerde duurzaamheidsanalyses (bv. product analyses)

Duurzame ontwikkeling wordt omschreven als een manier om natuurlijke hulpbronnen door de huidige generatie dusdanig te laten gebruiken, dat de noden van de toekomstige generaties niet in het gedrang komen. Duurzaamheid is vandaag de dag een hot topic, waardoor de wetenschappelijke wereld extra aandacht en middelen krijgt voor duurzaamheidsonderzoek. Duurzaamheid beschrijft een systeemeigenschap en duidt op de mogelijkheid van een systeem om schokken en spanningen te overwinnen in de zoektocht naar een balans tussen de interacties van ecologische, sociale en economische aspecten. Bij het streven naar duurzaamheid moet een route gevolgd worden, die van de ene stabiele toestand naar de volgende gaat, waarbij factoren die duurzaamheid verstoren verwijderd of gereduceerd worden.

In hun zoektocht naar meer duurzaamheid bepaalden vele landen nationale strategieën met streefdoelen en indicatoren. Verscheidene pogingen werden ondernomen om de landbouwsector in het algemeen en de voedingssector in het bijzonder duurzamer te maken. Een uniforme standaard, die aangeeft wat duurzaam produceren inhoudt, bestaat echter niet. Meer nog, er is geen consensus omtrent welke set indicatoren men moet gebruiken om duurzaamheid te meten. Daarom is er behoefte aan een kader voor indicatoren met betrekking tot duurzaamheid.

Het indelen van het duurzaamheidsconcept in drie pijlers (of dimensies) heeft de neiging om mogelijk rivaliserende belangen te accentueren i.p.v. onderlinge verbanden te benadrukken. Hierdoor wordt integratie uiterst moeilijk en uitdagend en mogelijks veroorzaakt het afwegingen tussen dimensies, vaak ten nadele van het milieu. In dit manuscript wordt de mogelijkheid en noodzaak onderzocht om duurzaamheid meetbaar te maken en te kwantificeren a.d.h.v. 1 samengestelde indicator of index.

Soja is één van de belangrijkste voedercomponenten en Brazilie één van de voornaamste producenten en exporteurs van sojameel. De impact van deze handel tussen Latijns-Amerika en de EU heeft vooral in Europa heel wat vragen doen rijzen. Critici wijzen voornamelijk op de gevolgen van klimaatsverandering en ontbossing in Latijns-Amerika. Terzelfdertijd draagt de agribusiness bij tot de sterke economische groei in Latijns-Amerikaanse landen. Braziliës positie in de markt van soja enerzijds en daarbij de duurzaamheidsvraagtekens anderzijds, maken een evaluatie van het duurzaamheidseffect van deze globale waardeketen uiterst belangrijk.

De duurzaamheidsanalyses werden volgens drie onderzoekslijnen (3 cases) uitgevoerd in een benadering om op die manier een geïntegreerde analyse te bekomen waarin alle duurzaamheidsdimensies vervat zitten. Omdat de meeste duurzaamheidsanalyses de neiging hebben enkel de focus te leggen op milieuduurzaamheid, is er geopteerd - gezien het huidige

belang ervan – in de eerste lijn te beginnen met de impact op klimaatsverandering, toegepast op vleesproductie.

De tweede lijn analyseert bijkomend de economische dimensie, bovenop milieu. Een belangrijke casus hierbij vindt men terug bij huishoudelijk afval, meer specifiek in het behandelen (inzamelen en verwerken) ervan. De derde lijn integreert alle duurzaamheidsdimensies, om zo tot een algemene tool voor duurzaamheidsanalyses te komen, toegepast op een algehele waardeketen. In lijn met deze drie onderzoekslijnen werden verscheidene onderzoeksvragen geformuleerd en behandeld, gestaafd op theorie, empirisch bewijs en bestaande wetenschappelijke hiaten.

De gegevens werden verzameld via zowel primaire als secundaire bronnen. Er is gebruik gemaakt van verscheidene onderzoeks- en besluitvormende methodes. Er werd een gestructureerde vragenlijst opgesteld omtrent de problematiek van afvalinzameling bij alle Vlaamse gemeentes (N = 308). Het onderzoek over koolstofvoetafdruk is gebaseerd op literatuur, gestructureerde interviews en levenscyclusanalyses. Om de geïntegreerde duurzaamheid te analyseren werd een samengestelde indicator gehanteerd, gebruik makend van primaire en secundaire data.

Het onderzoek brengt in de levenscyclus van vleesproductie twee kritieke punten aan het licht, die het meest bijdragen tot de uitstoot van broeikasgassen : de productie van veevoeder enerzijds, mestproductie en –verwerking anderzijds. Hiervoor is het mogelijk opportuniteiten te definiëren om de koolstofvoetafdruk te reduceren en tezelfdertijd meer duurzaamheid te bekomen. Het is voornamelijk veevoederproductie dat zwaar weegt op de koolstof voetafdruk. De samenstelling van veevoeder hangt niet alleen af van de impact die het heeft op de klimaatsverandering, maar meer van factoren zoals beschikbaarheid, prijs en eigenschappen van de componenten. Men kan echter niet zomaar alle bestanddelen met negatieve impact op het klimaat bannen, want dit zou de economische duurzaamheid van de landbouw bedreigen.

Alsdusdanig moeten afwegingen gemaakt worden tussen de uitstoot van broeikasgassen en andere duurzaamheidsaspecten. Een maatregel tot vermindering heeft slechts een positief effect zodra alle andere aspecten ook bijdragen tot meer duurzaamheid. Het is belangrijk om te onderlijnen dat carbon footprint een goede indicator is voor de uitstoot van broeikasgassen, zich richtend op 1 milieu indicator, maar dat het geenszins een indicator is voor de algemene impact op het milieu.

De tweede lijn van het onderzoek richt zich op huishoudelijk afval en de factoren die een impact hebben op het ontstaan ervan. De belangrijkste factoren voor het verminderen van het restafval van de huishoudens in het Vlaamse gedeelte van België werden geïdentificeerd a.d.h.v. het gebruik van binaire logistische regressie. Het behalen van de verplichte doelstelling van 150 kg per jaar per inwoner werd als afhankelijke variabele gebruikt. De onafhankelijke variabelen, nodig om de kansen te verklaren waarmee de gemeentes deze doelstelling kunnen halen, werden afgeleid uit literatuur. Vier van de twaalf verklarende variabelen in het model leveren een belangrijke bijdrage.

Door ophalen van restafval duurder te maken, door omschakelen van wekelijks naar tweewekelijkse ophaling, en door afzonderlijk ophalen van GFT, verhogen de kansen dat gemeenten hun doelstellingen rond restafval zullen bereiken. Binaire logistische regressie is aldus een geschikte tool om de hoofdfactoren te bepalen voor een afvalbeleid dat bijdraagt tot meer duurzaamheid.

Uit de geïntegreerde duurzaamheidsanalyse toegepast op de sojaketen blijkt dat een raamwerk van indicatoren noodzakelijk is. Daarom is een opeenvolging van toepassingen gedaan om zulk een kader te verkrijgen. Dit kader werd dan aangewend als startpunt om een geïntegreerd instrument voor duurzaamheidsevolutie te ontwikkelen. De geïntegreerde duurzaamheidstool laat toe op een praktische wijze het concept duurzaamheid bij soja producerende bedrijven te evalueren. Gebaseerd op 9 duurzaamheidsindicatoren die de drie duurzaamheidsdimensies omvatten, is dit programma ontwikkeld om zowel landbouwers als andere belanghebbenden te laten afwegen om duurzaamheid te ondersteunen in de landbouw en in de sojateelt in het bijzonder. De uiteindelijke monometrische duurzaamheidsscore omvat de geaggregeerde dimensionele scores, waarbij elke dimensie een gelijk aandeel heeft.

Uit de onderzochte cases in Brazilië is gebleken dat biologische sojaproductie de beste duurzaamheidscore haalde voor de weerhouden indicatoren, gevolgd door GM en niet GM soja productiemethodes. Bekijken we de individuele dimensies, dan is GM meest winstgevend, en biologisch meest milieuvriendelijk.

Uit de analyse komt ook naar voor dat Europeanen als belanghebbenden een voorkeur tonen voor onderwerpen zoals opwarming van de aarde, water consumptie, bedrijfswinsten en werkomstandigheden. De Latijns-Amerikanen ondersteunen dezelfde economische en sociale belangen, maar verkiezen zich te richten op een meer efficiënt landgebruik i.p.v. op klimaatsverandering en de gevolgen voor het milieu. Beide continenten willen de handel met elkaar versoepelen door het harmoniseren van hun duurzaamheidscriteria. De EU wil echter minder afhankelijk zijn van de import van eiwitten. Als Latijns Amerika de handel met Europa wil bestendigen, moet het zich houden aan de duurzaamheidsrichtlijnen als men harmonisatie wil, dan moet de duurzaamheidsperceptie voor dezelfde gewassen (bijv. soja) aan weerszijden van de oceaan worden onderzocht.

Dit PhD manuscript biedt een belangrijk uitgangspunt voor belanghebbenden in het domein van duurzaamheidsanalyses, preventie van milieu-impact en daarbij het gezondheidsaspect. Het kan het academisch debat stimuleren en het grotere publiek bewust maken om duurzaamheid naar een hoger kwaliteitsniveau te tillen.

Curriculum Vitae

Personal details

Full name:	Ray M.L. Jacobsen
Date of Birth:	November 6th, 1984
Nationality:	Belgian & Swedish
Office address:	Ghent University Faculty of Bioscience Engineering Dept. of Agricultural Economics (marketing and chain management) Coupure Links 653, Gent 9000 Belgium
Office Phone	(+32) 9 264 59 45
Cell phone	(+32) 474 500 288
Email	ray.jacobsen@ugent.be ray.jacobsen@skynet.be

Education

June 2009	**University degree in Business Economics** Ghent University, Belgium – Faculty of Economics *M.Sc. thesis: Comparison of on and offshore wind turbines* Promoter: Prof. dr. Johan Albrecht
June 2008	**University degree in Bioscience Engineering, Environmental Technology** Ghent University, Belgium – Faculty of Bioscience Engineering *M.Sc. thesis: Increasing the dry matter content of pre dewatered sludge cakes* *Promoter: dr.ir.Hans Saveyn*
Jan. – June 2007	ERASMUS: student exchange program 4th year of Bioscience Engineering, SLU, Uppsala, Sweden
June 2002	Secondary School : Latin / Sciences-Mathematics (8hrs) Sint-Paulus, Ghent, Belgium

Work Experience

Jan. 2010–to date	**Ph.D. Researcher**
	Department of agricultural economics, Ghent university (BE)
	Job responsibilities include: (1) research on (amongst other) sustainability and waste management, (2) peer-review paper writing (3) participation in international scientific conferences, seminars and workshops with oral contributions, (4) supervision of (inter)national M.Sc and B.Sc students, (5) proposal writing to attract (inter)national funding, (6) conference management, (7) European project management, (8) market studies for agri-food companies, (9) development of business plan to valorize animal byproducts, (10) auditing of subsidized private company, (11) sustainability impact assessments

Teaching Experience

Dec. 2011-2012	**Guest Lecturer**: food losses, food waste and sustainability
	3rd bachelor year in bioscience engineering, food science

Professional Training

March 21-25, 2011	**Food Supply Chain Management, Ph.D. course**
	Wageningen University, The Netherlands

Research Funding

Jan. 2010–May 2014	EU FP-7 project SALSA
May 2011	EU project VALUE
2012	Application of CF methodology on Flemish agriculture

· International peer-reviewed journals A1

Jacobsen, R., Driesen, T., Bungenstab, D., Gellynck, X. (2014). Analyzing stakeholder perceptions of an integrated sustainability approach: the case of Latin American-EU soy chains. Submitted in April 2014 for International journal of agricultural sustainability (under review)

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• Book chapters

Jacobsen, R., Vandermeulen, V., Van Huylenbroeck, G., Gellynck, X. (2015). Carbon footprint: calculations and sensitivity analysis for cow milk produced in Flanders, a Belgian region. The carbon footprint handbook. p.281-302. Boca Raton, FL, USA.

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PRESENTATIONS

May 2014	Waste Management, 2014, Ancona (Italy)
December 2013	Vegaplan, Sustainability seminar, Brussels, Belgium
April 2013	Sustainability tool presentation, Campo Grande, Brazil
May 2012	Green growth conference, Sheffield, United Kingdom
May 2011	Business plan presentation, Farmer Union, Leuven, Belgium
March 2011	Waste management and technology conference, Philadelphia, USA

Supervision (tutor) of Master and Bachelor students

Norah Benmeridja, M.Sc. student (2014-2015), Ghent University, Belgium

> " Sustainability assessment of food/feed chains at processing and consumer level"
> Promoter: Prof.dr. X. Gellynck

Peter Hudak, IMRD exchange M.Sc. student (2013-2014), Ghent university, Belgium

> "Impacts of biofuel policies on rural areas of developing countries"
> Promoter: Prof.dr. X. Gellynck

Alexander Verclyte, M.Sc. student (2013-2014), Ghent university, Belgium

> "Conversion of pellets to bio-oil by means of pyrolysis"
> Promoter: Prof.dr. X. Gellynck

Jonas Thys, M.Sc. student (2012-2013), Ghent university, Belgium

> "Communication of carbon footprint labels"
> Promoter: Prof.dr. X. Gellynck

Maarten-Jan Aelbrecht, Pieter Candry, Margaux Claeys, Mathias Claus, B.Sc. students (2012-2013), Ghent university, Belgium

> "Sustainability claims of food products"
> Promoter: Prof.dr. X. Gellynck

Flavia Villela Ferreira, IMRD exchange M.Sc. student (2012-2013), Ghent university (BE)

> "Comparative analysis of certification schemes and impacts across supply chain agents in Brazil"
> Promoter: Prof.dr. X. Gellynck

Lise Dereu, Fien Minnens, Myriam Baecke, B.Sc. students (2011-2012), Ghent university, Belgium.

"Valorization of animal byproducts in Belgian food chains."

Promoter: Prof.dr. X. Gellynck

Charlotte Boterdaele, Kimberly De Ruyck, Annelien Tack, Aurelie Van Glabeke, B.Sc. students (2010-2011), Ghent university, Belgium.

"Folate and folic acid during pregnancy: prevention of neural tube defects."

Promoter: Prof.dr. X. Gellynck

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ANNEX

Data accompanying figure 13.

COST	AVGINC		P_i (COST)		P_i (AVGINC)
0	20	−1.428	0.19341	0.93	0.717075
10	21	−1.148	0.240855	0.775	0.684602
20	22	−0.868	0.295671	0.62	0.650219
30	23	−0.588	0.357094	0.465	0.6142
40	24	−0.308	0.423603	0.31	0.576885
50	25	−0.028	0.493	0.155	0.538673
60	26	0.252	0.562669	0	0.5
70	27	0.532	0.629949	−0.155	0.461327
80	28	0.812	0.692536	−0.31	0.423115
90	29	1.092	0.748758	−0.465	0.3858
100	30	1.372	0.797703	−0.62	0.349781
110	31	1.652	0.839161	−0.775	0.315398
120	32	1.932	0.873471	−0.93	0.282925
130	33	2.212	0.901322	−1.085	0.252561
140	34	2.492	0.923579	−1.24	0.224436

Data accompanying table 31.

Dimension/indicator	EU	LA
Environment *	13.7	17.4
(1) *	12,9	16,8
(2)	12,1	13,4
(3)*	12,9	16,8
(4)	12,4	14,6
(5) *	11	13,7
Social	12,9	16,7
(6)	14,1	14,2
(7)	13,9	14,9
(8)	13,3	15,3
(9)	13,4	14,7
(10)	13,3	13
(11)	12,3	12,7
Economic *	13,1	16,5
(12) *	11,5	15,5
(13) *	12,1	14,8
(14) *	10,9	16,1
(15)	13,2	14,9

1. Interviews with experts in the soy chain

For the quantitative data collection the Latin American partners should select representative cases (chains), which are relevant to be compared in the analysis. For Argentina we suggest to use representative production systems being: **GM soy, non-GM soy and organic** soy farming. We also suggest to make a distinction between **family farming and industrial** (large-scale) farming.

These representative cases are also used for comparison and evaluation in this qualitative survey. In the interviews we want to focus on the bottlenecks within and the differences among the above mentioned production systems.

Interviewees

We try to focus on a balanced mix of experts (aim at 4 to 5 interviewees including representatives of academic, NGO or governmental institutions, familiar with the soy supply chain. Each of these experts can than respond to the questions relating to his/her expertise. (so e.g. experts on labour issues don't have to answer the questions on ecological criteria)

Try to look for experts in
Social issues: labour conditions
Economics: supply chain management, agricultural economics, business economics,
Environment: agricultural production, natural resource management, conservation
Governance: Institutional/legal context, business ethics, community relations,...

Please give a short description of the background and expertise of the interviewees and clearly indicate which experts were commenting on which parts of the survey.

1. Ecological criteria

Biodiversity

Please make an evaluation for the different representative production systems (GM, non-GM, organic and family farming) about...
The direct or indirect impact of soy cultivation on the biodiversity?

Through clearing natural vegetation including native forests for expansion?
Through agricultural practices (use of agrochemicals, monocropping, rotation)?
Through conservation areas on farm (legal reserve, etc.)?

Land use and soil quality on production level

Please make an evaluation for the different representative production systems (GM, non-GM, organic and family farming) about...
Processes of soil degradation and measures being taken:

Process of land degradation	Description
Erosion	soil erosion caused by heavy rainfall, or by gully erosion (erosion by run-off in small or larger channels in the land) or wind
subsoil compaction	the deterioration of soil structure due to compaction of the soil by pressure

	from wheels, tracks, rollers or by the passage of animals
salinization	the accumulation of soluble salts of sodium, magnesium and calcium in the soil due to high salt content of the groundwater or due to insufficient drainage or irrigation with salt-rich water
acidification	the decrease in soil pH due to e.g. natural acidification of the land, high application of ammonium-based nitrogen fertilizers, or continual removal of plant residues from the land
nutrient depletion	the relative decrease of available soil nutrients to the plants due to insufficient fertilizer use, intensive land management, unbalanced fertilizes and nutrient losses due to wind and water erosion
loss of soil organic matter	the loss of organic matter components (micro-organisms, fresh and partially decomposed residues and well-decomposed stable organic material – humus-) due to the continual removal of plant residues, heavy tillage or the absence of cover crops, rotations or organic fertilizers (compost and manure)

Waste disposal at all stages in the chain

Please make an evaluation for the different representative production systems (GM, non-GM, organic and family farming) about…

critical stages in the soy chain where waste is being produced: production level, crushing level,…? (agrochemicals, …)

- Do they pose a risk on air/water/soil pollution?

Are you aware of innovations in the waste management at production/crushing/storage/… level?

2. Economic criteria

Investments being made at all stages of the supply chain

Please make an evaluation for the different representative production systems (GM, non-GM, organic and family farming) about…

- Business investments in the following aspects, please verify in which stage of the supply chain:
  - research and development (to increase efficiency of operations, to increase productivity, to reduce losses, to decrease environmental impact, to diversify products,…)?
  - employee education (improving skills of the employees, capacity building,…)?
  - measures and facilities that improve sustainability performance (decreasing environmental impact, improving social wellbeing of workers, improving economic resilience)? Please verify with some examples.

Local economy

Please make an evaluation for the different representative production systems (GM, non-GM, organic and family farming) about…

- length of the chain from producer to consumers (stakeholders involved in the chain)?
- the value added locally, regionally? And in which stages is the highest value added (looking at profit margins?)

- the length of the chain in relation of the value added? (i.e. intermediaires and middlemen participating in the chain without adding value to the product?)

Economic vulnerability

Please make an evaluation for the different representative production systems (GM, non-GM, organic and family farming) about...

The dimensions regarding economic vulnerability:

- the unavailability of or inability to get **access to cash** to pay your debts: i.e. debts with suppliers or lenders
- non-payments or **suspended payments of buyers** resulting in periods of insufficient cash flow/working capital
- unavailability of **essential inputs,** i.e. the unavailability of or inability to get access to essential inputs like land, water, seeds, fertilizers, electricity, fuel, etc. interrupting your production/business activities
- unavailability of **workforce**, i.e. the unavailability to attract and keep qualified employees or workforce capable of doing specific tasks in your business, interrupting your production/business activities
- **Climatic of political factors** interrupting production/business activities, i.e. climatic factors (erratic weather conditions, flooding, prolonged periods of drought, etc.) and political factors (legitimate disputes on land ownership, legal restrictions, inadequate law enforcement,...)
- Dependence upon the most dominant **supplier** of inputs
- Dependence upon the most dominant **buyer** of products
- The **inability to sell** your products, i.e. the unavailability of buyers or the inability to get access to a markets due to problems of infrastructure, transport, market information, quality requirements, legal requirements, food safety regulations,...

Alternative lenders for a loan?
Alternative buyers?
Alternative input suppliers?

3. Social criteria

Labour rights

Please make an evaluation for the different representative production systems (GM, non-GM, organic and family farming) about...

- incidents of child labour during the last 5 years? Explain
- incidents of forced labour during the last 5 years? Explain
- incidents of human rights violations during the last 5 years? Explain
- the right to form or adhere to an association defending workers' rights (i.e. labour union, workers' association, etc.)?
- Please verify the actions taken by the company or the workers' association to address the freedom of association and bargaining?

Working conditions, health & safety

Please make an evaluation for the different representative production systems (GM, non-GM, organic and family farming) about…

- The share of workforce with legally recognized work contracts?
- The share of workforce with temporary or seasonal work contracts?
- The share of workforce with pension or security benefits?
- access to facilities at the work place (clean water, sanitary facilities, etc.)
- The share of workforce doing dangerous work?
    - Access to adequate training?
    - access to protective gear and medical assistance (masks, protective clothes, …)?
- Accidents, injuries or illnesses at the work place (frequency and severity)
- The extent and effectiveness of actions taken by the companies in the chain to address the physical work environment

Capacity building

Please make an evaluation for the different representative production systems (GM, non-GM, organic and family farming) about…

Attracting and employing qualified workforce and subcontractors

- Attracting regionally hired workers?
- Attracting migrant workers?
- Employing women?
- Difference in age of workforce?

- Access to training and education at the workplace

Equity

Please make an evaluation for the different representative production systems (GM, non-GM, organic and family farming) about…

- What is the wage gap for comparable work between: men and women
- What is the wage gap for comparable work between: permanent and temporary staff
- What is the wage gap for comparable work between local and migrant workers
- What is the average number of training days per worker per year? (#/year)

- Please mention if you are aware of any form of discrimination in the company and verify? (on access to trainings and career development; harassment against women, minorities, migrants etc.; recruitment; remuneration; personnel development,…)

Food & Nutrition security

Please make an evaluation for the different representative production systems (GM, non-GM, organic and family farming) about…

- The number of production sites located in food insecure regions?
- Contribution to local food security ?
- Contribution to global food security?