Exergy-based natural resource accounting in sustainability assessment of agricultural production systems

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Thesis submitted in fulfillment of the requirements
for the degree of Doctor (PhD) in Applied Biological Sciences

Titel van de doctoraatsthesis in het Nederlands:

Exergie-gebaseerde berekening van grondstoffenverbruik binnen duurzaamheidsevaluatie van landbouwproductiesystemen

Cover design: Sophie Huysveld (photos from www.istockphoto.com)

Please refer to this work as follows:

Huysveld, S. (2016) Exergy-based natural resource accounting in sustainability assessment of agricultural production systems. *PhD Thesis.* Ghent University, Belgium.

ISBN-number: 978-90-5989-908-7

This work was financially supported by the Institute for Agricultural and Fisheries Research (ILVO, bursaal: 2011/09/01 – 2011/12/31) and the Institute for the Promotion of Innovation by Science and Technology in Flanders (IWT, SBO – 111297: 2012/01/01 – 2015/12/31).

SUMMARY

Concerns about the impact of human activities on the environment gradually increased during the past half century. The high living standard in developed regions has been built upon higher exploitation of natural resources, of which fossil resources are the best known example. Environmental concerns related to agricultural activities started to rise after the Green Revolution, a very prolific period for agricultural research and development, leading to major crop yield increases. These increases, achieved with higher material and energy inputs (fertilisers, pesticides, irrigation, machinery, etc.), were associated with a diverse range of environmental burdens (climate change, water pollution, etc.). In the search for mitigation of these impacts, environmental impact assessment studies have been increasingly performed. To cover all phases of production chains, assessments that consider the life cycle perspective, i.e. Life Cycle Assessments (LCAs), are used. Initially, these assessments were mainly focused on emission problems. This has resulted in many adequate end-of-pipe techniques for waste treatment and emission reduction. This emission-oriented approach gradually shifted towards more resource-oriented approaches and the adoption of clean technologies to prevent pollution. Given the increasing scarcity of natural resources and the value that they represent for human activities, resource-oriented approaches are highly relevant. Assessment methods based on the concept of exergy have proved to be particularly suitable for overall natural resource accounting and efficiency assessment. Both material and energy flows can be quantified on a single scale, i.e. exergy joule (J_{ex}). Exergy analysis, however, has been elaborated in the energy, chemical and metallurgical industries primarily and, therefore, it needs further development to assess overall natural resource use and its efficiency in an agricultural context. The **general objective** of this PhD thesis was to improve the framework of exergy-based natural resource accounting for its application within sustainability assessment of agricultural production systems, and to provide insight into its value by case study illustrations.

This PhD thesis starts with a general introduction (**Chapter 1**), including three sections. The first section deals with sustainable agriculture, and includes a historical overview of the meaning of sustainable agriculture, followed by a presentation of the current concerns, trends and challenges, mainly from an environmental viewpoint. Over the next decades, agriculture will be challenged by a number of developments. Due to the ongoing growth of the world population, global demand for food is projected to increase. While people in the developed world generally already have high intake levels of animal-based food products, increasing urbanization and income growth in less developed regions of the world will lead to dietary changes towards a higher proportion of animal-based food products. This will drive an increased demand for animal feed. Growth in livestock production rises environmental concerns, because environmental problems caused, directly and indirectly, by livestock production occur at every scale from local to global. Additionally, agriculture will be challenged in the next decades by a rising demand for biomass in the emerging bioeconomy, which is an important strategy towards a more sustainable production of energy and materials that makes us less dependent on finite stocks of fossil resources. This rise in demand for biomass, however, will put more pressure on the limited amount of available bio-productive land in the world, leading to a growing competition for land between food, feed, biomaterials and bioenergy. Moreover, increasing biomass yields to avoid area expansion into natural habitats may induce other environmental problems and threaten long-term productivity of the soil.

The second section of the first chapter elaborates on environmental sustainability assessment, and more specifically on LCA. After explaining the four-step framework of the LCA methodology, an overview of different types of resource-oriented assessments is given, followed by a focus on exergy-based resource accounting, including an explanation of the exergy concept and providing insight into its main applications.

The third section of the first chapter provides the aims and the outline of this PhD thesis. The focus of this PhD thesis is twofold. Thematically, this work focuses on two major challenges within the current debate on sustainable development of agriculture, i.e. (i) the growing demand for bio-based products to substitute their fossil-based counterparts in a bioeconomy, and (ii) the increasing environmental concerns about intensive livestock production, which is narrowed down to dairy farms in this thesis.

Methodologically, this work considers the exergy accounting methodology to evaluate (cumulative) overall natural resource use and its efficiency. This third section also formulates five specific objectives that will be addressed in Chapters 2 to 4, in order to achieve the general objective of this PhD thesis.

Chapter 2 fills the gap in scientific literature about how to calculate a cumulative overall natural resource efficiency in an agricultural context by developing an improved exergy-based framework, called Cumulative Overall Resource Efficiency Assessment (COREA). Guidelines about how to account for land resources in the calculation of overall natural resource efficiency were lacking, although it is essential to take them into account in an agricultural context. Moreover, in the context of the bioeconomy, this is very relevant because bio-based products potentially decrease consumption of fossil resources compared to their fossil-derived counterparts, but they are more demanding for bio-productive land use. The most appropriate way to account for bio-productive land resources as an input during the quantification of efficiencies was identified by analysing accounting principles for land resources of existing resource accounting methods (RAMs). While some existing RAMs did not include land resources, others had different accounting principles. A precondition of an adequate RAM for the purpose of efficiency calculation is that efficiencies higher than the upper limit on efficiency (i.e. 100%) are not achievable. The exergy-based resource accounting method *Cumulative Exergy Extraction from the Natural Environment (CEENE),* which takes into account land, water, minerals, metals, nuclear energy, fossil fuels, abiotic renewable energy and atmospheric resources, was concluded to be the most appropriate method for the quantification of a cumulative overall natural resource efficiency. With respect to land resources, the CEENE method has two versions (CEENE v2007 and CEENE v2013) that account in a different way for land resources. Because CEENE v2013 accounts for the potential natural net primary production (NPP) of the occupied land, efficiencies higher than 100% are theoretically achievable for human-made systems, because the actual NPP of agricultural cultivation can be higher than the potential natural NPP at a given location. CEENE v2007 accounts for 2% of the exergy content of the solar radiation on occupied land, which equals the upper limit on the gross primary production (GPP) of natural ecosystems. Because it was not clear whether this approach is sufficient to avoid that

efficiencies higher than 100% are reached in case of human-made systems, a scientifically sound upper limit for primary biomass production in human-made systems was sought by appealing to photosynthesis research. Two appropriate fractions of the solar radiation on occupied land were identified: (1) 4.8% is the theoretical maximum efficiency to convert solar surface radiation into harvestable (aboveground) biomass and (2) 2.3% is the global actually observed maximum efficiency to convert solar surface radiation into harvestable (aboveground) biomass. So, the developed COREA framework, based on the CEENE v2007 method, takes into account land resources by accounting for one of these two well-defined fractions of the exergy content of solar radiation on occupied land in human-made systems. Regarding the original CEENE v2007 method, we concluded that, with a status quo of the currently observed maximum achieved efficiency, efficiencies higher than 100% are not achievable with this method.
Furthermore, Chapter 2 also elaborates on the choice of the temporal system boundary of the studied primary biomass production system. A distinction should be made between monoculture systems, which usually grow only during a limited period of the year with the most favourable local conditions, and both perennial systems, which grow over several years, and multiple-cropping systems, which tend to grow several crops over a longer period thanks to a well-planned crop rotating system. From a resource efficiency viewpoint, it is most appropriate to account for an entire year of land occupation in all cases, which is then fully assigned to one (in case of monoculture or perennial systems) or more crop products (in case of multiple-cropping systems).

The effect of using different accounting principles for land resources and temporal system boundaries was illustrated with case studies, i.e. three cases at crop level and two cases at bio-based product level. Comparing the bio-based products with their fossil-based counterparts in terms of cumulative overall natural resource efficiency revealed higher efficiencies for the fossil-based products. This could be explained by a discrepancy in the way land resources and fossil resources were taken into account. While a fraction of the current solar exergy consumption of crops was taken into account, the ancient solar exergy consumption by fossil resources was not. In the final version of the COREA framework, this ancient solar exergy consumption was taken into account in order to address the non-renewable character of fossil resources. This resulted in higher efficiencies for the bio-based products.

Because many agricultural systems have become high input/high output systems under the influence of the Green Revolution, an evaluation of the overall natural resource use is very relevant to improve their environmental performance. **Chapter 3** demonstrates a generic exergy-based framework for the evaluation of the overall natural resource use of agricultural systems at both the process level and the life cycle level, by means of a case study of one specialized dairy farm in the region of Flanders (Belgium). At the process level, exergy analysis of the cattle herd was performed. Milk was produced with an exergy efficiency of 15.2%. More than half of the resources consumed by the dairy farm's herd was irreversibly lost. The remaining went for almost two-thirds to manure and methane emissions, while only one-third went to milk and animals awaiting slaughter. This analysis showed that the process of milk production has a rather low efficiency in converting resources into the desired product. The reduction of exergy losses in favour of an increase in milk yield requires a further increase of animal efficiency, which is subject to a biological limit. Besides milk production, the chemical exergy in the animal feed is expended in the biological metabolism (e.g. regulating constant body temperature, excretion of waste products, etc.), movement, growth and reproduction. Other potential improvements from a resource efficiency viewpoint could be sought in better utilizing the exergy-rich outputs manure and methane.

At the life cycle level, an overall natural resource footprint was calculated using the CEENE v2013 method. For the purpose of resource footprinting, CEENE v2013 is regarded as more appropriate compared to CEENE v2007: the potential natural NPP of occupied land is a better proxy for the resource value of land, because in addition to solar radiation other local conditions such as temperature, water availability and soil type are reflected by the potential natural NPP of occupied land. The supply of feed was by far the most resource-intensive part of the studied dairy production chain. With respect to the type of resources, land resources took the largest share in the resource footprint, followed by fossil resources. Comparison of different feed types for this case study on the basis of the overall natural resource footprint, showed that concentrates were on average 2.5 times more resource-intensive per kg dry matter than roughages, while wet by-products were 34 and 73% less resource-intensive than roughages and concentrates, respectively.

Besides representing the majority of natural resources extracted throughout the supply chain of the dairy farm, feed is the most important cost at dairy farms. It therefore plays a key role in the challenge of dairy farmers to produce in an environmentally sustainable, yet competitive way. In **Chapter 4**, it was investigated whether and how dairy farms in the region of Flanders could simultaneously reduce feed costs and overall natural resource use in the feed supply chain without reducing farm revenues. In other words, it was identified whether a specific farm could achieve an economic-exergetic win-win or whether this farm was in an economic-exergetic trade-off situation. To achieve this objective, exergy-based resource accounting using the CEENE v2013 method was integrated with frontier analysis, a methodology based on economic production theory. In this analysis, revenues from milk and meat (animals awaiting slaughter) were considered as a combined output that had to be maintained. Based on the data of the dairy farm population, frontier methods construct a 'best practice' efficiency frontier, representing how feed inputs can together be used most efficiently. How efficiently they are used, compared to the frontier, is expressed by a technical efficiency score. The frontier envelops the dairy farm population and the less technical efficient a farm is, the further it is located from that frontier. There is a clear difference between efficiencies quantified by frontier analysis (Chapter 4) and the exergy efficiency (Chapters 2 and 3). While the first type of efficiency reflects the distance from the optimum in an existing population, the exergy efficiency reflects the distance from the thermodynamic optimum.

Three commonly used frontier approaches were applied to the same dataset of 103 specialized dairy farms in Flanders. Overall, the results showed that for almost all farms cost and overall natural resource savings could simultaneously be made. These improvements could mainly be obtained by increasing technical efficiency (proportionally minimizing both feed inputs), rather than by substituting the feed inputs (kilograms of purchased concentrates and by-products versus costs for on-farm produced roughages) in cost or CEENE minimizing proportions. The optimal allocation of the feed inputs was reflected from both a cost and a CEENE allocative efficiency viewpoint. Increasing both technical and cost or CEENE allocative efficiency led to the maximum achievable savings in terms of feed costs or overall natural resource use of the feed supply chain, respectively. While increasing technical efficiency always led to

an economic-exergetic win-win, not all farms could achieve an economic-exergetic win-win by input substitution. When the implied substitution to reduce costs was opposite to the implied substitution to reduce CEENE, an economic-exergetic trade-off occurred. Whether an economic-exergetic win-win could be achieved by substitution was farm-specific. Although frontier analysis was very suitable to analyse farm-specific win-wins and trade-offs, further research in correctly constructing the frontier is needed, because it influences the quantified improvement margins and the diagnosis of win-win and trade-off situations. The frontier methodology, therefore, still has to take some substantial steps in further methodological development in order to be reliable for farm-specific decision support.

In order to better understand the underlying characteristics that may explain dairy farm economic and exergetic performances, frontier analysis was combined in a next step with analysis of Key Performance Indicators (KPIs), which are traditionally used by farmers and their advisors. Combination of frontier analysis with analysis of Key Performance Indicators (KPIs) allowed identification of improvable KPIs. An example is the costs for on-farm roughage production per ha, which was significantly lower at farms with high cost and overall natural resource efficiencies. The improvable KPIs can be used as starting points in benchmarking exercises to steer farmers towards appropriate changes in their farm management.

Consulting farm advisors and other agricultural experts with the results of this work provided additional insights that were valuable to this research and future research. An important feedback for future research was the need to visualize the effects of improving KPIs on the farm performances through simulation. Feedback also included the need for analyses over longer time periods in order to see the evolution of farm performances in relation to their KPIs and to analyse the effects of strategic decisions on long-term farm performances.

Chapter 5 includes a general discussion of the results obtained during this thesis. First, this final chapter provides insight into the value of the exergy accounting methodology within sustainability assessment of agricultural production systems. The strengths of the exergy accounting methodology are illustrated with results from the case studies in the previous chapters. A critical view on the exergy accounting methodology follows with

some suggestions for potential further development. Second, the final chapter discusses efforts that were made to translate research into practice in order to support the decision-making of farmers. Finally, concluding remarks with respect to both thematic and methodological issues are provided.

SAMENVATTING

Onze bezorgdheid over de impact van menselijke activiteiten op het milieu is geleidelijk aan toegenomen tijdens de afgelopen halve eeuw. De hoge levensstandaard in de ontwikkelde regio's ging gepaard met een grotere exploitatie van natuurlijke grondstoffen, waarvan fossiele grondstoffen het bekendste voorbeeld zijn. De bezorgdheden over de impact van landbouwactiviteiten op het milieu begonnen toe te nemen na de Groene Revolutie, een zeer vruchtbare periode op vlak van landbouwonderzoek en -ontwikkeling die geleid heeft tot grote toenames in gewasopbrengst. Deze opbrengststijgingen werden bereikt door een groter gebruik van materialen en energie, vervat in meststoffen, gewasbeschermingsmiddelen, irrigatie, machines, enz., en gingen gepaard met een brede waaier aan milieuproblemen zoals klimaatopwarming, watervervuiling, enz. In de zoektocht om deze impacten op het milieu te verminderen, werden milieu-impactstudies in toenemende mate uitgevoerd. Om rekening te houden met alle fasen van de productieketens, worden evaluaties uitgevoerd die de levenscyclus beschouwen, zogenaamde levenscyclusanalyses (LCA's). Aanvankelijk waren deze evaluaties vooral gericht op het bestrijden van emissies, maar dit verschoof geleidelijk aan naar meer grondstoffen-georiënteerde benaderingen en de invoering van schone technologieën om vervuiling te voorkomen. Gegeven de toenemende schaarste aan natuurlijke grondstoffen en de waarde die zij hebben voor menselijke activiteiten, zijn grondstoffen-georiënteerde benaderingen zeer relevant. Evaluatiemethoden gebaseerd op het concept van exergie hebben bewezen bijzonder geschikt te zijn voor het kwantificeren van het totale gebruik van natuurlijke grondstoffen. Zowel materiaal- als energiestromen kunnen gekwantificeerd worden op één enkele schaal, namelijk exergie joules. Omdat exergieanalyse voornamelijk ontwikkeld is voor toepassing in de energiesector en in chemische en metallurgische industrieën, is een verdere ontwikkeling van de methode nodig om het totale grondstoffengebruik en zijn efficiëntie te evalueren in een landbouwcontext. De **algemene doelstelling** van deze doctoraatsthesis was het verbeteren van het methodologische kader van exergie-gebaseerde kwantificering van natuurlijk

grondstoffengebruik voor toepassing binnen duurzaamheidsevaluaties van landbouwproductiesystemen, en om inzicht te verschaffen in zijn waarde door middel van gevalsstudies.

Deze doctoraatsthesis start met een algemene inleiding (**Hoofdstuk 1**), ingedeeld in drie delen. Het eerste deel gaat over duurzame landbouw, en omvat een historisch overzicht van de betekenis van duurzame landbouw, gevolgd door een uiteenzetting van de huidige problemen, tendensen en uitdagingen, vooral vanuit milieustandpunt bekeken. In de komende decennia zal de landbouw geconfronteerd worden met een aantal ontwikkelingen. Door de toenemende groei van de wereldbevolking wordt verwacht dat ook de mondiale vraag naar voedsel zal toenemen. Terwijl mensen in ontwikkelde regio’s over het algemeen al een hoge inname van dierlijke voedingsproducten hebben, zullen toenemende verstedelijking en inkomensstijging in de minder ontwikkelde regio's leiden tot veranderingen in het dieet in de richting van een groter aandeel dierlijke voedingsproducten. Bijgevolg zal ook de vraag naar diervoeders toenemen. Een verdere groei in dierlijke productie versterkt onze milieubezorgdheden, omdat de directe en indirecte milieuproblemen die hierbij ontstaan zich manifesteren op elk niveau, van lokaal tot mondiaal. Daarnaast zal de landbouw in de komende decennia worden geconfronteerd met een stijgende vraag naar biomassa door de opkomende bio-economie. Deze bio-economie is een belangrijke strategie naar een meer duurzame productie van energie en materialen, en maakt ons minder afhankelijk van eindige fossiele grondstofvoorraden. Maar, de stijgende vraag naar biomassa zal meer druk leggen op de beperkte hoeveelheid beschikbare bioproductieve landoppervlakte in de wereld. Dit zal op zijn beurt de concurrentie om land tussen humaan voedsel, diervoer, biomaterialen en bio-energie versterken. Het verhogen van biomassaopbrengsten, in een poging om landuitbreiding in natuurlijke habitats te vermijden en de totale primaire productie te verhogen, kan bovendien leiden tot andere milieu-impacten en kan de productiviteit van de bodem op lange termijn in het gedrang brengen.

Het tweede deel van het inleidende hoofdstuk gaat dieper in op milieuduurzaamheidsevaluaties, en meer specifiek op LCA. Na het uitleggen van het 4-stappenkader van de LCA-methodologie, is een overzicht gegeven van verschillende grondstoffen-georiënteerde evaluatiemethodes, gevolgd door een deel over

exergie-gebaseerde kwantificering van grondstoffengebruik, waarbij het concept exergie wordt uitgelegd en inzicht wordt gegeven in zijn belangrijkste toepassingen.
Het derde deel van het inleidende hoofdstuk beschrijft de doelstellingen en de indeling van deze doctoraatsthesis. De focus van deze doctoraatsthesis is tweeledig. Thematisch focust dit werk op twee belangrijke uitdagingsgebieden binnen de huidige duurzame ontwikkeling van de landbouw, i.e. (i) de stijgende vraag naar bio-gebaseerde producten om hun fossiele alternatieven te vervangen in een bio-economie, en (ii) de toenemende milieubezorgdheden over intensieve dierlijke productie, waar we ons in deze doctoraatsthesis toespitsen op melkveebedrijven. Methodologisch beschouwt dit werk de exergiemethodologie om het totale natuurlijke grondstoffengebruik en zijn efficiëntie te evalueren. Dit derde deel formuleert ook vijf specifieke doelstellingen, die behandeld zullen worden in Hoofdstukken 2 tot 4, om de algemene doelstelling van deze doctoraatsthesis te realiseren.

Hoofdstuk 2 vult de lacune in de wetenschappelijke literatuur over hoe een cumulatieve efficiëntie van totaal natuurlijk grondstoffengebruik in een landbouwcontext te berekenen, door middel van de ontwikkeling van een verbeterd exergie-gebaseerd kader, de zogenaamde *Cumulative Overall Resource Efficiency Assessment (COREA)*. Richtlijnen over hoe landgebruik mee te nemen in de berekening van de efficiëntie van totaal natuurlijk grondstoffengebruik ontbraken, hoewel het essentieel is om dit in rekening te brengen in een landbouwcontext. In de context van de bio-economie is dit zeer relevant omdat bio-gebaseerde producten het potentieel hebben om het gebruik van fossiele grondstoffen te verminderen, maar ze hebben een grotere vraag naar bioproductieve landoppervlakte. De meest geschikte manier om bioproductieve landoppervlaktes mee te nemen in de kwantificering van efficiëntie werd geïdentificeerd door het analyseren van bestaande grondstoffenmeetmethoden. Terwijl sommige grondstoffenmeetmethoden landgebruik niet in rekening brengen, hebben andere verschillende benaderingen. Een voorwaarde voor een geschikte methode voor het berekenen van een cumulatieve efficiëntie van totaal natuurlijk grondstoffengebruik is dat efficiënties hoger dan 100% niet realiseerbaar mogen zijn. De exergie-gebaseerde grondstoffenmeetmethode *Cumulative Exergy Extraction from the Natural Environment (CEENE)*, die land, water, mineralen, metalen, nucleaire energie, fossiele grondstoffen, abiotische hernieuwbare energie en atmosferische hulpbronnen in rekening brengt, werd geïdentificeerd als de meest geschikte methode voor de berekening van een cumulatieve efficiëntie van totaal natuurlijk grondstoffengebruik. Wat betreft landgebruik, bestaan er twee versies van de CEENE methode (CEENE v2007 en CEENE v2013) die landgebruik op een verschillende manier in rekening brengen. Omdat CEENE v2013 de potentieel natuurlijke netto primaire productie (NPP) van het gebruikte land in rekening brengt, zijn efficiënties hoger dan 100% theoretisch haalbaar voor niet-natuurlijke systemen, omdat de NPP bij landbouwproductie hoger kan zijn dan de potentieel natuurlijke NPP op een gegeven locatie. CEENE v2007 brengt 2% van de exergie-inhoud van zonnestraling op gebruikt land in rekening, wat gelijk is aan de bovengrens voor bruto primaire productie (BPP) van natuurlijke systemen. Omdat het niet zeker was dat deze benadering voldoende is om te vermijden dat efficiënties hoger dan 100% realiseerbaar zijn in niet-natuurlijke systemen, werd een wetenschappelijk onderbouwde bovengrens voor primaire productie in niet-natuurlijke systemen gezocht door beroep te doen op fotosyntheseonderzoek. Twee geschikte fracties van zonnestraling op gebruikt land werden geïdentificeerd: (1) 4,8% is de theoretische maximale efficiëntie waarmee planten zonnestraling omzetten in oogstbare (bovengrondse) biomassa en (2) 2,3% is de mondiaal werkelijk waargenomen maximale efficiëntie van planten om zonnestraling om te zetten in oogstbare (bovengrondse) biomassa. Zo neemt het ontwikkelde COREA kader, gebaseerd op de CEENE v2007 methode, landgebruik in niet-natuurlijke systemen mee in rekening door middel van een van deze twee goed gedefinieerde fracties van de exergie-inhoud van zonnestraling op de gebruikte landoppervlakte. Wat betreft de originele CEENE v2007 methode kunnen we besluiten dat, met een status quo van de huidige werkelijk waargenomen maximale efficiëntie, efficiënties hoger dan 100% niet bereikbaar zijn met deze methode.

Daarnaast gaat Hoofdstuk 2 ook in op de keuze van de temporele systeemgrens van het bestudeerde primaire biomassaproductiesysteem. Een onderscheid dient gemaakt te worden tussen monocultuursystemen, die doorgaans groeien gedurende een beperkte periode van het jaar met gunstige lokale omstandigheden, en zowel meerjarige systemen, die over verschillende jaren groeien, en meervoudige teeltsystemen, die verschillende gewassen over een langere periode telen op basis van een gewasrotatieplan. Vanuit het oogpunt van grondstoffenefficiëntie, is het in rekening

brengen van een volledig jaar vereist in alle gevallen, waarbij dit jaar dan wordt toegekend aan één (in het geval van monocultuursystemen en meerjarige systemen) of meerdere gewassen (in het geval van meervoudige teeltsystemen).
Het effect van verschillende meetmethoden voor landgebruik werd geïllustreerd met gevalstudies, namelijk drie gevallen op gewasniveau en twee gevallen op het niveau van het finale bio-gebaseerde product. Vergelijken van deze bio-gebaseerde producten met hun fossiele alternatieven in termen van cumulatieve efficiëntie van totaal natuurlijk grondstoffengebruik onthulde hogere efficiënties voor de fossiel-gebaseerde producten. Dit kon verklaard worden door de tegenstrijdigheid in de manier waarop landgebruik en gebruik van fossiele grondstoffen in rekening werden gebracht. Terwijl het huidige gebruik van zonne-exergie door gewassen werd meegenomen, werd het eeuwenoude gebruik van zonne-exergie door fossiele grondstoffen niet meegenomen. In de finale versie van het COREA kader werd dit eeuwenoude gebruik van zonne-exergie wel in rekening gebracht om het niet-hernieuwbare karakter van fossiele grondstoffen correct weer te geven. Dit resulteerde in hogere efficiënties voor bio-gebaseerde producten.

Omdat vele landbouwsystemen, onder invloed van de Groene Revolutie, hoge input/hoge output-systemen zijn geworden, is de evaluatie van het totale gebruik van natuurlijke grondstoffen zeer relevant om hun milieuprestaties te verbeteren. **Hoofdstuk 3** demonstreert een algemeen exergie-gebaseerd kader voor de evaluatie van totaal natuurlijk grondstoffengebruik van landbouwsystemen, zowel op procesniveau als op levenscyclusniveau, door middel van een gevalstudie van een gespecialiseerd Vlaamse melkveebedrijf. Op procesniveau werd een exergieanalyse ter hoogte van de kudde uitgevoerd. Melk werd geproduceerd met een exergie-efficiëntie van 15,2%. Meer dan de helft van de verbruikte grondstoffen door de kudde ging onherroepelijk verloren. De resterende verbruikte grondstoffen werden voor bijna tweederde omgezet in mest en methaanemissies, terwijl een derde naar melk en slachtdieren ging. Deze analyse toonde aan dat het melkproductieproces een eerder lage efficiëntie heeft in het omzetten van grondstoffen in het beoogde product. De reductie van exergieverliezen ten gunste van een stijging in melkopbrengst vereist een verdere toename van de dierlijke efficiëntie, die onderworpen is aan een biologische

limiet. Naast melkproductie, wordt de chemische exergie in het diervoeder verbruikt in het biologische metabolisme (bv. regelen van lichaamstemperatuur, excretie van afvalstoffen, enz.), beweging, groei en reproductie. Een andere mogelijke verbetering vanuit het oogpunt van grondstoffenefficiëntie zou kunnen gezocht worden in het beter valoriseren van de exergie-rijke stromen mest en methaan.
Op levenscyclusniveau werd een totale natuurlijke grondstoffenvoetafdruk berekend met de CEENE v2013 methode. Voor het berekenen van een grondstoffenvoetafdruk is de CEENE v2013 methode beter geschikt dan de CEENE v2007 methode: de potentieel natuurlijke NPP van gebruikt land is een betere benadering van de grondstofwaarde van land, omdat naast zonnestraling andere lokale omstandigheden zoals temperatuur, waterbeschikbaarheid en bodemtype weerspiegeld worden door de potentieel natuurlijke NPP van gebruikt land. Voedervoorziening was veruit het meest grondstoffenintensieve deel van de bestudeerde melkproductieketen. Op vlak van type grondstoffen vertegenwoordigde landgebruik het grootste aandeel van de grondstoffenvoetafdruk, gevolgd door fossiele grondstoffen. Vergelijking van verschillende types voeders voor de gekozen gevalstudie toonde aan dat krachtvoeders per kg droge stof gemiddeld 2,5 keer meer grondstoffenintensief waren dan ruwvoeders, terwijl natte bijproducten 34 en 73% minder grondstoffenintensief waren dan ruwvoeders en krachtvoeders, respectievelijk.

Voeder is, naast vertegenwoordiger van het grootste aandeel van het natuurlijke grondstoffenverbruik doorheen de toevoerketen van het melkveebedrijf, ook de grootste kost op melkveebedrijven. Daarom speelt het een belangrijke rol in de uitdaging van melkveehouders om te produceren in een milieuvriendelijke, maar ook competitieve manier. In **Hoofdstuk 4** werd onderzocht of en hoe melkveebedrijven in Vlaanderen tegelijkertijd voederkosten en totaal natuurlijk grondstoffengebruik kunnen reduceren zonder een verlies aan bedrijfsopbrengsten. Met andere woorden, er werd geïdentificeerd of een bepaald bedrijf een economisch-exergetische win-win kon behalen of dit bedrijf te maken had met een economisch-exergetisch conflict (*trade-off*). Om dit doel te bereiken, werd exergie-gebaseerde kwantificering van grondstoffen via de CEENE v2013 methode gecombineerd met grenslijnanalyse, een methodologie op basis van economische productietheorie. In deze analyse werden opbrengsten van melk en vlees (slachtdieren) samen als één constant te houden output beschouwd. Gebaseerd op gegevens van een populatie van melkveebedrijven, construeren grenslijnmethoden een 'beste praktijk' grenslijn, die voorstelt hoe voederinputs samen het meest efficiënt ingezet kunnen worden. Hoe efficiënt zij gebruikt worden, ten opzichte van de grenslijn, wordt uitgedrukt in een technische efficiëntiescore. De grenslijn omsluit de populatie van melkveebedrijven en hoe minder technische efficiënt een bedrijf is, hoe verder het zich bevindt van de grenslijn. Er bestaat een duidelijk verschil tussen efficiënties gekwantificeerd door grenslijnanalyse (Hoofdstuk 4) en de exergie efficiëntie (Hoofdstukken 2 en 3). Terwijl de eerste efficiëntie de afstand van het optimum in een bestaande populatie meet, weerspiegelt de exergie efficiëntie de afstand van het thermodynamische optimum.

Drie veelvuldig gebruikte grenslijnbenaderingen werden toegepast op dezelfde gegevensreeks van 103 gespecialiseerde Vlaamse melkveebedrijven. Over het algemeen toonden de resultaten aan dat bijna alle bedrijven tegelijkertijd kosten- en grondstoffenbesparingen zouden kunnen realiseren. Deze verbeteringen zouden hoofdzakelijk bekomen kunnen worden door het verhogen van de technische efficiëntie (proportioneel beide voederinputs minimaliseren), eerder dan door substitutie van voederinputs (de hoeveelheid aangekochte krachtvoeders en bijproducten, uitgedrukt in kilogram, versus de kosten voor op het bedrijf geteeld ruwvoeder) in verhoudingen die kosten of grondstoffengebruik (CEENE) minimaliseren. De optimale verhouding van voederinputs werd weerspiegeld door de kosten of CEENE allocatieve efficiëntie. Het verhogen van zowel de technische als de allocatieve efficiënties leidde tot de maximaal bereikbare besparingen op het vlak van voederkosten en totaal natuurlijk grondstoffengebruik in de voederproductieketen. Terwijl het verhogen van de technische efficiëntie altijd leidde tot een economisch-exergetische win-win, konden niet alle bedrijven een economisch-exergetische win-win behalen door inputsubstitutie. Wanneer de voorgestelde substitutie om kosten te reduceren omgekeerd was ten opzichte van de voorgestelde substitutie om grondstoffengebruik te reduceren, was er sprake van een economische-exergetische trade-off. Of een economisch-exergetische win-win gerealiseerd kon worden door substitutie was bedrijfsafhankelijk. Hoewel grenslijnanalyse zeer geschikt is om bedrijfsspecifieke win-wins en trade-offs te analyseren, is verder onderzoek naar het correct construeren van de grenslijn nodig,

omdat het de mogelijke verbetermarges en de diagnose van win-win en trade-off situaties beïnvloedt. De grenslijnmethodologie moet daarom nog verdere methodologische vooruitgang boeken om de betrouwbaarheid van grenslijnanalyse voor bedrijfsspecifieke beslissingsondersteuning te verbeteren.
Om een beter inzicht te verkrijgen in de onderliggende kenmerken die de economische en exergetische performantie van melkveebedrijven zouden kunnen verklaren, werd grenslijnanalyse in een volgende stap gecombineerd met de analyse van Kritische Prestatie Indicatoren (KPI's). Deze laatste worden traditioneel gebruikt worden door landbouwers en hun adviseurs. Combinatie van grenslijnanalyse en analyse van KPI's liet toe om verbeterbare KPI's te identificeren. Een voorbeeld is de kosten voor op het bedrijf geteeld ruwvoeder uitgedrukt per hectare, die significant lager waren voor bedrijven met hoge efficiënties op vlak van voederkosten en totaal natuurlijk grondstoffengebruik in de voederproductieketen. De verbeterbare KPI's kunnen gebruikt worden als vertrekpunten in vergelijkingsoefeningen om landbouwers te ondersteunen richting de juiste aanpassingen in hun bedrijfsmanagement.

Hoofdstuk 5 omvat een algemene discussie van de resultaten bekomen in deze thesis. Vooreerst verstrekt dit laatste hoofdstuk inzicht in de waarde van de exergiemethodologie binnen duurzaamheidsevaluatie van landbouwproductiesystemen. De sterke punten van de exergiemethodologie worden geïllustreerd door middel van de resultaten van de gevalstudies in de vorige hoofdstukken. Een kritische kijk op de exergiemethodologie volgt met enkele suggesties voor mogelijke verdere ontwikkeling. Ten tweede bespreekt dit laatste hoofdstuk gemaakte inspanningen om onderzoeksresultaten te vertalen naar de praktijk, met het oog op ondersteuning van beslissingsvorming van landbouwers. Tot slot volgen enkele afsluitende opmerkingen met betrekking tot zowel thematische als methodologische aspecten.

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LIST OF ABBREVIATIONS

Abbreviation	Description
CAE	Cost Allocative Efficiency
CDP	Cumulative Degree of Perfection
CE	Cost Efficiency
CED	Cumulative Energy Demand
CEENE v2007	Cumulative Exergy Extraction from the Natural Environment version 2007
CEENE v2013	Cumulative Exergy Extraction from the Natural Environment version 2013
CEENE v2014	Cumulative Exergy Extraction from the Natural Environment version 2014
CEnC	Cumulative Energy Consumption
CExC	Cumulative Exergy Consumption
CExD	Cumulative Exergy Demand
CF	Characterization Factor
CHP	Combined Heat and Power
COREA	Cumulative Overall Resource Efficiency Assessment
CP	Crude Protein
CRS	Constant Returns to Scale
DDGS	Distillers Dried Grains with Solubles
DEA	Data Envelopment Analysis
DM	Dry Matter
DSS	Decision Support System
EA	Exergy Analysis
EAE	Environmental Allocative Efficiency
EE	Environmental Efficiency
ELCA	Exergetic Life Cycle Assessment

Abbreviation	Description
FPCM	Fat-and-Protein-Corrected Milk
GCV	Gross Calorific Value
GE	Gross Energy
GHG	Greenhouse gases
GPP	Gross Primary Production
KPI	Key Performance Indicator
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LOCF	Land Occupation Characterization Factor
NPP	Net Primary Production
OMCA	solar system boundary level based on the Observed Maximum Convertible part of surface solar radiation into Aboveground biomass
PAR	Photosynthetically Active Radiation / solar system boundary level based on the Photosynthetically Active part of total surface solar Radiation
PCE	Protein Conversion Efficiency
PVC	polyvinyl chloride
RAM	Resource Accounting Method
SED	Solar Energy Demand
SEF_i	Solar Energy Factors for each type of resource flow i based on the Baseline Emergy Budget
SFA	Stochastic Frontier Analysis
TE	Technical Efficiency
TMC	solar system boundary level based on the Theoretical Maximum Convertible part of surface solar radiation into total biomass
TMCA	solar system boundary level based on the Theoretical Maximum Convertible part of surface solar radiation into Aboveground biomass
TMR	Total Mixed Ration

Abbreviation	Description
TOT	solar system boundary level based on total surface solar radiation
VCM	vinyl chloride monomer
VRS	Variable Returns to Scale

CHAPTER 1

GENERAL INTRODUCTION, AIMS AND OUTLINE

CHAPTER 1: GENERAL INTRODUCTION, AIMS AND OUTLINE

1.1 Sustainable agriculture

Nowadays, sustainable agriculture receives a widespread interest; from farmers, over researchers and policy makers, to food industries and consumers. The pivotal place of sustainable agriculture in future developments is widely agreed upon. This diversity of interests, however, has blurred the concept of sustainable agriculture. Many different and sometimes contrasting interpretations exist about what should be included under the “umbrella” of sustainable agriculture. Surely, sustainable agriculture involves more than only one goal, it is a complex collection of objectives, which have economic, environmental and social motives. Trade-offs between different goals are, not surprisingly, part of this, making agricultural sustainable development not straightforward (Conway and Barbier, 1990). This section starts with a historical overview of the meaning of sustainable agriculture, followed by a presentation of the current concerns, trends and challenges, mainly from an environmental viewpoint.

1.1.1 A history of sustainable agriculture

Thinking about agricultural development is not peculiar to the present time. If we literally consider sustainable agriculture, we can trace incentives to “sustain” agriculture since its inception 10 000 years ago, which was called the Neolithic Revolution. Hunter-gatherers started to colonize attractive habitats and domesticate plants and animals (Bogucki, 2008). Agriculture allowed people to live at one place and, therefore, it was the main ingredient for civilization. Agricultural evolution always has been guided by the circumstances, the concerns and the needs of a particular time period. Because they are changing with time, agricultural development thinking has also changed with time (Harwood, 1990).

Since the early 1900s, two parallel agricultural developments evolved, i.e. industrial and alternative agriculture. Both movements had different views on how agriculture should be practiced. While industrial agriculture was conducted by the so-called *systematic agriculturalists*, who looked to the emerging agricultural support industries as their guide, alternative agriculture evolved from the so-called *scientific or natural agriculturalists*, who looked to nature as their guide (Harwood, 1990; Zimdahl, 2012).

The increased demand for food by a growing world population was a major driver for industrial agricultural development (Hazell and Wood, 2008). Industrial agriculture was supported by industries of machinery, fertilizers and pesticides. Mechanization spread rapidly in the first decades of the 1900s and lead to area expansions (Harwood, 1990). The roots of the chemical innovation in agriculture can be traced in the influential publication of Justus von Liebig, called *'Die organische Chemie in ihrer Anwendung auf Agricultur und Physiologie' (Organic Chemistry in its Application to Agriculture and Physiology),* in 1840 (Kirschenmann, 2004). Synthetic nitrogen became available after World War I, in which the Haber-Bosh process was developed for the manufacture of explosives (Lotter, 2003). The use of industrially produced fertilizers was followed by the use of pesticides; the latter knew a rapid expansion after World War II (Harwood, 1990; Zimdahl, 2012). The emerging use of the insecticide dichloro-diphenyl-trichloroethane (DDT) after 1945 for application in agriculture, but also to fight human illnesses like malaria, is a well-known example. Many years later, DDT was banned during the 1970s for its disastrous effects on the environment and on human health (Swanson, 2012). The use of both synthetic inputs, fertilizers and pesticides, resulted in rapid increases in crop yield. Industrialization of agriculture also stimulated specialization towards mono-cropping systems (Harwood, 1990; Zimdahl, 2012).

Alternative agricultural movements evolved as an answer to concerns about the rapidly expanding industrialization of agriculture. Alternative agriculture only selectively made use of industrial innovations, like mechanization, new crop varieties and soil nutrient testing. Three major movements can be distinguished in alternative agriculture in the 20th century, i.e. biodynamic agriculture, humus farming and organic agriculture (Harwood, 1990).

The biodynamic movement was launched by a series of agricultural lectures given by Steiner in 1924 (Bio-Dynamic Farming and Gardening Association, 1993). Steiner and his followers pointed the danger of synthetic fertilizers and pesticides for the biological health of the soil and the health of animals and people who come in touch with them. Principles that characterize biodynamic agriculture and gardening are diversification, composting, avoiding chemicals, local production and distribution, and using sound techniques, traditional or new. Steiner's ideas also included the recognition of cosmic and terrestrial forces on biological organisms (Harwood, 1990).

The humus farming movement, which peaked in the 1940s and early 1950s, also contested the use of synthetic fertilizers and focused on the major importance of the humus content of the soil in order to maintain its productivity. Advanced techniques for composting and compost use were established. *'The Field Book of Manures' or 'the American Mulch Book'*, written by Browne in 1855, was the first influential work within humus farming. Many years later, *'An Agricultural Testament'*, written by Howard in 1943, was a new milestone for humus farming (Harwood, 1990; Kirschenmann, 2004).

The biodynamic movement and the humus farming concept were the forerunners of what we know today as organic agriculture. While the biodynamic movement had a more spiritual background, which looked to the farm as a living organism, the humus farming movement introduced scientific knowledge about the soil. The term organic agriculture was coined by Northbourne in 1940 in his book *'Look to the Land'*, who stated the importance of biodiversity and warned for the harmful effects of synthetic inputs and large-scale monoculture on soil fertility. A decentralized and chemical-free agriculture was advocated (Harwood, 1990; Lotter, 2003; Paull, 2006). Very influential works for the development of the organic movement were, in the United States, Rodale's '*Pay dirt: farming and gardening with composts'* in 1945, and in Europe, Howard's '*The Soil and Health: A Study of Organic Agriculture'* in 1947. Many of the issues debated during the development phase of organic agriculture are still discussion points in today's debate on agricultural sustainability.

Despite of the alternative agricultural movements, industrial agriculture had become widespread in developed countries by the late 1950s. The success of the industrial innovation was overwhelming and low prices of fertilizers and pesticides stimulated crop specialization (Harwood, 1990). In the 1960s and 1970s, agricultural development thinking was preoccupied with the problem of feeding a rapidly growing world population. This gave rise to the so-called Green Revolution, a very prolific period for agricultural research and development, knowledge transfer and the spread of new technologies and high yielding crop varieties in high production potential areas (Conway and Barbier, 1990). Very influential was the work of Norman Borlaug, the so-called "father of the Green Revolution", who won the Nobel Peace Prize in 1970, for his contributions to world food security by supplying high-yielding and disease-resistant wheat varieties (Swaminathan, 2009). In 1971, an international consortium of funders and agricultural research centers, the *'Consultative Group on International Agricultural Research' (CGIAR)*, was established to reduce poverty and hunger, to improve human health and nutrition, and to prevent environmental degradation (http://www.cgiar.org). Besides the technological innovations and the stimulated homogeneity by genetically uniform high-yielding varieties, the increased use of fertilizers, pesticides, mechanization and irrigation (Figure 1.1) contributed to the yield rises during the Green Revolution (Hazell and Wood, 2008).

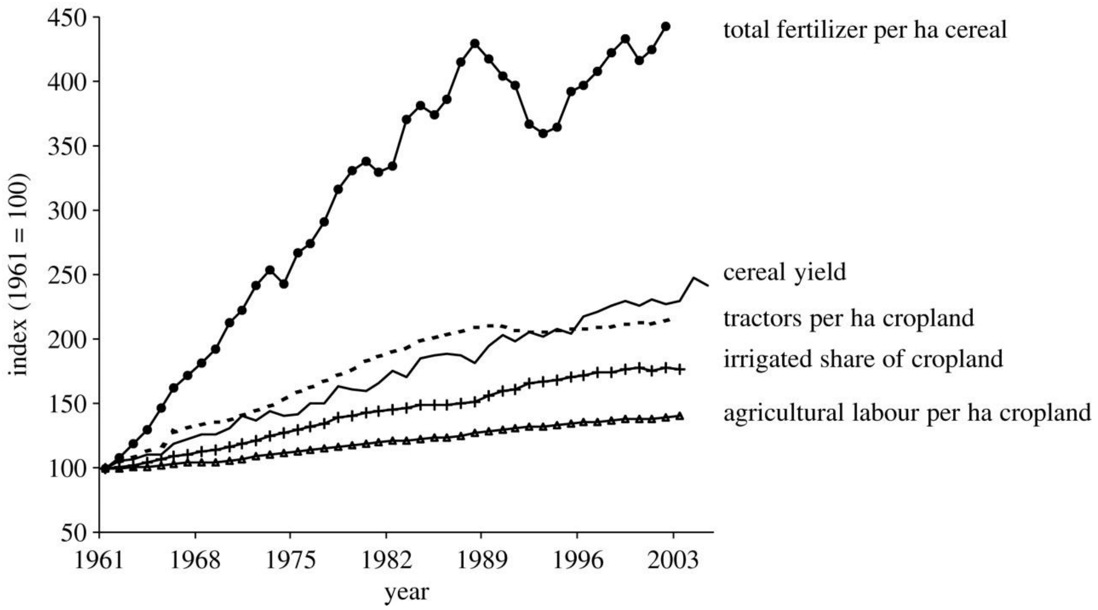


Figure 1.1 Global trends in the intensification of crop production (index 1961-2002/2005). Retrieved from Hazell and Wood (2008). Adapted from Cassman & Wood (2005), updated from FAOSTAT (2006, tractor and fertilizer data to 2002, land use to 2003, production to 2005).

Although the world population rapidly increased from the 1960s, world food production increased even faster, resulting in a steady rise of per capita food production (Figure

1.2). This has been accompanied with a downward trend in world food prices (except the world food crisis in the early 1970s due to tremendously increased oil prices) until the flattening out since the late 1980s. Starting from 2003, world food prices have risen and have become much more volatile, which was caused by several aspects: supply shocks, low stocks, rising energy prices and an increased global demand. While producers and net exporting countries may benefit from higher food prices, these higher prices increase food insecurity of poor consumers and may negatively affect net importing countries (FAO, 2009a). Since 2012, FAO reports lower and less volatile prices due to higher stocks and lower energy prices (FAO, 2015a).

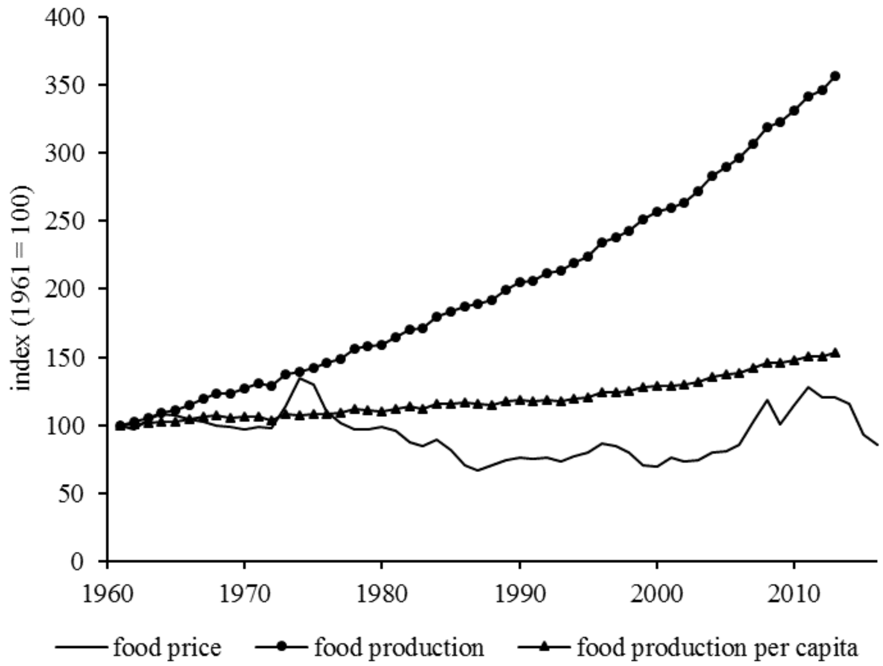


Figure 1.2 Global trends in food production and price (index 1961-2013/2016). Data from FAOSTAT (2016).

The Green Revolution had a major impact on food self-sufficiency and food security in developing countries in the 1970s and 1980s, where impressive yield increases of the major cereal staples (wheat, maize and rice) were achieved (Harwood, 1990). The percentage of people that live in famine worldwide declined from 26 to 14% between 1969-1971 and 2000-2002 (FAO, 2009b), and is estimated to be further reduced to 11% in 2016. According to the latest estimates, about 795 million people are currently undernourished (FAO, 2015b). Although the Green Revolution enormously reduced the number of undernourished people worldwide, it has also shown some major shortcomings in terms of equity, stability, and sustainability.

Technological innovations have mainly been implemented in regions with the most favourable agroclimatic conditions and by larger rather than smaller and poorer farms. Substandard conditions in terms of soil quality and access to water for irrigation have been large barriers for successful implementation (Conway and Barbier, 1990). In addition to a lack in investment capital and a limited access to infrastructure and knowledge, this can partly explain why in Africa the Green Revolution was not as successful as in the rest of the developing world (Figure 1.3). A very limited implementation of new technologies and a low application of modern inputs led to periods of decline or stagnation in food production per capita (Hazell and Wood, 2008). Although famine is more associated with poverty and poor access to food than inadequate food production (Matson et al., 1997), it has become clear that the Green Revolution has failed to ban hunger from the world. Hunger is mainly concentrated in South Asia and sub-Saharan Africa (IFAD, 2010).

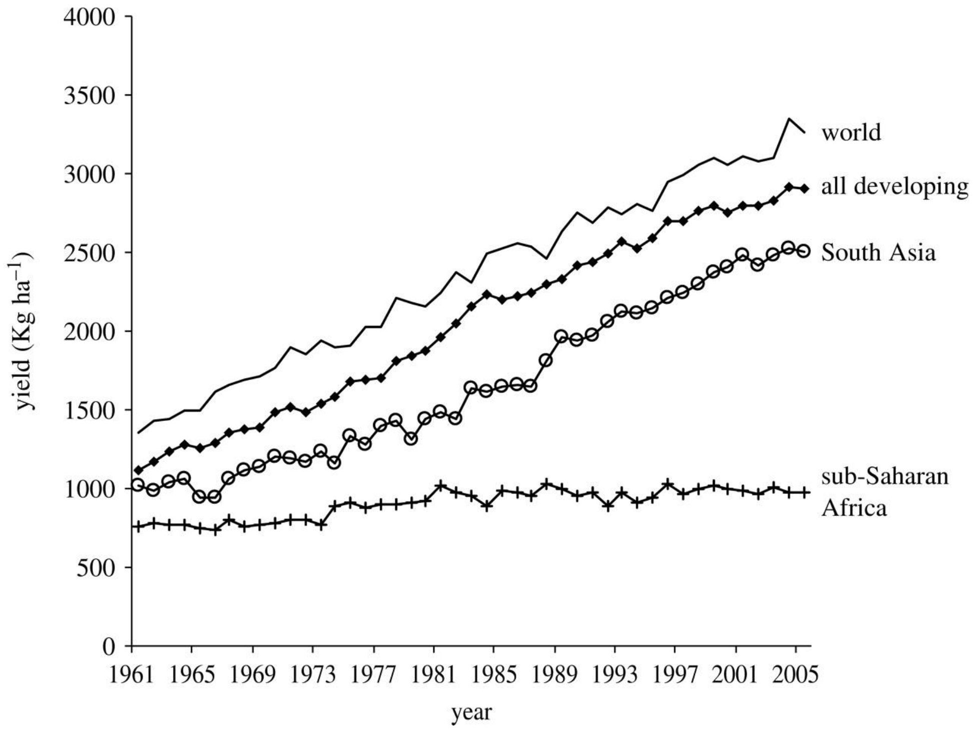


Figure 1.3 Global trends in cereal yield (kg ha-1) by region (1961-2005). Retrieved from Hazell and Wood (2008). Adapted from FAOSTAT (2006).

In terms of stability, the effect of the Green Revolution was also not exclusively positive. Increased output variability and increased incidence of diseases and weed problems have been associated with the widespread adoption of mono-cropping systems. Crop yields, which were increased by the implementation of modern inputs, appeared to be more sensitive to fluctuations in input use caused by shortages or price increases (Conway and Barbier, 1990).

In the 1970s and 1980s, awareness was gradually increasing about the negative effects of agricultural intensification on the environment; residues of pesticides were traced in food, nutrients were accumulating in ground and surface waters, increasing levels of soil erosion and degradation were noticed, poor irrigation management led to salinization, etc. Agricultural policies had been focusing too much on short-term growth; fertilizers were replacing soil quality management and herbicides were used for weed control instead of crop rotations. At the same time, people became aware about the limits of the natural resource base; irrigation was putting a high pressure on water resources and by the energy shortage of the early 1970s people realized that industrial agriculture was greatly dependent on fossil resources. It had become clear that all these environmental problems could endanger long-term productivity (Conway and Barbier, 1990; FAO, 2011b).

The abovementioned problems in terms of equity, stability and sustainability were extensively acknowledged in the report *'Our Common Future'* of the World Commission on Environment and Development (WCED) in 1987. In this highly influential report, sustainable development was defined as 'development that meets the needs of the present without compromising the ability of future generations to meet their own needs'. From this definition it is clear that sustainable development is a process of change, in which present generations should orient their decisions and activities from a long term perspective. Human exploitation of natural resources is occurring at a rate beyond the Earth's estimated carrying capacity and that is a threat for the future. Renewable resources like forests and fish stocks and non-renewable resources like oil reserves are overexploited. Our future welfare should be less dependent on non-renewable resources and current generations should find more sustainable material and energy sources. Moreover, human development is limited by the ability of the biosphere to absorb the harmful effects of human activities, e.g. greenhouse gas emissions. Besides the gap between now and later, the definition on sustainable development also highlights the urgent need to reduce the gap between rich and poor. The economic growth in the wealthier parts of the world consumes too many resources and produces too many harmful emissions at the expense of people in less prosperous parts of the world (Brundtland et al., 1987). Although the WCED report already emphasized the broad areas of concern, i.e. environmental, economic and social issues, Elkington introduced the *'Triple Bottom Line'* concept to state that sustainable development should result in benefits in the three P-areas: People, Planet and Profit (Elkington, 1999). This viewpoint intended to change the perception that profitability could not go hand in hand with environmental and social benefits. Elkington's work has been valuable to facilitate a more practical implementation of sustainable development in a business context.

So far, the debate about how sustainable agriculture should look like is still ongoing. In the first place, because achieving a higher sustainability is a complex task. That sustainable agricultural development should take into account the three sustainability dimensions is widely agreed upon. That a long-term viewpoint should be adopted in making decisions is supported by many. That agricultural production should be performed in a way that makes efficient use of natural resources and that eliminates or minimizes adverse effects on the environment is also broadly recognized (FAO, 2011b; Pretty, 2008; Tilman et al., 2002). But how the sustainable development idea should be translated into concrete improvement paths is not straightforward and, therefore, still under debate. In the second place, the debate about sustainable agriculture depends on the context of time and place-bound conditions and needs. Sustainable agricultural development should thus be case and region-specific (FAO, 2011b; Pretty, 2008; Tilman et al., 2002). The next section gives a general (not region-specific) overview of current concerns, trends and challenges, mainly from an environmental viewpoint.

1.1.2 Current concerns, trends and challenges

1.1.2.1 Livestock production

Over the next decades, agriculture will be challenged by a number of developments. World population continues to grow. By 2030, there will be more than one billion people more to feed, reaching 8.5 billion people. By 2050 and 2100, there will be more than two and almost four billion people more to feed, respectively, according to the medium-variant projection of the United Nations (2015). This growth will mostly take place in less developed regions of the world, where increasing urbanization and income growth will be additional drivers for an increased food demand. Simultaneously, these factors will lead to dietary changes towards a higher proportion of animal-based food products and a higher consumption of processed foods (FAO, 2011b; Thornton, 2010). In developed regions, which already have high intake levels of animal-based food products, consumption levels of animal-based food products grow only slowly or stagnate. These patterns are induced by consumer awareness about negative health effects of high intake levels of red meat and animal fats, e.g. cardio-vascular diseases and cancer. Increasing consumer concerns about animal welfare and negative environmental impacts of livestock production could further decrease the consumption of animal-based food products in developed regions in the next decades (FAO, 2006).

Livestock production is pulled by the consumption of livestock products; production is booming in developing regions, while it is growing slowly in the developed world (Figure 1.4 and Figure 1.5).

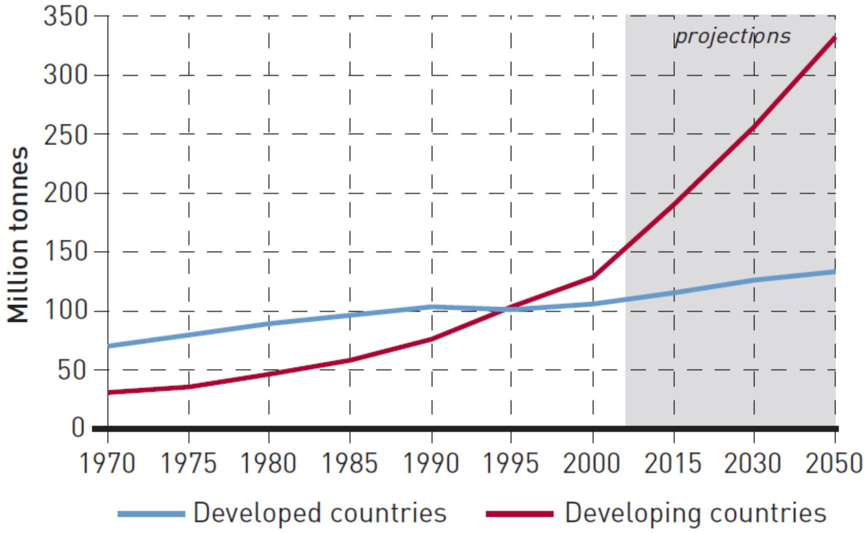


Figure 1.4 Past and projected meat production in developed and developing countries from 1970 to 2050. Retrieved from FAO (2006).

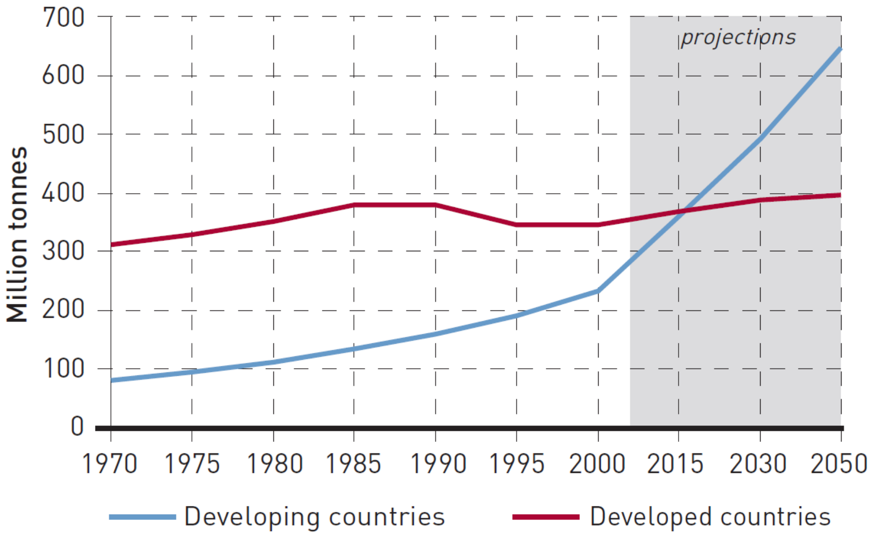


Figure 1.5 Past and projected milk production in developing and developed countries from 1970 to 2050. Retrieved from FAO (2006) (modified).

The largest increases since the 1980s occurred in developing countries that experienced the most rapid economic growth, particularly in Brazil, China and India. Whereas China contributed to the largest growth in meat production in the developing countries, India accounted for the highest rise in milk production. Remarkable is the impressive growth of poultry and pigs all over the world, while ruminant meat production has only grown relatively little in developing countries and declined in developed countries. The ongoing rapid growth in livestock production in developing regions will drive an increased demand for feed, particularly cereals and protein-rich processing by-products. While China and India will increasingly need to import feed, Brazil and Argentina can rely on their own expanded feed production (FAO, 2006).

Particularly the growth in livestock production is currently causing rising environmental concerns. The significant environmental impacts, at every scale from local to global, of the livestock sector were extensively acknowledged in the *'Livestock's long shadow'* report of the Food and Agriculture Organization of the United Nations (FAO, 2006). From land use change and land degradation, over water depletion and water pollution, climate change and air pollution, to loss of biodiversity, the environmental problems caused, directly and indirectly, by livestock production occur on a massive scale.

Livestock production uses 78% of all agricultural land and one-third of all arable land, which corresponds to 30% of the world's land surface. Of this land, about 13% is occupied for crop production, 36% is pastures with relatively high productivity and 51% is extensive pastures with relatively low productivity (FAO, 2006). Both by area expansion and intensification, livestock production has contributed/is contributing to significant environmental problems. By area expansion, livestock production has been (and is) a major player in land use change, such as deforestation, particularly of the Amazon forest in South-America, but also in sub-Saharan Africa and Southeast Asia. About 70% of the deforested Amazon land is used as pasture and the remaining is occupied by feed crops, mainly soybeans, whose processing by-product, soybean meal, is a major protein source for livestock feed. Most of the increase in feed (and food) demand during the past decades, however, has been met by intensification of land use rather than by land area expansion (FAO, 2006; Thornton, 2010) (Figure 1.6). High yields have been attained by an increased use of machinery and irrigation, and fossil-based inputs such as fuel, fertilizers and pesticides (Pretty, 2008). All over the world, these inputs have contributed to water pollution, biodiversity loss and harmful gaseous emissions.

By area expansion and intensification, livestock production is a major driver of land degradation. Besides deforestation, overgrazing of pastures, particularly in arid and semi-arid environments of Africa and Asia, but also in subhumid areas in Latin America, is a major hotspot of land degradation. By tillage and grazing, livestock production also contributes to soil compaction and erosion, which are significant problems in both developed and developing regions. Land degradation reduces in the first place land productivity. Furthermore, land degradation has other environmental consequences, such as biodiversity loss due to habitat destruction and depletion of water resources by changing soil texture and removal of vegetation cover (FAO, 2006).

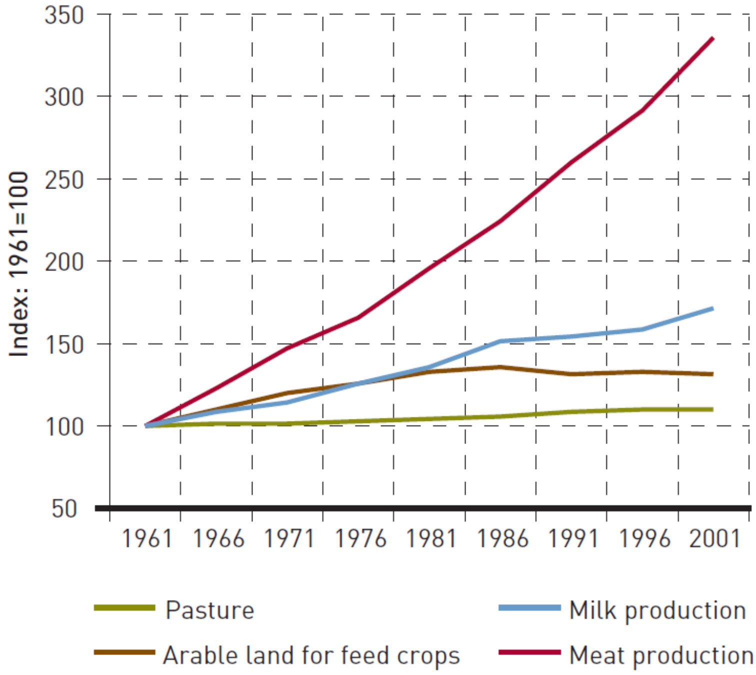


Figure 1.6 Global trends in land use area for livestock production and total production of meat and milk. Figure retrieved from FAO (2006). Data from FAOSTAT (2006).

In contrary to land use, livestock production has a rather modest contribution of 8% to global anthropogenic freshwater use, mainly (indirectly) for irrigating feed crops (FAO, 2006). Nevertheless, the agricultural sector as a whole accounts for 70% of global human freshwater use, which substantially differs among different world regions (Europe 21%; America 51%, Oceania 60%; Asia 81%; Africa 82%). These differences can mainly be explained by the climate and the place of agriculture in the economy (FAO, 2016). Industrial and domestic freshwater demand account for 20 and 10% of global anthropogenic freshwater use, respectively (FAO, 2006). The fact that livestock products would generally have far higher freshwater consumptions than crop-based products was previously stated (Mekonnen and Hoekstra, 2010), but cannot be generalized because the type of freshwater and the degree of local freshwater stress determine the environmental relevance of freshwater use. Livestock systems often use substantial amounts of so-called green water, which is soil moisture that originates from natural rainfall, but the consumption of this type of water generally does not contribute to local freshwater scarcity (Ridoutt et al., 2012). Nevertheless, freshwater scarcity is an increasing problem, because by 2025 64% of the world's population is projected to live in water-stressed basins (Rosegrant et al., 2002). Increasing water scarcity is likely to compromise future food production, because the available freshwater will have to be divided between agricultural, domestic and industrial uses (FAO, 2006). Global freshwater demand is projected to increase with 22% in the period 1995-2025 under the 'business as usual scenario' (Figure 1.7), but it will increase much more rapidly in developing regions (+27%) than in developed regions (+11%).

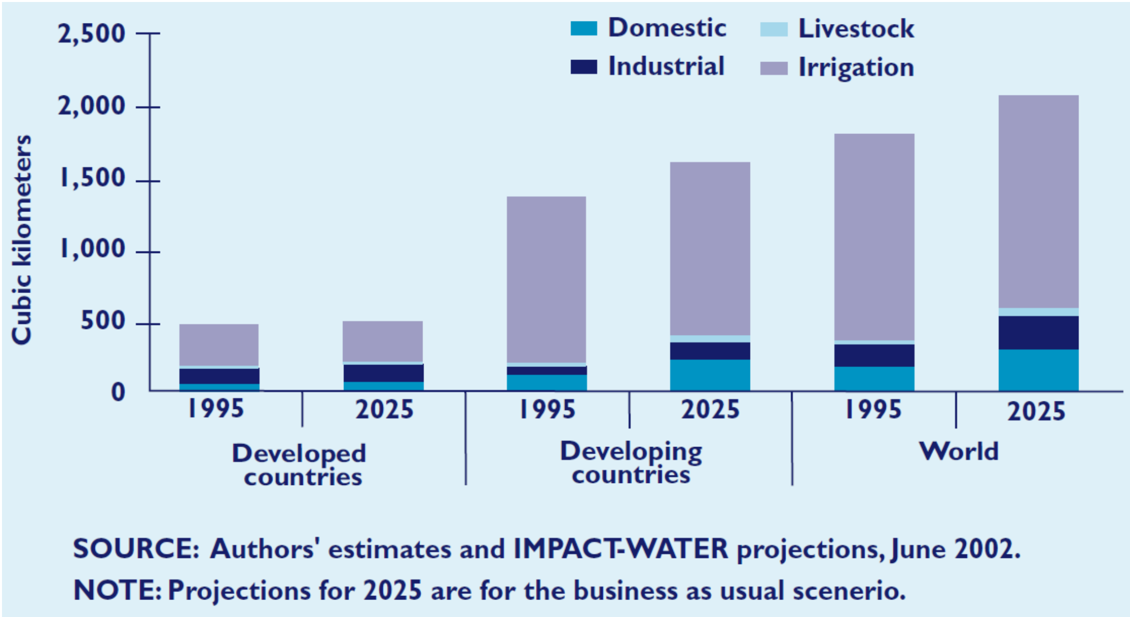


Figure 1.7 Water consumption by sector, 1995 and 2025. Water use by '*Livestock*' includes only direct water consumption; irrigation water for feed crops is included in '*Irrigation*'. Retrieved from Rosegrant et al. (2002).

Although irrigation will remain the world's by far largest freshwater user, it is estimated to increase globally with only 4% between 1995 and 2025, while domestic and industrial freshwater demands are projected to increase with 71 and 50% in that period, respectively. This dramatic rise will mainly occur in developing regions due to population and income growth and will put extra pressure on local water reserves. In developing regions, the increase in demand for irrigation water will rise substantially in sub-Saharan

Africa, with 27%, and in Latin America, with 21%. The rapid growth in livestock production in developing countries will more than double the direct water consumption by livestock, while it will grow with 19% in the developed world between 1995 and 2025 (Rosegrant et al., 2002).

Livestock production is probably the largest sectoral source of water pollution (FAO, 2006). Large amounts of nitrogen and phosphorous end up in the environment by leaching, surface run-off, subsurface flow and soil erosion, causing eutrophication of water bodies. Major sources of these nutrients are manure, applied as fertilizer on agricultural land used for feed production, and nutrient-rich wastewater from production sites. Also the increased use of mineral fertilizers and pesticides in feed production have largely contributed to water pollution. Besides nitrogen pollution of water bodies, livestock production is responsible for a major share (about 64%) of global anthropogenic emissions of ammonia into the atmosphere and deposition in the environment, causing eutrophication of waterways and acidification of soils (FAO, 2006). A major source of ammonia emissions is manure, during storage and after application on agricultural land. Especially regions with a high density of intensive livestock production systems with large numbers of animals concentrated in relatively small areas face large nutrient surpluses. These intensive production systems are located in both developed regions, such as the United States, Europe and Japan, and developing regions, such as Latin America (e.g. Brazil, Ecuador, etc.) and Southeast Asia (e.g. China, Indonesia, Thailand, etc.) (FAO, 2006).

In addition to ammonia emissions, livestock production also substantially contributes to the emission of greenhouse gases (GHGs) into the atmosphere, which are driving global warming. Livestock production is estimated to contribute to 14.5% of anthropogenic GHG emissions worldwide (Gerber et al., 2013). In terms of the three most important emitted GHGs by livestock production, i.e. carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O), livestock's contribution to global anthropogenic emissions differs: 5% in terms of CO_2, 44% in terms of CH_4 and 53% in terms of N_2O. Methane emissions form the largest part of the livestock sector's GHG emissions with 44%; nitrous oxide and carbon dioxide contribute almost equally to the remaining part, 29 and 27%, respectively. The majority of the livestock sector's GHG emissions comes from feed production and processing (47%) and enteric fermentation (39%), followed by manure management (10%) (Figure 1.8). Emissions from total energy consumption, added up along the livestock supply chains, account for 20% of the total sector's emissions (Gerber et al., 2013).

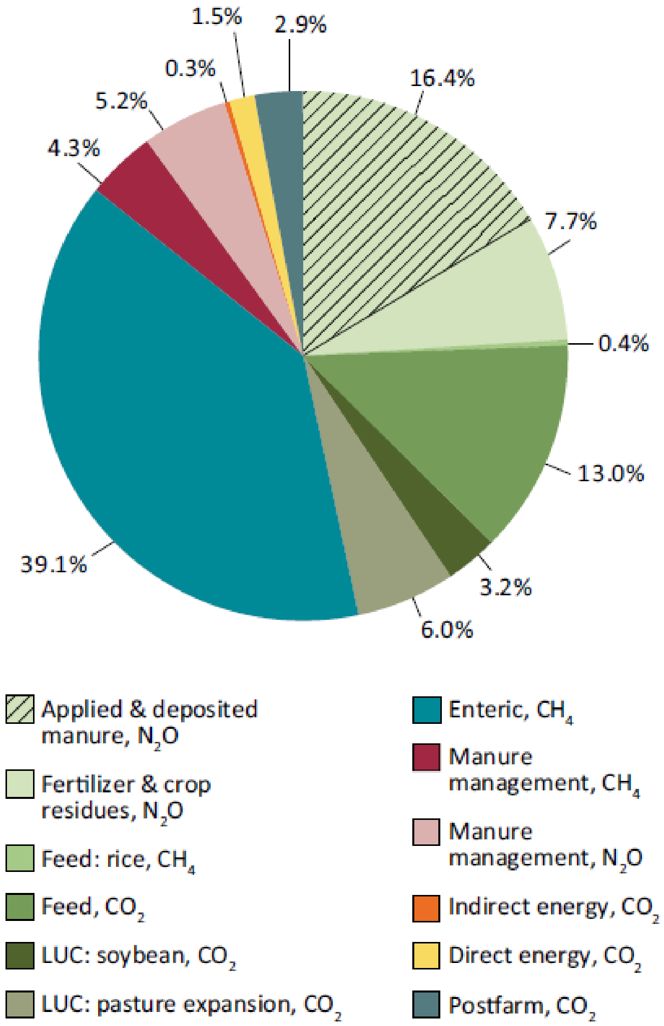


Figure 1.8 Global emissions from livestock supply chains by category of emissions. Figure retrieved from Gerber et al. (2013). *Indirect energy* is related to the construction of the animal production buildings and equipment. *Direct energy* is related to energy use for heating, ventilation, etc. on the animal production site.

Methane emissions come from enteric fermentation in ruminant animals (cattle, buffalo, sheep and goat) and from anaerobic decomposition of organic material during manure storage and processing (Gerber et al., 2013). Nitrous oxide emissions occur through both a direct pathway and two indirect pathways. The direct pathway involves the formation of nitrous oxide via combined nitrification and denitrification of nitrogen present in manure during storage and of nitrogen applied on agricultural land in the form of manure or synthetic fertilizers. Indirect nitrous oxide emissions are generated after deposition of volatilised nitrogen (ammonia and nitrogen oxides) on soils and surface waters, and after leaching or run-off of nitrogen from agricultural soils (IPCC, 2006). Carbon dioxide emissions originate from the oxidation of carbon in soils and vegetation after expansion of feed crops and pastures into natural habitats (land use change), and from the use of fossil fuels along the entire livestock supply chain. Changes in soil and vegetation carbon stocks caused by expansion of feed crops into grasslands or carbon stock changes within one land use type were not included in Gerber et al. (2013) due to lack of global databases and models, but can be significant in both positive and negative way. In the European Union (EU), permanent grasslands may represent a source or sink of GHG emissions, equal to 3 ± 18% of GHG emissions from the EU's ruminant sector (Opio et al., 2013), but uncertainties are very high.

Beef and cattle milk are the livestock products that contribute most to the sector's GHG emissions with 41 and 20%, respectively. They are followed by pig meat (9%), buffalo meat and milk (8%), chicken meat and eggs (8%), and small ruminant meat and milk (6%). Expressed per kg edible protein produced, beef is the livestock product with the highest average emission intensity (over 300 kg CO_2-eq per kg of protein). Beef is followed by small ruminant meat (165 kg CO_2-eq per kg of protein) and small ruminant milk (112 kg CO_2-eq per kg of protein). Cattle milk, pork, chicken meat and eggs have the lowest emission intensities (all below 100 kg CO_2-eq per kg of protein). These emission intensities vary largely among producers, indicating ample room for improvement (Gerber et al., 2013).

Loss of biodiversity is currently another major environmental concern, because biodiversity is an important condition for ecosystem resilience, i.e. the ability to adapt to changes such as climate change and to continue to provide ecosystem services in the future (Diaz et al., 2001). It is a complex problem to study because it is the result of many environmental changes that are caused by multiple agents. Quantification of livestock's contribution to this problem, therefore, is difficult. Nevertheless, the livestock sector is regarded as a major player in the current biodiversity crisis by its important contribution to many environmental issues that are driving biodiversity loss and ecosystem services changes (habitat change, climate change, pollution, etc.) (FAO, 2006).

The presented overview clearly demonstrates why the projected increasing global livestock production raises large environmental concerns. The significant contribution of the livestock sector to many environmental impacts, the substantial variations among producers and the fact that best practices and technologies are not widely used, imply that a large potential for improvement is present in this sector (FAO, 2006; Gerber et al., 2013).

1.1.2.2 Bioeconomy

Besides the projected increasing global demand for food and feed, another major development by which agriculture will be challenged over the next decades is the rising demand for biomass in the emerging bioeconomy. The major industrialized regions, the United States and Europe, see the bioeconomy as an important strategy to reduce dependence on finite fossil resources, which is a major cause of climate change (European Commission, 2012; United States White House Office, 2012). The overall rise in demand for biomass, however, will put more pressure on the limited amount of available bio-productive land in the world. The competition for land between food, feed, biomaterials and bioenergy is a growing concern and a major challenge to be addressed in the coming decades (Harvey and Pilgrim, 2011; Thornton, 2010).

To meet its annual demand for food, feed, biomaterials and bioenergy, the European Union (EU) has a high demand for cropland. Bringezu et al. (2012) calculated that the EU is a net importer of cropland; the EU used one-third more cropland than globally available on a per capita basis in 2007. With the projected increase in world population and rising living standards in developing countries, the EU is expected to exceed its fair share of acceptable resource use even more by 2030 under the assumption of constant consumption levels (Bringezu et al., 2012). The challenge to bring European consumption levels within the planetary boundaries, and to achieve a competitive economy that respects resource constraints and has much lower environmental impacts, was acknowledged in the European Commission's *'Roadmap to a Resource Efficient Europe'* (2011d).

According to Tilman et al. (2009), the huge challenge that the world is facing can be called the 'food, energy and environment trilemma' and is illustrated in Figure 1.9. To meet the rising demand for biomass, agriculture could further expand area into natural habitats and/or intensify production in order to obtain higher yields. Area expansion into natural habitats and other (direct and indirect) land use changes (e.g. conversion from grasslands to cropland) are usually responsible for net GHG emissions and thus are drivers of climate change, in addition to other environmental problems such as biodiversity loss. Agricultural area expansion, therefore, is not regarded as a sustainable option to meet the rising demand for biomass (Smith et al., 2014).

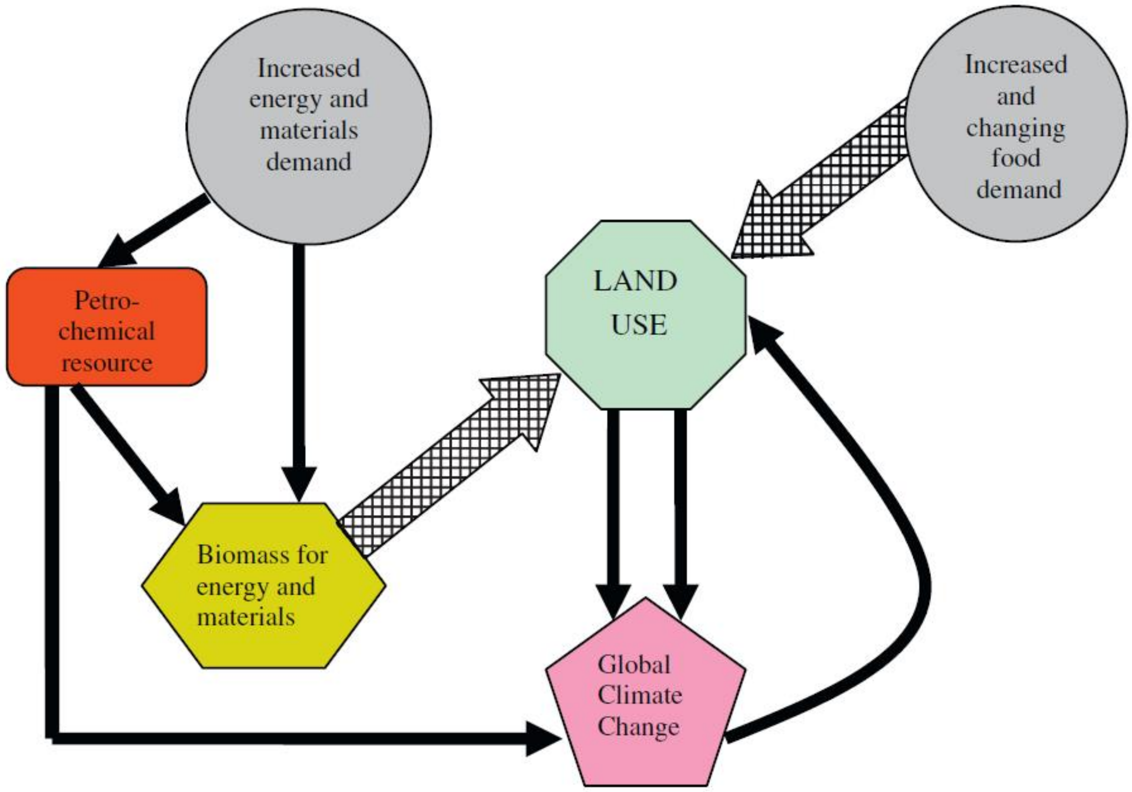


Figure 1.9 The new competitions for land use: interactions and feedback. Retrieved from Harvey and Pilgrim (2011).

While the future potential of yield increases by intensification is rather uncertain (Bringezu et al., 2012; Tilman et al., 2002), intensification also presents risks of increasing GHG emissions from agriculture. Within one land use type, different management practices, related to tillage, irrigation, rotation, fertilizing, residues, etc., influence GHG emissions from land use (IPCC, 2006). Poorly implemented intensification has adverse effects on long-term productivity and is associated with other environmental problems such as nutrient pollution, soil degradation, pesticide pollution, etc. (Smith et al., 2014).

To address the trilemma challenge, a broad consensus exists about the need for sustainable intensification (FAO, 2011b; Garnett et al., 2013; Godfray et al., 2010; Pretty, 2008; Smith, 2013). Definitions for sustainable intensification were suggested by several authors; summarizing, it comes to producing more product from the same land area, but also broader in terms of other natural resources, and it requires conservation of the natural resource base and an increased resource efficiency, while reducing environmental impacts and preventing damage to ecosystem services that support human health and wellbeing of current and future generations (FAO, 2011b; Smith, 2013).

The meaning and objectives of the term ‘sustainable intensification’, however, are subject to debate and criticism because the concept would be too narrowly focused on increasing production or would be even a contradiction in terms (Garnett et al., 2013). It is clear that intensification as it has occurred in the past, with increased use of fossil-based inputs such as fuel, mineral fertilizers and pesticides, cannot be a sustainable pathway for the future (Smith, 2013). Sustainable intensification should be more than the ‘business as usual’ scenario with only marginal efficiency gains (Garnett et al., 2013). For many, the word ‘intensification’ is also linked to negative agricultural developments in terms of biodiversity and animal welfare (Freibauer et al., 2011; Garnett et al., 2013). Because broad consensus exist on bringing agricultural expansion to a stop, sustainable intensification should be perceived as closing the yield gap, meaning eliminating the difference between the actual attained yield and the attainable yield given the location-specific conditions, in those regions, particularly developing countries, where production is still below the ‘sustainable threshold’ (European Commission, 2015; Garnett et al., 2013; Smith et al., 2014). The latter term can be understood as a collection of environmental tipping points at which the limits of the planet in terms of natural resource provision and pollutant absorption are exceeded. Environmental thresholds indicate the proximity to dangerous levels of environmental damage. By crossing thresholds negative irreversible consequences are likely to occur (Ecologic Institute and SERI, 2010).

Many of today’s agricultural systems in developed regions compromise future capacity to produce food and other agricultural commodities, because they have exceeded the agro-ecosystem carrying capacity (Buckwell et al., 2014; Freibauer et al., 2011). In these cases, 'sustainable extensification' could be proposed (van Grinsven et al., 2015), meaning that yield reduction could be considered to restore the equilibrium between production and preservation of ecosystem functionality. This highlights the importance of paying attention to the context and conditions within which actions towards a more sustainable agriculture should be implemented. In some cases, major focus should be on investigating the potential to increase production in a sustainable way, while in other cases focus should be more on bringing production within the ecological limits, which may or may not be realized with a reduction in yield (Garnett et al., 2013). The need to respect the ecological limits of primary resource supply should be considered in a broad sense, beyond agriculture, including other sectors that provide renewable biomass resources for the bioeconomy, like forestry, fisheries and aquaculture.

In addition to a long-term and context-specific vision on increasing yields, resource efficiency and resilience are seen as key strategies for a sustainable bioeconomy. To make future agriculture more resilient to increasing instability (economic, political and environmental), diversity in terms of species, between regions, and between and within farming systems should be maintained or fostered. Technological advances such as precision farming and introduction of new and improved species, whether or not by biotechnological advances, are regarded as promising ways towards increasing resource efficiency. Important resource savings can also be achieved by a better utilisation of waste streams through a cascading approach in a circular economy. The cascading use of biomass, in which use for high-value products receives priority over uses of lower value, is an important strategy for an optimized resource efficiency of biomass use (European Commission, 2015). The preferred cascading order is food-feed-biomaterial-bioenergy (De Meester, 2013; Scarlat et al., 2015). The concept of circularity is based on reuse and recycling (European Commission, 2015). Waste reduction is an important strategy, as about one-third of total food produced worldwide would be wasted. In developed regions, a significant amount of food is wasted at the consumption stage (FAO, 2011a), showing that improvements should not only be sought at the supply side of the food chain.

Many agree that strategies to address the complex challenges of the bioeconomy should also focus on the demand side, which involves efforts to change consumer behaviour and consumption. Regarding food consumption, a reduced consumption of animal-based products in Western diets, especially meat, is often suggested as an important strategy towards more sustainable and healthy diets (European Commission, 2015; Garnett et al., 2013; Smith, 2013).

Effectively addressing the complex challenges of this era involves widespread support and efforts from governments, farmers and consumers (Smith, 2013). In times of increasing economic instability, it is a key priority to provide decent incomes to primary producers and to provide incentives, especially for smallholder farmers in developing regions, to produce in a (more) sustainable way (FAO, 2011b). Additionally, investment in agricultural research and innovation, particularly to unravel trade-offs that likely occur between food security, energy security and environmental problems, plays a key role (European Commission, 2015).

1.2 Sustainability assessment

To foster the transition towards more sustainable practices and products, the field of sustainability assessment has emerged and is a rapidly developing research area with a large diversity in methodologies. These methodologies are developed to assist decision-makers with deciding which actions they should take towards a more sustainable society. Because achieving a higher sustainability is a complex task, the assessment of this concept certainly is just as challenging. Categorising the assessment methodologies can be done based on various aspects. According to Ness et al. (2007), three main aspects can be considered. First, the temporal characteristic of the methodology, i.e. does it evaluate developments in the past (descriptive assessment) or in the future (change-oriented assessment). Second, the focus of the methodology, i.e. at product level (micro level) or at policy level (macro level). Third, the extent to which the methodology integrates the three sustainability dimensions, i.e. environmental, social and/or economic aspects.

Given the major environmental challenges with which agriculture, and society in general, will have to deal over the next decades, the next subsection further focuses on environmental sustainability assessment, and more specifically on the Life Cycle Assessment (LCA) methodology. LCA is used to investigate the environmental sustainability of a product and is regarded as an appropriate methodology for this purpose, because it considers the life cycle perspective, i.e. covering the entire production chain, and it can assess environmental sustainability in a comprehensive way, i.e. covering a wide range of environmental problems. Furthermore, it can be performed in both a retrospective and a prospective way. The reader who is not familiar with conducting LCA is encouraged to read the next subsection, while it might not be necessary for the experienced LCA practitioner. The second subsection ('1.2.2 Resource-oriented assessment') is strongly encouraged for all readers of this dissertation.

1.2.1 Life Cycle Assessment (LCA)

The roots of LCA date back to the early 1970s, when energy analyses to study energy efficiency were broadened to include growing awareness about resource requirements, pollution and waste generation. Until the 1990s, LCAs were performed without a common theoretical framework, which hampered a major breakthrough. Since the 1990s, a decade of strong methodological development and harmonization began. During this period the Society of Environmental Toxicology and Chemistry (SETAC) had a leading and coordinating role in the organization of workshops and forums (Guinee et al., 2011) and published a 'code of practice' (SETAC, 1993). In 1994, the International Organization for Standardization (ISO) started to engage in LCA (Guinee et al., 2011), which resulted in the publication of a series of standards and technical reports, referred to as the 14040 series (Heijungs and Guinée, 2012). Together with the United Nations Environment Programme (UNEP), SETAC established in 2002 the Life Cycle Initiative, whose aim is to promote LCA and to facilitate knowledge exchange. In the early 2000s, several national LCA networks were established and there was a growing interest at policy level, like the U.S. Environmental Protection Agency and the European Commission. The latter launched the European Platform for LCA in 2005 (Guinee et al., 2011), which published the International Reference Life Cycle Data System (ILCD) handbook (European Commission, 2010c). The first decade of the 21st century was a period of elaboration, both in depth and width, with diverging approaches as a result (Guinee et al., 2011).

The ISO international standards provide a generic framework for LCA, without standardizing LCA methods in detail (Guinee et al., 2011). They were initially established to study environmental aspects and impacts, but the framework can as well be valid to study economic and social sustainability aspects (ISO, 2006a). Life Cycle Costing (LCC) is the economic variant of environmental LCA (Swarr et al., 2011). Guidelines for social LCA (S-LCA) exist as well (UNEP/SETAC Life Cycle Initiative, 2009), but this technique has received less attention in the past. Interest in S-LCA, however, is now rapidly growing. Integrating the three techniques to obtain a more comprehensive sustainability assessment results in Life Cycle Sustainability Analysis (LCSA), a coherent framework that is still in an early stage of development (Kloepffer, 2008; UNEP/SETAC Life Cycle Initiative, 2011).

This section focuses further on the framework of environmental LCA. ISO has defined it as ‘a compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle’. The term product can refer to both goods and services (ISO, 2006a; ISO, 2006b). The entire product’s life cycle involves several phases; from natural resource extraction, via production, distribution and use phases, to waste management (i.e. from cradle to grave) (Finnveden et al., 2009). The ISO standards divide the LCA framework into four main phases, i.e. (i) Goal and scope definition, (ii) Inventory analysis, (iii) Impact assessment and (iv) Interpretation. Although these phases are performed in the order mentioned, LCA is an iterative process (Figure 1.10) (ISO, 2006a; ISO, 2006b).

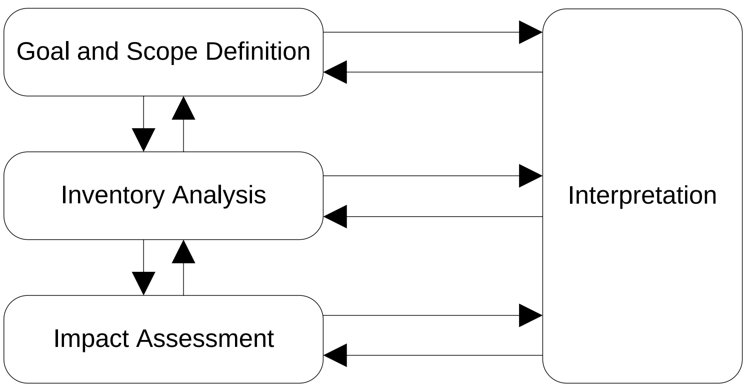


Figure 1.10 Four stages of an LCA. Modified from ISO (2006a).

In the first phase, goal and scope of the LCA must be clearly defined. The goal definition includes the intended application of the study, the reasons for performing the LCA, the intended audience and the (non-)comparative character of the study. LCAs can be performed to compare the environmental performance of two or more product systems or to analyse a single product system. In the scope definition, a number of major choices, which influence the following steps of the LCA procedure as well as the results of the study, are described. Scope definition includes the product system(s) to be studied, choice of the functional unit, description of the system boundaries, selection of the impact categories, etc. The functional unit is a quantitative measure of the function of the product(s). It acts as a reference to which all inputs and outputs of the product system(s) can be scaled and it enables a comparison between product systems on a common basis. System boundaries are described to specify which unit processes are part of the studied product system and to delimit the life cycle. A unit process is defined by ISO (2006a) as 'the smallest element considered in the life cycle inventory analysis for which input and output data are quantified'. Starting from the extraction of natural resources (the 'cradle'), the system boundary can either be set at the production facility gate (i.e. a cradle-to-gate study) or further in the life cycle (distribution stage, consumer stage, etc.). Accounting for the complete life cycle, including end-of-life management (i.e. the 'grave'), results in a cradle-to-grave study. Some studies only focus on a smaller part of the life cycle; a gate-to-gate system boundary is set when studying the processes within one production facility (European Commission, 2010c; Heijungs and Guinée, 2012; ISO, 2006a; ISO, 2006b).

The second LCA phase, the inventory analysis, is usually the most time-consuming step, because the life cycle inventory (LCI) has to be compiled through data collection and calculation procedures. Data of different types of flows, i.e. product flows, waste flows and elementary flows[i], are collected. The product system is usually split into a foreground system and a background system in order to distinguish between

[i] material or energy entering the system being studied that has been drawn from the environment without previous human transformation, or material or energy leaving the system being studied that is released into the environment without subsequent human transformation (ISO, 2006a; ISO, 2006b)

foreground processes for which specific data has to be collected and background processes for which average or generic data can be used. The foreground processes are also regarded as 'those processes under direct control or decisive influence of the producer of the good or the operator of the service', in contrary to background processes. Foreground data are preferably collected or measured at the site of the studied production facility (primary data). Only when these data are not available or not representative, secondary data (e.g. retrieved from literature) can be used. Background data are retrieved from literature or from LCI databases (European Commission, 2010c). Examples of LCI databases are *ecoinvent* (http://www.ecoinvent.org), the European reference Life Cycle Database (ELCD) (http://eplca.jrc.ec.europa.eu), the U.S. Life Cycle Inventory Database (USLCI) (http://www.nrel.gov/lci) and, more specifically for agricultural products, the Agri-footprint database (http://www.agri-footprint.com), the World Food LCA Database (WFLCD) (www.quantis-intl.com/wfldb), etc.

Different modelling principles and methods exist to compile the LCI. Two modelling principles are distinguished, i.e. attributional and consequential modelling. The choice to perform the LCA in an attributional or consequential way is usually already decided in the first phase, because this choice influences the entire scope of the study. Attributional LCA makes an inventory of the inputs and outputs of all relevant unit processes of the product system(s) under study. This type of LCA describes the potential environmental impacts of the studied life cycle as it was, as it is or as it is estimated to be in the future. In contrary, consequential LCA describes how the potential environmental impacts will change in consequence of decisions made in the core of the product system. Consequential LCA, therefore, only makes an inventory of the inputs and outputs of unit processes that will change as a result of these decisions. A typical question in consequential LCA is how an additional demand of the studied product will change the dynamic technosphere in which it is embedded. Consequential LCA thus considers market effects and requires additional information to describe these effects (European Commission, 2010c; Finnveden et al., 2009). The question whether one type of LCA is more appropriate than the other is under debate. According to Weidema (2003), consequential LCA is more appropriate than attributional LCA because consequences beyond the studied product system have to be taken into account to grasp the complete picture. The description of short-term and long-term market effects, however, is very complex and involves large uncertainties (Curran, 2012; Finnveden et al., 2009). Ekvall (2005) concludes that both modelling principles have methodological limitations and address different research needs, and, therefore, there is no superior type of LCA. The choice between attributional and consequential LCA should depend on the main purpose of the study and in some cases it could be relevant to perform both types of LCA (Ekvall et al., 2005).

Closely related to the discussion about the most appropriate modelling principle, is the discussion about allocation procedures in case of multifunctional processes. ISO (2006a; 2006b) defined allocation as ‘partitioning the input or output flows of a process or a product system between the product system under study and one or more other product systems’. Attention should be paid to allocation procedures in three main cases, i.e. (i) when a process produces next to a product also co-products (multi-output problem), (ii) when several waste streams are treated by one process (multi-input problem) and (iii) when waste streams are recycled into a new product (Finnveden et al., 2009). To deal with the problem of multifunctionality, ISO (2006b) gives a preferred order:

1. Avoid allocation wherever possible by dividing the process into sub-processes with only one product and collecting data for these sub-processes, or by system expansion to include the additional functions of the co-products;
2. When allocation cannot be avoided, perform allocation in a way that reflects the underlying physical causalities between the inputs and outputs;
3. When physical relationships between inputs and outputs are absent, allocation should be performed based on other relationships such as the economic value of products.

In practice, the ISO guidelines are implemented with a high degree of freedom in interpretation (Curran, 2012). Dividing a multifunctional process in single-product sub-processes is often not possible in practice. Avoiding allocation by system expansion is an inherent part of consequential LCAs. Although allocation is most commonly applied in attributional LCAs, a variant of system expansion, ‘the avoided burden approach’ or ‘the substitution approach', is applied in some cases for co-product allocation. This approach subtracts the environmental impacts of an alternative product system with the same function as the co-product from the total environmental impact of the studied product system (Finnveden et al., 2009). In attributional LCAs, the allocation procedure is chosen on a case-by-case basis, in which all types of allocation (based on mass, energy, economic value, etc.) are applied, and economic allocation is the most commonly applied (Lundie et al., 2007). Because different allocation procedures can significantly influence the LCA results, Lundie et al. (2007) argue that sector-specific allocation guidelines are very useful to improve the methodological consistency of LCA studies. One example is the biological allocation procedure advised by the International Dairy Federation (IDF) to streamline LCAs of milk (IDF, 2010).

In addition to modelling principles, three main methods for LCI compilation can be distinguished, i.e. process-based, input-output (IO) based and hybrid forms of the preceding ones. While process-based methods calculate the inventory of processes and their products, i.e. at the micro level, IO-methods are used to calculate the inventory of sectors and nations, i.e. at the macro level. Both types of methods have their strengths and weaknesses. IO-methods are more complete than process-based methods, but IO-data are less detailed and less accurate than process-based data (Suh and Huppes, 2005). Process-based methods are still most commonly used in LCA studies. Hybrid forms of process-based and IO-methods are promising to fill data gaps in attributional LCAs and to provide a more complete picture, but further research and development is required. One example is the fact that average data generated by IO-methods are not adequate for consequential LCAs, in which marginal data are used for modelling consequences (Finnveden et al., 2009).

In the third phase of the LCA framework, i.e. the life cycle impact assessment (LCIA) phase, the compiled LCI is used to evaluate the potential environmental impacts of the studied product system. According to ISO (2006b), this phase consists of mandatory and optional elements. Mandatory are the selection of impact categories, classification and characterization. Optional are normalisation and weighting.

During classification, the elementary flows, i.e. the emissions to and extracted resources from the natural environment, are assigned to impact categories to which they contribute. For example, emissions of carbon dioxide contribute to climate change, emissions of ammonia contribute to acidification, etc. Characterization involves modelling of the potential impact of each elementary flow in a quantitative way according to the relevant environmental mechanism or cause-effect chain. Substance-specific characterization factors are calculated and multiplied with the inventory data to express the potential environmental impact of each elementary flow in a common unit of the impact category. For example, to express the impact category climate change in a common unit, the greenhouse gas carbon dioxide (CO_2) is used as reference substance with a global warming potential (GWP) equal to 1. All other contributing substances to this impact category are expressed in CO_2-equivalents by normalizing their GWP to that of CO_2. Characterization can be performed at midpoint or endpoint level, depending on the location of the chosen indicator along the impact pathway (see example for the impact category climate change in Figure 1.11).

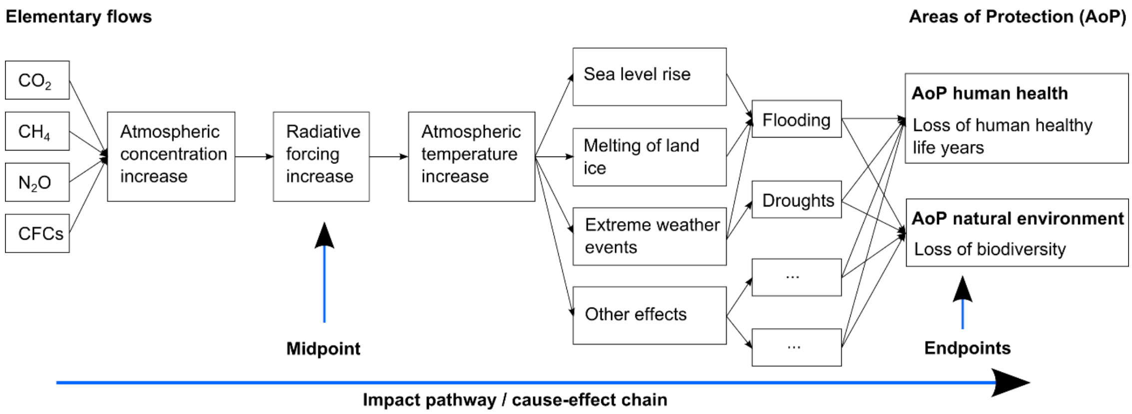


Figure 1.11 Simplified impact pathway / cause-effect chain for global warming connecting elementary flows from the inventory to the Areas of Protection (AoP), with indicated location of midpoints and endpoints. Adapted from Hauschild and Huijbregts (2015).

At midpoint level, impacts are indicated at an intermediate point along the impact pathway between emissions or resource extractions and the endpoint level, i.e. the end of the cause-effect chain. Midpoint indicators are defined at the location where a common mechanism exists for the main contributing substances within a specific impact category. For example, an appropriate midpoint indicator for climate change is the increase in radiative forcing of the atmosphere (Figure 1.11) (European Commission, 2010b; Hauschild and Huijbregts, 2015; Hauschild et al., 2013).

Characterization at endpoint level requires modelling of the entire impact pathway. While midpoint indicators are used to express the relative impacts of elementary flows within one impact category, endpoint indicators are used to express damage to the main areas that society wants to sustain or protect (European Commission, 2010b; Hauschild et al., 2013). The so-called areas of protection (AoP), proposed by Udo de Haes et al. (1999), that are usually included in LCIA are human health, natural environment and natural resources. Less often considered is a fourth AoP, i.e. man-made environment. Figure 1.12 gives a non-exhaustive overview of midpoint impact categories and their link to the areas of protection at endpoint level. The endpoint approach has the goal to assist in understanding and interpreting midpoint impacts by making a more concrete link with the sustainability concept through the AoPs. For example, in case of climate change, greenhouse gas emissions are linked to their effects on ecosystems and humans, which are endpoints for the AoP natural environment and the AoP human health, respectively (Figure 1.12) (European Commission, 2010b; Hauschild et al., 2013).

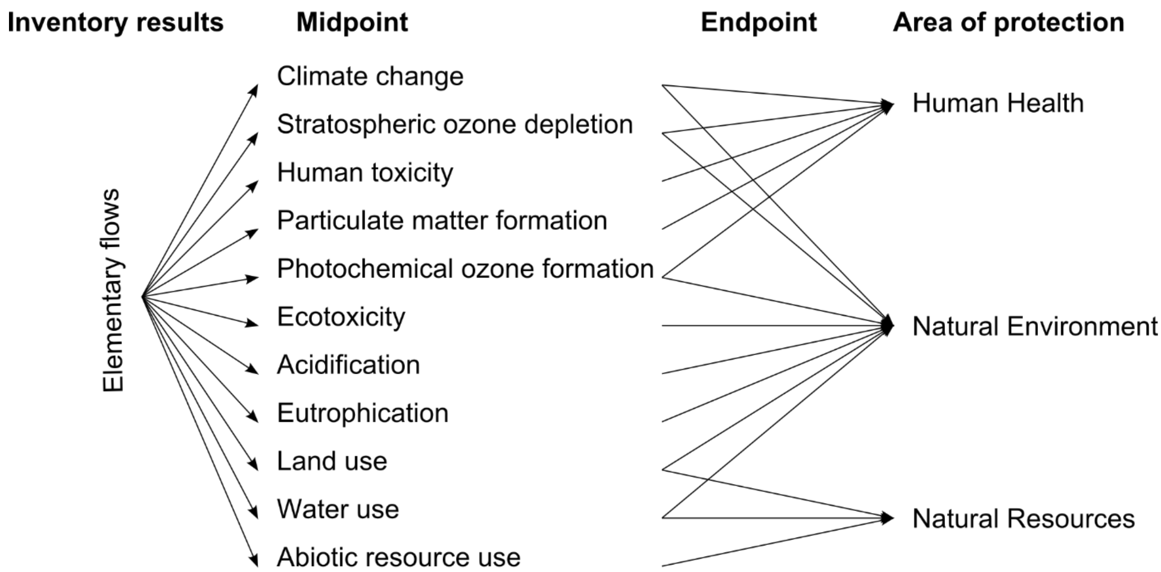


Figure 1.12 LCA impact categories at midpoint level and their relationship with damages to the areas of protection at endpoint level. Adapted from European Commission (2010b).

Quantification of the damage to the AoP natural environment is focused on biodiversity loss, for which the Potentially Disappeared Fraction of species (PDF) is a commonly used endpoint indicator (Goedkoop et al., 2013; Goedkoop and Spriensma, 2001), recommended by the European Commission (2010b). The PDF represents the fraction of species that has a high probability of no occurrence in a region due to unfavourable conditions. For the AoP human health, the Disability Adjusted Life Years (DALY), representing the potential number of healthy life years lost, is commonly used as endpoint indicator (Goedkoop et al., 2013; Goedkoop and Spriensma, 2001), and recommended by the European Commission (2010b). Damage to the AoP natural resources is less well-defined and the distinction with the other AoPs is not always clear. Current endpoint approaches focus on the reduced availability and exploitability of resources used by humans in the future, respectively known as resource depletion and resource scarcity (European Commission, 2010b). Two examples of existing approaches for quantification of damage to the AoP natural resources are the ‘surplus energy’ concept (Goedkoop and Spriensma, 2001) and the ‘surplus cost’ concept (Goedkoop et al., 2013). These concepts are based on the idea that future resource extractions will increasingly require additional efforts in terms of energy and costs, respectively. Recommendations of mature methods by the European Commission for quantification of damage to the AoP natural resources, however, are absent, showing that this area needs further elaboration, which is the topic of discussion in Dewulf et al. (2015).

The last decade was a very prolific period in the development of life cycle impact assessment methods, both in width and in depth. These developments, however, are associated with a growing need for harmonisation and guidance to achieve a higher consistency and quality in the LCIA methods (Hauschild et al., 2013). In the framework of their International Reference Life Cycle Data System (ILCD) Handbook, the European Commission (2010a; 2010b; 2011c) has evaluated existing LCIA methods at midpoint and endpoint level with the aim to identify the best existing practice. An important conclusion of this evaluation is the higher scientific consensus about midpoint methods compared to endpoint methods, which are in a larger need for further development. Compared to midpoint modelling, endpoint approaches require more data and involve more modelling assumptions, usually resulting in higher uncertainties (European Commission, 2010b).

Optional steps in LCIA are normalisation and weighting, which can be performed to facilitate the interpretation of the results. Normalisation and weighting can be applied at both midpoint and endpoint level. Normalisation expresses the magnitude of impact scores relative to reference information (e.g. a global or regional reference). The relative significance of different impact scores according to the goal of the study can be expressed through weighting. Weighting criteria have a normative character and can be set based on public values or policy priorities (European Commission, 2010b; ISO, 2006b). The advantage of weighting is to provide a fully aggregated result, which can be useful for decision-making when trade-offs between different impact categories occur. When weighting is applied, however, ISO (2006b) emphasizes that the different impact scores should remain available to prevent loss of information.

The last phase of the LCA framework is the iterative interpretation phase. During this phase intermediate (LCI and LCIA) results are interpreted, which can lead to a refinement or revision of the initial scope of the study. Good interpretation requires knowledge about methodological choices and assumptions made during the study. Additional analyses (e.g. sensitivity analysis and uncertainty analysis) can support the interpretation phase. While a sensitivity analysis can be performed to determine how changes in data and methodological choices affect the LCA results, an uncertainty analysis determines how data and model uncertainties affect the reliability of the LCA results. At the end of the study, this phase aims to provide a clear and understandable presentation of the results, to answer the questions that have been raised in the goal definition of the study and to provide recommendations for decision-makers (ISO, 2006b).

1.2.2 Resource-oriented assessment

Initially, environmental impact assessments were mainly focused on emission problems. This has resulted in many adequate end-of-pipe techniques for waste treatment and emission reduction. This emission-oriented approach gradually shifted towards more resource-oriented approaches and the adoption of clean technologies to prevent pollution. Given the increasing scarcity of natural resources and the value that they represent for economic activities, resource-oriented process and life cycle assessments are highly relevant (De Meester et al., 2009; Dewulf et al., 2008). In this context, several methodologies with a life cycle perspective that focus on resource use were developed.

Different classifications of resource-oriented methods can be found in literature. A distinction is often made between methods that address (i) land use, (ii) water use and (iii) other abiotic resource uses (metals, minerals, fossil energy, nuclear energy, atmospheric resources (e.g. argon) and flow energy resources (e.g. wind energy)) (Swart et al., 2015). Surprisingly, biotic resources, defined as materials derived from presently living organisms (e.g. tropical hardwood, wild fish, etc.) excluding biotic resources reproduced by a human-controlled production process (e.g. agriculture, aquaculture, wood plantations, etc.), have received much less attention (Klinglmair et al., 2014; Swart et al., 2015). Furthermore, it can be noted that land use, although classified as abiotic by Swart et al. (2015), is neither as clearly to be characterized as biotic or abiotic (Klinglmair et al., 2014).

Another distinction is often made between methods that account for overall natural resource use along the life cycle (resource accounting methods) and methods that address the scarcity of resources at midpoint or endpoint level (resource depletion methods). Resource accounting methods (RAMs) use an inherent property of resource flows (e.g. mass, energy, exergy, etc.) as a basis for characterization, which allows them to sum up different types of resources used in the life cycle in a common unit (European Commission, 2011c; Swart et al., 2015). Methods that characterize resources in terms of mass (e.g. Material Intensity Per Unit Service (MIPS) (Spangenberg et al., 1999)) or energy (e.g. Cumulative Energy Demand (CED) (Frischknecht et al., 2007; VDI, 1997)), however, have an important drawback, because they cannot quantify both material and energy flows in a common unit (kg vs. kJ). Moreover, some resources can fulfil both functions, e.g. in the chemical industry fossil fuels can be used as both feedstock and energy source (Van der Vorst et al., 2010). The thermodynamically-based concept of exergy, defined as the maximum amount of work that can be obtained from a resource (Dewulf et al., 2008), overcomes this limitation, because both material and energy flows can be quantified in one common unit, i.e. exergy joule (J_{ex}). Examples of exergy-based

RAMs are the Cumulative Exergy Demand (CExD) (Bösch et al., 2007) and the Cumulative Exergy Extraction from the Natural Environment (CEENE) (Dewulf et al., 2007a). While resource depletion methods are regarded as more relevant to quantify environmental impacts at midpoint level and environmental damages at endpoint level, exergy-based resource accounting methods are considered as valuable from another perspective because (i) they characterize resources in a relatively more robust and certain way (European Commission, 2011c), based on objective thermodynamic laws and, therefore, (ii) they can be very adequate for addressing overall resource use and efficiency, both at process level and at the life cycle level (Dewulf et al., 2008).

The main purpose of this section is to provide a relatively broad overview of currently available resource use-oriented methods. Because this overview includes exergy-based resource accounting methods among other methods, first, a more detailed explanation on the concept of exergy and its applications is provided in the next subsection. The second subsection covers successively methods that address (i) abiotic resource use, (ii) biotic resource use, (iii) water use and (iv) land use.

1.2.2.1 Exergy-based resource accounting

When explaining the concept of *exergy*, the difference with the widely known term *energy* needs to be addressed first. People experience energy in many of their daily activities. Energy comes in many forms, such as electrical, thermal and mechanical energy, but also chemical energy in materials. The human body itself is an example of a biological system that converts the chemical energy of food into other forms of energy, such as heat and work (Dincer and Rosen, 2013c). The part of these energy forms that people value is the useful part, as not every quantity of energy has the ability to produce work or to cause a change (Dewulf et al., 2008). There is a difference between one joule of electricity and one joule of heat. Also, there is a difference between one joule of heat at 100°C and one joule of heat at 25°C. These examples explain the difference between energy and exergy. Exergy is the useful part of energy and allows a distinction between different qualities of energy (Stougie, 2014).

To explain the difference between energy and exergy scientifically, the laws of thermodynamics can be used. The first law of thermodynamics (FLT) is the law of conservation of energy, which states that, although energy can change forms, energy can neither be created nor destroyed. This law gives no information about the direction in which processes can spontaneously occur. A transfer of heat from a low-temperature body to a high-temperature body without the input of external energy would be possible only on the basis of the FLT, not on the basis of the second law of thermodynamics (SLT), which implies that heat transfer can only occur spontaneously in the direction of temperature decrease. The SLT thus (i) provides information on the direction in which processes can spontaneously occur and (ii) allows a distinction between different qualities of energy. The SLT states that exergy is destroyed during real or irreversible processes, because irreversibilities cause that the original quality of the resource input cannot be fully recovered. The distinction between reversible, or ideal, and irreversible, or real, processes can be made on the basis of entropy. The SLT states that real processes can only occur in the direction of increased entropy, while ideal processes do not generate entropy. The destroyed exergy by real processes is proportional to the generated entropy. Entropy is a measure of the amount of disorder within a system. Because disordered states are more probable than ordered states and because the natural direction of a change in the state of a system is from a state of low probability to one of higher probability, the natural or spontaneous direction of a change of the state of a system is from order to disorder, or in other words from low entropy to high entropy. It can be confusing, however, that the entropy in an open system can decrease, and this because of the exchange of energy across the system boundary. The entropy of the overall system always increases according to the SLT. An example is freezing water; the entropy of the water is decreased to increase order of the water molecules and to obtain ice by removal of heat. This heat increases the entropy of the substance to which the heat is transferred. Additionally, the electricity used by the freezer will ultimately be degraded to heat (Dincer and Rosen, 2013c).

The term exergy comes from the Greek words *ex* (out of) and *ergon* (work) (Dincer and Rosen, 2013c), referring to its definition 'the maximum work potential of a material or an energy flow, when bringing it into equilibrium through reversible processes with the reference natural environment'. Only reversible processes are considered when bringing a flow to the reference conditions of the natural environment, because they reflect the most ideal (thermodynamic optimal) path, and, therefore, they yield the theoretical maximum amount of work (Szargut et al., 1988). At the same time, exergy also reflects the minimum work necessary to produce a substance in its specified state (temperature, pressure) and chemical composition and concentration in a reversible way from common components (i.e. reference substances) in the natural environment (Morris and Szargut, 1986). It is clear that the exergy content of a substance is dependent on the properties of both that substance and the natural environment. Because the latter is not in thermodynamic equilibrium, a reference environment with zero exergy must be specified in terms of temperature (e.g. 25 °C defined by Szargut et al. (1988)), pressure (e.g. 1 atm defined by Szargut et al. (1988)), and chemical composition and concentration (by means of reference substances) in order to calculate exergy contents. While differences in temperature and pressure reflect physical exergy, a different chemical composition and/or concentration reflect chemical exergy. For example, when considering a copper deposit, the copper in the deposit occurs in a different chemical structure (e.g. $CuFeS_2$) and is much higher concentrated than the reference substance for copper, which is copper dissolved in seawater (Cu^{2+}) (Swart et al., 2015). Some important characteristics of exergy can be deduced (Dincer and Rosen, 2013c):

- The exergy content of a substance is equal to zero when it is in complete equilibrium with the reference environment; this means no difference in terms of temperature, pressure, nor chemical composition or concentration.
- The more a substance deviates from the reference environment, the higher its exergy content.

Exergy destruction during a real process causes that the output exergy is always lower than the input exergy, which is illustrated in Figure 1.13 (Dewulf et al., 2008). In addition to exergy destruction due to entropy generation, part of the input exergy can be lost in the form of wastes. As a result, the actual process performance is lower than the ideal, or thermodynamic optimal, performance. To improve the performance, both internal irreversibilities and wastes need to be addressed. Also, when heat is part of the output, it could be recovered to reduce loss of exergy. Exergy analysis of processes and systems thus provide insights into the magnitude, the types and the locations of exergy losses.

To quantify how well resources are transformed into the desired products, the exergy efficiency can be calculated as the ratio of the exergy in the product(s) over the input exergy. The exergy efficiency can be regarded as an overall resource efficiency and a measure of approach to ideality (Dincer and Rosen, 2013c).

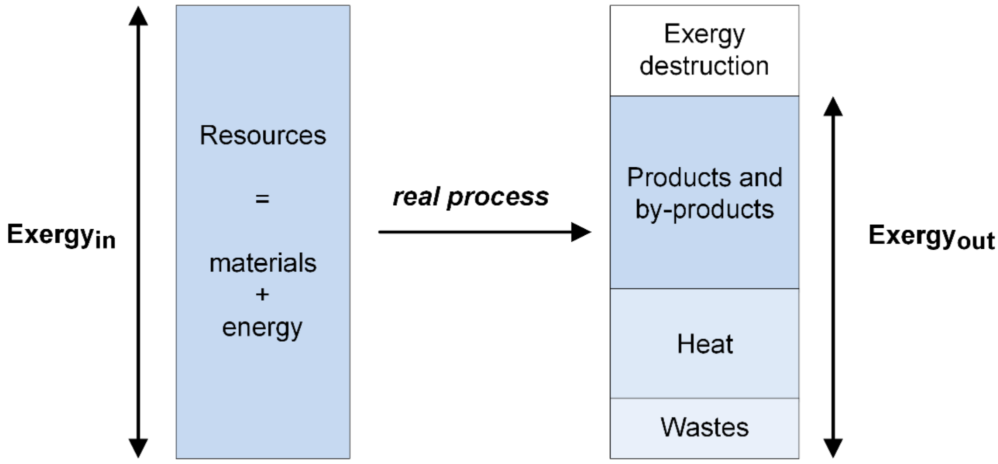


Figure 1.13 Exergy destruction during a real process. Adapted from Dewulf et al. (2008).

Exergy analysis can be extended beyond a single process to consider all processes in the supply chain of a product. The exergy concept, therefore, can be used to quantify cumulative overall resource use and its efficiency. Cumulative exergy consumption (CExC) equals the sum of the exergy contained in all natural resources used throughout the supply chain of a product. Dividing the exergy content of the product by the CExC of its supply chain gives the resource efficiency of the entire supply chain, which is called the Cumulative Degree of Perfection (CDP) (Szargut et al., 1988).

Integration of the CExC concept in the conventional LCA framework results into Exergetic Life Cycle Assessment (ELCA). The four-phase framework of conventional LCA is similar for ELCA, except the inventory analysis, which can be more detailed because of the quantification of all material and energy flows in exergy terms. ELCA aims to reduce cumulative exergy losses and thus improve the resource efficiency of the complete life cycle (Dincer and Rosen, 2013a). Exergy-based resource accounting methods (RAMs), such as the Cumulative Exergy Demand (CExD) (Bösch et al., 2007) and the Cumulative Exergy Extraction from the Natural Environment (CEENE) (Dewulf et al., 2007a), were developed within the ELCA framework and were operationalized for the process-based LCI database *ecoinvent*. These RAMs enable the calculation of a life cycle's overall resource footprint, expressed in exergy joules (J_{ex}), by aggregating the exergy content of an extensive range of natural resources (water, metals, minerals, fossil energy, nuclear energy, abiotic renewable energy, atmospheric resources and biotic and/or land resources). CExD and CEENE have some methodological differences, such as their approach to account for biotic resources reproduced by a human-controlled production process; while CExD accounts for the exergy content of the harvested biomass, CEENE accounts for the exergy deprived from the natural environment due to land use. Regarding biotic resources extracted from natural systems, both methods account for the exergy content of the extracted biomass. Two CEENE versions with a different conceptual approach for land use accounting currently exist, i.e. CEENE v2007 (Dewulf et al., 2007a) and CEENE v2013 (Alvarenga et al., 2013c). CEENE v2007 uses the exergy content of the solar radiation that can be metabolized through photosynthesis by natural ecosystems, per unit area and time, as a proxy for land occupation. This solar exergy is considered as no longer available to nature due to land occupation by human-controlled systems (e.g. agriculture). Site-dependent factors such as climate and soil quality are not taken into account by CEENE v2007. To tackle this limitation, CEENE v2013 accounts for the occupied land through the exergy content of the potential natural net primary production (NPP) on that land.

Thanks to a different approach, ELCA is a valuable complement to conventional LCA: it reveals additional insights and helps to better understand the causes of inefficient production chains (Cornelissen and Hirs, 2002; Dincer and Rosen, 2013a; Rosen et al., 2012). Figure 1.14 illustrates the qualitative relation between the exergy efficiency and the environmental impact of a process, and between the exergy efficiency and the sustainability of a process. This figure is valuable when considering the extreme values of exergy efficiency, i.e. 0% and 100%. Approaching an exergy efficiency of 100%, environmental impacts would be absent because resource conversions occur without exergy loss, either by entropy generation or waste emissions. Approaching an exergy efficiency of 0% shows that sustainability cannot exist without an efficient conversion of resources (Rosen and Dincer, 2001). Considering Figure 1.14, it is very important to stress that the presented relations should be evaluated within one process (e.g. a pharmaceutical process) and not in a comparison between different processes (e.g. a pharmaceutical process versus an agricultural process). Also 'sustainability' in Figure 1.14 should be narrowed down to environmental sustainability. Furthermore, exergy efficiency cannot be used as the only indicator to evaluate whether one process is more environmentally sustainable than another. Emissions, for example, also play an important role in the environmental sustainability of a process and their impact on the environment cannot really be reflected by their exergy content.

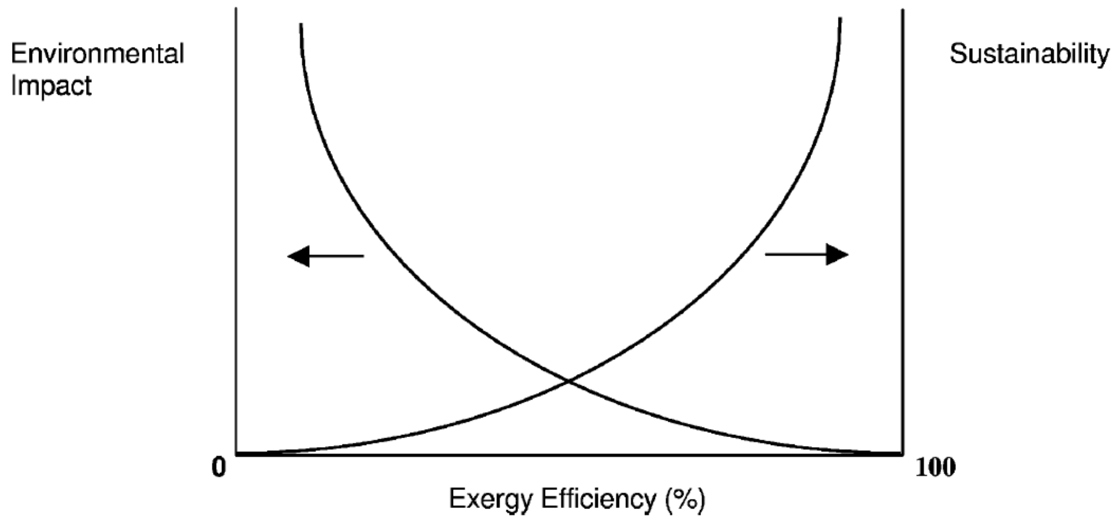


Figure 1.14 Qualitative illustration of the relation between the environmental impact and sustainability of a process, and its exergy efficiency. Retrieved from Rosen and Dincer (2001).

Exergy analysis has primarily been developed in the energy, chemical and metallurgical industries (Kotas, 1985; Sciubba and Wall, 2007; Szargut et al., 1988). Due to the growing recognition of its usefulness, it is increasingly applied on biological systems as well as technological systems. Applications on biological systems include exergy analyses of photosynthesis in green plants (Bisio and Bisio, 1998; Lems et al., 2010; Petela, 2008; Reis and Miguel, 2006) and exergy analyses of biochemical processes at the level of the living cell (Lems et al., 2003; Lems et al., 2007; Lems et al., 2009). Exergy analyses of industrial processes and systems, however, are still far more often applied (BoroumandJazi et al., 2013; Dincer and Rosen, 2013b; Luis, 2013; Stougie, 2014). Exergy analysis has also been applied on processes in the food industry (Fang et al., 1995; Tekin and Bayramoglu, 2001; Zisopoulos et al., 2015a; Zisopoulos et al., 2015b), and Apaiah et al. (2006) demonstrated the usefulness of exergy analysis to study entire food supply chains. While the exergy concept is still rarely used to study entire supply chains of food products (Degerli et al., 2015; Nhu et al., 2015; Ozilgen and Sorguven, 2011; Sorguven and Ozilgen, 2012), it is more frequently applied to examine the life cycle of bioenergy and biomaterials (Alvarenga et al., 2013a; Brehmer et al., 2008; Christopher and Dimitrios, 2012; De Meester et al., 2011; De Meester et al., 2012; Dewulf et al., 2000; Dewulf et al., 2005; Liao et al., 2011; Taelman et al., 2013). Furthermore, exergy has been used to analyse the exergetic performance of whole countries (Rosen, 1992; Rosen and Dincer, 1997; Schaeffer and Wirtshafter, 1992), and even the Earth (Hermann, 2006).

Various extensions of exergy analysis have been developed (Dewulf et al., 2008). A first example is situated in the context of natural systems. The Eco-Exergy (EE) concept quantifies the exergy value of living organisms by taking into account the information in their DNA in addition to their chemical composition (Jorgensen et al., 2005; Jorgensen et al., 2010). The EE concept is used to study the development of ecosystems and their dynamics (Jorgensen and Nielsen, 2014; Jorgensen, 2007). Second, various extensions of the traditional Cumulative Exergy Consumption (CExC) have been developed. One example is the Ecological Cumulative Exergy Consumption (ECEC) that extends the CExC by accounting for the contribution of ecosystem services (e.g. rain, wind, pollination, etc.). ECEC therefore takes into account the solar, tidal and deep earth exergy consumed by ecological processes (Hau and Bakshi, 2004). Another example is situated in the field of economic analysis. Extended Exergy Accounting (EEA) calculates an exergy value for production costs such as capital and labour. Conversion factors for capital and labour hours are calculated by dividing the total net primary exergy input of a society, which is time and case specific, by the corresponding monetary circulation or number of working hours in the society, respectively. In addition to capital and labour, EEA takes into account the exergy use in abatement processes of emissions (Sciubba, 2001). Although it makes sense to include the exergy use for transformation of emissions to streams that cannot pollute or harm the environment anymore, this approach cannot replace the emission-oriented impact assessment methods developed in the conventional LCA framework, because the abatement exergy cannot really reflect the environmental impact of emissions (Dewulf et al., 2008).

1.2.2.2 Overview of resource-oriented methods

This section covers successively methods that address (i) abiotic resource use, (ii) biotic resource use, (iii) water use and (iv) land use.

Abiotic resource use

Methods that evaluate abiotic resource use can be divided in methods that account for overall natural resource use along the life cycle (resource accounting methods) and methods that address the scarcity of resources at midpoint or endpoint level (resource depletion methods).

Resource accounting methods were already discussed at pages 32 and 33. Because abiotic resource use can consist of both material use (e.g. minerals and metals) and energy use (e.g. fossil energy, wind energy, etc.), exergy-based resource accounting methods are particularly suitable to account for overall abiotic resource use (Swart et al., 2015).

The abiotic depletion potential (ADP) (Guinée et al., 2002) is an example of a commonly used framework to assess abiotic resource use at midpoint level (Equation 1.1). This framework is based on the use-to-availability ratio of the considered abiotic resource relative to the one of the reference substance antimony (Sb).

$$ADP_i = \frac{\frac{DR_i}{(R_i)^2}}{\frac{DR_{ref}}{\left(R_{ref}\right)^2}} \tag{1.1}$$

With ADP_i the abiotic depletion potential of resource i, R_i the ultimate reserve of substance i (kg), DR_i the extraction rate of resource i (kg/year), R_{ref} the ultimate reserve of the reference substance (kg) and DR_{ref} the extraction rate of the reference substance (kg/year). The use of ultimate reserves in this framework, however, has been subject to debate. Ultimate reserves are the total amount of the considered substance available on Earth. Because ultimate reserves are so large, their use in this framework implies that there would be no scarcity issue (Swart et al., 2015). Because only the reserves that can eventually be extracted are relevant, The European Commission (2011c) recommends to use ultimately extractable reserves, for which characterization factors are available from van Oers et al. (2002). Ultimately extractable reserves include deposits that meet certain minimal requirements to become potentially economically exploitable in a long-term perspective, taking into account possible improvements in mining technology. The ADP approach was implemented in the CML method for metals, minerals, fossil energy, atmospheric resources and nuclear energy (van Oers, 2012).

At endpoint level, abiotic resource depletion is often assessed by accounting for the future consequences of resource extractions, i.e. additional efforts in terms of energy and costs to extract resources in the future. Examples are the 'surplus energy' concept used in the Eco-Indicator 99 framework (Goedkoop and Spriensma, 2001) and the 'surplus cost' concept in the ReCiPe method (Goedkoop et al., 2013). To assess abiotic resource depletion appropriately, Swart et al. (2015) concluded that further developments are needed to address uncertainty issues, such as in the estimation of the actual amount and quality of available stocks.

Biotic resource use

Although biotic resources (extracted from natural systems, see definition page 32) received relatively little regard within LCA, they can be evaluated by similar methods as abiotic resources. Mass-, energy- or exergy-based resource accounting methods include biotic resources by accounting for their mass, energy or exergy content. Regarding depletion of biotic resources, a biotic depletion potential could be calculated in a similar way as the ADP, taking another reference, e.g. the reserve of African elephants (Guinée et al., 2002). More recently, midpoint impact assessment methods were developed to assess biotic depletion by overfishing (Emanuelsson et al., 2014; Langlois et al., 2014). These methods are based on the concept of Maximum Sustainable Yield (MSY), which is the highest wild fish catch that can be sustained in the long term.

Water use

Assessment of water use usually focuses on freshwater consumptive use, which is used freshwater that is not released into the same watershed from which it was withdrawn. Freshwater degradative use, which considers an alteration of the quality of the used water, is much less considered as such and usually replaced by emission-oriented methods (e.g. eutrophication, ecotoxicity, etc.) (Berger and Finkbeiner, 2010).

Methods that account for water use at the inventory level can be distinguished from methods that account for water use at the impact assessment level. While the most straightforward approach at the inventory level only accounts for the volume of *blue* water use, other approaches also account for *green* and/or *grey* water uses. Blue water consumption includes uses of ground and surface water. Green water is precipitation on land that does not run-off or recharges aquifers and is stored in the soil or temporarily stays on top of the soil and vegetation. Grey water use equals a virtual amount of water that is required to dilute the used water until it reaches commonly agreed quality standards (Berger and Finkbeiner, 2010). The water footprint method introduced by Hoekstra (2011) takes into account the three types of water uses, however, this approach has been subject to much debate. Especially the inclusion of green and grey water in water footprints is often contested. Consumption of green water generally does not contribute to local freshwater scarcity, which suggests that it should not be included in impact assessment. Regarding grey water, water pollution could be assessed more suitably in other (emission-oriented) impact categories (Milà i Canals et al., 2009; Pfister et al., 2009).

The use of water as a material flow is taken into account in mass- and exergy-based resource accounting methods, whereas it is not addressed by energy-based resource accounting methods.

At the impact assessment level, the withdrawal-to-availability (WTA) ratio is a commonly used indicator for local water scarcity. WTA is defined as the ratio of total annual (blue) freshwater withdrawal for human uses in a specific region (W) to the annually available renewable water supply in that region (A) (Berger and Finkbeiner, 2010). Renewable water resources can be distinguished from non-renewable water resources, i.e. deep aquifers that have a negligible rate of recharge on the human time scale (FAO, 2003). Pfister et al. (2009) introduced the water stress index (WSI), which is based on the WTA ratio but takes into account seasonal variations in water availability. By multiplying the WSI with blue water consumption, midpoint impacts are obtained. Pfister et al. (2009) also proposed endpoint indicators for the three AoPs human health, natural environment and natural resources according to the Eco-Indicator 99 framework (Goedkoop and Spriensma, 2001). To quantify damage to the AoP human health, Pfister et al. (2009) consider the impact pathway of malnutrition due to lack of irrigation water, which is based on the WSI index and the calculation of the annual number of malnourished people. Damage to the AoP natural environment is taken into account by considering local water shortage constraints for natural net primary production, and comparing the blue water consumption with the precipitation quantity in a certain area. Damage to the AoP natural resources is quantified by multiplying the surplus energy needed for replacing depleted freshwater by means of seawater desalination with the fraction of water consumption contributing to freshwater depletion.

In addition to differentiating different input freshwater sources, the use of water can be classified into evaporative and non-evaporative use, referring to how the used water returns to nature. While non-evaporative water use involves water that is returned to the water basin after use and that is then available to other users, evaporative water use refers to dissipated water that is not immediately available after use (Milà i Canals et al., 2009). Based on all these distinctions, Milà i Canals (2009) suggests two midpoint impact categories for freshwater use. One is freshwater depletion (FD), which could be linked to the AoP natural resources at endpoint level, while another is freshwater ecosystem impact (FEI), which could be linked to the AoP natural environment. FD assesses the reduced availability of freshwater in case its use exceeds the renewability rate of the respective water body, therefore, only the evaporative groundwater use and the use of non-renewable 'fossil' water (both evaporative and non-evaporative use) are taken into account. Contribution of these water uses to FD is quantified according to the abiotic depletion potential (ADP) framework (Guinée et al., 2002), which is based on the use-to-availability ratio of the considered abiotic resource (i.e. water in this case) relative to the one of the reference substance antimony (Sb) (see Equation 1.1).

The second midpoint impact category freshwater ecosystem impact (FEI) assesses the ecological water scarcity in a certain region and takes into account evaporative blue water use and changes in water availability due to land use change. Contribution of these water uses to FEI is quantified according to the WTA ratio but 'reserving' part of the renewable freshwater supply for sustaining the local ecological functions (Berger and Finkbeiner, 2010; Milà i Canals et al., 2009).

Land use

Quantification of the occupied area during a time period (expressed in m^2*year) is the most straightforward way of accounting for land use, because this approach only involves data collection without further impact assessment. Besides land occupation, land transformation or land use change (LUC) (expressed in m^2), i.e. change from one land use type to another, is often considered (Mattila et al., 2012).

Other, more complex, methods focus on environmental consequences linked to land occupation and land transformation, such as impacts on soil quality, biotic production potential (i.e. long-term ability of land to produce biomass) and biodiversity. Regarding soil quality, changes in soil organic matter (SOM) and soil organic carbon (SOC) are suggested as midpoint indicators (Brandão et al., 2011; Milà i Canals et al., 2007b). To characterize land use impacts on the biotic production potential, i.e. an important endpoint for the AoP natural resources, Brandão and Milà i Canals (2013) show that the change in SOC can be used as an indicator, because SOC relates to a range of soil properties responsible for soil resilience and fertility. At the endpoint level for the AoP natural environment, several methods consider land use impacts on species diversity loss (Goedkoop et al., 2013; Goedkoop and Spriensma, 2001; Koellner and Scholz, 2007; Koellner and Scholz, 2008).

Other examples of methods that focus on land use are methods based on the Ecological Footprint (Ewing et al., 2008; Huijbregts et al., 2008; Venetoulis and Talberth, 2007; Wackernagel and Rees, 1996) and methods based on the human appropriation of net primary production (HANPP) (Alvarenga et al., 2013b; Haberl et al., 2007; Taelman et al., 2016). The Ecological Footprint is defined as 'the biologically productive land and water area a population requires to produce the resources it consumes and to absorb part of the waste generated by fossil and nuclear energy consumption' (Wackernagel and Rees, 1996). Results of the Ecological Footprint are easy to communicate, because they can be compared with the actual land available on the Earth. While the Ecological Footprint addresses the overshoot of the Earth's carrying capacity, the HANPP indicator addresses the intensity of land use, which is related to the risk of biodiversity loss (Haberl et al., 2004). HANPP makes use of net primary production (NPP), which is the net amount of plant biomass produced through photosynthesis per unit of time and area. The HANPP indicator measures the difference in the NPP left for ecosystems between a reference natural state and the current land use, obtaining the NPP loss or increase due to human intervention (e.g. harvest of biomass, change of land use type). The HANPP result can thus be positive (NPP loss) or negative (NPP increase). In case of irrigated land or intensive agricultural land use, the actual NPP can be higher than the potential NPP of the natural vegetation (Haberl et al., 2007).

Furthermore, methods exist that account for land use from a thermodynamic point of view. The thermodynamically-based concept of exergy is used to quantify the exergy deprived from nature due to human-controlled land use. This approach has been operationalized in the Cumulative Exergy Extraction from the Natural Environment (CEENE) method (Dewulf et al., 2007a), of which to date two versions with a different conceptual approach for land use accounting exist, i.e. CEENE v2007 (Dewulf et al., 2007a) and CEENE v2013 (Alvarenga et al., 2013c) (see also section 1.2.2.1). When considering the three areas of protection (AoP natural resources, AoP natural environment and AoP human health), the application of conventional exergy-based resource accounting should be seen especially in the first area ‘natural resources’. Recently, however, Taelman et al. (2016) developed two exergy-based indicators, based on an actual NPP loss, to assess land use impacts on biodiversity within the AoP natural environment. NPP has already been used as proxy for damage assessment in the AoP natural environment (Costanza et al., 2007; Nunez et al., 2013; Pfister et al., 2009), due to its correlation with damage on vascular plant species biodiversity. According to Taelman et al. (2016), the actual loss of NPP can be calculated on the basis of two concepts: HANPP and naturalness. The naturalness concept is based on descriptive (qualitative) conditions and measures the difference in ‘naturalness’ between a reference natural state and the current land use. For both indicators, Taelman et al. (2016) calculated spatially differentiated characterization factors in exergy terms.

Because of the complexity of land use impacts, a scientific debate is still ongoing about which types of land use impacts should be quantified and which indicators are most suitable (Michelsen and Lindner, 2015; Milà i Canals et al., 2007a; Taelman et al., 2016).

1.3 Aims and outline of this thesis

Because exergy analysis has primarily been elaborated in the energy, chemical and metallurgical industries, it needs further development to assess overall natural resource use and its efficiency in an agricultural context. The *general objective* of this PhD thesis is to improve the framework of exergy-based natural resource accounting for its application within sustainability assessment of agricultural production systems, and to provide insight into its value by case study illustrations.

Given the context described in the previous sections, the *focus* of this PhD thesis is twofold. *Thematically*, this work focuses on two major challenges within the current debate on sustainable development of agriculture, i.e. (i) the growing demand for bio-based products to substitute their fossil-based counterparts in a bioeconomy, and (ii) the increasing environmental concerns about intensive livestock production, which is narrowed down to dairy farms in this thesis. *Methodologically*, this work considers the exergy accounting methodology to evaluate (cumulative) overall natural resource use and its efficiency.

To achieve the general objective, five *specific objectives* are formulated and will be addressed in Chapters 2 to 4.

1. Given the competition for land between food, feed, biomaterials and bioenergy, optimizing the use of bio-productive land is essential to meet future demand for biomass. While some existing resource accounting methods (RAMs) do not include land resources, others have different accounting principles. Guidelines about how to account for land resources in the calculation of overall natural resource efficiency are lacking. The first specific objective, therefore, is to identify the most appropriate way to account for bio-productive land resources as an input during the quantification of overall natural resource efficiency, in order to develop an improved framework, and to show, by means of case studies of primary biomass products, how this efficiency score is affected by different existing and newly developed accounting approaches (Chapter 2).

2. The second specific objective is to further improve the developed framework for quantification of overall natural resource efficiency, by including the non-renewable character of fossil resources, and to show, by means of case studies of final bio-based products and their fossil-derived counterparts, how this modification affects their efficiency score (Chapter 2).
3. Because many agricultural systems have become high input/high output systems under the influence of the Green Revolution, evaluation of overall natural resource use is very relevant to improve their environmental performance. Although exergy analysis is a well-known tool for resource efficiency evaluation of technological systems in industries, it is much less applied in an agricultural context. The third specific objective, therefore, is to demonstrate a generic exergy-based framework for evaluation of overall natural resource use of agricultural systems at both the process level and the life cycle level, by means of a case study of a dairy farm (Chapter 3).
4. Analysis of the overall natural resource use of a dairy farm's supply chain in Chapter 3 identifies feed as the by far most resource-demanding input. Because feed is also the most important cost at dairy farms (Hemme et al., 2014), the fourth specific objective is to investigate whether feed costs and overall natural resource use in the feed supply chain can simultaneously be reduced, without reducing farm revenues. Because improvement options may be farm-specific, the aim is to identify whether a specific farm can achieve an economic-exergetic win-win or whether this farm is in an economic-exergetic trade-off situation. To achieve this objective, exergy-based resource accounting is integrated with frontier analysis, a method based on economic production theory, which has already shown its usefulness for economic-environmental optimization (Van Meensel et al., 2010a) (Chapter 4).
5. The fifth specific objective is to identify underlying characteristics that may explain dairy farm economic and exergetic performance and to facilitate communication and validation of the identified economic-exergetic improvement paths by analysis of Key Performance Indicators (KPIs), which are traditionally used by farmers and their advisors (Chapter 4).

Finally, Chapter 5 discusses what can be learned from the case studies with respect to both thematic and methodological issues; conclusions are drawn and perspectives for further research are provided. A schematic representation of the different chapters of this PhD thesis is depicted in Figure 1.15.

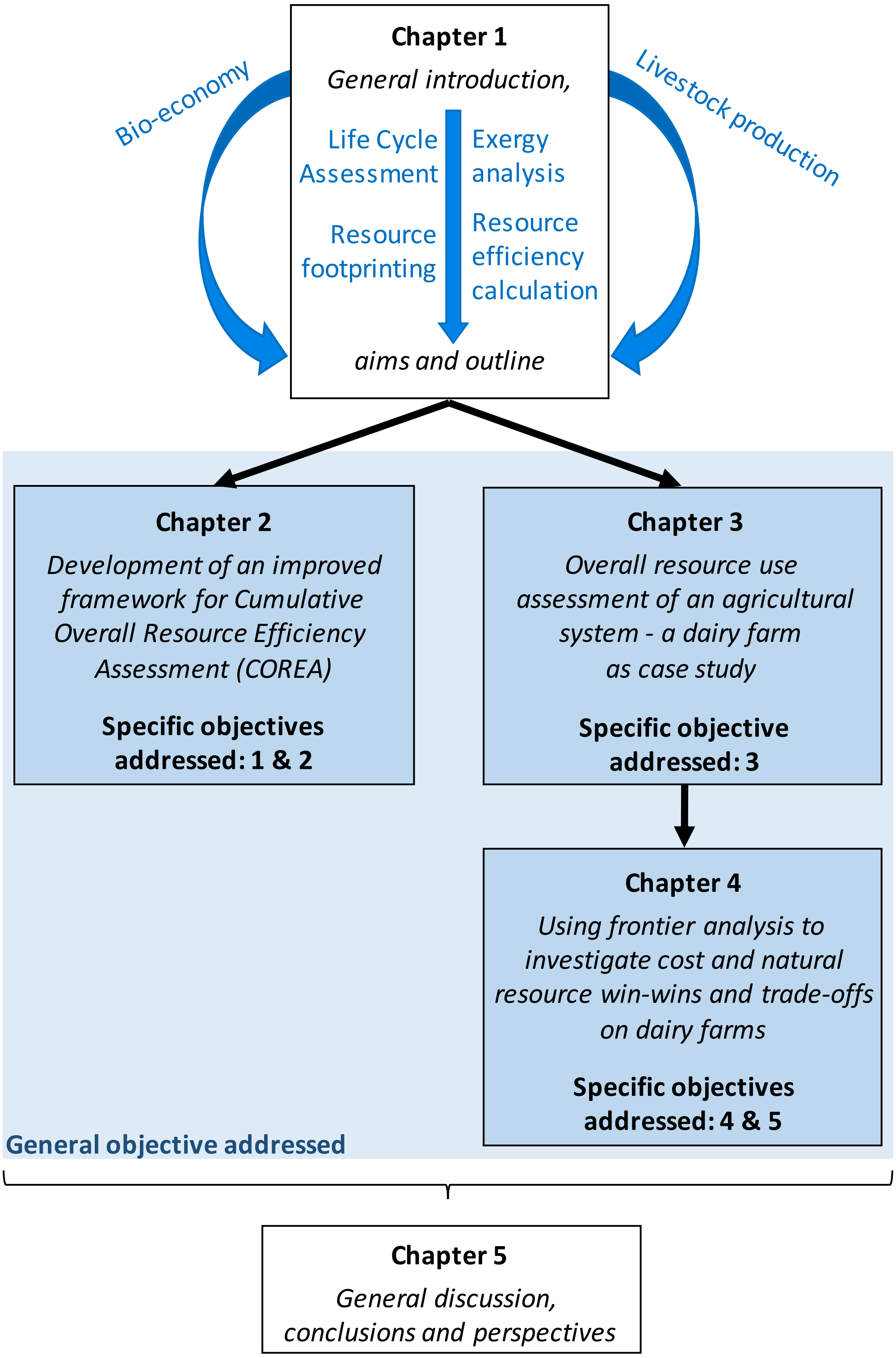


Figure 1.15 Schematic representation of the different chapters of this PhD thesis.

CHAPTER 2

DEVELOPMENT OF AN IMPROVED FRAMEWORK FOR CUMULATIVE OVERALL RESOURCE EFFICIENCY ASSESSMENT (COREA)

Redrafted from:

Huysveld, S., De Meester, S., Van linden, V., Muylle, H., Peiren, N., Lauwers, L. and Dewulf, J. (2015). Cumulative Overall Resource Efficiency Assessment (COREA) for comparing bio-based products with their fossil-derived counterparts. *Resources, Conservation and Recycling* 102, 113-127.

CHAPTER 2: DEVELOPMENT OF AN IMPROVED FRAMEWORK FOR CUMULATIVE OVERALL RESOURCE EFFICIENCY ASSESSMENT (COREA)

Abstract

Bio-based products potentially decrease consumption of non-renewable fossil resources compared to their fossil-derived counterparts, but are more demanding for bio-productive land use. Although thermodynamics-based resource accounting methods are available for calculating overall resource efficiency from a life cycle perspective, their accounting for bio-productive land resources as an input during the quantification of efficiencies is unclear. This work aims to fill the gap in scientific literature about how to calculate a cumulative overall resource efficiency indicator by developing a framework, called Cumulative Overall Resource Efficiency Assessment (COREA). COREA (i) takes into account bio-productive land resources and (ii) addresses the non-renewable character of fossil resources. To account for bio-productive land resources, two methodological questions need to be addressed: 1) 'how to define the system boundary of the solar energy input in the primary biomass production system?' and 2) 'how to choose the temporal system boundary of this system?'. Resource efficiencies are calculated for three cases at crop level and two cases at bio-based product level. To account for the non-renewable character of fossil resources, we propose an accounting approach that includes the ancient solar energy consumption of fossil resources. This methodological choice is illustrated through comparing the resource efficiencies of the two bio-based products with their fossil-based counterparts. The results showed that the bio-based products only had a higher resource efficiency than their fossil-derived counterparts if fossil resources were considered as ancient consumers of solar energy.

2.1 Introduction

2.1.1 Land use efficiency is a key element towards a renewables-based economy

Increasing resource efficiency is a major challenge in our society's sustainable development (European Commission, 2011a). Some natural resources, defined as 'objects of nature which are extracted by man from nature and taken as useful input to man-controlled, mostly economic, processes' (Udo de Haes et al., 2002), are extracted from finite stocks. Their continuing extraction will unavoidably result in depletion (e.g. fossil fuels). Other resources are renewable, but their use is subject to competition because of limiting factors (e.g. land availability) (Swart et al., 2015). In both cases, a key feature of sustainable processes is the optimized conversion of resources into products. This optimization can be performed at different levels: from single processes (gate-to-gate perspective) to complete production chains (life cycle perspective). In recent decades, Life Cycle Assessment (LCA) has become a widely used tool to evaluate the environmental sustainability of products along the production chain (Guinée et al., 2002). With the rising trend towards a renewables-based economy, bio-based products are increasingly compared with their fossil-derived counterparts from a life cycle perspective (Adom et al., 2014). Normally, bio-based products substitute for non-renewable energy and materials, but research also revealed that this may happen at the expense of additional use of other resources, like land, water and minerals, and associated environmental impacts, such as eutrophication (De Meester et al., 2011). Given the food-feed-fuel competition, bio-productive land is limited to fulfil the demand for biomass, which is expected to increase in a more renewables-based economy (UNEP, 2014). Optimising the use of bio-productive land is essential to meet future demand for biomass.

In order to quantify the fossil resource savings of bio-based products at the expense of additional land use, the metric *land use efficiency* can be used (e.g. Bos et al. (2012)). This metric (expressed in GJ/ha) is defined as the ratio between the savings in non-renewable energy use and the additional land use of a bio-based product compared to its fossil-based alternative (Pawelzik et al., 2013). This metric, however, does not reflect the actual efficiency of the conversion of resources into products. Moreover, it does not take into account natural resources such as metals, minerals and water. A complete resource accounting method (RAM) should be chosen, but scientific literature shows a gap in guidelines about how to calculate an overall resource efficiency indicator, taking into account all different resources including bio-productive land resources.

2.1.2 Indicators for resource efficiency

A diversity of resource efficiency indicators has been developed in the past. This research situates in the field of environmental science and engineering, in which biophysical and no monetary metrics are used. For the sake of clarity, we use in this work the definition of resource efficiency in *sensu stricto*, meaning that only resources and no emissions are taken into account, in contrast to resource efficiency metrics in *sensu lato* (Huysman et al., 2015).

To design production chains towards a higher resource efficiency, we first take a look at the existing indicators from process engineering. Process efficiencies are often based on the thermodynamic laws. According to the first law, mass and energy are conserved during every process: they cannot be destroyed or created (Dincer and Rosen, 2013c). The mass and energy efficiency indicators quantify how much of the input mass and energy, respectively, is embedded in the useful outputs. Only taking into account either mass or energy is a shortcoming of these metrics when aiming to calculate overall resource efficiency (Van der Vorst et al., 2010). This limitation can be overcome with the exergy concept as a quantifier for both the amount and quality of material and energy flows in one common unit, i.e. joules of exergy (J_{ex}) (see also section 1.2.3 in Chapter 1). The exergy concept originates from the second law of thermodynamics, which states that every process transforms resources into work, heat, and/or products, by-products and wastes, and generates entropy. The sum of the exergy embodied in these outputs is lower than the input of exergy in the resources, because part of the initial exergy dissipates through irreversible entropy production. The quality of resources thus decreases in every transformation step. Exergy therefore takes into account both the quality and the quantity of resources, while energy only includes their quantity (Dewulf et al., 2005; Szargut et al., 1988). Quantification of both material and energy flows on one single scale makes the calculation of an overall resource efficiency metric rather straightforward. The process exergy efficiency η is defined as the ratio between all useful outputs (products) and all required inputs (resources) of the process, all quantified in exergy (Dewulf and Van Langenhove, 2005).

Second, towards an overall resource efficiency from a life cycle perspective, we can appeal to Szargut et al. (1988), who extended exergy analysis beyond a single process and introduced the concept of Cumulative Exergy Consumption (CExC). The CExC is calculated by the sum of exergy contained in all resources extracted from the natural environment ('the cradle') throughout the supply chain of a product or service. The CExC concept enables the calculation of a cumulative resource efficiency, called Cumulative Degree of Perfection (CDP), which equals the ratio of exergy contained in a product (Ex_p) to the CExC of its supply chain (Szargut et al., 1988) (Equation 2.1).

$$CDP_{ex} = Ex_p\ (J_{ex})/CExC\ (J_{ex}) \tag{2.1}$$

For comparison, the Cumulative Energy Requirement Analysis (CERA) (Boustead and Hancock, 1979; Pimentel et al., 1973) is solely based on the first law of thermodynamics and focuses only on primary energy use (expressed in energy joules (J_{en})) and not on material use. Using CERA, the CDP can be calculated similarly by the ratio of the gross calorific value of a product (En_p) to the Cumulative Energy Consumption (CEnC) of its supply chain (Equation 2.2).

$$CDP_{en} = En_p\ (J_{en})/CEnC\ \ (J_{en}) \tag{2.2}$$

The methodological framework for calculating exergy efficiency (both η at process level and CDP at life cycle level) has been elaborated for non-bio-based processes in the mainly fossil-based chemical and metallurgical industries (Szargut et al., 1988). Guidelines about how to account for land resources in overall resource efficiency assessment of bio-based processes are lacking in scientific literature and are very relevant in the context of the upcoming bio-based economy. Exergy analyses of photosynthesis, the basic process of primary biomass production, have rather rarely been applied (Petela, 2008), but have been performed in Bisio and Bisio (1998), Reis and Miguel (2006), Petela (2008) and Lems et al. (2010). These analyses account fully or partially for the input of solar radiation on occupied land. When the entire amount of solar radiation is taken into account, crops achieve dramatically low efficiencies (Dewulf et al., 2005). Because the photosynthetic process can inherently utilize only a portion of the solar spectrum, i.e. the photosynthetically active radiation (PAR), a distinction is often made between the total solar radiation and its PAR fraction (Bisio and Bisio, 1998; Petela, 2008; Reis and Miguel, 2006). In addition to the non-PAR fraction of the solar radiation, other inherent natural losses are occurring during the conversion of solar energy into biomass (Zhu et al., 2010). Therefore, a useful resource efficiency indicator for optimization of human-controlled processes needs to distinguish between inherent natural inefficiencies and inefficiencies that could be tackled by human intervention.

2.1.3 Development of a framework to calculate a cumulative overall resource efficiency indicator

The research objective is to develop a framework for the calculation of a cumulative overall resource efficiency, and thus called *Cumulative Overall Resource Efficiency Assessment (COREA)* framework, that (i) takes into account bio-productive land resources and (ii) addresses the non-renewable character of fossil resources. For the first challenge, we combine knowledge from two different research domains, i.e. Life Cycle Assessment (LCA) and photosynthesis research. We start to describe the available resource accounting methods (RAMs) that were developed in the past decades for application within the LCA framework, with a focus on land resources accounting. We critically analyse available thermodynamics-based RAMs, with different levels of comprehensiveness in terms of overall resource accounting and different conceptual rationales, for calculating a useful resource efficiency indicator. Then, we address two questions about the system boundary definition of the primary biomass production. First, how to define the system boundary of the input of solar energy into the primary biomass production system? This question is addressed with photosynthesis research of Zhu et al. (2010), who quantified the minimum energy losses in each step of the conversion of solar energy into biomass. Second, how to define the temporal system boundary of the primary biomass production? As land use equals the occupation of a piece of land during a given period, this temporal system boundary will play an important role in the CDP calculation. To support this discussion, we calculate resource efficiencies for three cases at crop level and two cases at bio-based product level.

To further improve comparison of bio-based with fossil-based products, we include the non-renewable character of fossil resources in the framework. When thermodynamics-based RAMs account for the energy or exergy content of fossil resources that are extracted from finite stocks, the ancient consumption of solar energy during the formation of fossil resources is overlooked. Based on the work of Dukes (2003), who quantified this ancient solar energy consumption, we introduce an accounting approach for fossil resources that reflects their non-renewability. To support this discussion, we compare the two bio-based products with their fossil-based counterparts.

The focus of this research is on primary biomass production in human-made systems (agriculture), not in natural systems (e.g. rainforest), nor is the focus on solar-based technologies such as photovoltaics.

2.2 Towards a cumulative overall resource efficiency indicator

2.2.1 Accounting for bio-productive land resources

2.2.1.1 Appealing to Life Cycle Assessment (LCA) research

Land use is reported as one of the key methodological issues in LCA studies of bio-based materials (Pawelzik et al., 2013). Approaches to account for land use and land use-related environmental impacts in LCA developed in recent years are not always suitable for calculating overall resource efficiency, so, we first give a brief overview.

Land use generally refers to land occupation whereas *land use change (LUC)* is similar to land transformation (Mattila et al., 2012). In the context of land occupation, we distinguish between methods accounting for the occupied land from a resource viewpoint and methods addressing the environmental impacts linked to land occupation. The first group considers land as a limited resource, while the second group focuses on soil quality and biodiversity. Mattila et al. (2012) distinguish three categories of land use indicators: 1) resource depletion, 2) soil quality and 3) biodiversity. To address soil quality, Milà i Canals et al. (2007b) and Brandão et al. (2011) developed a calculation method for the soil organic carbon (SOC) indicator, expressed in kg C per m^2*year. Examples of impact assessment methods that address biodiversity are Eco-indicator 99 (EI99) (Goedkoop and Spriensma, 2001), Impact 2002+ (Jolliet et al., 2003), Solar Exergy Dissipation (Wagendorp et al., 2006), Ecosystem damage (EDP) (Koellner and Scholz, 2007; Koellner and Scholz, 2008), ReCiPe v1.08 at the endpoint level (Goedkoop et al., 2013), and the work of de Baan et al. (2013). In the first category of methods, i.e. resource depletion, Mattila et al. (2012) classified methods such as the

Ecological Footprint (Ewing et al., 2008) and methods that use inventory data (expressed in m^2*year) as midpoint impact category results, e.g. CML (Guinée et al., 2002). However, an important share of the available resource accounting methods (RAMs) that account for land occupation was not considered in Mattila et al. (2012). The ignored methods are based on thermodynamics and seem in particular suitable for the calculation of overall resource efficiencies, because they enable to quantify both the product and the required resources on a common scale.

Among the thermodynamics-based RAMs, we can distinguish energy and exergy accounting methods, based on the first and the second law of thermodynamics, respectively. These methods were developed for application within the LCA framework and can be used to calculate a cumulative overall resource efficiency or Cumulative Degree of Perfection (CDP) (Huysman et al., 2015). Understanding the rationales of different thermodynamics-based RAMs and examining their effect on the CDP is essential for interpretation of the CDP results.

Regarding land resources, two major accounting approaches can be distinguished among the thermodynamics-based RAMs (Alvarenga et al., 2013c). The first approach does not account for land occupation but for the biomass output, i.e. the energy or exergy content of the harvested biomass. Thermodynamics-based RAMs applying this approach are the Cumulative Energy Demand (CED) (Frischknecht et al., 2007; VDI, 1997) and the Cumulative Exergy Demand (CExD) (Bösch et al., 2007). The second approach accounts for the surface area and time (m^2*year) needed to produce the biomass. Thermodynamics-based RAMs with this approach are the Cumulative Exergy Extraction from the Natural Environment (CEENE) (Dewulf et al., 2007a), of which to date three versions exist, i.e. CEENE v2007 (Dewulf et al., 2007a), CEENE v2013 (Alvarenga et al., 2013c) and CEENE v2014 (Taelman et al., 2014), and the Solar Energy Demand (SED) (Rugani et al., 2011). CEENE v2014 is an extended version of CEENE v2013, because CEENE v2014 also accounts for marine area occupation. As CEENE v2013 and CEENE v2014 have the same accounting approach for land resources, CEENE v2014 is not further considered in this work.

The focus of this work is on thermodynamics-based RAMs, which were operationalized for the process-based life cycle inventory database *ecoinvent*. Briefly, to calculate cumulative energy or exergy consumption values in general, the energy or exergy contained in the natural resources used throughout the supply chain is quantified. For each RAM, conversion factors, defined as the energy or exergy content of the considered resource reference flow (J_{en} or J_{ex}) per unit of the reference flow as it is defined in *ecoinvent*, were established. The cumulative energy or exergy value of a described product in *ecoinvent* is then calculated by the summation (over all resource reference flows) of the products of the conversion factor of the reference flows (J_{en} or J_{ex}/unit resource) and the cumulative amount of these reference flows necessary to obtain that product. Considering the land occupation reference flows of *ecoinvent*, all land occupied by human-made systems was taken into account, except occupied land that is not bio-productive (construction site, dump site, industrial area, mineral extraction site, traffic area and urban area). For more detailed information, we refer to the scientific papers that explain the rationale of these RAMs (Alvarenga et al., 2013c; Bösch et al., 2007; Dewulf et al., 2007a; Rugani et al., 2011; VDI, 1997). Table 2.1 gives on overview of the resources considered in the thermodynamics-based RAMs.

Table 2.1 Type of resources considered by the thermodynamics-based resource accounting methods (RAMs).

	CED	CExD	CEENE v2007	CEENE v2013	SED
Land resources	(a)	(a)	X	X	X
Water resources		X	X	X	X
Mineral resources		X	X	X	X
Metal resources		X	X	X	X
Fossil resources	X	X	X	X	X
Nuclear resources	X	X	X	X	X
Renewable energy resources[b]	X	X	X	X	X

[a] CED and CExD do not directly account for land occupation, but they account for the harvested biomass.
[b] Renewable energy resources include hydropower and wind energy in the case of all methods. In the case of SED, renewable energy resources also include geothermal energy. In the case of CED and CExD, renewable energy resources also include solar energy (in the context of solar-based technologies). In the case of CEENE v2007, CEENE v2013 and SED, solar energy (in the context of solar-based technologies) is included in the land resources category. In order to avoid double counting, it is not included in the category renewable energy resources.

Cumulative Energy Demand (CED) and Cumulative Exergy Demand (CExD)

The CED method only includes energy carrying resources, whereas the CExD method also considers non-energetic resources such as water, metals and minerals (Table 2.1). The CED and CExD methods do not directly consider land occupation; they indirectly account for a part of the solar radiation on occupied land, namely the share that is embedded in the harvestable part of the produced biomass. In doing so, the specific gross calorific value (in case of CED) or the specific exergy value (in case of CExD) of the harvested biomass is multiplied by the amount of the harvested biomass (Table 2.2). Equations 2.3 and 2.4 show how CED and CExD can be used to calculate the CDP, respectively.

$$CDP_{CED} = \frac{En_p\ (J_{en})}{En_b\ (J_{en}) + En_f\ (J_{en}) + En_{ne}\ (J_{en}) + En_{re}\ (J_{en})} \tag{2.3}$$

with

En_p: energy content of the product (J_{en})

En_b: energy content of biomass (J_{en})

En_f: energy content of fossil resources (J_{en})

En_{ne}: energy content of nuclear energy resources(J_{en})

En_{re}: energy content of renewable energy resources (J_{en})

$$CDP_{CExD} = \frac{Ex_p\ (J_{ex})}{\begin{array}{c} Ex_b\ (J_{ex}) + Ex_f\ (J_{ex}) + Ex_{ne}\ (J_{ex}) + Ex_{re}\ (J_{ex}) \\ +Ex_w\ (J_{ex}) + Ex_{mi}\ (J_{ex}) + Ex_{me}\ (J_{ex}) \end{array}} \tag{2.4}$$

with

Ex_p: exergy content of the product (J_{ex})

Ex_b: exergy content of biomass (J_{ex})

Ex_f: exergy content of fossil resources (J_{ex})

Ex_{ne}: exergy content of nuclear energy resources (J_{ex})

Ex_{re}: exergy content of renewable energy resources (J_{ex})

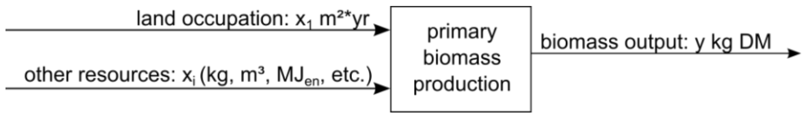
Ex_w: exergy content of water resources (J_{ex})

Ex_{mi}: exergy content of mineral resources (J_{ex})

Ex_{me}: exergy content of metal resources (J_{ex})

Table 2.2 How land resources are taken into account in five thermodynamics-based resource accounting methods (RAMs) in the context of human-made systems (e.g. agriculture).

Graphic representation of the inventory data of the primary biomass production process



Type of approach for land resources accounting	Resource accounting method (RAM)	Reference	How land resources are taken into account
Accounting for the biomass output (and not for land occupation)	**CED**	Frischknecht et al. (2007); VDI (1997)	specific energy content (gross calorific value) of the harvested biomass (expressed as MJ_{en}/kg DM) is multiplied by *y* kg DM example: 18.0 MJ_{en}/kg DM for maize silage (see section 2.3.4)
	CExD	Bösch et al. (2007)	specific exergy content of the harvested biomass (expressed as MJ_{ex}/kg DM) is multiplied by *y* kg DM example: 18.7 MJ_{ex}/kg DM for maize silage (see section 2.3.4)
Accounting for land occupation	**CEENE v2007**	Dewulf et al. (2007a)	2% of the solar surface irradiance (expressed as MJ_{ex}/m²*year) is multiplied by x_1 m²*year example: 68.14 MJ_{ex}/m²*year for average Western European conditions
	CEENE v2013	Alvarenga et al. (2013c)	site-specific potential natural net primary production (NPP) (expressed as MJ_{ex}/m²*year) is multiplied by x_1 m²*year global range: 0-64 MJ_{ex}/m²*year examples: - Germany 26.5 MJ_{ex}/m²*year - Brazil 38.8 MJ_{ex}/m²*year
	SED	Rugani et al. (2011)	6.17 * 10^4 MJ_{se}/m²*year (equals the ratio between the annual baseline emergy budget and the total land area in the world) is multiplied by x_1 m²*year

Cumulative Exergy Extraction from the Natural Environment (CEENE v2007 and CEENE v2013)

Table 2.1 shows that CEENE v2007 and CEENE v2013 account for the full range of resources. Regarding land resources, CEENE v2007 and CEENE v2013 use a different conceptual framework for assigning an exergy value to the surface area and time needed to produce the biomass: top-down versus bottom-up. While the top-down approach starts from the solar radiation exergy on occupied land (CEENE v2007), the bottom-up approach is based on the potential bioproductivity of the occupied land (CEENE v2013). CEENE v2007 uses the solar radiation that can be metabolized through photosynthesis by natural ecosystems, per unit area and time, as a proxy for land occupation. According to the rationale of the CEENE v2007 method, this solar exergy is considered as no longer available to nature due to land occupation by human-made systems (e.g. agriculture). In practice, the fraction of the solar radiation that is taken into account, has been set equal to 2% of the average surface solar irradiation for Western European conditions (i.e. 68.14 MJ_{ex}/m^2*year) (Table 2.2). This fraction is chosen as an upper limit for natural ecosystems, which merely attain 2.0% metabolization, of which about half is conserved and the other half is consumed through respiration (Dewulf et al., 2007a). In this way CEENE v2007 accounts for the gross primary production (GPP). For regions outside Western-Europe, the value of 68.14 MJ_{ex}/m^2*year should be modified based on local solar irradiance data.

Site-dependent factors such as climate and soil quality were not taken into account by CEENE v2007. To tackle this limitation, Alvarenga et al. (2013c) accounted for the occupied land through the exergy content of the potential natural net primary production (NPP) on that land (Table 2.2). Equations 2.5 and 2.6 show how CEENE v2007 and CEENE v2013 can be used to calculate the CDP, respectively.

$$CDP_{CEENE\ v2007} = \frac{Ex_p\ (J_{ex})}{Ex_{bpl\_SR\_2\%}\ (J_{ex}) + Ex_f\ (J_{ex}) + Ex_{ne}\ (J_{ex}) + Ex_{re}\ (J_{ex}) + Ex_w\ (J_{ex}) + Ex_{mi}\ (J_{ex}) + Ex_{me}\ (J_{ex})} \quad (2.5)$$

$$CDP_{CEENE\ v2013} = \frac{Ex_p\ (J_{ex})}{Ex_{bpl\_PNNPP}\ (J_{ex}) + Ex_f\ (J_{ex}) + Ex_{ne}\ (J_{ex}) + Ex_{re}\ (J_{ex}) + Ex_w\ (J_{ex}) + Ex_{mi}\ (J_{ex}) + Ex_{me}\ (J_{ex})} \quad (2.6)$$

with

Ex_p: exergy content of the product (J_{ex})

$Ex_{bpl\_SR\_2\%}$: 2% of the solar radiation exergy on occupied bio-productive land (J_{ex})

Ex_{bpl_PNNPP}: exergy content of potential natural net primary production of occupied bio-productive land (J_{ex})

Ex_f: exergy content of fossil resources (J_{ex})

Ex_{ne}: exergy content of nuclear energy resources (J_{ex})

Ex_{re}: exergy content of renewable energy resources (J_{ex})

Ex_w: exergy content of water resources (J_{ex})

Ex_{mi}: exergy content of mineral resources (J_{ex})

Ex_{me}: exergy content of metal resources (J_{ex})

Solar Energy Demand (SED)

Table 2.1 shows that SED accounts for the full range of resources. Conceptually, the SED method differs from the other thermodynamics-based RAMs, because SED delineates its system boundary between the Sun and the natural environment, while the other examples delineate their system boundary between the natural environment and the technosphere (the human-industrial system) (Alvarenga et al., 2013c). SED thus quantifies the solar energy needed to produce all required resources (expressed in solar energy joules (J_{se})). The embodied solar energy is also called *emergy*. Except for some methodological differences (Rugani et al., 2011), the SED method shares the same conceptual rationale as the broader emergy concept that was introduced by Odum (1996). In order to calculate the SED of a product or service, solar energy factors (SEF_i) for each type of resource flow i are required (expressed in J_{se} per unit resource flow). Generally, SEF_i are calculated by dividing the annual baseline of emergy that flows in the geobiosphere by the annual flow of the resource i. Rugani et al. (2011) explains the rationale of the SED method and the supplementary material includes the list of SEFs for *ecoinvent* reference flows. Several values for the annual baseline emergy budget can be found in literature; the SED method applies the value of $9.26 * 10^{18}$ MJ_{se} per year (Rugani et al., 2011). Land resources are characterized within the SED method by one single non-site-specific characterization factor of $6.17 * 10^4$ MJ_{se}/m^2*year (Table 2.2). This value was obtained by dividing the annual baseline emergy budget by the total land area in the world, i.e. $1.50 * 10^{14}$ m^2 (Rugani et al., 2011). Equation 2.7 shows how SED can be used to calculate the CDP.

$$CDP_{SED} = \frac{En_p\ (J_{en})}{SE_{bpl}\ (J_{se}) + SE_f\ (J_{se}) + SE_{ne}\ (J_{se}) + SE_{re}\ (J_{se}) + SE_w\ (J_{se}) + SE_{mi}\ (J_{se}) + SE_{me}\ (J_{se})} \quad (2.7)$$

with

En_p: energy content of the product (J_{en})

SE_{bpl}: solar energy assigned to occupied bio-productive land based on the baseline emergy budget (J_{se})

SE_f: solar energy needed to produce fossil resources based on the baseline emergy budget (J_{se})

SE_{ne}: solar energy needed to produce nuclear energy resources based on the baseline emergy budget (J_{se})

SE_{re}: solar energy needed to produce renewable energy resources based on the baseline emergy budget (J_{se})

SE_w: solar energy needed to produce water based on the baseline emergy budget (J_{se})

SE_{mi}: solar energy needed to produce minerals based on the baseline emergy budget (J_{se})

SE_{me}: solar energy needed to produce metals based on the baseline emergy budget (J_{se})

Using the thermodynamics-based RAMs for the purpose of CDP calculation

We now address the question how adequate each of the available thermodynamics-based RAMs resembles for the purpose of calculating overall resource efficiency or Cumulative Degree of Perfection (CDP). As far as we know, to date published CDP results were calculated using CEENE v2007 (De Meester et al., 2011; Dewulf et al., 2010; Huysveld et al., 2013; Van der Vorst et al., 2009) and CEENE v2013 (Nhu et al., 2015).

CED and CExD, which account for the harvestable part of the produced biomass, are regarded as not adequate for the purpose of CDP calculation, because of two reasons. First, biomass produced in agriculture cannot be taken into account as a natural resource, because it is a flow produced by a human-made system (cfr. definition natural resources by Udo de Haes et al. (2002)). Second, because CED and CExD do not consider land occupation, they do not allow accounting for differences in crop yield (produced biomass per unit of area and time) and thus they are not able to address land use efficiency.

SED delineates the system boundary between the Sun and the natural system, instead of between the natural system and the human-industrial system. While the other discussed thermodynamics-based RAMs account for natural resources as they are available in the natural environment (except for land resources in the case of CED and CExD), SED quantifies the solar energy needed to produce all types of natural resources. The conceptual rationale of SED thus goes beyond the definition of natural resources of Udo de Haes et al. (2002) and, therefore, SED might be questioned as an appropriate RAM for the purpose of calculating a useful resource efficiency indicator for optimization of human-controlled processes (Huysman et al., 2015). Another reason why the SED method seems not appropriate is the way in which the solar energy factors (SEF_i) are calculated. Except for oil and gas resources, the entire emergy baseline is divided by the formation rate of the resource, irrespective whether this amount of solar energy was really required to produce this resource (Rugani et al., 2011). Indeed, the allocation approach of the SED method is uncommon: this method assigns the total emergy budget to each of its different resource categories. Finally, the current SED approach for land use accounting does not allow one to apply spatially-differentiated characterization factors for land occupation.

CEENE accounts for land occupation and this method is consistent with the definition of natural resources of Udo de haes et al. (2002). In the case of CEENE v2013, CDPs higher than the upper limit on efficiency (i.e. 100%) are theoretically achievable, because the actual NPP of agricultural cultivation can be higher than the potential NPP of the natural ecosystem at a given location (DeLucia et al., 2014). Calculating CDPs higher than 100% is obviously not scientifically sound. In the case of CEENE v2007, which accounts by definition for the upper limit on the gross primary production (GPP) of natural ecosystems, it is not yet clear whether or not this approach is sufficient to avoid that CDPs higher than 100% are achievable in case of human-made systems. Before answering this question (see section 2.2.1.4), we first answer two important methodological questions when accounting for bio-productive land resources: 1) ‘how should we define the boundary of the solar energy input in the primary biomass production system?' and 2) 'how should we choose the temporal system boundary of this system?'.

2.2.1.2 Defining the boundary of the solar energy input in the primary biomass production system

Appealing to photosynthesis research

The maximum yield (Y_{max}) that crops can achieve under ideal conditions, i.e. optimal management and absence of (a)biotic stresses, can be calculated by multiplying the total surface solar irradiance across the growing season (S_t) by the maximum values of the light interception efficiency ($\varepsilon_{i,max}$), the conversion efficiency ($\varepsilon_{c,max}$) and the partitioning efficiency ($\varepsilon_{p,max}$) (Equation 2.8).

$$Y_{max} = S_t \times \varepsilon_{i,max} \times \varepsilon_{c,max} \times \varepsilon_{p,max} = S_t \times \varepsilon_{total,max} \tag{2.8}$$

The surface solar irradiance is site-dependent and can be spatially differentiated depending on the location. The light interception efficiency equals the fraction of the surface solar irradiance intercepted by the plant. The maximum light interception efficiency ($\varepsilon_{i,max}$) is close to 95% (Katerji et al., 2008). The partitioning efficiency, often called harvest index in the case of grains, quantifies how much of the total biomass energy is embedded in the harvestable part of the crop (Zhu et al., 2010). For the latter an absolute maximum value ($\varepsilon_{p,max}$) was not found in literature, but the highest partitioning efficiency that we have found in literature is 85% in the case of palm fruit production in Malaysia (Alvarenga et al., 2013b), considering the entire above-ground biomass (excluding weeds and lost biomass) as harvestable. The conversion efficiency is defined as the ratio of the produced chemical energy in biomass over a given period to the solar radiation energy intercepted by the plant canopy over the same period (Zhu et al., 2010). To identify the inherent natural energy losses during the conversion of solar energy into chemical energy in biomass, we appeal to the quantified levels of efficiency in energy transduction by Zhu et al. (2010) (Figure 2.1).

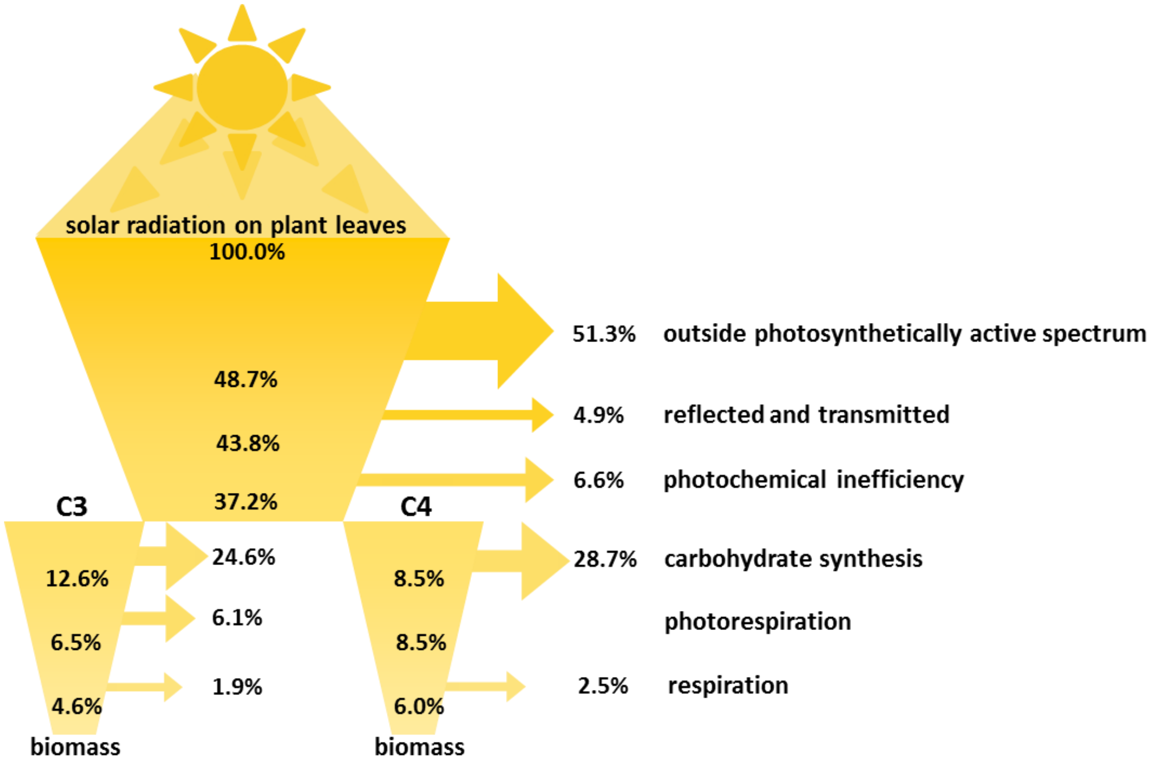


Figure 2.1 Representation of the minimum energy losses from the solar irradiation intercepted by plant leaves to the storage of chemical energy in the plant biomass. A distinction is made between C3 and C4 photosynthesis. This figure was redrafted from Zhu et al. (2010).

In addition to the non-PAR fraction of the solar radiation (51.3%), 4.9% of the total incident solar radiation is not absorbed by the chlorophyll in the plant leaves due to reflection and transmission. Another 6.6% is lost because of the ‘photochemical inefficiency’, i.e. heat loss due to relaxation of higher excited states of chlorophyll. Due to the second law of thermodynamics, the energy available for charge separation in the photosynthetic reaction centre is limited; 13.8% of the total incident solar radiation is lost during carbohydrate synthesis. A distinction had to be made between C3 and C4 photosynthesis. The C4 photosynthetic pathway has additional losses in the carbohydrate synthesis (14.9%) compared to the C3 pathway (10.8%) due to different requirements of adenosine triphosphate (ATP) molecules. ATP molecules store and transfer chemical energy within cells. C3 species, however, have energy losses due to photorespiration (6.1% of the total incident solar radiation), while C4 species have not (or almost not). Photorespiration is the non-desired process in which oxygen (O_2) is used instead of carbon dioxide (CO_2). Respiration for maintenance and growth is the final energy loss in both plant types. A minimum energy loss of 30% of the energy available prior to respiration was assumed based on experimental measurements (Zhu et al., 2010). Without major changes to the photosynthetic mechanism, all these losses are unavoidable. After quantification of these losses, Zhu et al. (2010) established the theoretical limit on the efficiency ($\varepsilon_{c,max}$) with which photosynthesis can convert solar energy into biomass under ideal conditions (i.e. optimal management and absence of (a)biotic stresses). A maximum conversion efficiency of solar energy into chemical energy in biomass of 4.6 and 6.0% was obtained for C3 and C4 species, respectively, at 30°C and 380 ppm atmospheric CO_2 concentration.

Considering the three theoretical maximum efficiencies $\varepsilon_{i,max}$, $\varepsilon_{c,max}$ and $\varepsilon_{p,max}$ (Equation 2.8), the total surface solar irradiance across the growing season (S_t) can be multiplied by a theoretical maximum total efficiency ($\varepsilon_{total,max}$) of 4.8% (considering the ultimate maximum conversion efficiency (6%) in the case of C4 species) (Table 2.3). This value is useful as efficiency reference to measure the distance reduction from the potential optimum that can be achieved by human intervention without altering the photosynthetic mechanism, i.e. a distance-to-target indicator. By taking into account 4.8% as an absolute upper limit for human-made systems, inherent natural inefficiencies can be excluded from the system boundary of primary biomass production and considered as part of the natural system.

According to Zhu et al. (2008), the highest observed conversion efficiencies are 2.4% for C3 crops and 3.7% for C4 crops across an entire growing season. The maximum observed conversion efficiency of 3.7% was seen in the production of the temperate perennial C4 grass *Miscanthus x giganteus* in south-eastern England (Beale and Long, 1995). With an interception efficiency of 83% and a partitioning efficiency of 74.5%, *Miscanthus x giganteus* was able to convert the solar surface radiation across its growing season in the second year (from April 24th until September 21th) into aboveground biomass with a total efficiency of 2.3% (Table 2.3). In addition to the theoretical maximum total efficiency of 4.8%, the actually observed maximum total efficiency of 2.3% can also be useful as efficiency reference, but in this case to measure the distance reduction from the actually observed optimum.

Table 2.3 Overview of the total efficiency (ε_{total}) and its constituent parts, i.e. the light interception efficiency (ε_i), the conversion efficiency (ε_c) and the partitioning efficiency (ε_p), for five solar system boundary levels.

solar system boundary level	abbreviation of level	light interception efficiency (ε_i)	conversion efficiency (ε_c)	partitioning efficiency (ε_p)	total efficiency (ε_{total})
		%	%	%	%
total surface solar radiation	**TOT**	95	100.0	100.0	95.0
photosynthetically active part of surface solar radiation	**PAR**		48.7		46.3
theoretical maximum convertible part of surface solar radiation into total biomass	**TMC**		6.0		5.7
theoretical maximum convertible part of surface solar radiation into aboveground biomass	**TMCA**		6.0	85.0	4.8
observed maximum convertible part of surface solar radiation into aboveground biomass	**OMCA**	83	3.7	74.5	2.3

By identifying the total efficiencies 4.8 and 2.3% as two useful distance-to-target efficiency levels, the first methodological question towards the development of the *Cumulative Overall Resource Efficiency Assessment (COREA)* framework, 'how should we define the boundary of the solar energy input in the primary biomass production system?', is answered. Depending on the purpose of the comparison, we recommend to use one of these two total efficiency levels. To show the effect of using these two approaches compared to the use of other solar system boundary levels with higher total efficiencies (Table 2.3), we calculate, in section 2.4, CDPs for three cases at crop level and two cases at bio-based product level. The use of bio-productive land is taken into account by the conceptual approach of the CEENE v2007 method, i.e. multiplying the solar radiation on occupied land at a given location with the total efficiency (ε_{total}), of which the value depends on the chosen solar system boundary (Table 2.3) (in case of CEENE v2007, the total efficiency equals 2%). For the sake of clarity, we will hereafter present the original CEENE v2007 method (Dewulf et al., 2007a) with the subscript '2%' (CEENE v2007$_{2\%}$) and the approaches based on the different solar system boundary levels introduced in this work as CEENE v2007$_{TOT}$, CEENE v2007$_{PAR}$, CEENE v2007$_{TMC}$, CEENE v2007$_{TMCA}$ and CEENE v2007$_{OMCA}$.

During which period the surface solar radiation should be taken into account, or in other words how to choose the temporal boundary of the primary biomass production system, is the second methodological question towards the development of the COREA framework.

2.2.1.3 Choosing the temporal boundary of the primary biomass production system

When crop efficiencies are calculated in photosynthesis research, the temporal system boundary of the primary biomass production system consists of the growing season of the studied crop (Equation 2.8). In other words, these crop efficiencies are obtained by only taking into account the surface solar radiation during the growing season. Even though the bio-productive land is not used for the cultivation of another crop, the portion of the year outside the growing season is not taken into account. From a resource efficiency point of view, however, it is more appropriate that an entire year of land occupation (i.e. 365 days) is taken into account and fully assigned to one (in case of monoculture systems or perennial systems) or more crop products (in case of multiple-cropping systems). In this way the land use efficiency as well as the crop efficiency are taken into account in the resource efficiency assessment of the primary biomass production system.

Accounting for land occupation in LCA research is usually done based on the cultivation period of the studied crop, i.e. from the moment of soil cultivation until the harvest of the crop. In the case of spring-sown crops (e.g. maize, sugar beets, etc.), however, the period during which the land is considered to be occupied can be broader than the actual cultivation period of the studied crop. Before cultivation of the spring-sown crop, a catch crop can be sown to cover the soil during winter in order to reduce soil erosion and nutrient loss, from which the spring-sown crop will benefit. The catch crop is then not harvested but ploughed into the soil. In this case, the period during which the catch crop, also called *green manure*, is present, is also included in the temporal system boundary of the succeeding main crop (Nemecek and Kägi, 2007). When the catch crop is however harvested, the occupied land can be allocated between the main crop and the catch crop. In order to perform a fair allocation, occupation of the occupied land should take into account the lower production potential of the soil during winter, by accounting for the seasonal variation of the surface solar radiation. Although the catch crop is harvested, its function can still be mainly the reduction of adverse effects on the soil instead of productivity. This approach, therefore, still might assign a too large proportion of the occupied land to the catch crop. For autumn-sown crops (e.g. wheat, barley, etc.), the crop itself covers the soil during winter and the occupied land is considered as the actual cultivation period of the studied crop (Nemecek and Kägi, 2007). In case of a planned crop rotating system, the occupied land should be allocated between the different crops, preferably also taking into account the seasonal variation of the surface solar radiation. In fact, when a piece of land is not occupied between two cultivations, i.e. fallow land, it should be allocated between the preceding and/or subsequent cultivation in order that an entire year of land occupation is taken into account. In case of perennial crops and grasses, which are not replanted or resown after each harvest, the inventory data are usually collected for multiple years until replanting or resowing. Based on these data, one-year average data are then calculated (Nemecek and Kägi, 2007).

In section 2.4, we show the effect of different temporal system boundaries on the calculated CDPs for three cases at crop level and two cases at bio-based product level.

2.2.1.4 *Are efficiencies higher than 100% achievable using CEENE v2007$_{2\%}$ in case of human-made systems?*

Previously we wondered whether the CEENE v2007$_{2\%}$ approach, which accounts for the upper limit on the GPP of natural ecosystems, is sufficient to avoid that CDPs higher than 100% are achievable in case of human-made systems. To answer this question, we can compare the 2% fraction of the surface solar radiation taken into account by CEENE v2007$_{2\%}$ with the actually observed maximum total efficiency of 2.3%. This efficiency was achieved over the growing season of *Miscanthus x giganteus* in the second year of cultivation; in other words, this efficiency is obtained when only taking into account the surface solar radiation from April 24th until September 21th (Beale and Long, 1995). As *Miscanthus x giganteus* is a perennial grass, it is appropriate from a resource efficiency perspective that an entire year of land occupation (i.e. 365 days) is taken into account. As the growing season of *Miscanthus x giganteus* corresponds with about 70% of the annual solar surface radiation, based on the profile of the solar surface radiation over an entire year, *Miscanthus x giganteus* was thus able to convert only 1.6% of the annual solar surface radiation into aboveground biomass. Comparing 1.6% with the fraction of the solar surface radiation taken into account by CEENE v2007$_{2\%}$, we can conclude that, with a status quo of the currently observed maximum achieved efficiency, efficiencies higher than 100% are not achievable with CEENE v2007$_{2\%}$.

2.2.2 Accounting for the ancient solar energy consumption of fossil resources to address their non-renewable character

The final element towards the C*umulative Overall Resource Efficiency Assessment (COREA)* framework is the inclusion of the non-renewable character of fossil resources. The CEENE method (similar for all three versions, i.e. CEENE v2007$_{2\%}$, CEENE v2013 and CEENE v2014) accounts for the exergy content of fossil resources that are extracted from finite stocks; fossil resources are considered as primary natural resources. In this way, the ancient consumption of solar energy during the formation of fossil resources is overlooked. Dukes (2003) estimated the amount of photosynthetically stored carbon that was required to form coal, oil and gas. Based on these estimations, Dukes (2003) was able to calculate the amount of solar energy that was required to form these fossil fuels (assuming that the PAR radiation is converted into plant matter with an average photosynthetic efficiency of 1.7% in natural systems, that plant matter is 45% carbon and that the energy content of plant matter is 20 MJ per kg). Using these (average) data and taking into account 4.8% (TMCA) or 2.3% (OMCA) of the total solar radiation, we calculated characterization factors (CFs) for hard coal, brown coal, peat, oil and gas of 2.8 GJ_{ex} (TMCA) or 1.3 GJ_{ex} (OMCA) per kg, 1.6 GJ_{ex} (TMCA) or 0.8 GJ_{ex} (OMCA) per kg, 0.9 GJ_{ex} (TMCA) or 0.4 GJ_{ex} (OMCA) per kg, 2273.2 GJ_{ex} (TMCA) or 1069.8 GJ_{ex} (OMCA) per kg and 1865.4 GJ_{ex} or 877.8 GJ_{ex} (OMCA) per m^3, respectively (see Supplementary material A1 in Appendix A). In section 2.4, we illustrate the effect of this methodological choice by comparing the resource efficiencies of two bio-based products with their fossil-based counterparts.

2.2.3 Summary of the COREA framework

Summarizing, in this work we developed the COREA framework for the calculation of a cumulative overall resource efficiency indicator or Cumulative Degree of Perfection (CDP), i.e. CDP_{COREA}. CDP_{COREA} is calculated by the ratio between the exergy content of the considered product (Ex_p) and the cumulative exergy consumption of its supply chain that is quantified according to the COREA framework ($CEENE_{COREA}$) (Equation 2.9). For the resource categories water, minerals, metals, nuclear energy and renewable energy, $CEENE_{COREA}$ accounts in the same way as all three existing versions of the CEENE method (CEENE v2007*2%*, CEENE v2013 and CEENE v2014). For fossil resources, $CEENE_{COREA}$ takes into account the ancient solar energy consumption by fossil fuels (Ex_{f_ASEC}). Bio-productive land resources are included in $CEENE_{COREA}$ by accounting for 4.8% (TMCA) or 2.3% (OMCA) of the total surface solar radiation, depending on the purpose of the efficiency analysis (see section 2.2.1.2) (Equation 2.10).

$$CDP_{COREA} = \frac{Ex_p\ (J_{ex})}{CEENE_{COREA}(J_{ex})} \tag{2.9}$$

$$\begin{aligned} where\ CEENE_{COREA}\ (J_{ex}) \\ &= Ex_{bpl\_SR\_4.8\%}\ (TMCA)\ or\ Ex_{bpl\_SR\_2.3\%}\ (OMCA)(J_{ex}) \\ &+ Ex_{f\_ASEC}\ (J_{ex}) + Ex_{ne}\ (J_{ex}) + Ex_{re}\ (J_{ex}) + Ex_{w}\ (J_{ex}) \\ &+ Ex_{mi}\ (J_{ex}) + Ex_{me}\ (J_{ex}) \end{aligned} \tag{2.10}$$

with

Ex_p: exergy content of the product (J_{ex})

$Ex_{bpl\_SR\_4.8\%}$: 4.8% of the solar radiation exergy on occupied bio-productive land (J_{ex})

$Ex_{bpl\_SR\_2.3\%}$: 2.3% of the solar radiation exergy on occupied bio-productive land (J_{ex})

Ex_{f_ASEC}: exergy of ancient solar energy consumption by fossil resources (J_{ex})

Ex_{ne}: exergy content of nuclear energy resources (J_{ex})

Ex_{re}: exergy content of renewable energy resources (J_{ex})

Ex_w: exergy content of water resources (J_{ex})

Ex_{mi}: exergy content of mineral resources (J_{ex})

Ex_{me}: exergy content of metal resources (J_{ex})

2.3 Materials and methods

2.3.1 Case studies

2.3.1.1 Case study 1

The first case study deals with bioenergy, i.e. electricity produced by an anaerobic digester. Life cycle inventory (LCI) data were retrieved from De Meester et al. (2012). The digestion plant, with a capacity of about 20000 tonnes of biomass inputs per year, was located in Germany. At the moment of the data collection, the digester was mainly fed by maize silage, supplemented with smaller amounts of rye silage and poultry manure. While De Meester et al. (2012) collected the inventory data of silage maize production in Germany, they retrieved the inventory data of rye from the *ecoinvent v2.2* database (*'rye IP, at farm (CH)'*). As this *ecoinvent* process deals with rye cultivation for the purpose of grains, we modified these data in this work in order to better reflect the production of rye silage (see Supplementary material A2 in Appendix A). LCI data of maize and rye silage production can be found in the Supplementary material A2 in Appendix A.

The overall functional unit in this case study was 1 kWh of electricity produced. Electricity produced by a natural gas power plant in Germany was selected as fossil-based alternative. LCI data were retrieved from the *ecoinvent v2.2* database (*'electricity, natural gas, at power plant (DE)'*) (Swiss Centre for Life Cycle Inventories, 2010).

2.3.1.2 Case study 2

The second case study comprises a bio-based material, i.e. bio-ethanol-based polyvinyl chloride (PVC) produced from sugarcane, which was cultivated in the region of Sao Paulo in Brazil in 2010. LCI data were retrieved from Alvarenga et al. (2013a). The bio-based PVC production consists of 5 major stages, i.e. sugarcane production, bio-ethanol production, bio-ethylene production, vinyl chloride monomer (VCM) production and PVC resin production. Brazilian sugarcane production usually consists of a 6-year cultivation cycle with five harvests and a gradual decrease of the productivity over the years (Macedo et al., 2008). LCI data of sugarcane production can be found in the Supplementary material A2 in Appendix A.

The overall functional unit in this case study was 1 kg of bio-ethanol-based PVC resin at factory gate. We compared the bio-based PVC in terms of CDP with fossil-based PVC, of which LCI data were also retrieved from Alvarenga et al. (2013a).

2.3.2 Land occupation characterization factors (LOCFs)

Table 2.4 gives an overview of the year average values of the land occupation characterization factors (LOCFs) calculated with the applied approaches in this work.

For CED and CExD no values are presented because these methods do not account for land occupation (see section 2.2.1.1). In case of SED, only one site-generic LOCF is available. For the different CEENE-based approaches, LOCFs were calculated for the geographic areas considered in the case studies: Germany (case study 1), region of Sao Paulo in Brazil (case study 2) and Western-Europe (case studies 1 and 2). In case study 1, we applied LOCFs of Germany for the land occupied by silage maize production and rye production. In case study 2, we applied LOCFs of the region of Sao Paulo in Brazil for the land occupied by sugarcane production. For all other bio-productive land occupied in the supply chain of the bio-based products, we used average LOCFs for Western-Europe. Also for the bio-productive land occupied in the supply chain of the fossil-based alternatives, we used average LOCFs for Western-Europe.

Table 2.4 Overview of the year average values of the land occupation characterization factors calculated with different approaches.

	resource accounting method	**land occupation characterization factor**	**unit**
Available thermodynamics-based RAMs	CED (Frischknecht et al., 2007; VDI, 1997)	n/a	MJ_{en}/m^2*year
	CExD (Bösch et al., 2007)	n/a	MJ_{ex}/m^2*year
	SED (Rugani et al., 2011)	$6.17 * 10^4$	MJ_{se}/m^2*year
	CEENE v2007$_{2\%}$ (Dewulf et al., 2007a)	70.2[a]; 122.4[b]; 68.1[c]	MJ_{ex}/m^2*year
	CEENE v2013 (Alvarenga et al., 2013c)	26.5[a]; 42.2[b]; 23.2[c]	MJ_{ex}/m^2*year
Different solar system boundary levels integrated in the CEENE v2007 method (see Table 2.3)	CEENE v2007$_{TOT}$	3334[a]; 5815[b]; 3237[c]	MJ_{ex}/m^2*year
	CEENE v2007$_{PAR}$	1624[a]; 2832[b]; 1576[c]	MJ_{ex}/m^2*year
	CEENE v2007$_{TMC}$	207[a]; 349[b]; 194[c]	MJ_{ex}/m^2*year
	CEENE v2007$_{TMCA}$	176[a]; 297[b]; 165[c]	MJ_{ex}/m^2*year
	CEENE v2007$_{OMCA}$	83[a]; 140[b]; 78[c]	MJ_{ex}/m^2*year

[a] Germany; [b] region of Sao Paulo in Brazil; [c] average for Western-Europe

In case of CEENE v2013, site-specific LOCFs were retrieved from the supplementary material of Alvarenga et al. (2013c). For CEENE v2007$_{2\%}$, site-specific LOCFs were obtained by multiplying year average surface solar irradiance values with 2%. A year average value for Western-Europe equal to 3407 MJ_{ex}/m^2*year was retrieved from Dewulf et al. (2007a). For Germany, a year average value of 3894 MJ/m^2*year was obtained from the World Radiation Data Centre (WRDC) database, using data from the Lindenberg station in 2010 (WRDC, 2010). Multiplying with an exergy-to-energy ratio of 0.9327 (Dewulf et al., 2008), a value of 3510 MJ_{ex}/m^2*year was calculated. For the region of Sao Paulo in Brazil, a year average value of 6121 MJ_{ex}/m^2*year was retrieved from Alvarenga et al. (2013a). In case of the other solar system boundary levels integrated in the CEENE v2007 method (CEENE v2007$_{TOT}$, CEENE v2007$_{PAR}$, CEENE v2007$_{TMC}$, CEENE v2007$_{TMCA}$ and CEENE v2007$_{OMCA}$,), site-specific LOCFs were obtained by multiplying these year average surface solar irradiance values with the total efficiency values (ε_{total}) presented in Table 2.3.

2.3.3 Temporal system boundary

The involved crops in the case studies belong to different crop types: silage maize is a spring-sown crop (case study 1), silage rye is an autumn-sown crop (case study 1) and sugarcane is a perennial grass (case study 2). For the crops in case study 1, the CDP calculations will be performed considering different temporal system boundaries (actual cultivation period vs. an entire year). The actual cultivation period of silage maize is a period of 134 days (from May 15 until September 25) (Nemecek and Kägi, 2007). By including green manure cultivation (from September 26 until May 14), we will account for an entire year of land occupation (i.e. 365 days). To account for the additional inputs required for green manure cultivation, we used inventory data for green manure from the *ecoinvent v2.2* database.

In the case of rye silage production (40.3% dry matter (DM)), the actual cultivation period was estimated at 264 days (from September 25 until June 15). This period is shorter than the cultivation period when rye is grown for grains (84% DM; from September 25 until August 5, i.e. 314 days) (Nemecek and Kägi, 2007). We therefore modified the period of land occupation in the *ecoinvent v2.2* process of rye from 314 days to 264 days (Supplementary material A2 in Appendix A). Cultivation of green manure during winter is not necessary in the case of an autumn-sown crop such as rye (see section 2.2.1.3). In addition to the actual cultivation period of rye, we also calculated the CDP considering an entire year of land occupation. In this situation, rye is not immediately followed by another crop and thus the fallow period (from June 16 until September 24) is also assigned to rye cultivation. When considering only the actual cultivation period of rye, we assume that rye is followed by another crop that will be harvested.

The LOCFs in Table 2.4 are year average values. However, in case of silage maize and rye, of which the actual cultivation period is shorter than one entire year, we accounted for the seasonal variation of the surface solar radiation using monthly radiation data of WRDC (2010). For the Lindenberg station (Germany) in 2010, we obtained the following surface solar radiation profile: 2% (January) - 4% (February) - 7% (March) - 13% (April) - 10% (May) - 18% (June) - 18% (July) - 12% (August) - 8% (September) - 6% (October) - 2% (November) - 2% (December). The effect of whether or not accounting for the seasonal variation of the surface solar radiation depends on the type of crop cultivation. For example in the case of silage maize, without accounting for the seasonal variation, we took into account 37% (i.e. 134 of 365 days) of the annual solar radiation on the area of land occupied by silage maize based on its actual cultivation period. Instead, including the seasonal variation of the surface solar radiation, we accounted for 59% of the annual surface solar radiation. Second, for the case of rye, whether or not accounting for the seasonal variation of the surface solar radiation has an opposite effect compared to the case of silage maize. Without accounting for the seasonal variation of the surface solar radiation, we took into account 72% (i.e. 264 of 365 days) of the annual surface solar radiation on the area of land occupied by rye based on its actual cultivation period. This fraction dropped to 55% when accounting for the seasonal variation of the surface solar radiation.

2.3.4 Calculation of the Cumulative Degree of Perfection (CDP)

In order to calculate the CDPs of the crops and the final products, the exergy or energy value of the defined functional unit is required in addition to the cumulative exergy or energy consumption (see Equation 2.1 and 2.2).

The specific exergy value and specific gross calorific value (GCV) of sugarcane (32.5% DM) were retrieved from the *ecoinvent v2.2* database (‘*sugarcane, at farm (BR)*’) and amount to 5.20 MJ_{ex}/kg and 4.95 MJ_{en}/kg, respectively. The specific exergy value and specific GCV of maize silage (35.9% DM) amount to 6.72 MJ_{ex}/kg and 6.47 MJ_{en}/kg, respectively, and were calculated based on the macronutrient composition. The chemical exergy of macronutrient molecules was calculated using the group contribution method (Szargut et al., 1988). The group contribution method can be used if the molecular formula of the substance is known. The chemical exergy can then be calculated by the sum of the chemical exergy values of the functional groups, which can be retrieved from Szargut et al. (1988). The GCV value was calculated based on the formula of Van Es (1975). The exergy value and GCV of rye silage (40.3% DM) amounted to 7.36 MJ_{ex}/kg and 7.23 MJ_{en}/kg and were calculated in a similar way as for maize.

The calculation of the exergy value of 1 kWh electricity (case study 1) is very straightforward and equals the energy content, i.e. 3.6 MJ_{en}. For electricity the exergy-to-energy ratio thus amounts to 1 (Dewulf et al., 2008). The chemical exergy value of 1 kg of PVC (case study 2) was calculated using the group contribution method and amounted to 19.7 MJ_{ex}. The gross calorific value of 1 kg of PVC was calculated from the elemental composition using the Milne formula (Milne et al., 1990) and amounted to 18.6 MJ_{en}.

2.4 Results and discussion

2.4.1 CDPs of primary biomass production systems

2.4.1.1 The different available thermodynamics-based RAMs (CED, CExD, CEENE v2007$_{2\%}$, CEENE v2013 and SED)

For all studied crops, application of CED, CExD and CEENE v2013 resulted in the three highest CDPs (Table 2.5). For maize silage in case study 1, the highest CDP was obtained by applying CEENE v2013. Also for rye silage in case study 1, application of CEENE v2013 resulted in the highest CDP but only when considering the actual cultivation period of rye silage as temporal system boundary. When considering an entire year of land occupation, the highest CDP for rye silage was obtained using CED. For sugarcane production in case study 2, using CED the highest CDP was calculated.

The CDPs calculated by means of CEENE v2013 exceeded the upper limit on efficiency (i.e. 100%) several times, which was expected (see section 2.2.1.1). In case of maize silage, irrespective of which temporal boundary was applied, the CDPs calculated by means of CEENE v2013 were higher than 100%. This can be explained by its very high yield (17.9 tonnes DM per ha in this case study; see Supplementary material A2 in Appendix A). For rye silage (DM yield of 10.5 tonnes per ha; see Supplementary material A2 in Appendix A), we also calculated a CDP higher than 100% when applying CEENE v2013 but only when considering the actual cultivation period. In case of sugarcane production (DM yield of 22.7 tonnes per ha; see Supplementary material A2 in Appendix A), the CDP calculated by means of CEENE v2013 was relatively high (83.1%), but did not exceed 100%. The reason for this is because the potential natural NPP for the Sao Paulo region in Brazil is quite high (Table 2.4). The results confirm our expectation that CEENE v2013 is generally not adequate for the purpose of calculating an overall resource efficiency or CDP of bio-based production chains.

Table 2.5 Overview of the calculated CDPs (expressed as percentages) of the involved crops in both case studies (silage maize and silage rye in case study 1; sugarcane in case study 2). CDPs that exceed the upper limit on efficiency (i.e. 100%) are <u>underlined</u>.

Case study 1: maize silage (Germany)		actual cultivation period	entire year of land occupation (including green manure)
Available thermodynamics-based RAMs	CED	96.3	95.2
	CExD	96.0	94.9
	SED	<0.1[a]	<0.1[a]
	CEENE v2007$_{2\%}$	73.1	43.8
	CEENE v2013	<u>191.4</u>	<u>115.9</u>
Different solar system boundary levels integrated in the CEENE v2007 method (see Table 2.3)	CEENE v2007$_{TOT}$	1.6	0.9
	CEENE v2007$_{PAR}$	3.2	1.9
	CEENE v2007$_{TMC}$	26.1	15.6
	CEENE v2007$_{TMCA}$	30.7	18.3
	CEENE v2007$_{OMCA}$	64.0	38.4
Case study 1: rye silage (Germany)		**actual cultivation period**	**entire year of land occupation (including fallow period)**
Available thermodynamics-based RAMs	CED	89.7	
	CExD	86.7	
	SED	<0.1[b]	<0.1[b]
	CEENE v2007$_{2\%}$	43.3	25.3
	CEENE v2013	<u>110.3</u>	66.3
Different solar system boundary levels integrated in the CEENE v2007 method (see Table 2.3)	CEENE v2007$_{TOT}$	1.0	0.5
	CEENE v2007$_{PAR}$	2.0	1.1
	CEENE v2007$_{TMC}$	15.7	9.0
	CEENE v2007$_{TMCA}$	18.4	10.6
	CEENE v2007$_{OMCA}$	38.1	22.2
Case study 2: sugarcane (Brazil)		**entire year of cultivation (in a 6-year cycle)**	
Available thermodynamics-based RAMs	CED	96.3[c]	
	CExD	96.3[c]	
	SED	<0.1	
	CEENE v2007$_{2\%}$	29.3	
	CEENE v2013	83.1	
Different solar system boundary levels integrated in the CEENE v2007 method (see Table 2.3)	CEENE v2007$_{TOT}$	0.6	
	CEENE v2007$_{PAR}$	1.3	
	CEENE v2007$_{TMC}$	10.3	
	CEENE v2007$_{TMCA}$	12.2	
	CEENE v2007$_{OMCA}$	25.6	

[a-b] Values considering an entire year of land occupation are lower but differences are smaller than 0.05%; [c] CDP calculated by means of CExD is lower but difference is smaller than 0.05%.

Using CEENE v2007$_{2\%}$, the CDPs were in a range from 61 to 65% lower than the CDPs calculated by CEENE v2013. This is logic because the CEENE v2007$_{2\%}$ LOCFs were higher than the CEENE v2013 LOCFs for all locations (Table 2.4). The CDPs calculated by means of CEENE v2007*2%* did not exceed 100% in any case studies, which was also expected (see section 2.2.1.4).

Using CED and CExD, high CDPs (>86%) were calculated for all studied crops because these methods do not account for land occupation in a direct way. By taking only the energy or exergy content of the harvested biomass into account, they exclude photosynthesis, the basic process of primary biomass production. As they do not include the whole supply chain of the produced biomass, calculating the efficiency by means of these methods does not really make sense.

By means of SED, very low CDPs (<0.1%) were calculated for all studied crops. The large difference in the calculated CDPs between SED and the other thermodynamics-based RAMs is due to the different conceptual rationale of the SED method compared to the other thermodynamics-based RAMs. In section 2.2.1.1, we already explained why SED is not considered as an appropriate RAM for the purpose of calculating an overall resource efficiency or CDP. To visualize the difference between SED and the other methods, Figure 2.2 shows the relative contributions of the different resource categories to the five available thermodynamics-based resource indicators (CED, CExD, CEENE v2007*2%*, CEENE v2013 and SED) for all studied crops, when considering an entire year of land occupation as temporal system boundary. The alternative figure when considering the actual cultivation period is very similar to Figure 2.2 and can be found in the Supplementary material A3 in Appendix A.

The contribution of land resources to the SED results was small compared to the other approaches (in the case of CED and CExD, land resources are indirectly taken into account in the category biomass) (Figure 2.2). The share of land resources to the total SED amounted to 18, 15 and 24% for maize, rye and sugarcane, respectively, while it was in a range from 91 to 99% in the case of the other thermodynamics-based RAMs. The majority of the cumulative resource consumption in terms of SED was due to non-renewable resources, i.e. mineral (on average 39% for all studied crops), fossil (21%) and metal resources (21%). The fact that these resource categories generally dominate SED results of agricultural products was also reported in the introductory paper of the SED method (Rugani et al., 2011).

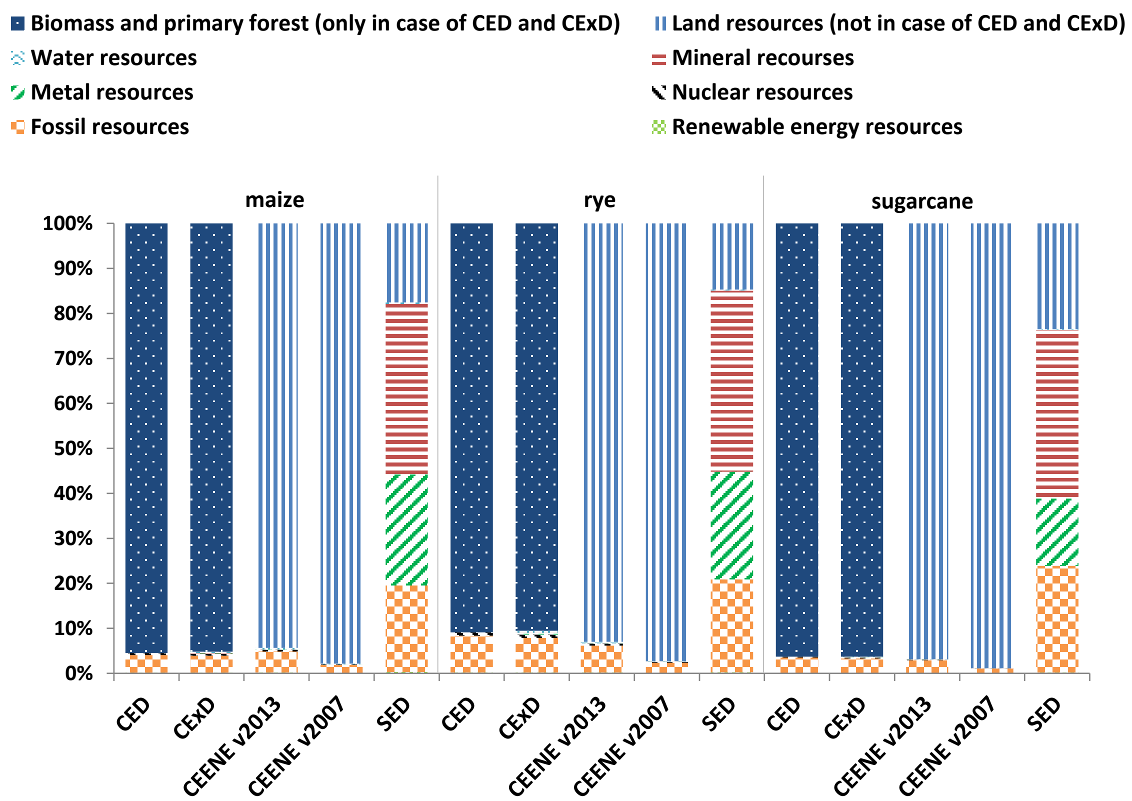


Figure 2.2 Relative contributions of the different resource categories to the five available thermodynamics-based resource indicators (CED, CExD, CEENE v2007$_{2\%}$, CEENE v2013 and SED) for the crops in both case studies, when considering an entire year of land occupation as temporal system boundary. *Renewable energy resources* include hydropower and wind energy in the case of CEENE v2007$_{2\%}$ and CEENE v2013. In the case of SED, *Renewable energy resources* include hydropower, wind energy and geothermal energy. In the case of CExD and CED, *Renewable energy resources* include hydropower, wind energy and solar energy.

2.4.1.2 The boundary of the input of solar-based energy into the primary biomass production system (CEENE v2007$_{TOT}$, CEENE v2007$_{PAR}$, CEENE v2007$_{TMC}$, CEENE v2007$_{TMCA}$ and CEENE v2007$_{OMCA}$)

The effect on the calculated CDP of the system boundary of the input of solar-based energy into the primary biomass production system can be seen in Table 2.5. Obviously, the higher the portion of solar radiation that was taken into account, the lower the CDP. CDPs calculated by means of CEENE v2007$_{OMCA}$ are more or less twice as high as the CDPs calculated using CEENE v2007$_{TMCA}$. This highlights indeed that there is much room for improvement without altering the photosynthesis mechanism. Zhu et al (2008), however, noted that the maximum observed conversion efficiencies for C3 and C4 crops are still 3 to 4 times larger than the average conversion efficiencies achieved by major crops in the U.S., mainly because of non-ideal conditions. In order to reduce the gap with the theoretical maximum conversion efficiency ($\varepsilon_{c,max}$), DeLucia et al. (2014) reported that an improvement in water use efficiency, in addition to a low nitrogen requirement, will be necessary to achieve the full potential of primary biomass production.

2.4.1.3 The temporal system boundary of the primary biomass production system

The effect on the calculated CDP of considering different temporal system boundaries can be seen in Table 2.5 for the crops in case study 1, i.e. silage maize and silage rye. In case of rye, there is no effect on the CDP calculated using CED or CExD when considering the entire year of cultivation instead of the actual cultivation period. The reason is again that these methods do not account for land occupation in a direct way. In contrast to rye, there is a small effect on the CDP calculated for silage maize using CED or CExD because the additional inputs required for green manure cultivation during winter were taken into account when considering the entire year of cultivation. Whether we should account for the actual cultivation period or the entire year of land occupation depends on the cropping system design, e.g. monoculture followed by a green manure period or a fallow period vs. a multiple-cropping system. When silage maize cultivation in a monoculture system is either (i) followed by a green manure that is not harvested or (ii) followed by a fallow period, the entire year of land occupation should be allocated to the harvested silage maize. In contrast, when silage maize cultivation is operated in a multiple-cropping system (e.g. maize-rye-grass, maize-grass-maize, etc.), the entire year of land occupation should be allocated among the harvested products while taking into account the seasonal variation of the surface solar radiation. In other words, the more efficient the land surface is used, the higher the resource efficiency of the cropping system. However, when optimising land use efficiency, it is extremely important that the effects on other environmental aspects, such as soil fertility, and nutrient and water availability are simultaneously evaluated.

2.4.2 Share of the primary biomass production system in the resource consumption of the entire supply chain of the bio-based products

Before presenting the CDPs of the final bio-based products (electricity in case study 1; PVC in case study 2), we take a closer look at the share of the foreground primary biomass production stage (silage maize and silage rye in case study 1; sugarcane in case study 2) in the resource consumption of the entire supply chain of the bio-based products. A detailed table that shows the shares for each resource category separately and for the total resource consumption can be seen in the Supplementary material A4 in Appendix A. Also more information on the most remarkable observed similarities and differences between all applied approaches can be found in the Supplementary material A4 in Appendix A. To calculate these results, the ancient solar energy consumption of fossil fuels was not yet taken into account. Results that take into account the ancient solar energy consumption of fossil fuels are discussed in the section 2.4.3.1.

First, in the case study of electricity produced by anaerobic digestion, the foreground primary biomass production stage predominated the cumulative overall resource consumption among all applied approaches, in a range from 72.6 to 99.7%, when considering an entire year of land occupation. The lowest share (72.6%) was seen in the SED results, while the share in the other approaches ranged from 94.1 to 99.7%. This major share is mainly due to the land resources category. Focusing only on land resources (the category biomass in the case of CED and CExD), the contribution of the primary biomass production stage was nearly 100% for all applied approaches. Also in the mineral and metal resource categories this contribution was high, i.e. >82% and >77%, respectively. This can mainly be explained by the consumption of mineral fertilizers and the production of agricultural machinery. For fossil resources, the contribution of the primary biomass production stage amounted to 55 à 56% among all applied approaches (with lower SED results: 43%), while the downstream production stages contributed to the remaining 44 à 45% (for the SED results: 58%). Of this remaining part, the anaerobic digestion stage accounted for the major share, i.e. about 76% among all applied approaches (with higher SED results: 81%). In the primary biomass production stage, the production of machinery and fuel for field work operations consumed about 52% of the fossil resources among all applied approaches (with slightly higher SED results: 54%), followed by green manure cultivation (25% for all applied approaches, with slightly lower SED results (23%)) and the production of mineral fertilizers (17% for all applied approaches). For water, nuclear energy and renewable energy resources, the contribution of the primary biomass production stage was lower than half of the cumulative resource consumption of the entire supply chain, except for the renewable energy resources in case of the SED method (58%). Considering the actual cultivation period of the involved crops in case study 1 instead of an entire year of cultivation, the contribution of the primary biomass production stage always decreases (see Supplementary material A4 in Appendix A).

Second, in the case study of the bio-based PVC, the primary biomass production stage predominated the cumulative resource consumption among all applied approaches except the SED, in a range from 78.0 to 99.5%. The primary biomass production stage contributed only to 15.3% of the cumulative overall resource consumption in case of the SED results. Instead, the vinyl chloride monomer (VCM) production and the bio-ethanol production contributed to 60.7 and 16.9%, respectively. The large share of the VCM production stage was mainly due to mineral resource use (chlorine consumption) and the high impact factors assigned to mineral resources in the SED method. Focusing only on land resources (the category biomass in the case of CED and CExD), the contribution of the primary biomass production stage was very high (>99.5%) for all applied approaches. For fossil resources, the contribution of the primary biomass production stage amounted to about 17% for all applied approaches (with slightly higher SED results: 20%), while the downstream production stages contributed to the remaining 83% (with slightly lower SED results: 80%). The VCM production stage accounted for the major part thereof (72% for all applied approaches), followed by the bio-ethylene production (more or less 17% for all applied approaches), PVC resin production (9% for all applied approaches) and the bio-ethanol production (4% in the case of SED; 2 à 3% in the case of the other approaches). Compared to the first case study, we can see that the share of the primary biomass production stage in the second case study was much lower in all resource categories except the land resources category (the category biomass in the case of CED and CExD).

2.4.3 CDPs of the bio-based products compared to their fossil-derived counterparts

The CDPs of the final bio-based products and their fossil-based counterparts are presented in Table 2.6 (to calculate these results, the ancient solar energy consumption of fossil fuels was not yet taken into account).

Table 2.6 Overview of the calculated CDPs (%) of the final bio-based products and their fossil-based counterparts in both case studies. In the column of the bio-based electricity, the first values were calculated considering an entire year of land occupation as temporal system boundary for silage maize and silage rye, while the second values between parentheses were calculated considering the actual cultivation period of both crops. CDPs of bio-based products that exceed the corresponding CDPs of their fossil-based counterparts are underlined.

		Case study 1		Case study 2	
		bio-based electricity	**fossil-based electricity**	**bio-based PVC**	**fossil-based PVC**
Available thermodynamics-based RAMs	**CED**	25.4 (25.6)	35.3	22.3	38.2
	CExD	24.2 (24.5)	37.1	21.8	38.1
	SED	<0.1[a]	<0.1[a]	<0.1[a]	<0.1[a]
	CEENE v2007$_{2\%}$	10.7 (17.7)	34.9	7.8	36.6[e]
	CEENE v2013	27.3 (43.5)	35.0[c]	19.1	36.9
Different solar system boundary levels integrated in the CEENE v2007 method (see Table 2.3)	**CEENE v2007$_{TOT}$**	0.2[d] (0.4)	33.6	0.2[d]	25.4
	CEENE v2007$_{PAR}$	0.5 (0.8)	34.3	0.4	30.3
	CEENE v2007$_{TMC}$	3.9 (6.5)	34.9[b]	2.9	36.0
	CEENE v2007$_{TMCA}$	4.5 (7.6)	34.9[b]	3.4	36.1
	CEENE v2007$_{OMCA}$	9.4 (15.6)	35.0[c]	6.9	36.6[e]

[a-e] Values are different but differences are smaller than 0.5%.

The differences between the CDPs calculated by means of different approaches were larger for the bio-based products compared to those observed for the fossil-based products. The choice whether land occupation is directly taken into account and, if so, how and to which extent it is taken into account, has a larger influence on the CDP of the bio-based products. Consequently, good knowledge about how the CDP has been calculated, is therefore particularly important for interpretation of resource efficiency results when bio-based products are involved. Even though we concluded in section 2.2.1.2 that CEENE v2007$_{TMCA}$ and CEENE v2007$_{OMCA}$ are useful and scientifically sound for the purpose of calculating an overall resource efficiency of bio-based products, we present in Table 2.6 the CDPs calculated by means of all approaches in order to show the effect of these different approaches on the calculated CDP of the final products.

Almost all approaches ranked the fossil-based products in favour of their bio-based alternatives. Exceptions are the SED method, irrespective of which temporal system boundary was applied in case study 1, and the CEENE v2013 method, when only the actual cultivation period of the involved crops was considered as temporal boundary. In the latter case, the bio-based product was 1.2 times more efficient than its fossil-based alternative. In case of the SED method, the bio-based product was 2.1 and 2.4 times more efficient when considering an entire year of cultivation as temporal boundary and when considering only the actual cultivation period, respectively. As aforementioned, SED and CEENE v2013 are, like CED and CExD, considered as not adequate for the purpose of calculating an overall resource efficiency, we can conclude for the case studies in this work that the fossil-based products are ranked in favour of their bio-based counterparts in terms of their overall resource efficiency. Using CEENE $v2007_{TMCA}$, the bio-based product in case study 1 was between 7.7 and 4.6 times less resource efficient than its fossil-based alternative, depending on the considered temporal boundary. These values dropped to 3.7 and 2.2 times less resource efficient when using CEENE $v2007_{OMCA}$. The bio-based product in case study 2 was 10.7 and 5.3 times less resource efficient than its fossil-derived counterpart, when using CEENE $v2007_{TMCA}$ and CEENE $v2007_{OMCA}$, respectively.

2.4.3.1 Addressing the non-renewable character of fossil resources

After implementing the fossil resources characterization factors (CFs) that take into account their ancient solar energy consumption (see section 2.2.2), we have calculated the $CDP_{COREA(TMCA)}$ and $CDP_{COREA(OMCA)}$ of the final products using Equations 9 and 10. Due to the high CFs for fossil fuels in this approach, the fossil resources category predominated the total resource consumption along the production chain of both bio-based and fossil-based products (see Supplementary material A5 in Appendix A). Their CDP results therefore become very small (<0.1%) (Table 2.7). The effect of this alternative accounting approach for fossil resources on the comparison of the bio-based products and their fossil-based counterparts is large. Using $CEENE_{COREA(TMCA)}$, the fossil-based product in case study 1 was between 18.6 and 15.6 times less resource efficient than the bio-based product, when considering an entire year of cultivation as temporal boundary and when considering only the actual cultivation period, respectively. Similar values were obtained using $CEENE_{COREA(OMCA)}$. The fossil-based product in case study 2 was about 3.5 times less resource efficient than the bio-based counterpart, when using both $CEENE_{COREA(TMCA)}$ and $CEENE_{COREA(OMCA)}$. Accounting for the ancient solar energy consumption of fossil fuels definitely reflects their non-renewability, which is an increasingly important aspect to be taken into account in resource efficiency assessments. While the focus of this research is on resource efficiency, it is important to note that other aspects such as greenhouse gas emissions should be taken into account in order to have an overall view on the environmental sustainability of a product. For example, Font de Mora et al. (2012) compared three types of biodiesel and showed that the biodiesel with the lowest total fossil exergy consumption in its supply chain had the highest emissions of greenhouse gases during its production.

Table 2.7 Overview of the calculated $CDP_{COREA(TMCA)}$ and $CDP_{COREA(OMCA)}$ (%) of the final bio-based products and their fossil-based counterparts in both case studies. In the column of the bio-based electricity, the first values were calculated considering an entire year of land occupation as temporal system boundary for silage maize and silage rye, while the second values between parentheses were calculated considering the actual cultivation period of both crops. CDPs of bio-based products that exceed the corresponding CDPs of their fossil-based counterparts are underlined.

	Case study 1		Case study 2	
	bio-based electricity	**fossil-based electricity**	**bio-based PVC**	**fossil-based PVC**
$\mathbf{CDP_{COREA(TMCA)}}$	<u>1.14e-2</u> (<u>1.36e-2</u>)	7.30e-4	<u>3.10e-3</u>	8.84e-4
$\mathbf{CDP_{COREA(OMCA)}}$	<u>2.41e-2</u> (<u>2.88e-2</u>)	1.55e-3	<u>6.59e-3</u>	1.88e-3

2.5 Conclusions and perspectives

To support a transition towards a sustainable renewables-based economy, it is important to optimize the conversion of natural resources into bio-based products. Optimising bio-productive land use efficiency is one of the key features of sustainable land use, in addition to preserving soil fertility, nutrient and water availability. The challenge to use the limited available bio-productive land in a sustainable way as well as to reduce our reliance on declining stocks of non-renewable fossil resources calls for adequate indicators. The Cumulative Overall Resource Efficiency Assessment (COREA) framework, developed in this work, fills an important gap in scientific literature about how to calculate an overall resource efficiency indicator, while (i) taking into account bio-productive land resources and (ii) addressing the non-renewable character of fossil resources. Of key importance to this indicator is a full coverage of the different types of natural resources and a distance-to-target approach to measure the distance reduction from the potential optimum in biomass yield that can be achieved by human intervention without changing the photosynthetic mechanism. The overall resource efficiency indicator is useful to support sustainability assessment of bio-based products, both at the full chain level and at the level of the primary biomass production stage. A higher degree of spatial differentiation in life cycle inventory data on land use and taking into account environmental constraints for an optimal primary production (e.g. temperature, precipitation, steep slopes in mountain regions, soil type) could further improve its practical applicability.

Figure 2.3 presents an overview of the specific objectives addressed in Chapter 2.

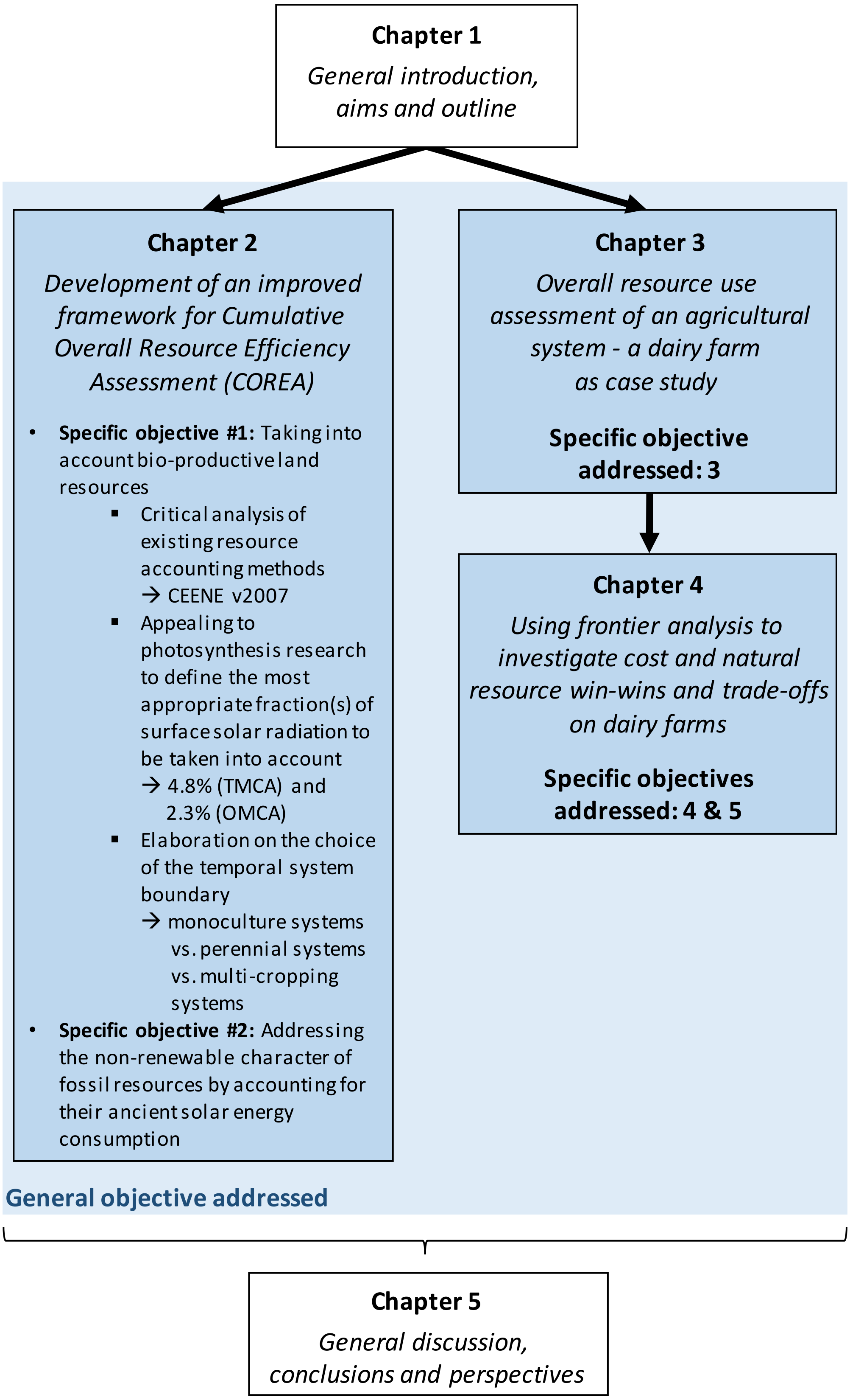


Figure 2.3 Overview of the specific objectives addressed in Chapter 2.

CHAPTER 3

RESOURCE USE ASSESSMENT OF AN AGRICULTURAL SYSTEM FROM A LIFE CYCLE PERSPECTIVE – A DAIRY FARM AS CASE STUDY

Redrafted from:

Huysveld, S., Van linden, V., De Meester, S., Peiren, N., Muylle, H., Lauwers, L. and Dewulf, J. (2015). Resource use assessment of an agricultural system from a life cycle perspective - a dairy farm as case study. *Agricultural Systems* 135, 77-89.

CHAPTER 3: RESOURCE USE ASSESSMENT OF AN AGRICULTURAL SYSTEM FROM A LIFE CYCLE PERSPECTIVE - A DAIRY FARM AS CASE STUDY

Abstract

Despite the great pressure on global natural resources, few LCA studies focus on overall resource consumption and the efficiency of the use of those resources. Moreover, an overall resource use assessment for agricultural systems is highly relevant because many of these systems have become high input/high output systems in order to achieve higher productivity. In this study, we propose a framework to evaluate overall resource consumption of agricultural systems at the process level using Exergy Analysis (EA) and at the life cycle level using Exergetic Life Cycle Assessment (ELCA). We evaluate the applicability and usefulness of this approach based on a case study of an intensive confinement-based dairy farm in the region of Flanders, Belgium. The EA showed that more than half of the resources consumed by the dairy farm's herd was irreversibly lost, as a consequence of the second law of thermodynamics. The remaining went for almost two-thirds to manure (54%) and methane emissions (9%), while only one-third flowed to end-products, i.e. milk (32%) and the animals awaiting slaughter (2%). The ELCA identified the feed supply as by far the most demanding part of the dairy production chain, representing 93% of the resource footprint. Overall, concentrates were on average 2.5 times more resource-intensive per kg dry matter than roughages, while wet by-products were 34 and 73% less resource-intensive than roughages and concentrates, respectively. Mainly land (77%) and fossil resources (17%) were required throughout the life cycle. About 36% (in terms of m^2*year) of the occupied land was located off-farm. Slightly less than one-quarter of the fossil resources were used on-farm as fuel and electricity. The on-farm use of groundwater accounted for about half of the total *blue* water use across the life cycle. With this work, we show the usefulness of the proposed

framework to evaluate overall resource consumption of dairy farms and to identify on-farm and off-farm improvement opportunities. This framework has the potential to support research on whole-farm improvement strategies such as pasture-based systems and low-input farming, and to compare populations of contrasting milk production systems.

3.1 Introduction

The global stocks of natural resources, all of which support our human activities, are under pressure. Natural resources include water, minerals, metals, land, fossil resources, etc. We are consuming natural resources at an unsustainable rate that exceeds the carrying capacity of the Earth (Global Footprint Network, 2012). Since the 1980s, the global annual extraction of resources has increased by almost 50% (from 40 billion tonnes to 58 billion tonnes) and it is expected to rise further to 100 billion tonnes by 2030 (SERI, GLOBAL 2000 and Friends of the Earth Europe, 2009). Due to the increasing standard of living in developing countries, the global resource extraction is even expected to rise about 25% faster than the growth of the worldwide population, which is projected to increase from around 6 billion today to 8.3 billion in 2030 (FAO, 2002). The European Commission's publication entitled *A resource-efficient Europe - Flagship initiative under the Europe 2020 strategy* (European Commission, 2011a) also supports the notion that the sustainable development of our society should rely on increased efficiency of resource use. Striving for higher resource use efficiency is especially relevant for Europe, because it is the continent with the largest net-import of natural resources (SERI, GLOBAL 2000 and Friends of the Earth Europe, 2009).

Agriculture should also face the challenge of increasing its resource use efficiency. The Food and Agriculture Organisation (FAO), in its 2011 book *Save and Grow*, states that 'to feed a growing world population, we have no option but to intensify crop production. But farmers face unprecedented constraints. In order to grow, agriculture must learn to save.' During past decades, the increase in agricultural productivity, the so-called *Green Revolution*, has mainly been achieved by an increased material and energy input (fertilisers, pesticides, irrigation, machinery powered by fossil fuels, etc.) and has been accompanied by environmental burdens (greenhouse gas emissions, eutrophication, acidification, etc.). Along with the rising environmental concerns, especially about livestock farming (FAO, 2006; Gerber et al., 2013), livestock systems have increasingly been studied using Life Cycle Assessment (LCA). LCA is a commonly accepted method to evaluate the environmental sustainability of a product throughout its entire life cycle (Guinée et al., 2002). Animal-derived food products, especially red meat and dairy products, tend to have higher environmental impacts than plant-based foods (Heller et al., 2013; Meier and Christen, 2013; Vanham et al., 2013). Many LCA studies have been performed on livestock products such as beef, chicken, eggs, milk and pork (de Vries and De Boer, 2010). Frequently studied environmental aspects can be classified into two types of impact categories: (1) emissions, e.g. global warming, eutrophication and acidification, and (2) resource use, e.g. land use and primary energy use. Primary energy use includes both non-renewable energy resources, such as fossil and nuclear energy, and renewable energy resources, such as solar energy, wind energy, hydropower, etc. Although in the past emissions-related impacts were more frequently evaluated in LCA studies than resource use aspects, many recent LCA studies on livestock products have quantified both primary energy use (MJ_{en}) and land use (m^2) (e.g. da Silva et al. (2014), O'Brien et al. (2012)). Also recently, water consumption has gained more attention, especially in studies on milk production (e.g. de Boer et al. (2013), Sultana et al. (2014)). Some of the studies that investigated energy use also focused on the efficiency with which these energy resources were used (Meul et al., 2007; Vigne et al., 2013). However, a more extended resource assessment can be achieved when land occupation and non-energetic resources, i.e. water, metals and minerals, are addressed in addition to energy carrying resources (Dewulf et al., 2007a). An assessment of the full range of resources is needed to avoid environmental problem-shifting in resource consumption. The study of De Meester et al. (2011) is a good illustration of how important it is to analyse overall resource use. Their study revealed that the production of fuel bioethanol in a biorefinery to replace petrol can save 27% of fossil resources, but this comes at the cost of 93% extra land, water and minerals. An integrated assessment of overall resource consumption is observed as a gap in existing LCA research of livestock systems.

Such an integrated assessment of resource consumption considers energy resources and non-energetic resources at the same time. In order to calculate overall resource consumption and efficiency, one needs a single quantifier for both material and energy flows. The exergy concept, which originates from the second law of thermodynamics, is stated to be an appropriate quantifier for both the amount and quality of material and energy flows in one common unit, i.e. joules of exergy (J_{ex}) (European Commission, 2011c; Liao et al., 2012; Science Europe, 2015) (see also section 1.2.3 in Chapter 1). In this work, we introduce a generic framework that uses the exergy concept to evaluate the overall resource consumption of agricultural systems. To build this framework, we have chosen specialised dairy farms in Flanders (the northern region of Belgium) as a starting base; then we have drawn a generic process flow diagram. The main reason for choosing dairy farms is that these farms include both plant and animal production, which interact by feed production and manure utilisation. The process flow diagram can therefore be used as a blueprint for other agricultural systems with only minor modifications or deletions (e.g. on-farm feed production is usually not present at pig farms). In the light of the trend towards more intensively managed and more specialised dairy farms during the past decade in Europe (CEAS Consultants, 2000), and more specifically in Flanders (Van der Straeten et al., 2012), we chose to evaluate this framework in a case study of one specific intensive confinement-based dairy farm in Flanders.

The generic framework is characterised by a thorough input/output analysis of the dairy farming system, meaning that the system was not considered as a *black box*. Dairy farms are rather complex systems that are composed of several subsystems with interactions among them. For that reason, we considered internal flows of the dairy farm (e.g. on-farm produced roughages and manure) in order to thoroughly understand the system. The resource efficiency of the cattle herd was calculated after quantifying all its input and output flows in exergy terms. This approach, called an Exergy Analysis (EA) (Szargut et al., 1988), indicates how efficiently inputs are converted into products. An EA therefore allows the identification of improvement opportunities from a resource point of view. The boundaries of such an EA can be enlarged to include the supply chains of the dairy farm. Application of the exergy concept to LCA results into Exergetic Life Cycle Assessment (ELCA) (De Meester et al., 2009). In our study, an overall natural resource consumption footprint of the dairy farm was quantified using the exergy-based life cycle resource accounting method, named *Cumulative Exergy Extraction from the Natural Environment* (CEENE), developed by Dewulf et al. (2007a). This method adds up a comprehensive range of natural resources in exergy terms. The usefulness of considering internal flows instead of a *black box* analysis was also demonstrated by Vigne et al. (2013), who introduced a generic energy use assessment framework for comparing contrasting dairy systems in different regions around the world. Whereas Vigne et al. (2013) focused on fossil energy, solar energy, energy contained in biomass, and energy from human and animal labour, we focused in this study on overall natural resource use, including use of energy carriers (fossil resources, nuclear energy and abiotic renewable energy), non-energetic resources (water, minerals and metals) and land.

3.2 Materials and methods

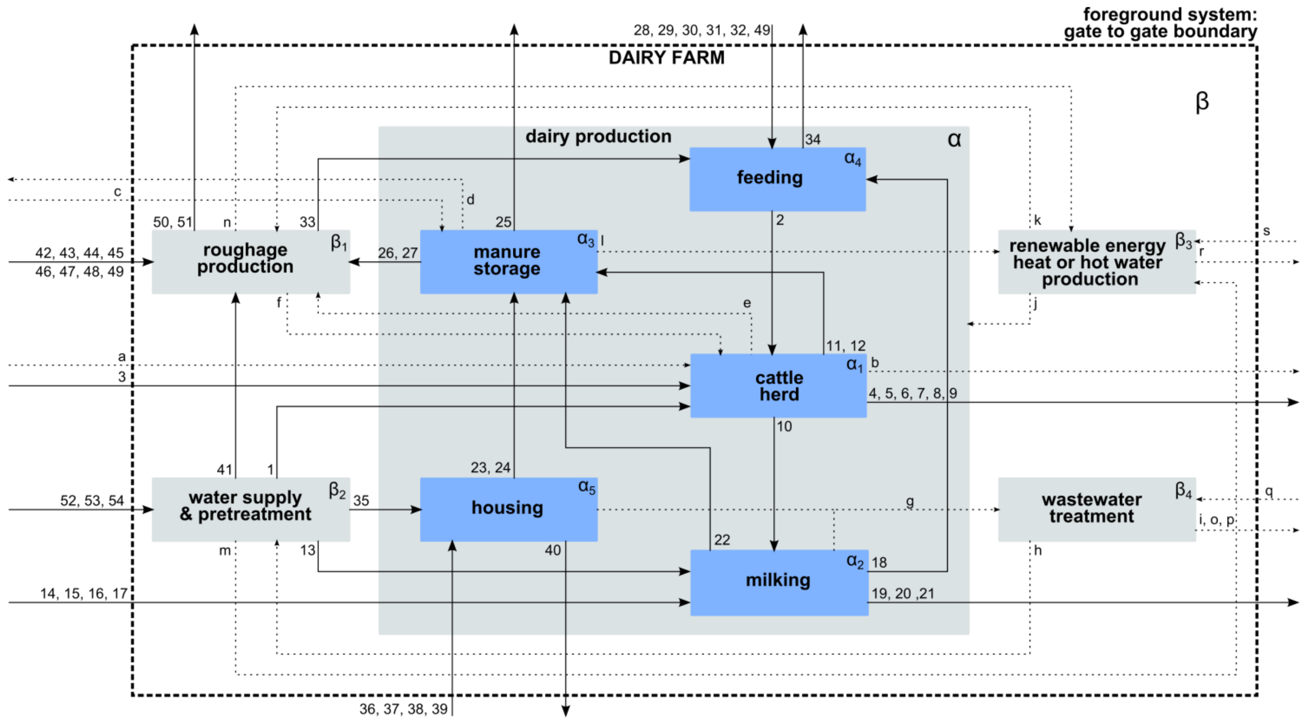
3.2.1 Scope definition

We have performed a case study on a confinement-based specialised dairy farm in Flanders. The boundary of the study involves the life cycle from cradle to farm gate; the functional unit was defined as 1 kg fat-and-protein-corrected milk (FPCM) (4% fat and 3.3% protein content (IDF, 2010)). The foreground system was defined as the entire dairy farm, i.e. the production unit within the gate-to-gate boundary (Figure 3.1), including on-farm feed (roughage) production and manure utilisation. The background system was defined as the part of the production chain outside the dairy farm boundary, including all human-industrial processes (agricultural, industrial and transport) necessary to produce and deliver the inputs to the dairy farm. Regarding the handling of co-products, more information can be found in section 3.2.4, ‘Allocation procedure’.

3.2.2 The foreground system

3.2.2.1 Description of the foreground system

Starting with a detailed analysis of specialised dairy farms in Flanders, we drew a generic process flow diagram (Figure 3.1). Based on the nomenclature for system boundaries used by Dewulf et al. (2007b), the foreground system (β) was divided into a core subsystem (α) and subsystems (β_i) that support the core activity. In doing so, the foreground system was divided into five subsystems: the α-core subsystem *dairy production* and the β_i-supporting subsystems *roughage production* (β_1), *water supply and pretreatment* (β_2), *renewable energy/hot water/heat production* (solar panels, solar boilers and anaerobic digesters) (β_3) and *wastewater treatment* (β_4). The α-core subsystem *dairy production* was divided into five processes: *cattle herd* (α_1), *milking* (α_2), *manure storage* (α_3), *feeding* (α_4) and *housing* (α_5). In this work, this generic framework was applied to one case in detail. For this case, all identified flows for which data were collected, are presented as solid lines and designated by a number (1-54) in Figure 3.1. The flows not present at the dairy farm under study are presented as dashed lines and designated by a letter (a-s). The β_3- and β_4-subsystems were not present at the dairy farm under study.



1 drinking water
2 feed preparation
3 medicines
4 methane emissions (rumen fermentation)
5 dead animals
6 culled cows
7 surplus calves
8 latent heat production
9 sensible heat production
10 milk
11 liquid manure
12 solid manure (excl. bedding material)
13 rinse water milk installation and tank
14 electricity
15 desinfecting teat dip
16 cleaning agent milk installation and tank
17 milking parlour
18 home-produced milk for calves
19 milk sold to dairy plant
20 milk discarded to sewerage
21 wastewater from rinsing milk installation and tank to sewerage
22 wastewater from cleaning the milking parlour
23 waste bedding material
24 wastewater from cleaning stables
25 emissions from manure storage
26 liquid manure
27 solid manure (incl. waste bedding material)
28 fuel
29 purchased roughage
30 purchased concentrates
31 purchased wet by-products
32 purchased other feedstuff (milk powder, vitamins, etc.)
33 home-grown roughage
34 emissions from machinery operation
35 water for cleaning stables and machinery
36 land occupation of stables, sheds, trench silos and paved yard
37 cleaning agent
38 electricity
39 purchased bedding material
40 wastewater from cleaning agricultural machinery to surface water
41 water for spraying pesticides
42 occupation of land
43 water (rainwater in this case)
44 fuel used by farmer
45 fuel used by contract workers
46 seed
47 pesticides
48 mineral fertilisers
49 on-farm mobile machinery
50 sale of crop products (grass seeds in this case)
51 emissions (from burning fuel, from fertilser application, etc.)
52 water (groundwater pumped in this case)
53 electricity
54 chemical product water pretreatment
a purchase of cattle
b sale of (living) cattle
c purchase of manure
d sale of manure
e manure to pastures (in case of grazing)
f grazed grass (in case of grazing)
g wastewater to treatment
h treated water for reuse
i treated water that can be discharged
j produced electricty/heat/hot water
k digestate (use as fertiliser)
l manure to digester
m water to solar boiler
n crop products to digester
o sludge from wastewater treatment
p emissions (to be specified)
q resource use (to be specified)
r emissions (to be specified)
s resource use (to be specified)

Figure 3.1 Generic process flow diagram of specialised dairy farms in Flanders. For the chosen case, all identified flows for which data were collected, are presented as solid lines and designated by a number (1-54). The flows that are not present at the dairy farm under study, are presented as dashed lines and designated by a letter (a-s).

The studied farm was a confinement-based specialised dairy farm where the Holstein Friesian cattle were kept indoor throughout the year. Milking took place in a tandem milking parlour three times a day. The farmer produced two types of total mixed feed rations (TMR). To obtain a TMR, the farmer weighs and mixes different feed ingredients to achieve a feed mixture that meets the nutritional requirements of the animal group for which the feed ration is intended. Major feed ingredients can be divided in roughages and concentrates (based on their composition) and by-products of industries. Roughages are feeds with a high fibre content (e.g. grass and maize silage) and are mainly produced on-farm. Concentrates are feeds characterised by a higher dry matter content and a higher digestibility; they are usually purchased (FAO, 1993). A distinction can be made between energy concentrates (e.g. cereals) and protein concentrates (e.g. soybean meal). Concentrates can consist of one ingredient (e.g. soybean meal) or several ingredients that are mixed to obtain a balanced compound feed, for example in terms of protein content (FAO, 2014). In addition to roughages and concentrates, by-products of the food industry (e.g. pressed sugar beet pulp) and the bio-ethanol industry are also very often used in feeds. At the farm under study, the first type of TMR (TMR1) was produced for the young cattle older than 6 months and included mainly roughages. The second type of TMR (TMR2) was produced for the lactating dairy cows and included roughages, concentrates and wet by-products. The cattle younger than 6 months were fed TMR2, while the dry dairy cows were fed TMR1 in the first weeks of their dry period and TMR2 in the last weeks.

At the studied farm, roughage production consisted of grass and maize silage. These crops are produced under rainfed conditions. Between two maize cultivations, ryegrass was grown and ensilaged for feed. For the period under study (see 3.2.2.2, 'Data inventory of the foreground system'), the farmer also purchased an extra amount of maize silage, which was equal to 55% of the amount of on-farm produced maize silage. Purchased concentrates included three compound feeds (38% Crude Protein (CP), 20% CP and 18% CP), soybean meal and rapeseed meal. Purchased by-products were pressed sugar beet pulp, brewers grains and an animal feed by-product of the bio-ethanol industry, also known as *Distillers Dried Grains with Solubles* (DDGS).

The farm had two stables. One stable, which housed the lactating dairy cows, contained cubicles with wood sawdust as bedding material and was equipped with grid floors above a manure pit. This pit captured the major amount of the cow's urine and faeces ('liquid manure'). Additionally, a minor amount of 'solid manure', i.e. faeces and urine mixed with bedding material, was produced in the cubicles. The other stable, which housed the dry dairy cows and the young cattle, contained straw compartments. The cattle younger than 6 months and the dry dairy cows produced only solid manure mixed with straw. The young cattle older than 6 months had access to a grid floor above a manure pit; they consequently produced liquid manure in addition to solid manure mixed with straw. Wastewater from cleaning the milking places of the cows (daily) and the cubicles in the stable (once a year) contained cattle excrements and therefore flowed to the liquid manure pit. Wastewater from rinsing the milk installation and tank flowed to the sewerage. Wastewater from cleaning agricultural machinery ended up in surface water.

3.2.2.2 Data inventory of the foreground system

Data related to the foreground system were gathered on-site in close collaboration with the farmer. The majority of the data were retrieved from the farm accountancy files for the one-year period from November 1st, 2010 to October 31st, 2011. These accountancy files are essential for the calculation of the annual economic result but they also contain information expressed in physical units. Table 3.1 summarises a few characteristics of the farm for the period under study.

Table 3.1 Characteristics of the dairy farm under study for the period November 1st, 2010 to October 31st, 2011.

Characteristic (unit)	Value
Average number of milking cows (-)	53
Average number of young cattle (-)	35
Milk delivered to dairy plant (kg FPCM/year)	558 753
Culled cattle (kg/year)	8 314
Surplus calves (kg/year)	1 500
Area for grass production (ha)	18
Area for maize production (ha)	11
Proportion (%) of wet by-products in purchased feed (kg DM/kg DM)	46
Proportion (%) of concentrates in purchased feed (kg DM/kg DM)	33
Proportion (%) of maize silage in purchased feed (kg DM/kg DM)	21
Consumption ratio (%) of purchased feed over on-farm produced feed (kg DM/kg DM)	56
Consumption ratio (%) of concentrates (purchased) and wet by-products (purchased) over roughage (both on-farm produced and purchased) (kg DM/kg DM)	39

Total fuel use by the farmer (source of data: farm accountancy files) was distributed over the different demand sides based on Van linden and Herman (2014). Data from Van linden and Herman (2014) were also used to estimate fuel use by contract workers, who performed some of the activities such as harvesting maize. Total on-farm groundwater use (source of data: farm accountancy files) was distributed over the different demand sides based on Remmelink et al. (2013) and Derden et al. (2005).

In performing the Exergy Analysis at the level of the cattle herd, methane emissions from enteric fermentation were taken into account and calculated based on IPCC (2006) using a Tier 2 modelling approach, which is the intermediate method in terms of complexity and data requirements. Additionally, the amount of latent and sensible heat production from the cattle was calculated based on CIGR (2002).

In order to calculate the exergy content of input and output flows according to the methods described in Szargut et al. (1988) and Dewulf et al. (2008), additional data on their composition were needed. We obtained data on the composition of most feed ingredients from the farmer and *Productschap Diervoeder* (2007). Data about the composition of the animals were retrieved from Andrew et al. (1994) and Diaz et al. (2001). A macronutrient composition of the liquid manure (excl. wastewater), composed of both faeces and urine, was obtained from Van Horn et al. (1994). This composition was considered as representative based on the chemical analysis results (Dry Matter, Total N, etc.) of the liquid manure in the pit (including wastewater). Data on the composition of solid manure were obtained from the Phyllis 2 database (ECN, 2014).

3.2.3 Background system: data inventory

For the background system, the *ecoinvent* v2.2 database (Swiss Centre for Life Cycle Inventories, 2010) was used for most of the life cycle inventory (LCI) data, such as data on electricity, diesel, seed, pesticides, mineral fertilisers, etc. Table 3.2 lists the data sources for the major ingredients of the purchased feeds.

Table 3.2 Data inventory sources for production (and processing) processes of the major ingredients of the purchased feeds.

Feed ingredient	Origin	Data source
Soybean meal	Brazil and Argentina	*ecoinvent - soybean meal, at oil mill (BR)*
Soybean hulls	Brazil and Argentina	data about agricultural production: *ecoinvent - soybeans, at farm (BR)* processing data: van Zeist et al. (2012a)
Rapeseed meal	Germany	*ecoinvent - rape meal, at oil mill (RER)*
Sunflower meal	Argentina	data about agricultural production: *ecoinvent - sunflower conventional, Castilla-y-Leon, at farm (ES)* processing data: van Zeist et al. (2012a)
Palm kernel meal	Malaysia	*ecoinvent - palm kernel meal, at oil mill (MY)*
Maize germ meal[1] and maize glutenfeed[2]	Belgium[1] and France[2]	data about agricultural production: *ecoinvent - grain maize IP, at farm (CH)* processing data: van Zeist et al. (2012c)
Sugar cane molasses	Brazil	data about agricultural production: *ecoinvent - sugarcane, at farm (BR)* processing data: Renouf et al. (2011) and van Zeist et al. (2012b)
Vinasse	Belgium	*ecoinvent - vinasse, at fermentation plant (CH)*
Barley	n/a	*ecoinvent - barley grains conventional, Barrois, at farm (FR)*
Wheat	n/a	*ecoinvent* - wheat IP, at feed mill *(CH)*
Pressed sugar beet pulp	France	*ecoinvent - pulps, from sugar beet, at sugar refinery (CH)*
Brewers grains	Belgium	Novozymes (2009)
Animal feed product of the bio-ethanol industry	Belgium	*ecoinvent - DDGS, from corn, at distillery (US)* for which we replaced the input *corn, at farm (US)* into *grain maize IP, at farm (CH)*

[1] Maize germ meal comes from Belgium; [2] Maize glutenfeed comes from France.

Representative figures for the composition, in terms of both nutrients and ingredients, of the purchased compound feeds and information about the origin of these ingredients were retrieved for the period under study (November 1st, 2010 to October 31st, 2011) from the *Qualifeed* database (DSM Nutritional Products NV, 2013). This database provides through linear programming, on a monthly basis, the composition of compound livestock feeds, taking into account the market price of the feed ingredients, the nutritional requirements and constraints of the compound feed. As the farmer had no quantitative information about the composition in terms of ingredients and no information about their origin, we consider *Qualifeed* as an appropriate data source. An average composition of the three compound feeds used at the dairy farm was calculated for the period under study. Inventory data for the purchased extra amount of maize silage were approached by using data of the on-farm produced maize silage. Transport of feed ingredients from their origin of production to the feed mill and subsequently to the dairy farm was taken into account based on *ecoinvent* data on transport systems. As regards on-farm infrastructure, LCI data of the milking parlour and machinery for agricultural field operations were retrieved from the *ecoinvent* v2.2 database. Sperm for artificial insemination of dairy cows and heifers, originating from a specialised breeding bull company outside the foreground system, was not included in the impact assessment.

3.2.4 Allocation procedure

Allocation was defined by the ISO 14044 guideline (ISO, 2006b) as 'partitioning the input or output flows of a process or a product system between the product system under study and one or more other product systems'. Regarding the allocations performed in the foreground system, physical (non-economic) criteria were used. At the α_1-level (cattle herd) and the α_5-level (housing), biological allocation of the CEENE input was performed between the produced milk (90.1%), the animals culled (8.4%) and the surplus calves (1.5%), according to the International Dairy Federation (IDF, 2010) guide. Biological allocation reflects the physiological feed requirements of dairy cattle to produce milk and meat. This allocation method is based on a causal relationship between the feed energy and milk and meat production. The allocation factors for milk and meat, respectively, can be calculated with Equations 3.1 and 3.2 (IDF, 2010):

$$AF_{milk} = 1 - 5.7717 \times \frac{M_{meat}}{M_{milk}} \tag{3.1}$$

$$AF_{meat} = 1 - \ AF_{milk} \tag{3.2}$$

where M_{meat} is the sum of the live weight of all cattle sold (including bull calves and culled mature animals) and M_{milk} is the fat-and-protein-corrected milk (FPCM) sold.
At the α_4-level (feeding), (absolute) mass allocation of the CEENE associated with fuel use for mechanical feed distribution was performed between all types of feed. At the

β_1-level (roughage production), allocation was not necessary because we were able to collect separate data for the different crop production systems (grass and maize silage production). In the background system, economic allocation, as advised by the IDF (2010) guide, was performed for co-product feed ingredients of the purchased feeds (Table 3.2).

3.2.5 Exergy analysis (EA)

The exergy of a resource equals the minimum work necessary to produce that resource in its specified state (temperature, pressure) and composition in a reversible way from common materials in the reference environment (Szargut et al., 1988) (see also section 1.2.3 in Chapter 1). From the definition it is clear that exergy is both a function of the resource and of the environment. The natural environment is not in thermodynamic equilibrium, which implies that a reference environment with zero exergy must be defined in order to calculate the exergy of a resource. The reference environment applied in our study was defined by Szargut et al. (1988) with a reference temperature T_0 of 298.15 K, a reference pressure P_0 of 1 atm and average geophysical chemical characteristics. The most common components of the natural environment (litho-, hydro- and atmosphere) were selected as reference species and were assigned a zero exergy level, the so-called dead state. Examples are SiO_2 in the external layer of the earth's crust, Cl^- in seawater and water vapour in the atmosphere (Morris and Szargut, 1986).

The total exergy of a resource can generally be divided into four components: (i) physical exergy, (ii) chemical exergy, (iii) potential exergy and (iv) kinetic exergy (Szargut et al., 1988). Potential and kinetic exergy are usually negligible in EA, except for hydropower and wind energy. Calculation of the physical and chemical exergy generally makes up the largest part of exergy calculations.

The physical exergy is equal to the maximum amount of work that can be obtained when the substance under consideration is brought from its actual state (T, P) to the reference state (T_0, P_0) by physical processes involving only thermal interaction with the environment (Kotas, 1985). The physical exergy of the substance can be calculated from its enthalpy (h) and entropy (s) at its initial T and P and at environmental T_0 and P_0 (Dewulf et al., 2008) (Equation 3.3).

$$EX_{ph} = (h - T_0\, s) - (h_0 - T_0 s_0) \tag{3.3}$$

The chemical exergy reflects the exergy content of the resource at T_0 and P_0. It is equal to the minimum amount of work necessary to synthesise, and to deliver in the reference state (T_0, P_0) the substance under consideration from the defined reference substances by means of processes involving heat transfer and exchange of substances with the environment only (Kotas, 1985). In other words, the chemical exergy of the substance is different from zero if it is not in chemical equilibrium with the dead-state environment. Chemical exergy values for the reference species, for chemical elements and many inorganic and organic substances can be retrieved from Morris and Szargut (1986). Based on the exergy value of the chemical elements, the chemical exergy of any substance can be calculated based on the exergy balance of the reversible standard (°; at T_0 and P_0) reaction of formation of the considered substance (Szargut, 2005). The chemical exergy of a substance is calculated by Equation 3.4,

$$EX_{ch}^{\circ} = \Delta G_f^{\circ} + \sum_k n_k EX_{ch,k}^{\circ} \tag{3.4}$$

where ΔG_f° is the standard free energy of formation of the substance, n_k the number of moles of the k^{th} element per unit of the substance and $EX_{ch,k}^{\circ}$ the standard chemical exergy of the k^{th} element. Other techniques, more commonly used in practice to calculate the chemical exergy, are the group contribution method and the exergy-to-energy ratios (Dewulf et al., 2008).

In addition to the abovementioned exergy components, the exergy of heat at temperature T, an ideal gas at T_0 and partial pressure P, electricity, radiation and nuclear energy can be calculated in a straightforward way (Dewulf et al., 2008).

When conducting an Exergy Analysis (EA), a gate-to-gate balance of a system or process is established based on the exergy content of all inputs and outputs. The exergy balance is used to calculate the exergy efficiency of the system or process. The product exergy efficiency η indicates which fraction of the input exergy ends up in the desired product (Equation 3.5).

$$\eta\ (\%) = 100 \times \frac{exergy\ product\ (J_{ex})}{\sum exergy\ inputs\ (J_{ex})} \tag{3.5}$$

In addition to the product exergy efficiency η, an exergy efficiency of product & by-products η' can be calculated (Equation 3.6).

$$\eta'(\%) = 100 \times \frac{exergy\ product\ and\ by-products\ (J_{ex})}{\sum exergy\ inputs\ (J_{ex})} \tag{3.6}$$

For the process α_1 (cattle herd), a protein conversion efficiency (PCE) was calculated (Equation 3.7). This efficiency addresses the conversion of dietary feed protein (consumed by all cattle at the dairy farm) to milk protein (produced by the dairy cows).

$$PCE\ (\%) = 100 \times \frac{protein\ content\ milk\ (g)}{\sum protein\ content\ of\ feeds\ (g)} \tag{3.7}$$

3.2.6 Exergetic Life Cycle Assessment (ELCA)

The *Cumulative Exergy Extraction from the Natural Environment version 2013 (CEENE v2013)* method was applied in this study to quantify the total exergy that is contained in the various natural resources that are retrieved from the environment and used throughout the cradle-to-farm-gate life cycle (see also section 2.2.1 in Chapter 2). Compared to other resource-based indicators such as the Cumulative Energy Demand (CED) (Frischknecht et al., 2007) and the Cumulative Exergy Demand (CExD) (Bösch et al., 2007), the CEENE method allows a more extended footprint of resources. Eight categories of resource use are distinguished in the CEENE method: abiotic renewable resources (wind and hydropower), fossil resources, metals, nuclear energy, land resources, minerals, water and atmospheric resources. The CEENE method adds land resources to both the CExD and the CED method, and adds water resources, minerals and metals to the CED method.

The rationale of the CEENE method (CEENE v2007) is explained by Dewulf et al. (2007a) and was partially modified by Alvarenga et al. (2013c), who created a more consistent accounting for land and biotic resources by the CEENE method. The resulting new version of the CEENE method (CEENE v2013) accounts for both land occupation and biomass harvested, without double counting due to a clear distinction between natural and human-made systems. For natural systems, the exergy contained in the harvested biomass was accounted for in the CEENE land resources category. For human-made systems, the occupied land was accounted for in the CEENE land resources category through the exergy contained in the potential natural net primary production (NPP) on that land. In this way, CEENE v2013 accounts for what is actually deprived from the natural environment. This new approach allowed to establish spatial differentiation factors for land use (e.g. Belgium: 26.9 MJ_{ex}/m^2*year; France: 28.0 MJ_{ex}/m^2*year; Brazil: 38.8 MJ_{ex}/m^2*year; Malaysia: 48.3 MJ_{ex}/m^2*year) in human-made systems (e.g. agriculture). In this case study, one-year use of the on-farm land available for maize production was distributed between the main crop (maize; May-September) and the (harvested) catch crop (ryegrass; October-April) by taking into account the seasonal variation of the surface solar radiation (67% for maize and 33% for ryegrass).

Regarding water resources, the CEENE method accounts for *blue* water only. *Blue* water is extracted from the environment in a forced way and refers to so-called human-induced water use. In LCA research, a water footprint usually accounts for one or more contributions, including *blue* (fresh surface and groundwater), *green* (rainfall that does not run off, but directly used and evaporated by non-irrigated agriculture, pasture and forests) and *grey* water (the volume of freshwater needed to assimilate emissions to freshwater) (FAO, 2003; Hoekstra et al., 2011). Like solar radiation, rainfall is a non-forced environmental input, which is only accessible through land occupation. The CEENE method therefore does not account for rainfall on agricultural fields, as is the case in the *ecoinvent* datasets.

3.3 Results and discussion

3.3.1 Exergy analysis (EA) (gate-to-gate)

Focusing on the core process of the dairy farm, Figure 3.2 illustrates the exergy input and output flows at the α_1-level (cattle herd) for the accounting year under study. The major exergy input is the total consumed feed (99%), which can be split into purchased feed (37%) and on-farm produced roughage (61%). Main exergy outputs are the manure produced (54%), which can be split into liquid manure (51%) and solid manure (3%), the milk produced (32%) and the methane produced by enteric fermentation (9%). About 11 MJ_{ex} or 52% of the input exergy is irreversibly lost at herd level, as a consequence of the second law of thermodynamics (see section 1.2.3 in Chapter 1). Besides producing the quantified exergy outputs, the chemical exergy in the animal feed is expended in the biological metabolism, movement, growth and reproduction (Blumberg, 2002). Milk was produced with an exergy efficiency of 15.2% at herd level (Figure 3.2). When taking the by-products, culled animals, and surplus calves, into account, the efficiency increases only slightly to 16.1%. When also taking manure into account (also a type of by-product because it is used as a fertilizer), the efficiency increases to 42.0%. The calculation of these efficiencies includes the feed consumption of all cattle (both dairy cows and young cattle together). This choice was made because the dairy farm continually renews the dairy herd by producing female "replacement" calves; this guarantees continuous milk production. The protein conversion efficiency (PCE), commonly used in dairy research, in contrast, is generally calculated by only accounting for the feed consumption of the dairy cows (Sebek and Temme, 2009). In our study, we prefer to calculate the PCE by including the feed consumption of all cattle for the reason mentioned above; we calculated a PCE of 18.8%.

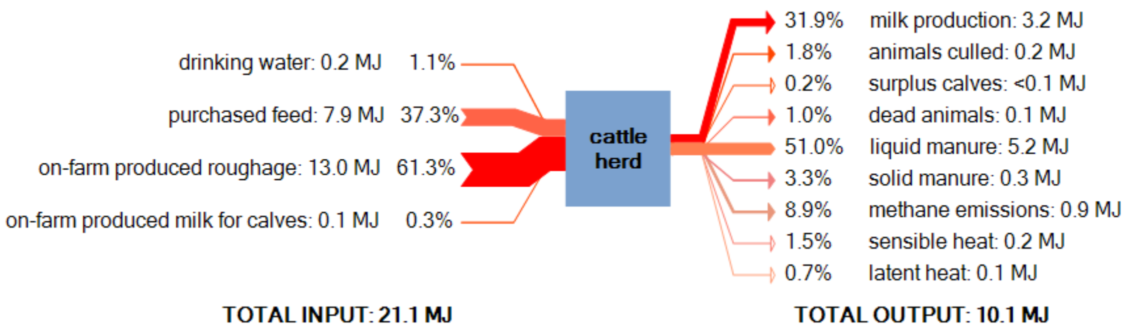


Figure 3.2 Sankey diagram of exergy flows crossing the α1-boundary (cattle herd) for the chosen case. Values are expressed in megajoules of exergy per kg FPCM sold to the dairy plant. The gases carbon dioxide and oxygen are not shown because their exergy value equals zero.

Another common calculation in dairy research is a gross energy (GE) balance. Like the PCE, this balance is usually calculated by only accounting for the feed consumption of the dairy cows. When we applied this calculation to our case study, we calculated that heat production, manure, milk and methane emissions represented 40, 31, 23 and 6% of the GE intake (feed), respectively. Similar figures were published by Van Horn et al. (1994). The difference in GE balance compared to the exergy balance lies mostly in the contribution of heat production. Heat production has only a very small share in the exergy output, because the temperature of the produced heat is rather low (body temperature). Heat at temperatures close to the reference temperature of 298 K (see

3.2.5, 'Exergy Analysis') does not contain much exergy, or in other words, it has a low ability to perform work.

Compared to the milk produced (3.2 MJ_{ex}), a large exergy output at herd level is embedded in manure (5.5 MJ_{ex}) and in methane emissions (0.9 MJ_{ex}). From a resource point of view, we should search for better ways to utilise these flows. In contrast to methane emissions, manure is not entirely lost to the environment, but it is applied as fertiliser on agricultural land. However, one opportunity would be to first digest the manure in an on-farm small-scale digester and then apply the remaining digestate, which retains the NPK nutrients, to the land. Anaerobic digestion of manure produces biogas, which could be burnt in a combined heat and power (CHP) installation. The successful implementation of a digester on a particular dairy farm depends on the profitability and the practical feasibility. The latter implies a continuous supply of fresh manure. Fresh manure is required for good biogas production and an amount of 2000 m^3 liquid manure per year is reported as a minimum to meet the continuous supply to the digester. This amount of manure corresponds with a herd size of 70 to 80 dairy cows (Goessens, 2012; Goessens, 2013). The farm under study only had 53 cows, thus successful implementation is hampered for that farm. Manure from young cattle is generally not considered because of several reasons, i.e. i) young cattle are often kept separately from the dairy cows (in another stable), which reduces the practical feasibility of using this amount of manure, and ii) young cattle are often housed in straw compartments, resulting in solid stable manure, which is generally not sent to the on-farm small-scale digester. A lower number of dairy cows (about 50), however, could become feasible when a manure scraper is present in the stable, because this allows immediate transport of fresh manure to the digester. In addition to the herd size of the farm, the profitability depends on several factors such as the presence of policy support for green power and the actual use of the electricity and heat produced on the farm. The latter depends in turn on the herd size, because the herd size indirectly determines the electricity demand.

Valorisation of methane emissions from enteric fermentation is certainly less straightforward compared to manure valorisation. Dijk et al. (2012) researched the possibilities to recover or remove methane from the atmosphere of the dairy stable. They determined that it was inefficient to recover methane from the stable atmosphere through adsorption on activated carbon because the amount of energy needed for methane recovery was approximately equal to the amount of energy that could be produced from the recovered methane. The low concentration of methane in the stable atmosphere (50 ppmv) also presents a technical problem. Removal of methane by a bio-filter would be a promising option to reduce global warming, because methane is oxidised to carbon dioxide, which has a 34 times lower global warming potential (with inclusion of climate-carbon feedbacks) than methane (IPCC, 2013). But from a resource point of view, oxidation of methane to carbon dioxide is not a satisfying solution. Another promising avenue of research to reduce global warming is the reduction of enteric methane emissions by adding methane-reducing feed supplements (Castro-Montoya et al., 2012; Machmuller, 2006; Staerfl et al., 2012). Despite that this mitigation strategy is promising, off-farm emissions from the production of the feed supplements must be included to ensure that greenhouse gas emissions are in fact reduced throughout the life cycle (Williams et al., 2014).

3.3.2 CEENE impact assessment: at life cycle level (cradle-to-farm-gate)

The total CEENE, i.e. the natural resource consumption over the cradle-to-farm-gate life cycle, amounted to 28.3 MJ_{ex} per kg FPCM sold for the chosen case. The CEENE resource footprint in terms of the different resource categories is presented in the bar chart of Figure 3.3. The on-farm roughage production (56%) and the feed purchased (37%) were the largest contributors to the total CEENE, followed by other inputs of the dairy production (7%) such as energy and groundwater use. We can conclude that, from a resource point of view, feed supply is by far the most demanding part of the dairy production chain, representing 93% of the total CEENE. With respect to the types of resources, land resources took the largest share (77%) in the total CEENE, followed by fossil resources (17%), nuclear resources (3%), water resources (2%) and abiotic renewable resources (1%) (Figure 3.3).

The large share of land resources in the total CEENE represented 24.1 MJ_{ex} per kg FPCM sold, which amounts to 0.88 m^2*year per kg FPCM sold after conversion. About 36% of the land resources (in terms of m^2*year) that were used, were indirectly used off-farm (0.32 m^2*year per kg FPCM sold). The use of land resources was almost entirely (96% in terms of m^2*year) related to the supply of feed (0.84 m^2*year per kg FPCM sold). On-farm roughage production contributed to the major part (0.56 m^2*year per kg FPCM sold), mainly because a higher proportion of on-farm produced roughage was included in the feed ration compared to the feed purchased (Table 3.1). Also, the low use of land resources of the purchased wet by-products and the purchased maize silage, which together made up the major part of the purchased feed (Table 3.1), compensated for the higher use of land resources per kg dry matter of the concentrates (see section 3.3.2.2, 'Feed purchased'). With regard to the purchased feed, approximately 72% of the off-farm occupied agricultural land was non-domestic (0.18 m^2*year per kg FPCM sold), i.e. outside Belgium in this case. About 61% of that non-domestic land use was located outside Europe (0.11 m^2*year per kg FPCM sold).

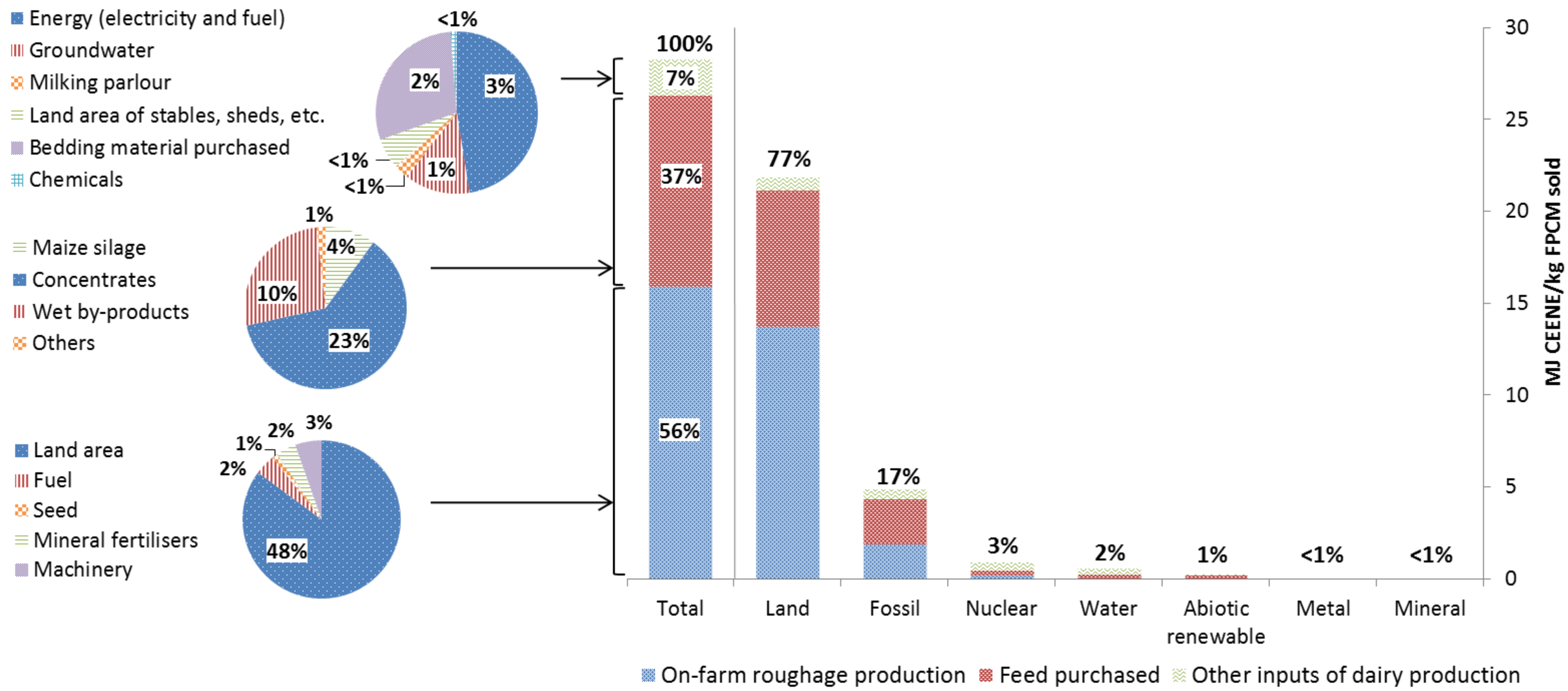


Figure 3.3 Representation of the share of the input flows to the dairy farm in the total CEENE value (expressed as MJ_{ex} CEENE/kg FPCM sold) for the chosen case. The share of the different resource categories in the total resource consumption footprint is also shown. *Chemicals* include lime, disinfectants and detergents for cleaning. *Others* include milk powder, micronutrients and feed additives. Inputs of pesticides and groundwater for spraying pesticides are not presented because their contribution was smaller than 0.1%.

With respect to fossil resources, there was a large share of indirect consumption; slightly less than one-quarter of the fossil resources that were used throughout the life cycle was related to on-farm energy use (fuel and electricity). Likewise, the major part of the fossil resources was used in the supply chain of the feed (89%), both grown on-farm (38%) and purchased (51%). The large indirect fossil resource consumption of high-input dairy systems in developed regions was also reported by Vigne et al. (2013), who highlighted the different modes of energy use of contrasting dairy systems in different regions around the world. Whereas the industrialized high-input systems heavily relied on fossil energy (in the form of mechanization, mineral fertilizers, concentrated feeds), the smallholder low-input systems were characterized by a high on-farm input of energy from human and animal labour. Intensification through mechanization and use of industrialized inputs clearly had an increasing effect on the efficiency of solar energy conversion into plant biomass in the high-input systems, compared to the smallholder systems with a low mechanisation rate and a poor access to industrialized inputs (Vigne et al., 2013).

Regarding water resources, the direct use of *blue* water (groundwater) on-farm accounted for half of the total water use across the life cycle. Of the indirect use of *blue* water, about 83% was consumed in the feed supply chain: of that amount, 27% was related to the roughage produced on-farm and 73% to the feed purchased. Some ingredients of purchased feeds, especially by-products such as maize glutenfeed, undergo several water-consuming processing steps during their production. A discussion on the comparison of the resource intensity per kg dry matter of the different types of feeds, i.e. concentrates, wet by-products and roughages, can be found in section 3.3.2.2, 'Feed purchased'.

To distinguish between renewable and non-renewable resources quantified by the CEENE method, a renewability parameter α can be calculated. This parameter reflects the renewable fraction of the overall resource consumption (Dewulf et al., 2000). For the chosen case, a value of 78% was obtained taking the CEENE categories abiotic renewable resources (wind and hydropower) and land resources into account. Land resources were included because we consider land occupation as representing the potential to capture solar radiation, a renewable resource. Water resources can also be considered as renewable and in that case the FAO (2003) defined them as the long-term average annual flow of rivers (surface water) and recharge of aquifers generated from precipitation. Non-renewable water resources were defined as deep aquifers, groundwater bodies that have a negligible rate of recharge on the human time-scale. When water resources were also included in the calculation of the renewability parameter, a value of 80% was obtained. However, it is not straightforward to distinguish which part of the water consumption is retrieved from non-renewable water resources and as a consequence contributes to water scarcity.

Further discussion of the CEENE results has been divided over three categories, i.e. (i) the on-farm roughage production, (ii) the feed purchased and (iii) other inputs of the dairy production.

3.3.2.1 On-farm roughage production

The large share of on-farm roughage production in the total CEENE (56%) is mainly due to pasture and arable land occupation (48% of total CEENE) (Figure 3.3). Regarding agricultural products, the total CEENE value is generally dominated by the land resources category (Dewulf et al., 2007a). While certain inputs can take only a relatively small part in the total CEENE, they can contribute in a more significant way to a separate CEENE resource category different from the land resources category. For each input flow to the dairy farm, Figure 3.4 shows a resource use profile, i.e. the share of the different CEENE categories in their total CEENE. Consequently, the sum of the percentages in one row must equal 100%.

	Land resources	Fossil resources	Nuclear resources	Water resources	Abiotic renewable resources	Metal resources	Mineral resources
On-farm roughage production	86%	12%	1%	<1%	1%	<1%	<1%
- Land area	100%	0%	0%	0%	0%	0%	0%
- Fuel	<1%	97%	1%	1%	<1%	<1%	<1%
- Seed	84%	12%	2%	1%	1%	<1%	<1%
- Mineral fertilisers	4%	88%	4%	2%	2%	<1%	<1%
- Pesticides	2%	69%	12%	13%	3%	<1%	<1%
- Groundwater for spraying pesticides	0%	0%	0%	100%	0%	0%	0%
- Agricultural machinery	3%	68%	16%	4%	8%	1%	<1%
Feed purchased	71%	24%	3%	2%	1%	<1%	<1%
- Maize silage	93%	6%	1%	<1%	<1%	<1%	<1%
- Concentrates	81%	16%	1%	1%	<1%	<1%	<1%
- Wet by-products	44%	47%	5%	3%	1%	<1%	<1%
- Others	14%	59%	12%	11%	3%	<1%	<1%
Other inputs of dairy production	37%	27%	20%	15%	1%	<1%	<1%
- Land area of stables, sheds, etc.	100%	0%	0%	0%	0%	0%	0%
- Energy (electricity and fuel)	1%	53%	41%	4%	1%	<1%	<1%
- Groundwater	0%	0%	0%	100%	0%	0%	0%
- Bedding material purchased	95%	4%	<1%	<1%	<1%	<1%	<1%
- Chemicals	7%	54%	19%	15%	5%	<1%	<1%
- Infrastructure: milking parlour	66%	27%	3%	2%	1%	<1%	1%
Total	77%	17%	3%	2%	1%	<1%	0%

Figure 3.4 CEENE resource use profile of inputs to the dairy farm. The sum of the percentages in one row must equal 100%. *Chemicals* include lime, disinfectants and detergents for cleaning. *Others* include milk powder, micronutrients and feed additives.

The resource use profile of the roughage produced on-farm is dominated by land resources (86%), followed by fossil resources (12%). When looking at the crop production inputs, we can see that the production of pesticides and mineral fertilisers is very fossil-intensive (69 and 88%, respectively), while seed production in particular requires land (84%). The resource use profile of agricultural machinery is mainly composed of fossil resources (68%) and nuclear resources (16%). On-farm roughage production consisted of grass and maize silage. The major part of the grass (92%) was harvested from the grasslands, while 8% was harvested between two maize cultivations. When we compare the overall resource intensity of the total production of grass silage and maize silage per kg dry matter (DM), the production of maize silage was half as resource intensive as the production of grass silage for the studied farm. The main reason for this difference was the high yield of silage maize, i.e. about 15 tonnes DM per ha over a growing period of five months, compared to the yield of the grasslands, i.e. an annual production of 12.6 tonnes DM per ha. If we would attribute the entire year of land use only to the main crop maize, instead of a distribution between maize and ryegrass (see 3.2.6, 'Exergetic Life Cycle Assessment'), maize silage would still be 24% less resource-intensive than the total amount of produced grass silage. Also, if we consider the other CEENE resource categories, the production of maize silage was between 2.6 and 5.7 times less resource-intensive than the production of grass silage. For example, in terms of fossil resources consumption, the use for maize production is 3.7 times lower per kg DM, mainly because maize was harvested in a single run, while the grasslands at the studied farm were mown 7 times per year. Thanks to a detailed (not *black box*) on-farm process-based analysis (see 3.2.2.1, 'Description of the foreground system'), the proposed framework in this work is considered as very appropriate to further investigate whole-farm strategies in terms of resource consumption, such as confinement-based versus pasture-based systems. In addition to research at the level of the individual farm, populations of contrasting milk production systems could be compared on the condition that both populations are representative in terms of optimized farm management.

When working towards a more renewables-based economy, one should seek improvements that reduce fossil resource consumption. On-farm roughage production demanded about 38% of the fossil resources that were used across the life cycle. Of that amount, direct fuel consumption for agricultural field processes accounted for one-third, while indirect use of fossil resources for the production of mineral fertilisers and agricultural machinery contributed both to one-third. Recently, Bardi et al. (2013) explored the possibilities to substitute fossil fuel use in agriculture with electricity produced from renewable sources, such as wind, photovoltaics, hydroelectricity and biomass. Note that it is very difficult, even nearly unthinkable, to generate electricity that is 100% renewable from a life cycle perspective. Biomass, for example, is generally considered as a renewable resource, but its production will probably still include fossil fuel use for the mechanical farm operations and the production of farming inputs such as mineral fertilisers. Bardi et al. (2013) concluded that several processes such as the production of nitrogen-based fertilisers, agricultural machinery operation (if a solution can be found for on-board energy storage), irrigation, etc. could be powered by renewable energy instead of fossil fuels. It is necessary, however, that farms also aim for a more efficient use of energy and other resources.

3.3.2.2 Feed purchased

The share of the feed purchased in the total CEENE (37%) is mainly due to concentrates (23% of total CEENE) and wet by-products (10% of total CEENE) (Figure 3.3).

Similar to the roughage produced on-farm, the feed purchased has a resource use profile that is dominated by land resources (71%), followed by fossil resources (24%) (Figure 3.4). In contrast to the roughage produced on-farm, the feed purchased had to be transported to the dairy farm. While transport of feed ingredients contributed to only 5% of the total CEENE of the feed purchased, it accounted for 56, 21 and 20% of the mineral, metal and fossil resources that were used, respectively. The large share of the category mineral resources is predominantly due to transportation via truck. This can be explained by the gravel needed for road construction.

Because the supply of feed has a major share in the resource consumption footprint, the environmental performance of the dairy farm could be improved by selecting feeds on the basis of the resource intensity of their production life cycle. Table 3.3 shows for our case study the relative comparison of the average resource footprint of roughages (both produced on-farm and purchased in our case study) with concentrates and wet by-products per kg dry matter (DM).

Table 3.3 Comparison of the average resource footprint (MJ_{ex}/kg dry matter) of three feed type categories, i.e. roughages (both produced on-farm and purchased in this case study), concentrates and wet by-products. For each CEENE resource category, the CEENE values of concentrates and wet by-products were expressed relatively to the CEENE value of roughages, which was set equal to one.

CEENE (MJ_{ex}/kg dry matter)	Land	Fossil	Nuclear	Water	Abiotic renewable	Metal	Mineral	Total
roughages	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
concentrates	2.3	3.4	3.2	6.9	2.6	0.9	12.5	2.5
wet by-products	0.4	2.4	3.0	4.1	1.5	0.3	3.8	0.7

Taking into account all resources, concentrates were on average 2.5 times more resource-intensive per kg DM than roughages, while wet by-products were 34 and 73% less resource-intensive than roughages and concentrates, respectively. Although wet by-products were less resource-intensive than roughages for the categories land and metal resources, they required more resources per kg DM for the categories fossil, nuclear, water, mineral and abiotic renewable resources. The low total resource consumption of wet by-products compared to roughages can mainly be explained by the low consumption of land resources. This is due to the usually very low economic value-based allocation factors to wet by-products (e.g. 3.8% for pressed sugar beet pulp). For all resource categories, concentrates were the most resource-intensive. This can mainly be explained by three reasons. First, compared to the roughages in our case study (maize and grass silage), major concentrate ingredients such as soybean meal are produced from crops that have lower yields (kg DM/ha*year) and that thus require more land per unit output. For example, according to the *ecoinvent* v2.2 database, soybeans are produced in Brazil with a yield of 2264 kg DM/ha over a growing period of six months, which is low compared to the roughage yields described in section 3.3.2.1, 'On-farm roughage production'. Second, because the CEENE method uses spatial differentiation factors for land use (see 3.2.6, 'Exergetic life Cycle Assessment'), these factors are higher for several concentrated feed exporting countries, such as Brazil and Malaysia, which have a higher potential natural NPP than the domestic country (Belgium in this case). Finally, compared to wet by-products, major concentrate ingredients usually have less low economic value-based allocation factors (e.g. 59% for soybean meal, 26% for rapeseed meal). Regarding the categories minerals and water, the very high resource consumption of concentrates compared to roughages can mainly be explained by the contribution of transport in the supply chain of concentrates. We recommend to further investigate the comparison of different feed types, taking into account also emissions-related impacts. Based on this comparison, we consider the inclusion of a higher proportion of roughages in the feed ration of dairy cows as an interesting farm strategy to further investigate.

Of course, in the selection of feed ingredients, many other factors such as nutritional parameters (e.g. positive effect of concentrates on milk yield), but also the market prices of the feeds play an important role. In Figure 3.5, we can see that the market affects the CEENE value of compound concentrates that were used at the dairy farm for the period under study (November 1st, 2010 to October 31st, 2011). The CEENE value of the compound concentrates varied throughout the year depending on the choice of the ingredients of the compound concentrates. This variation should be included in future optimisations of compound concentrate formulations.

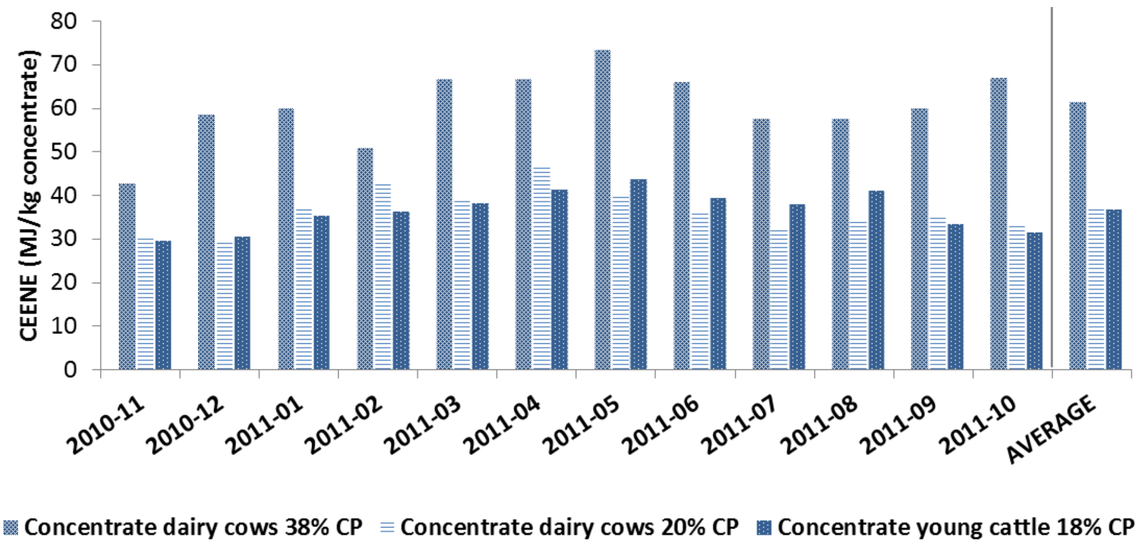


Figure 3.5 Effect of the market on the CEENE of three types of compound concentrates (expressed in MJ_{ex}/kg concentrate) for the period under study (November 1st, 2010 to October 31st, 2011).

3.3.2.3 Other inputs of dairy production

Energy consumption as electricity and fuel (excluding fuel consumption for on-farm roughage production) contributed to 3% of the total CEENE (Figure 3.3). The supply of this energy, which includes fuel for mechanical feed distribution and electricity for the milk installation and lightning, relies on fossil resources (53%) and nuclear resources (41%) (Figure 3.4). Energy consumption contributed to 10 and 46% of the fossil and nuclear resources that were used across the life cycle, respectively.

Although groundwater consumption only accounted for 1% of the total CEENE (Figure 3.3), it contributed to slightly less than half of the total *blue* water use throughout the life cycle. At the dairy farm under study, groundwater was used to provide drinking water for the animals (83%), to clean the milking parlour, to rinse the milking installation and tank (15%) and to clean the stables and other machinery (1%). Reduction of the on-farm groundwater consumption for the dairy farm under study could be possible by collecting rainwater. However, strictly speaking, this would not reduce the *blue* water consumption because "harvested" rainfall is also considered as *blue* water (Hoekstra et al., 2011). This is because most of the non-harvested rainfall would normally become run-off and replenish surface and groundwater. Other options to reduce the on-farm groundwater consumption for the chosen case is by investing in a water-saving milking installation that reuses part of its rinsing water and/or by reusing part of the rinsing effluent from the milking installation and tank for other applications. Through the installation of a three-way valve, the first, second and third water flows from rinsing the milking installation could be separated. The second and third rinse-water flows of the milking installation, as well as the rinsing effluent from the milking tank, could be reused to clean the milking parlour (the first rinse-water flow of the milking installation contains too much milk residue to be appropriate for reuse) (VMM, 2001; VMM, 2006). For the farm under study, total on-farm groundwater consumption could be reduced with 5% (calculations in Appendix B).

3.4 Conclusions and perspectives

In this study, we have demonstrated a framework to evaluate the overall resource consumption of agricultural systems at both the process level as well as the life cycle level using exergy-based resource accounting. We have performed a case study of an intensive confinement-based dairy farm in Flanders which has served as the first evaluation of the applicability and usefulness of this approach. For the chosen case, we have concluded that the feed supply chain and the animal efficiency play a key role in the improvement of the resource efficiency from a life cycle perspective. More than half of the resources consumed by the dairy farm's herd was irreversibly lost, as a consequence of the second law of thermodynamics. The remaining goes for almost two-thirds to manure and methane emissions, while only one-third goes to the milk and the animals awaiting slaughter. While manure and methane production will always remain inevitable in dairy production, better use of the exergy-rich outputs manure and methane could improve the environmental performance of the dairy farm. Anaerobic digestion of the manure could be an option, depending on farm characteristics that will determine the feasibility and the profitability of such an implementation. Valorisation of the methane is less straightforward because it cannot yet be recovered from the atmosphere of the stable. From a life cycle perspective, the supply of feed was by far the most resource-intensive part of the studied dairy production chain. With respect to the type of resources, land resources took the largest share in the resource footprint, followed by fossil resources. Because fossil resource stocks are finite and land competition is expected to increase in a more renewables-based economy (in addition to other drivers such as population growth), the challenge to achieve a higher resource efficiency is a major goal. But this goal will not be easy to achieve. A multidisciplinary approach is required. Evolution in the direction of this objective will require joint initiatives with research, policy, industry and farmers working together. Research that focuses on both resources and emissions should provide the necessary insights to steer dairy production in an environmentally sustainable direction. We recommend to further investigate the comparison of different feed types. For the chosen case in this work, concentrates were on average 2.5 times more resource-intensive per kg dry matter than roughages, while wet by-products were 34 and 73% less resource-intensive than roughages and concentrates, respectively. In practice, resource management is undoubtedly linked with the economic side of the story. In our study, we have seen that the influence of the market on the choice of the feed ingredients of compound concentrates affects the resource intensity of the production chain of those feeds. The framework proposed in this work is, therefore, very relevant in order to support research on whole-farm strategies to improve both the economic and environmental performance of dairy farms.

Figure 3.6 presents an overview of the specific objective addressed in Chapter 3.

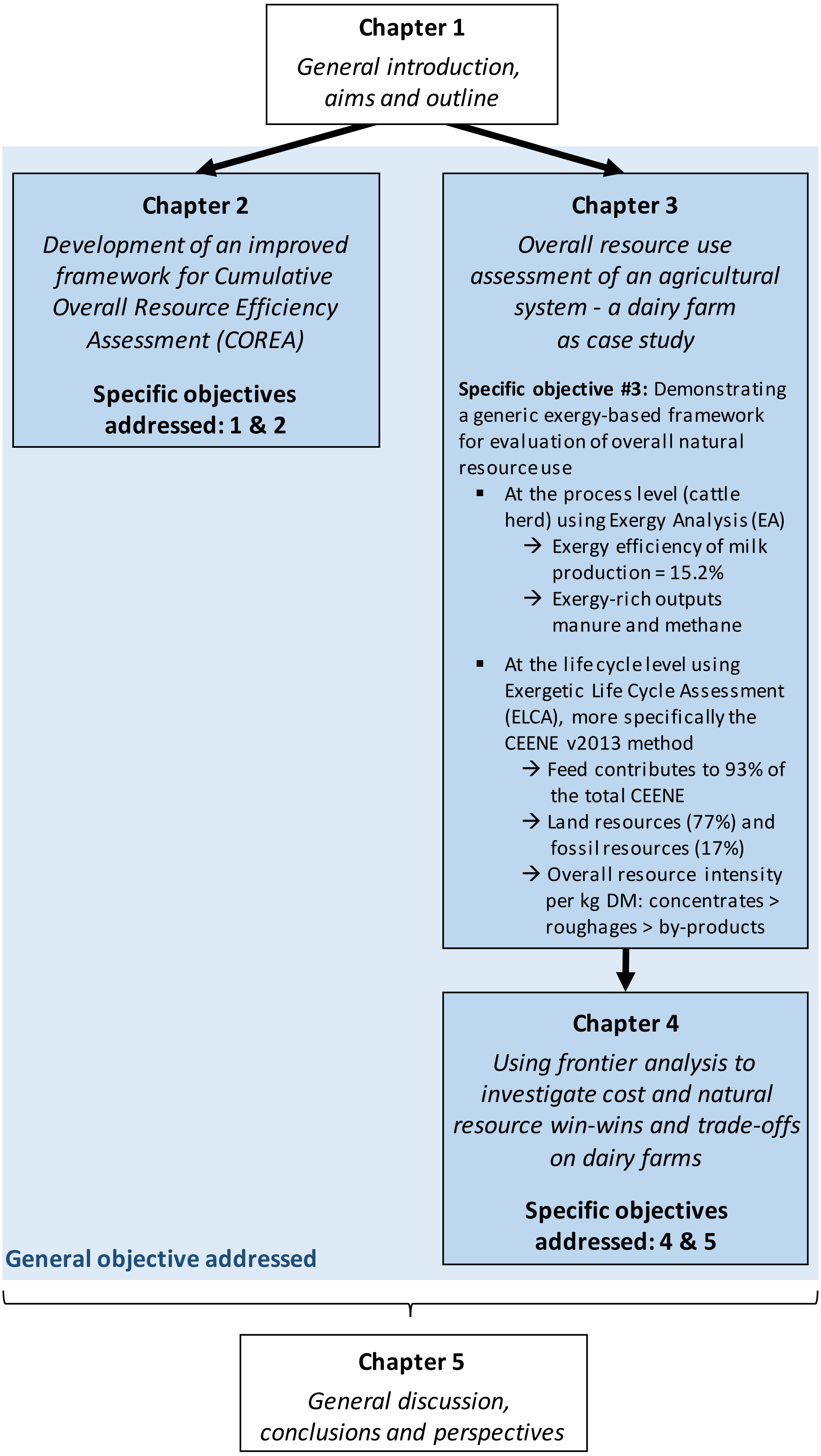


Figure 3.6 Overview of the specific objective addressed in Chapter 3.

CHAPTER 4

USING FRONTIER ANALYSIS TO INVESTIGATE COST AND NATURAL RESOURCE WIN-WINS AND TRADE-OFFS ON DAIRY FARMS

Redrafted from:

Huysveld, S., Van Meensel, J., Van linden, V., De Meester, S., Peiren, N., Muylle, H., Dewulf, J. & Lauwers, L. Using frontier analysis to investigate cost and natural resource win-wins and trade-offs on dairy farms. *Agricultural Systems*: under review.

CHAPTER 4: USING FRONTIER ANALYSIS TO INVESTIGATE COST AND NATURAL RESOURCE WIN-WINS AND TRADE-OFFS ON DAIRY FARMS

Abstract

Feed plays a key role in the challenge of dairy farmers to produce in an environmentally sustainable, yet competitive way: feed is the most important cost at dairy farms and it represents the majority of natural resources extracted throughout the supply chain of the dairy farm. In this chapter, we investigated whether and how dairy farms in the region of Flanders (Belgium) can simultaneously reduce feed costs and overall natural resource use in the feed supply chain (quantified in terms of the *Cumulative Exergy Extraction from the Natural Environment version 2013 (CEENE v2013)*) without reducing farm revenues. First, we used frontier analysis to identify realistic performance benchmarks, to distinguish win-win from trade-off situations and to calculate the achievable improvement margins. The results showed that cost and overall natural resource savings could simultaneously be made, mainly by increasing the technical efficiency (proportionally minimizing both feed inputs), rather than increasing the allocative efficiency (substituting feed inputs in cost and CEENE minimizing proportions). Second, we combined frontier analysis with analysis of Key Performance Indicators (KPIs) to acquire a better understanding of the underlying farm characteristics that may explain farm performances. The identified improvable KPIs can be used as starting points in benchmarking exercises to steer farmers towards appropriate changes in their farm management. Application of different frontier methods showed that the quantified improvement margins and the identification of win-wins and trade-offs were highly influenced by the shape of the constructed frontier. In order to improve the reliability of this approach for farm-specific decision support, further research in correctly constructing the frontier is needed.

4.1 Introduction

Dairy farmers face a major challenge to maintain the profitability of their business, while keeping it in harmony with the environment. Intensification of dairy farms has coincided with an increased resource input (material and energy) and has been accompanied by environmental burdens (e.g. greenhouse gas emissions, eutrophication, etc.) (Arsenault et al., 2009; Meul et al., 2012). In addition to rising environmental concerns, dairy farm income comes more and more under pressure due to multiple factors, e.g. increasing input costs, volatile output prices, unfavorable changing climatic conditions, etc. (UNCTAD, 2013).

Feed plays a key role in improving both the environmental and economic performance of dairy farms. Analysis of the overall natural resource use of a dairy farm's supply chain identified feed as the by far most resource-demanding input. Regarding different types of feed, concentrates were on average 2.5 times more resource-intensive per kg dry matter than roughage feed, while wet by-products were 34 and 73% less resource-intensive than roughages and concentrates, respectively (Huysveld et al., 2015b) (see Chapter 3). Intensification of dairy farms, which has led to a rise in milk yields, has been associated with an increased input of concentrates (Alvarez et al., 2008). In economic terms, feed is also of major importance on dairy farms. A comparison of the milk production costs in 46 countries, representing almost 90% of the global milk production, identified feed as the most important cost. The large contribution of feed in the total milk production costs was mainly driven by purchased feed costs (Hemme et al., 2014). In addition to natural resource and cost savings, an optimized conversion of natural resources into products could also help to reduce the production of pollutant emissions. The higher the use of raw materials per unit of product, the higher the probability of the formation of emissions (Stougie and van der Kooi, 2012). An example is the reduction of methane emissions from ruminants per unit product through an improved feed conversion (Waghorn and Hegarty, 2011). Life Cycle Assessment (LCA) studies of milk production also confirm the important role of feed in emission-related impacts (Cederberg and Mattsson, 2000; de Leis et al., 2015; Hospido et al., 2003; Thomassen et al., 2008). Increasing resource efficiency in feed production and consumption therefore appears a promising way for simultaneously targeting economic and environmental wins on dairy farms.

The objective of this work is to examine whether and how dairy farms can simultaneously reduce feed costs and overall natural resource use in the feed supply chain without reducing farm revenues. To achieve this objective, we integrate three methodologies, i.e. Exergetic Life Cycle Assessment (ELCA), frontier analysis and Key Performance Indicator (KPI) analysis, applying them on a set of 103 dairy farms in the region of Flanders (Belgium).

We rely on ELCA to quantify overall natural resource use in the feed supply chain, in particular on the exergy-based life cycle resource accounting method *Cumulative Exergy Extraction from the Natural Environment version 2013 (CEENE v2013)* (Alvarenga et al., 2013c; Dewulf et al., 2007a). This method has been elaborated for a case study of one dairy farm in Huysveld et al. (2015b) (see Chapter 3).

To investigate simultaneous reductions in feed costs and overall natural resource use in the feed supply chain, first, we integrate the CEENE method in frontier analysis. The integration of cumulative exergy use in frontier analysis was introduced by Hoang and Rao (2010), who applied it on the agricultural sectors in 29 OECD countries, and it was also applied by Maes and Van Passel (2014) on a greenhouse system for bell pepper production in Belgium. Frontier methods, frequently used in management science, analyse the transformation of input(s) into output(s) for a set of production systems with similar production technology (Coelli et al., 2005; Farrell, 1957); dairy farms in this work. Dairy farms that use their feed inputs most efficiently construct the best practice frontier. This frontier envelops dairy farms that uses their feed inputs less efficiently; the less efficient, the further the farm is located from that frontier. Frontier analysis is particularly suitable to address the objective of this work because of two reasons.

First, frontier analysis can be used to identify whether an economic-environmental win-win can be achieved on a specific farm, or whether an economic-environmental trade-off occurs (Van Meensel et al., 2010b). In this work, we focus on economic-exergetic win-wins and trade-offs. While a win-win reflects a simultaneous reduction of feed costs and cumulative overall natural resource use (CEENE) of the feed supply chain, a trade-off occurs when a reduction in feed costs goes along with an increased CEENE. After identification of an economic-exergetic win-win, an explicit improvement path can be determined.

Second, frontier analysis allows us to investigate two possible ways for achieving cost and natural resource savings, i.e. by increasing (i) technical efficiency and (ii) allocative efficiency (Coelli et al., 2005). A combination leads to the maximum achievable savings in terms of feed costs or in terms of cumulative overall natural resource use (CEENE) of the feed supply chain. By increasing technical efficiency, dairy farms move closer to the best practice frontier by proportionally minimizing both feed inputs. By increasing allocative efficiency, dairy farms move parallel with the frontier, maintaining their technical efficiency level, to an optimal proportion of their feed inputs by means of substitution; this optimal proportion minimizes feed costs (cost allocative efficiency) or the CEENE of the feed supply chain (CEENE allocative efficiency) at the considered technical efficiency level. Decomposition of cost efficiency and CEENE efficiency in technical and allocative components is an important feature of frontier analysis, because it enables to investigate the effect of substituting two main types of feeds, i.e. (i) on-farm produced roughage feed and (ii) purchased concentrates and by-products.

To acquire a better understanding of the underlying farm characteristics that may explain dairy farm economic and exergetic performances, we combine frontier analysis in a second step with analysis of Key Performance Indicators (KPIs). Examples of KPIs from dairy farming are average milk yield per cow, concentrate consumption per cow, etc. The integrated approach of frontier analysis and KPI analysis was introduced by Van Meensel et al. (2010a), who investigated cost-saving improvement paths that reduce nitrogen emissions on pig farms. Moreover, because farmers and their advisors traditionally use KPIs to measure farm performance, KPI analysis facilitates communication and validation of the outcomes of frontier analysis with practical experts. As a final step in our work, feedback on the results of the integrated approach is obtained by consulting farm advisors and agricultural experts.

This work is structured as follows. Next section (4.2) elaborates on the applied methods and the data sample. Section 4.3 presents the calculated efficiency scores and the identified economic-exergetic win-wins and trade-offs using frontier analysis, the identified improvable KPIs and the feedback from farm advisors. Section 4.4 discusses these results in both a thematic and a methodological way. Section 4.5 presents conclusions and perspectives.

4.2 Materials and methods

4.2.1 Cumulative Exergy Extraction from the Natural Environment (CEENE)

To quantify overall natural resource use of processes and entire production chains, we rely on the exergy concept (see section 1.2.3 in Chapter 1). Integrating the exergy concept in the Life Cycle Assessment (LCA) methodology results into Exergetic Life Cycle Assessment (ELCA), which can be used to calculate a production chain's overall resource footprint. In this work, the life cycle resource accounting method *Cumulative Exergy Extraction from the Natural Environment v2013 (CEENE v2013)* (Alvarenga et al., 2013c; Dewulf et al., 2007a) (see section 3.2.6 in Chapter 3) was applied to calculate the cumulative overall natural resource use of the dairy farm's purchased feeds and of the dairy farm's inputs for on-farm roughage production.

4.2.2 Frontier analysis

Frontier analysis can be used to identify farm-specific benchmarks for technical, economic and environmental performances. On the basis of the position of individual farms relative to these benchmarks, efficiency scores can be calculated and economic-environmental win-wins and trade-offs can be determined (Coelli et al., 2005; Coelli et al., 2007).

Frontier methods position individual farms against a best practice frontier, which is constructed by considering their technical performance, i.e. the transformation of input(s) into output(s). Because this construction is based on real data of a set of production systems with similar production technology, the identified benchmarks are realistic (Coelli et al., 2005). Identification of farm-specific technical, economic and environmental benchmarks through frontier analysis is influenced by two aspects: (i) the shape of the constructed frontier and (ii) the farm-specific input and output amounts. Additionally, the identified benchmarks for economic and environmental performance depend on the farm-specific input prices and the farm-specific environmental coefficients of the inputs (CEENE coefficients in this research), respectively. Before explaining frontier construction in more detail, the basic concept of benchmark identification and determination of economic-environmental win-wins and trade-offs through frontier analysis is explained by means of Figure 4.1, which presents an illustrative example where two inputs producing one output are considered. The best practice frontier is presented as a unit-isoquant, meaning that it is showing best practice input possibilities for producing one unit of output (Coelli et al., 2005).

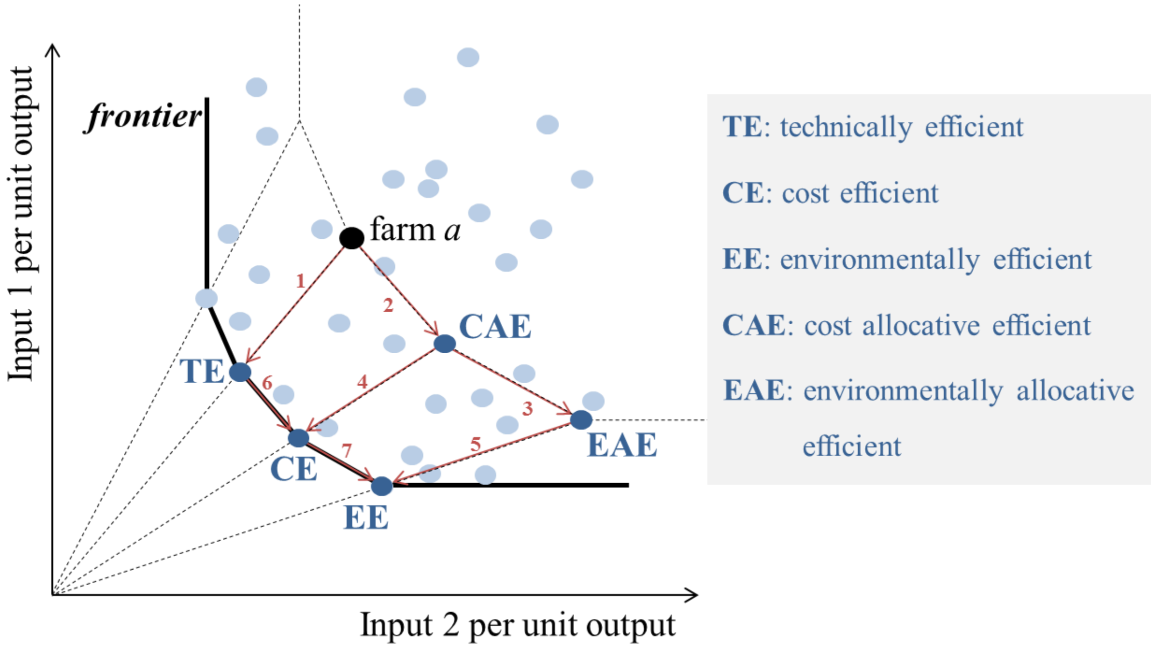


Figure 4.1 Illustrative example of the frontier (thick black line) and the identification of technical, economic and environmental performance benchmarks (dark blue dots) in the case where two inputs producing the output are considered. For farm a (black dot), paths towards performance benchmarks are in solid red arrows and numbered. Dashed black lines are alignment guides while drawing. Light blue dots represent other farms in the dataset.

Figure 4.1 illustrates for farm *a* the identification of its benchmark for technical performance (TE), located on the best practice frontier, by following path 1. This path covers the radial distance between farm *a* and the frontier (the shortest path between farm a and the frontier in the direction of the origin of the coordinate system). Technical efficiency is determined by comparing the technical performance of a specific farm (defined by its amounts of inputs 1 and 2 per unit output) to the farm-specific benchmark for technical performance (technically efficient targets for inputs 1 and 2 per unit output). This efficiency reflects the ability to use minimal amount of both inputs together to obtain a given amount of output. Efficiency scores can vary between 0 and

1, 1 indicating a point on the frontier and thus a fully technically efficient farm. All farms located on the frontier are technically efficient: given their farm-specific proportion of inputs, there is no other farm in the population that uses less of both inputs and that has the same input proportion. Farms can improve their technical efficiency by making a radial movement towards the frontier. This movement proportionally reduces both inputs (Coelli et al., 2005).

Frontier methods can also be used to measure cost and environmental efficiencies (Coelli et al., 2005; Coelli et al., 2007). They combine the technical efficiency score with cost or environmental allocative efficiencies, which reflect the ability to use inputs in cost or environmental effect minimizing proportions, given the respective prices or environmental coefficients (CEENE coefficients in this research) of the inputs. Benchmarks for cost or environmental allocative efficiency are identified by moving parallel with the best practice frontier, hence maintaining the technical efficiency level, to an input allocation that minimizes costs or environmental effects (Coelli et al., 2005; Coelli et al., 2007). Figure 4.1 illustrates for farm *a* the identification of the cost allocative efficient benchmark (CAE) by following path 2. The movement along this path substitutes input 1 by input 2. The environmental allocative efficient benchmark (EAE) is reached by moving further parallel with the frontier, i.e. path 3 in Figure 4.1. Benchmarks for cost efficiency (CE) and environmental efficiency (EE) are subsequently identified by making a radial movement from the cost and environmental allocative efficient benchmarks towards the frontier, i.e. paths 4 and 5 in Figure 4.1, respectively. From production theory, we know that CE is found where the lowest possible isocost line is tangent to the frontier. This isocost line shows all possible combinations of inputs for which the total cost is equal to the minimum cost. The slope of the isocost line is determined by the ratio of the input prices and thus farm-specific. The same applies for EE: in this research, EE is found where the lowest possible iso-CEENE line is tangent to the frontier. The larger the distance of a farm on the frontier from CE or EE, the more the farm deviates from the cost or environmental optimal input combination, and the lower the cost or environmental allocative efficiency score is.

The decomposition of cost and environmental efficiencies in technical and allocative components is an important feature of frontier analysis, because it enables a distinction between technical performance and the cost or environmental optimal input allocation

(Coelli et al., 2005; Coelli et al., 2007). At the CAE location, for example, farm *a* has a lower cost efficiency score than at the CE location, where it has a cost efficiency score equal to 1. The lower cost efficiency score at CAE is fully due to a lower technical efficiency and not to a lower cost allocative efficiency, because at both locations farm *a* has the same relative input allocation and thus the same cost allocative efficiency.

In this work, we performed frontier analysis with two input variables and one output variable. Farm revenues from milk and meat[ii] production (expressed in euro) were included as output variable (y). On-farm produced roughage feed (x_1, expressed in euro) and purchased concentrates and by-products (x_2, expressed in kg) were included as input variables. The input x_2 was expressed in kg to enable a decomposition between farm-specific amounts and farm-specific prices and CEENE coefficients of purchased concentrates and by-products. These prices and CEENE coefficients are farm-specific as a result of differences in concentrate and by-product composition between farms. The input x_1, however, could not be expressed in kg, because quantities of on-farm produced roughage feed were not available in the farm accountancies (see 4.2.4, 'Data'). Because estimation of roughage yield based on the available on-farm land area would introduce too much data uncertainty, the farm-specific costs for on-farm roughage feed production were included as input variable. As a consequence, the price of input x_1 amounted for all farms to 1 euro/euro. In contrast, the CEENE coefficients of input x_1 were calculated based on farm-specific data about roughage production, hence they were farm-specific. Frontier analysis was performed with only two input variables because of two main reasons, i.e. (i) the limited size of the dataset (103 farms) and (ii) in contrast to three input variables, two input variables allow a two-dimensional didactic representation.

In order to better grasp the quantification of efficiency scores, Equations 4.1 to 4.5 are presented. Equation 4.1 shows for the ith farm the relationship between the technical efficiency score (TE_i), the technically efficient input vectors (X_i^{te}) and the initial input vectors (X_i).

$$TE_i = \frac{X_i^{te}}{X_i} \tag{4.1}$$

[ii] animals awaiting slaughter

From the cost ($x_{1,i}^{ce}$ and $x_{2,i}^{ce}$) and environmentally ($x_{1,i,j}^{ee}$ and $x_{2,i,j}^{ee}$) efficient input targets, cost and environmental efficiencies are calculated, respectively, as:

$$CE_i = \frac{p_{1,i}x_{1,i}^{ce} + p_{2,i}x_{2,i}^{ce}}{p_{1,i}x_{1,i} + p_{2,i}x_{2,i}} \tag{4.2}$$

$$EE_{i,j} = \frac{c_{1,i,j}x_{1,i,j}^{ee} + c_{2,i,j}x_{2,i,j}^{ee}}{c_{1,i,j}x_{1,i} + c_{2,i,j}x_{2,i}} \tag{4.3}$$

with:

i: farm index (1-103; see 4.2.4, 'Data')

$x_{1,i}$: roughages (euro/year)

$x_{2,i}$: concentrates and by-products (kg/year)

$p_{1,i}$: price roughages (euro/euro); this equals 1 for all farms.

$p_{2,i}$: price concentrates and by-products (euro/kg)

$x_{1,i}^{ce}$: cost efficient roughage use (euro/year)

$x_{2,i}^{ce}$: cost efficient concentrates and by-products use (kg/year)

$c_{1,i,j}$: environmental (CEENE) coefficient roughages (MJ_{ex}/euro);

$c_{2,i,j}$: environmental (CEENE) coefficient concentrates and by-products (MJ_{ex}/kg)

$x_{1,i,j}^{ee}$: environmentally (CEENE) efficient roughage use (euro/year)

$x_{2,i,j}^{ee}$: environmentally (CEENE) efficient concentrates and by-products use (kg/year)

j: index for CEENE-total or one of the CEENE categories (land (LAN), water (WAT), minerals (MIN), metals (MET), fossil energy (FOS), nuclear energy (NUC) and abiotic renewable energy (REN))

Finally, the cost allocative and environmental allocative efficiencies can be calculated, respectively, as:

$$CAE_i = \frac{CE_i}{TE_i} \tag{4.4}$$

$$EAE_{i,j} = \frac{EE_{i,j}}{TE_i} \tag{4.5}$$

Besides identification of benchmarks and calculation of efficiencies, frontier analysis can be used to identify economic-environmental win-win and trade-off situations. For the illustrative example in Figure 4.1, following path 1 represent an economic-environmental win-win by improving the technical efficiency of farm *a*. Because cost and environmental efficiencies can be decomposed in technical and allocative components, increasing technical efficiency always simultaneously improves cost and environmental performances. At TE, following path 6 also represents an economic-environmental win-win, because farm *a* is moving closer, along the frontier, to both the cost and environmental optimal input allocations. At CE, following path 7 represents an economic-environmental trade-off, because farm *a*, although moving closer to EE, is moving further away from CE.

Benchmark identification by frontier analysis depends on the shape of the constructed frontier, which in turn depends on the applied frontier method. Because the applied frontier method affects the identified benchmarks, it also affects the determination of win-wins and trade-offs. The most commonly reported frontier methods in literature are Stochastic Frontier Analysis (SFA) and Data Envelopment Analysis (DEA) (Coelli et al., 2005). SFA fits a parametric continuous production frontier to given data, and specifies a two-part error term to account for both random errors and the degree of technical inefficiency. The functional form of the frontier has to be chosen by the researcher. DEA involves the use of linear programming to construct a non-parametric frontier that envelops the data points by piecewise connecting the best-performing farms in the dataset (cfr. Figure 4.1). Both DEA and SFA have advantages and disadvantages (Van Meensel et al., 2010b). In contrast to SFA, DEA is sensitive to outliers and corner solutions. Corner solutions refer to the fact that benchmarks on the frontier appear only on corner points of the frontier. DEA, however, has the major advantage compared to SFA that it does not require a predefined functional form. In this work, both SFA and DEA were applied, but the main focus of the results section is on the application of DEA, because DEA has some advantages that are essential for the objectives of this chapter: the frontier is constructed by piecewise connecting real farms, which also facilitates, in contrast to SFA, a graphical presentation of the identified improvement paths (cfr. Figure 4.1). Both characteristics support communication and validation of the results with practical experts. The effect on the determined improvement margins when applying SFA is quantified and discussed in the methodological discussion section of this work.

When performing DEA, an assumption about the *returns to scale* has to be made and this assumption also affects the constructed frontier. A distinction is made between constant returns to scale (CRS) and variable (decreasing/increasing) returns to scale (VRS). CRS assumes that a similar increase in input results into a similar increase in output regardless of the input level at which the input increase took place. VRS assumes that a similar increase in input results into a lower increase (decreasing returns to scale) or a higher increase (increasing returns to scale) in output at increasing input levels. As a consequence, technical efficiencies are equal or higher under VRS assumption (see Supplementary material C1 in Appendix C). Another consequence of performing DEA under VRS assumption is that a unit-isoquant graphical representation (cfr. Figure 4.1) can no longer be used. A unit-isoquant framework is only valid under CRS assumption, because under VRS assumption only farms with similar input levels can be compared. The focus of the results section, therefore, is on the application of DEA under CRS assumption, while the effect on the determined improvement margins when applying DEA under VRS assumption is quantified and discussed in the methodological discussion section of this work.

In case of both DEA and SFA, software packages (DEAP version 2.1 and FRONTIER version 4.1) were used to construct the frontier, to identify benchmarks and to calculate efficiency scores. More methodological background information about DEA and SFA can be found in the Supplementary materials C2 and C3 in Appendix C, respectively.

4.2.3 Key Performance Indicator (KPI) analysis

Frontier analysis is combined with KPI analysis because of two reasons. First, only on the basis of the outcomes of frontier analysis, it remains difficult to identify concrete improvement actions for farmers. KPI analysis can assist in providing additional, more concrete, advice. Second, KPIs facilitate validation of the results with experts in the dairy sector, because they are familiar with KPIs and not with frontier methods. In this work, the relation between the positioning of farms against the best practice frontier, when constructed with DEA under CRS assumption, and multiple KPIs was investigated. This was done by comparing KPIs between a reference group (10% of the farms from the dataset that were situated closest to the average farm) and another group that included farms that were situated closest to the coinciding cost and CEENE-total performance benchmarks of the average farm (see section 4.3.2). The average farm was not a real farm in the data sample. Values for the average farm were obtained by taking the average of the output variable and the average of the output-weighted input variables of the 103 farms in the data sample. Values for the prices and CEENE coefficients were obtained by taking the average for these coefficients of the 103 farms in the data sample. The nonparametric Wilcoxon two sample test was used to check whether KPI values significantly differed (*P<0.05; **P<0.01; ***P<0.001) between the reference group and the other group.

4.2.4 Data

Data of 103 specialized dairy farms in the region of Flanders (Belgium), affiliated with the same farm advisory company, were retrieved from their farm accountancy files for a one-year period in 2010-2011. The final sample of 103 farms results from an initial sample of 112 specialized dairy farms. Dairy farms with presence of beef cattle and suckler cows were not included in the initial sample. From the initial sample, 9 farms have been removed because of a low presence of young cattle due to off-farm rearing or because of substantial structural changes during the studied period. Table 4.1 summarizes the main characteristics of the dairy farms in the data sample.

Table 4.1 Characteristics of the 103 dairy farms in the data sample for a one-year period in 2010-2011.

Characteristic (unit)	Mean	Min.	Max.	Median	Interquartile range[b]
Average number of milking cows (-)	104	41	270	95	49
Average number of young cattle (-)	86	24	244	79	42
Milk sold (kg FPCM[a]/year)	912 978	263 156	2 439 105	855 406	436 936
Average milk yield (kg FPCM[a]/ cow.year)	8988	6476	10827	9015	1234
Total area for feed production (ha)	52	20	142	48	23
Area for grass production (ha)	28	9	81	25	12
Area for maize production (ha)	24	5	69	22	12

[a] FPCM: fat-and-protein-corrected milk (IDF, 2010); [b] The interquartile range is a measure of dispersion and equals the difference between the upper quartile (third quartile) and lower quartile (first quartile). The first quartile splits off the lowest 25% of data from the highest 75%. The third quartile splits off the highest 25% of data from the lowest 75%.

Data inventories of the output and the two input variables, and data about the input prices were established based on directly retrieved data from the farm accountancy files. With respect to the purchased concentrates and by-products, detailed data about their consumed quantity and their price were collected, separately for each type of concentrate (soybean meal, rapeseed meal, grains, high-protein compound concentrate, etc.) and for each type of by-product (beet pressed pulp, brewers grains, etc.). Both feed consumption data of dairy cows and young cattle were included. With respect to the on-farm produced roughage feeds, the farm-specific costs for on-farm roughage feed production (costs for land, mineral fertilizers, pesticides, fuel, machinery and contract work) were collected. This input variable was corrected for purchase and sale of roughage feeds in the accounting year, as well as for roughage feed stock changes between the beginning and the end of the accounting year.

Table 4.2 Descriptive statistics of the output and the two input variables, and their prices and CEENE-total coefficients, based on 103 dairy farms for a one-year period in 2010-2011.

	Symbol	Description	Mean	Min.	Max.	Median	Interquartile range[a]
Output and input variables	y	milk and meat (euro/year)	341 424	92 391	895 022	315 349	171 428
	x_1	roughages (euro/year)	74 445	26 509	216 533	69 983	37 274
	x_2	concentrates and by-products (kg/year)	299 231	79 782	947 338	277 806	149 310
Prices	p_1	roughages (euro/euro)	1	1	1	1	0
	p_2	concentrates and by-products (euro/kg)	0.23	0.16	0.31	0.23	0.04
CEENE-total coefficients	c_1	roughages (MJ_{ex}/euro)	219.5	139.7	431.3	210.8	59.5
	c_2	concentrates and by-products (MJ_{ex}/kg)	35.7	26.2	50.7	35.1	5.6

[a] The interquartile range is a measure of dispersion and equals the difference between the upper quartile (third quartile) and lower quartile (first quartile). The first quartile splits off the lowest 25% of data from the highest 75%. The third quartile splits off the highest 25% of data from the lowest 75%.

The data inventory of the CEENE coefficients of the two input variables was established based on resource use data of the inputs' supply chains. With respect to purchased concentrates and by-products, life cycle resource use data were mainly retrieved from *ecoinvent v2.2*, in addition to other literature sources. More detailed information about the CEENE calculation of purchased concentrates and by-products can be found in Huysveld et al. (2015b) (see Chapter 3), in which an in-depth case study of one specialized dairy farm was performed. Also with respect to the farm's inputs for on-farm roughage feed production, life cycle resource use data were mainly retrieved from *ecoinvent v2.2* (mineral fertilizers, pesticides, fuel, machinery). In addition to the collection of data about on-farm roughage production costs, physical data (ha of land, liters of fuel, kg of fertilizers, etc.) about on-farm roughage production were retrieved from the farm accountancy files. These physical data were then multiplied with their respective CEENE coefficients. The type of farm machinery used during field operations (by dairy farmers and contract workers) was estimated for all on-farm roughage feed cultivations based on Van linden and Herman (2014), and then life cycle resource use data for the production of these machineries were retrieved from *ecoinvent v2.2*. While data about the used quantity of fuel by the dairy farmers themselves could be retrieved from the farm accountancy files, the used quantity of fuel during contract work was estimated from the contract work costs based on Van linden et al. (2013). To account for the on-farm land area for roughage production, the CEENE value of 26.9 MJ_{ex}/m^2*year (Belgium) was used (Alvarenga et al., 2013c).

4.3 Results

4.3.1 Efficiency scores

Table 4.3 presents technical, cost and exergetic (CEENE-total) efficiency scores for the sample of 103 dairy farms. The average technical efficiency of the sample amounted to 0.768. Four farms were identified as technically efficient (TE=1); they construct the piecewise best practice frontier. The lowest technical efficiency in the sample was 0.524. About 89% of the farms were below the technical efficiency score of 0.90, while about 66% were below the technical efficiency score of 0.80. These results indicate room for improvement to save costs and natural resources, because increasing technical efficiency simultaneously improves economic and exergetic performances.

Table 4.3 Efficiency scores calculated with Data Envelopment Analysis (DEA) under constant returns to scale (CRS) assumption for the sample of 103 specialized dairy farms in Flanders during a one-year period in 2010-2011.

Efficiency type	Average	Min.	Max.	Median	Interquartile range[a]
Technical (TE)	0.768	0.524	1.000	0.753	0.140
Cost (CE)	0.743	0.523	1.000	0.738	0.118
Exergetic – CEENE-total (EE)	0.753	0.523	1.000	0.741	0.127
Cost allocative (CAE)	0.968	0.847	1.000	0.977	0.034
Exergetic allocative - CEENE-total (EAE)	0.980	0.847	1.000	0.992	0.019

[a] The interquartile range is a measure of dispersion and equals the difference between the upper quartile (third quartile) and lower quartile (first quartile). The first quartile splits off the lowest 25% of data from the highest 75%. The third quartile splits off the highest 25% of data from the lowest 75%.

Average cost and exergetic (CEENE-total) efficiency of the sample amounted to 0.743 and 0.753, respectively (Table 4.3). This shows that the farms in the data sample were on average more or less as cost efficient as they were CEENE-total efficient. Two of the four technically efficient farms were identified as CEENE-total efficient, while one of these two farms was identified as cost efficient. One farm was thus fully efficient in terms of both costs and CEENE-total. An overview of the cost and CEENE-total efficiencies of the four technically efficient farms, linked to their position on the frontier, is illustrated in Figure 4.2, showing that the highest cost and CEENE-total efficiencies among the technically efficient farms were achieved by the two most central points on the frontier.

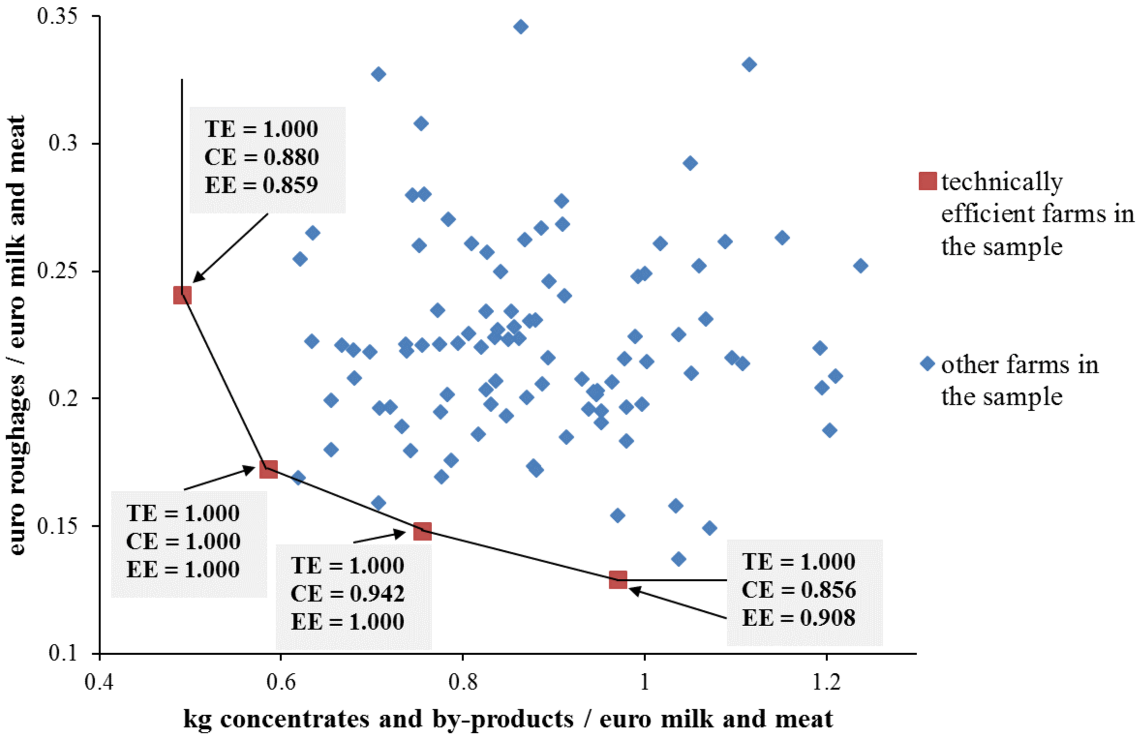


Figure 4.2 An overview of the cost and CEENE-total efficiencies of the four technically efficient farms, identified in the data sample of 103 specialized dairy farms in Flanders during a one-year period in 2010-2011 with DEA under CRS assumption, linked to their position on the frontier.

Average cost and CEENE-total allocative efficiencies were very high, both higher than 0.90 (Table 4.3). Technical efficiencies were substantially lower than the cost and CEENE-total allocative efficiencies. This shows that larger improvements could be obtained by increasing technical efficiency (using less of both inputs per unit output) rather than by substituting inputs in cost or CEENE-total minimizing proportions. The subdivision of the total CEENE in different resource categories allows one to look at one resource category in particular. Exergetic efficiency scores for each separate resource category can be found in the Supplementary material C5 in Appendix C. These scores were in the same range as the results for the total CEENE; the variation between the categories was small.

However, when looking for explicit economic-exergetic improvement paths in the next section, trade-offs between different resource categories could possibly occur.

4.3.2 Economic-exergetic win-wins and trade-offs

In addition to the calculation of efficiency scores, frontier analysis allows the identification of farm-specific improvement paths, yielding explicit targets for both inputs, given a constant output. Figure 4.3 illustrates this for the average farm. Three types of improvement paths can be distinguished: (1) proportionally minimizing both inputs up to the technical efficient benchmark, (2) substituting kilograms of concentrates and by-products by costs for roughages up to the cost allocative efficient input allocation, which also coincides for the average farm with the CEENE-total allocative efficient input allocation and (3) increasing technical efficiency and substituting kilograms of concentrates and by-products by costs for roughages up to the cost efficient input allocation, which again coincides for the average farm with the CEENE-total efficient input allocation. The coincidence of the cost and CEENE-total benchmarks was true for the average farm, but it was not true for each individual farm. Further on in this chapter, we elaborate on this farm specificity.

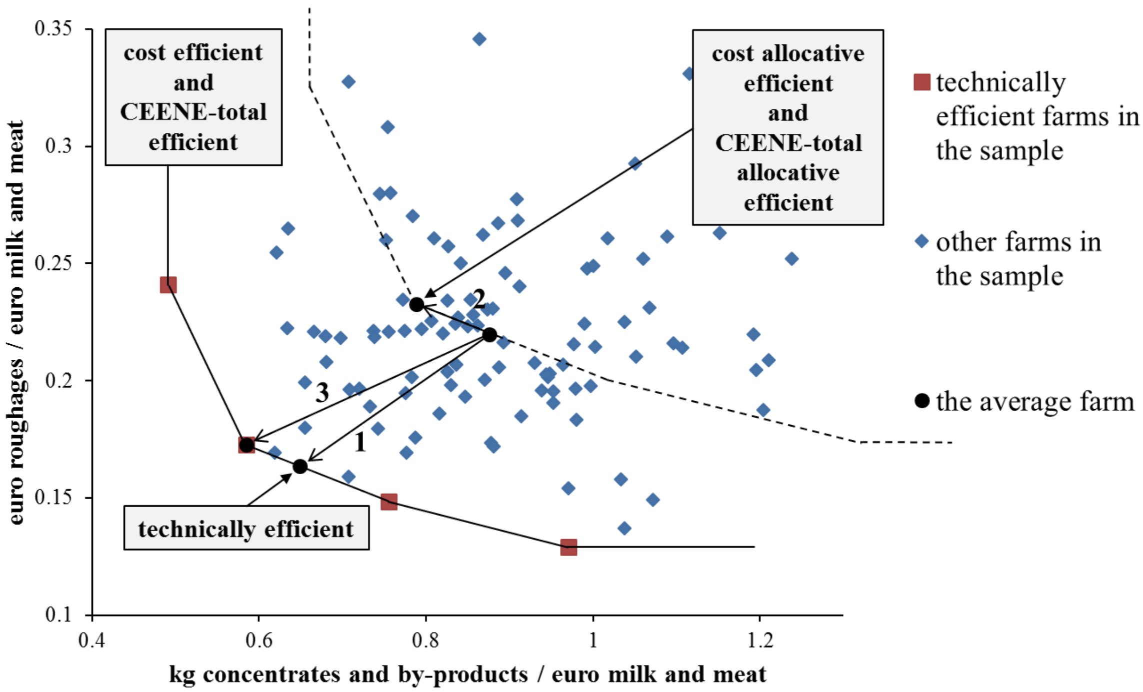


Figure 4.3 Improvement paths in terms of technical efficiency, cost (allocative) efficiency and CEENE-total (allocative) efficiency for the average farm in the data sample of 103 specialized dairy farms in Flanders during a one-year period in 2010-2011, based on application of DEA under CRS assumption.

Improvement path 1 in Figure 4.3 represents a technical optimization, i.e. using less of both inputs, in the same proportion, without reducing farm revenues. This optimization yielded both cost and natural resource savings for the average farm. All technically inefficient farms in the data sample, 99 farms in total, could achieve an economic-exergetic win-win by increasing their technical efficiency. Per euro earned, the average farm could decrease its costs and natural resource use (CEENE-total) with 10.9 eurocents and 20.47 MJ_{ex}, respectively, by becoming technically efficient. To better grasp the latter value, the total natural resource consumption of the average farm, considering the two feed inputs, amounted to 79.51 MJ_{ex} per euro earned. In other words, with 341424 euro annual revenues from milk and meat production, the average farm could reduce its costs with 37226 euro/year and its natural resource consumption with 6990 GJ_{ex}/year. This technical improvement corresponds for the average farm with a decrease of 19339 euro/year costs for roughage production and a reduction in consumption of 76941 kg/year concentrates and by-products (corresponding to 17887 euro costs). Cost reduction by moving towards the technical efficient frontier ranged in the data sample from zero eurocents for the four technically efficient farms to a maximum of 26.0 eurocents per euro earned. The maximum reduction of CEENE-total in the data sample amounted to 47.3 MJ_{ex} per euro earned, in the case where the farm's total natural resource consumption was 99.4 MJ_{ex}.

In the identification of improvement paths 2 and 3 in Figure 4.3, prices and CEENE coefficients of the inputs played a role because cost and CEENE minimizing benchmarks were targeted. Similarly to improvement path 1, path 2 simultaneously decreased both costs and natural resource use of the average farm, because its cost and CEENE-total allocative efficient benchmarks coincided. The achievable savings were, however, much smaller compared to the savings achievable by becoming technically efficient. Per euro earned, the average farm could reduce its costs and natural resource use (CEENE-total) with 0.8 eurocents and 0.32 MJ_{ex}, respectively, by substituting kilograms of concentrates and by-products by costs for roughages. For the average farm, this substitution corresponded with an increase of 4295 euro/year costs for roughage production and a reduction in consumption of 29491 kg/year concentrates and by-products (corresponding to 6856 euro costs).

Improvement path 3 is a combination of paths 1 and 2 and implies simultaneously a technical optimization and an optimal use of both inputs in cost and CEENE-total minimizing proportions. By following path 3, the average farm could achieve the largest economic-exergetic win-win. Per euro earned, the average farm could reduce its costs and natural resource use (CEENE-total) with 11.5 eurocents and 20.75 MJ_{ex}, respectively, by becoming cost and CEENE-total efficient. This improvement corresponds with a decrease of 16189 euro/year costs for roughage production and a reduction in consumption of 98970 kg/year concentrates and by-products (corresponding to 23009 euro costs).

Although the average farm could achieve an economic-exergetic win-win by the substitution of its inputs, this was not true for all individual real farms in the data sample. Whether a specific farm could achieve a win-win by input substitution depended on (i) the input proportion that this farm was using and (ii) the input proportion that corresponded with cost and CEENE-total minimization, given the farm-specific prices and CEENE coefficients of the inputs. Similarly as for the average farm, the cost and CEENE-total (allocative) efficient benchmarks coincided for 78 farms in the sample, thus in 76% of all cases. However, non-coincidence does not necessarily indicate an economic-exergetic trade-off. It is possible that the cost and CEENE-total (allocative) efficient benchmarks are not coinciding but that they imply the same input substitution (e.g. substituting kilograms of concentrates and by-products by costs for roughages), in which only the substituting quantities differ. Only when different substitutions are implied (substituting kilograms of concentrates and by-products by costs for roughages versus substituting costs for roughages by kilograms of concentrates and by-products), economic-exergetic trade-offs occur. Trade-offs between costs and CEENE-total occurred for 19 farms (18% of all cases) (Figure 4.4). In these cases, the cost (allocative) efficient benchmark implied a proportional decrease of the use of concentrates and by-products, while the CEENE-total (allocative) efficient benchmark implied a proportional increase of their use.

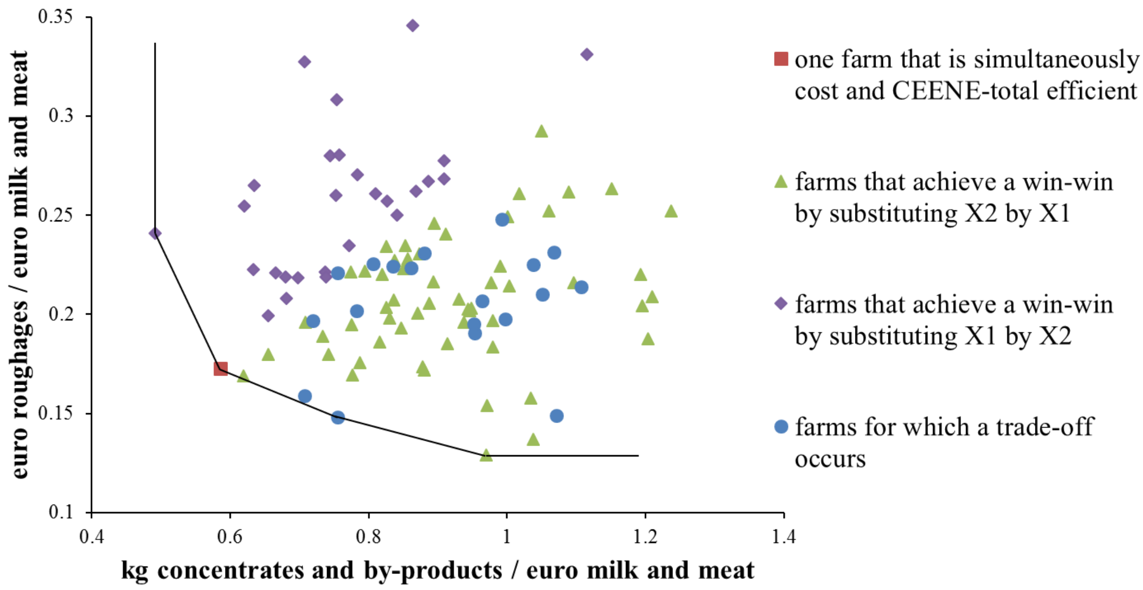


Figure 4.4 Representation of whether farms, in the data sample of 103 specialized dairy farms in Flanders during a one-year period in 2010-2011, can achieve a win-win in terms of costs and total natural resource use (CEENE-total) by substituting inputs, based on application of DEA under CRS assumption.

From the 25 farms (i.e. 103 – 78) that had non-coinciding cost and CEENE-total efficient benchmarks, 5 farms could still achieve a win-win through input substitution. Given that one farm in the sample was identified as simultaneously cost and CEENE-total efficient (Figure 4.4), 83 farms (i.e. 103 - 19 - 1 or 78 + 5) in the sample could achieve an economic-environmental win-win by substituting inputs. Figure 4.4 illustrates that 56 farms (54% of all cases) could achieve a win-win in terms of costs and CEENE-total by substituting kilograms of concentrates and by-products by costs for roughages, while 27 farms (26% of all cases) could realize this by substituting costs for roughages by kilograms of concentrates and by-products.

Table 4.4 presents descriptive characteristics of the cost and CEENE-total reductions for real farms in the data sample that could achieve a win-win by increasing technical efficiency and/or substituting inputs up to the win-win point for costs and CEENE-total on the frontier. The averages of the reductions that could be achieved by real farms in the data sample were very close to the previously mentioned achievable reductions by the (unreal) average farm. Maximum cost reduction, for example, amounted to 26.1 eurocents per euro earned, while this farm could achieve a CEENE-total reduction of 47.5 MJ_{ex} per euro earned, when the farm's total natural resource use was 99.4 MJ_{ex} per euro earned. While this farm had the lowest technical efficiency of the entire data sample, i.e. 0.524 (Table 4.3), it had very high cost and CEENE-total allocative efficiencies, i.e. both 0.998. Consequently, the majority of these reductions was achieved by increasing the technical efficiency.

Table 4.4 Descriptive characteristics of the cost and CEENE-total reductions for real farms in the data sample that could achieve a win-win by increasing technical efficiency and substituting inputs up to the win-win point for costs and CEENE-total on the frontier.

Win-win reductions	Average	Min.	Max.	Median	Interquartile range[a]
Costs (eurocents per euro earned)	11.1	0.3	26.1	10.7	6.6
CEENE-total (MJ_{ex} per euro earned)	20.0	0.4	53.8	18.9	11.6

[a] The interquartile range is a measure of dispersion and equals the difference between the upper quartile (third quartile) and lower quartile (first quartile). The first quartile splits off the lowest 25% of data from the highest 75%. The third quartile splits off the highest 25% of data from the lowest 75%.

Considering the different resource categories that make up the total CEENE, Figure 4.5 illustrates the efficiency benchmarks in terms of seven CEENE resource categories (land, water, minerals, metals, nuclear energy, fossil resources and abiotic renewable resources) for the average farm. An economic-exergetic trade-off was found in case of the average farm for the resource category land. For this resource category, moving towards the (allocative) efficient benchmark implied a substitution of costs for roughages by kilograms of concentrates and by-products. Although the (allocative) efficient benchmarks for the categories water and minerals also did not coincide with the benchmark for the total natural resource consumption (CEENE-total), they implied the same input substitution, in which the substituting quantities were larger, as the CEENE-total (allocative) efficient benchmark (i.e. substituting kilograms of concentrates and by-products by costs for roughages). The (allocative) efficient benchmarks for the categories fossil resources, nuclear energy and abiotic renewable energy coincided with the benchmark for the total natural resource consumption (CEENE-total).

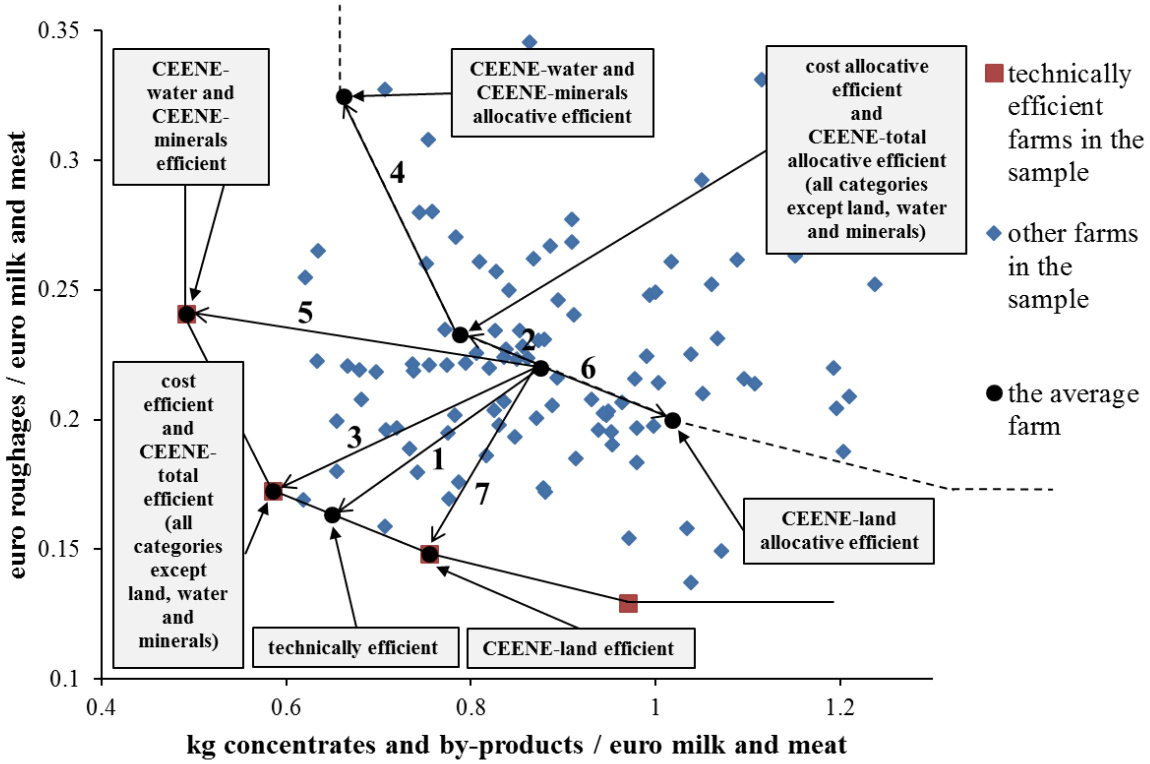


Figure 4.5 Improvement paths in terms of seven CEENE resource categories (land, water, minerals, metals, nuclear energy, fossil resources and abiotic renewable resources) for the average farm in the data sample of 103 specialized dairy farms in Flanders during a one-year period in 2010-2011, based on application of DEA under CRS assumption.

Although an economic-exergetic trade-off was found in case of the average farm for the resource category land, this was not true for all individual real farms in the data sample. In the entire data sample, economic-exergetic trade-offs were found in 67 cases, i.e. in 36 cases for the resource category land, in 29 cases for the category metals, in 22 cases for the category water, in 20 cases for the category minerals, in 3 cases for the category nuclear energy and in 1 case for the category fossil resources. Economic-exergetic trade-offs in terms of both the categories land and metals were found in 25 cases, while trade-offs in terms of both the categories water and minerals occurred in 16 cases. Considering the 19 cases in which a trade-off between costs and CEENE-total occurred, a trade-off between costs and CEENE-land was found in all these cases and between costs and CEENE-metals in 16 of these cases. Economic-exergetic trade-offs with resource categories different from land and metals all occurred in other cases than these 19 cases.

4.3.3 Analysis of Key Performance Indicators (KPIs)

In this section, we combine the results of the frontier analysis (DEA under the CRS assumption) with the analysis of Key Performance Indicators (KPIs) in order to acquire a better understanding of the underlying farm characteristics that may explain dairy farm economic and exergetic performances. KPIs of 10% of the farms from the sample that were situated closest to the average farm (group 1 in Figure 4.6) were compared with the KPIs of 10% of the farms closest to the coinciding cost and CEENE-total efficient benchmarks for the average farm (group 2). The average cost and CEENE-total efficiency for group 1 amounted to 0.728 and 0.734, respectively, while they equaled 0.900 and 0.909 for group 2. Table 4.5 shows whether KPI values significantly differed between both groups.

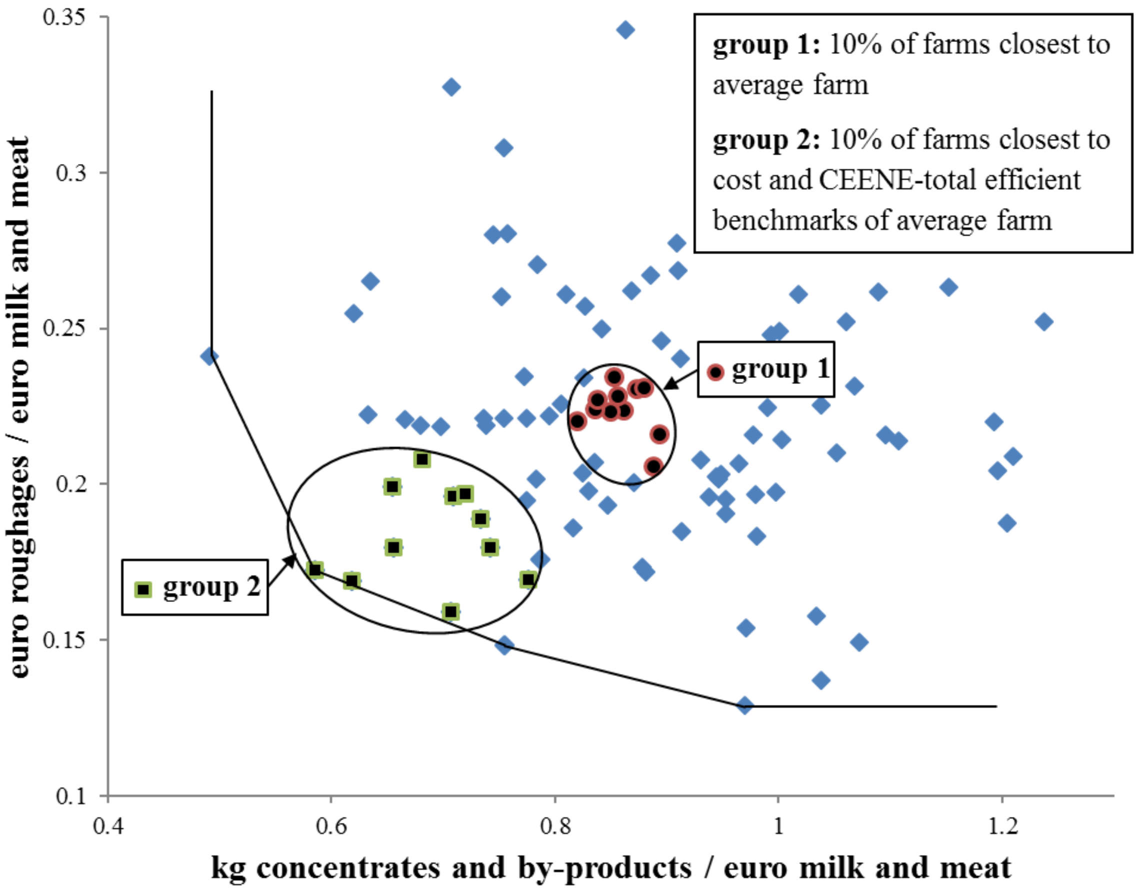


Figure 4.6 Representation of groups of farms, identified with DEA under CRS assumption, for comparison of key performance indicators.

Table 4.5 Comparison of Key Performance Indicators (KPIs) between the 10% of the farms closest to the average farm (group 1) and the 10% of the farms closest to the coinciding cost and CEENE-total efficient benchmarks for the average farm (group 2), identified with DEA under CRS assumption. A comparison between group 1 and group 2 excluding two farms with high replacement rates is also presented. The average value for each group is presented and the nonparametric Wilcoxon two sample test was used to check whether KPI values significantly differed between both groups.

Key Performance Indicators	Group 1	Group 2	Group 2 excluding two farms with high replacement rates
kg concentrates and by-products per kg FPCM[a] produced	0.31	0.25***	0.26***
euro roughages per kg FPCM[a] produced	0.081	0.068***	0.067***
average roughage cost (euro/ha)	1531	1241*	1238*
contract work (euro/ha)	415	327*	321*
area grass per total area (%)	54.33	47.69*	47.80*
area maize per total area (%)	45.76	51.99	51.80
area grass per area maize	1.27	0.93	0.94
euro milk and meat per dairy cow	3159	3516*	3612**
euro milk per dairy cow	2925	3193*	3281*
euro meat per dairy cow	234	323	331*
replacement rate	27.87	36.63*	33.23
average kg FPCM[a] produced per dairy cow	8725	9521*	9687*
farm size (ha)	52.40	52.53	56.77
farm size (number of dairy cows)	109	102	109
labor income per kg FPCM[a] produced	0.11	0.17**	0.17**

*P<0.05; **P<0.01; ***P<0.001; [a] FPCM: fat-and-protein-corrected milk (IDF, 2010)

The comparison of group 2 with group 1 showed in the first place significantly lower values for both inputs per kg of fat-and-protein-corrected milk (FPCM) produced, which could be expected. Second, the roughage production costs expressed per ha of total on-farm available land area for roughage production were significantly lower in the case of group 2. This suggests an optimized farm management in terms of roughage production. Group 2 also had significantly lower costs for contract work per ha of total on-farm available land area for roughage production. This means that farmers in group 2 outsourced less work than farmers in group 1, which may partially explain the lower roughage production costs in group 2. A limitation in this work, however, was the inclusion of contract labor costs while internal labor cost for the dairy farmer's work was not taken into account. Looking into the different cultivations, the ratio of grassland area over total available area was significantly lower in group 2. The ratio of grassland area over area for maize production was also lower in group 2, but only significantly at the 10% level. This outcome could be explained with the finding of Huysveld et al. (2015b) (see Chapter 3), who reported that the production of maize silage is half as natural resource-intensive as the production of grass silage. The major reason was the much higher maize yield compared to grassland yields. The consumption of fossil fuels was also lower in the case of maize, which is harvested in a single run, while grasslands are mown several times per year. Note, however, that the resource use intensity of grasslands depends on their use, i.e. for mowing or for grazing. When grasslands are used for grazing, fossil fuels are saved because the grass is not mechanically harvested. In terms of costs, the production of maize silage is on average 31% less costly than the production of grass silage per ton dry matter (LCV, 2012), but grazed grasslands are of course less expensive than the production of maize silage. In this study, no data were available about the grazing management of the farms in the data sample.

Third, expressed per cow, the farm revenues from milk and meat were significantly higher in group 2. Dividing the revenues between milk and meat, only the revenues from milk were significantly higher at the 5% level; revenues from meat were significantly higher at the 10% level. Group 2 had a significantly higher average milk yield per cow, implying that an optimized animal efficiency plays an important role in the dairy farm's economic and exergetic performance. Because purchased feed amounts per kg FPCM produced were significantly lower in group 2, this implies the strategy to optimize milk yield with as little as possible use of purchased feed. The proportion of by-products in the purchased feed was also lower in group 2, however, this was not significant. The replacement rate[iii] was also significantly higher in group 2 compared to group 1. This suggests that a higher replacement rate is required to be cost and CEENE-total efficient. However, consulting an expert revealed that the replacement rate is a very complex indicator to grasp and, therefore, less suitable as a univocal performance indicator. Within one farm, the replacement rate can fluctuate sharply from one year to another.

[iii] The average replacement rate in a particular year is the number of heifers that become a dairy cow during that year plus or minus the shrinkage or expansion of the dairy herd, respectively, minus the number of dairy cows that are sold on a voluntary basis (e.g. sale of cattle for breeding purpose), divided by the average total number of dairy cows present on the farm.

Some reasons can be an expansion of the dairy herd or a large number of sick cows. A closer look into group 2 revealed two farms with replacement rates of 50 and 54% (compared to the average for group 2 of 33%, excluding these 2 farms) and percentages of total forcedly disposed dairy cows (because of death, health problems or infertility) of 46 and 48% (compared to the average for group 2 of 28%, excluding these 2 farms). These numbers explain why a significantly higher replacement rate was found in group 2. Excluding these two farms from group 2, the replacement rate did no longer significantly differ between group 2 and group 1, which implies that a higher replacement rate was not a precondition to be cost and CEENE-total efficient. The average annual milk yield per cow in group 2, which was already significantly higher compared to group 1 before exclusion of these two farms, further increased to 9687 kg FPCM produced per cow per year, because the two excluded farms had a remarkably lower average annual milk yield per cow compared to the other farms in group 2. Accordingly, the average of group 2 for the revenues from milk further rose to 3281 euro per cow. After the exclusion of the two farms with high replacement rates, the average of group 2 for the revenues from meat, however, further increased to 331 euro meat per cow, which became significantly higher at the 5% level compared to group 1. The latter was mainly due to the presence of two (other) farms in group 2 with relatively high percentages (15 and 17%) of disposed dairy cows on a voluntary basis (e.g. sale of cows for breeding or disposal of cows with a low milk yield), which resulted in high revenues from meat.

Also interesting to note is that farm size, in terms of both available area and number of dairy cows, did not significantly differ between both groups. Finally, the indicator labor income[iv] per kilogram of produced FPCM was significantly higher in group 2, showing that an optimized feed management contributed to a better economic farm performance.

Other tested KPIs, which were not significantly different between both groups, were kilogram concentrates and by-products per roughage production costs, kilogram concentrates and/or by-products kilogram per dairy cow, produced FPCM per ha of

[iv] Labor income is the annual income of a farmer. It equals the farm revenues minus all costs (incl. paid salaries and paid interest of loans).

available area, average milk price, average price of concentrates and by-products, kilogram concentrates and/or by-products per ha of available area, proportion of by-products in the total amount of concentrates and by-products, proportion of soybean meal in the total amount of concentrates and by-products, ratio of young cattle number over dairy cattle number, average age of dairy cows, number of births per 100 dairy cows and number of dairy cows per ha (Supplementary material C6 in Appendix C).

KPI analysis was also performed between the groups of farms presented in Figure 4.4: (i) '*green*' farms: farms that could achieve a win-win in terms of costs and CEENE-total by substituting kilograms of concentrates and by-products by costs for roughages (reference group), (ii) '*purple*' farms: farms that could achieve a win-win in terms of costs and CEENE-total by substituting costs for roughages by kilograms of concentrates and by-products, and (ii) '*blue*' farms: farms that could not achieve a win-win in terms of cost and CEENE-total by input substitution (Table 4.6).

Compared to the other groups, '*green*' farms were characterized by a high milk production per ha, which was related to high purchased feed amounts per ha and a high number of dairy cows per ha. Per amount of milk produced, '*purple*' farms used significantly lower purchased feed amounts compared to the other groups, whereas their costs for roughage production were significantly higher. '*Purple*' farms did not have a significantly lower average milk production per dairy cow, which may be explained by the fact that the amount of concentrates per dairy cow was not significantly lower, in contrast to the significantly lower amount of by-products per cow. The lower proportion of by-products in the purchased feed was reflected by a significantly higher average purchased feed price, and also by a significantly higher average purchased feed CEENE. The latter is due to more resource-intensive concentrates compared to by-products (see Chapter 3). Although the high average purchased feed price and CEENE, '*purple*' farms could achieve a win-win by substituting costs for roughages by kilograms of purchased feed because of (i) the fact that their initial input of roughage costs was significantly higher compared to their initial input of kilograms of purchased feed and (ii) the frontier curvature and the location of corner points on the frontier. The technically efficient targets for the '*purple*' farms were all located between the upper left and the middle left corner point on the frontier. A win-win by substitution of kilograms of purchased feed by costs for roughages would require a higher price or CEENE-coefficient of the purchased feed in order that the isocost or iso-CEENE line would be tangent to the upper left corner point on the frontier (see Figure 4.4).

Table 4.6 Comparison of Key Performance Indicators (KPIs) between the '*green*' farms (reference group) and the '*purple*' and '*blue*' farms, presented in Figure 4.4, and identified with DEA under CRS assumption. The average value for each group is presented and the nonparametric Wilcoxon two sample test was used to check whether KPI values significantly differed between the reference group and the other two groups.

Key Performance Indicators	'*Green*' farms	'*Purple*' farms	'*Blue*' farms
kg concentrates and by-products per kg FPCM[a] produced	0.34	0.28***	0.33
euro roughages per kg FPCM[a] produced	0.075	0.094***	0.074
kg concentrates and by-products per euro roughages	4.54	2.97***	4.51
kg concentrates and by-products per dairy cow	3074	2424***	2868
kg concentrates per dairy cow	1835	1754	1722
kg by-products per dairy cow	1239	670***	1146
kg FPCM[a] produced per dairy cow	9185	8712	8793
average roughage cost (euro/ha)	1517	1503	1212**
average price concentrates and by-products (euro/kg)	0.22	0.25**	0.23
average CEENE-total concentrates and by-products (MJ_{ex}/kg)	35	38**	34
CEENE-total per euro roughages (MJ_{ex}/euro)	205	209	282***
FPCM[a] produced per available area (kg/ha)	20243	16138***	16510***
kg concentrates and by-products per ha	6799	4481***	5423**
kg concentrates per ha	4033	3234***	3172***
kg by-products per ha	2766	1247***	2251
kg by-products per kg total concentrates and by-products	0.40	0.26***	0.39
number of dairy cows per ha	2.21	1.85***	1.88**

*P<0.05; **P<0.01; ***P<0.001; [a] FPCM: fat-and-protein-corrected milk (IDF, 2010)

For '*blue*' farms, the cost (allocative) efficient benchmark implied the same substitution as for '*green*' farms, while the CEENE (allocative) efficient benchmark implied the opposite substitution as for '*green*' farms. '*Blue*' farms did not have a significantly lower average milk production per dairy cow, and their use of concentrates and/or by-products per dairy cow was also not significantly lower. Compared to '*green*' farms, '*blue*' farms had a significantly higher CEENE per on-farm roughage production costs, while the average CEENE of purchased feed was not significantly different. This explains why the CEENE (allocative) efficient benchmark for '*blue*' farms implied, in contrary to '*green*' farms, a substitution of costs for roughages by kilograms of purchased feed. The high CEENE per roughage production costs could mainly be attributed to a high on-farm area per roughage production costs in case of the '*blue*' farms. The latter is actually the inverse of roughage production costs per ha, which was significantly lower for the '*blue*' farms compared to the '*green*' farms.

Other tested KPIs, which were not significantly different between the groups of farms presented in Figure 4.4, were the ratio of grassland area over total available area, the ratio of area for maize production over total available area, the ratio of grassland area over area for maize production, contract work cost per total available area, the revenues from milk and/or meat per dairy cow, average milk price, replacement rate, farm size in terms of dairy cows or available area, labor income per kg milk produced, proportion of soybean meal in the total amount of concentrates and by-products, ratio of young cattle number over dairy cattle number, average age of dairy cows and number of births per 100 dairy cows (Supplementary material C6 in Appendix C).

4.3.4 Consulting farm advisors and agricultural experts

Advisors from the farm advisory company that supplied data for this research and agricultural experts were consulted to give feedback on the obtained results by frontier and KPI analysis. Visual presentation of the results in a two-dimensional graph seemed very helpful to communicate and discuss the research results. The farm advisors were not surprised to see the farms that were included in group 2, closely located to the cost and CEENE-total efficient benchmarks, and confirmed that these were well performing farms. The advisors were also not surprised, however, by the significantly higher replacement rate in group 2. Although they could have the best knowledge of the farms under study, they did not make us aware that the higher replacement rate in group 2 could be caused by farms with a high proportion of forcedly disposed cows due to health problems. The significantly lower proportion of grassland area in group 2 was immediately explained by them as due to the lower grass yield compared to the high yield of maize. The outcome that farm size did not seem to influence farm performances was expected by them. The fact that most cost and natural resource savings could be done by improving (technical) feed efficiency, rather than by substituting feed inputs, was perceived as interesting by the advisors. The advisors agreed with the strategy to optimize milk yield with as little as possible consumption of concentrates and by-products.

Consulting agricultural experts during other meetings provided additional insights that were valuable to this research and future research. First, one agricultural expert made us aware about the complex nature of the replacement rate and its increase when a farmer has to dispose a large number of sick cows. Second, in addition to the identification of improvable KPIs, agricultural experts wanted to visualize the effects of improving KPIs on the farm performances. Simulation of the effects of possible actions on the farm performances was perceived as a necessary following research step in knowing how to achieve improvement. Third, the need for analyses over longer time periods in combination with more background information about the farms (e.g. grazing management, breeding type of dairy cows, soil type, etc.) was mentioned. When farms could be analysed over several years, valuable insights could be gained about the evolution of their farm performances in relation to their KPIs.

4.4 Discussion

4.4.1 Methodological discussion

4.4.1.1 Influence of the applied frontier method

The presented results in the previous section were based on DEA under the constant returns to scale (CRS) assumption. When we want to use these results for farm-specific decision support, the question arises to what extent the results were influenced by the chosen returns to scale assumption (DEA CRS vs. DEA VRS) and the applied frontier method (DEA vs. SFA). When applying DEA under VRS assumption, 10 additional farms were identified as technically efficient. This higher number is logic because the VRS assumption takes into account that farms can also operate in an area of increasing or decreasing returns to scale. As a consequence, technical efficiencies are equal or higher under VRS assumption (Supplementary material C1 in Appendix C). Compared to the average technical efficiency in case of DEA under CRS assumption (0.768), the average technical efficiency under VRS assumption amounted to 0.823 (Supplementary material C7 in Appendix C). The calculation of the technical efficiency score under CRS ($TE_{i,\ CRS}$) and VRS assumption ($TE_{i,\ VRS}$) allows the calculation of the scale efficiency (SE_i) as the ratio of $TE_{i, CRS}$ to $TE_{i, VRS}$ (Coelli et al., 2005). In case a farm has scale inefficiency, $TE_{i, VRS}$ is higher than $TE_{i, CRS}$. In this work, scale inefficiency reflects that a farm is not operating at an optimal feed use level. On average, the scale efficiency amounted to 0.937, while it ranged from 0.614 to 1.000 and 18 farms had a scale efficiency lower than 0.90. Of the latter, 9 were operating in an area of increasing returns to scale, while 9 were operating in an area of decreasing returns to scale. When applying SFA, no farms in the sample were identified as fully technically efficient (TE=1) because a two-part error term is taken into account by SFA (Supplementary material C3 in Appendix C). The technical efficiencies calculated by SFA, however, were generally higher than the ones calculated with DEA, except in the cases where DEA assigned a TE score of 1 to technically efficient farms. The average technical efficiency when applying SFA amounted to 0.927 (Supplementary material C7 in Appendix C). Comparing the allocative efficiencies between the different approaches (DEA CRS vs. DEA VRS vs. SFA), no general trend could be observed about the approach that resulted in the highest allocative efficiencies (Supplementary material C7 in Appendix C). Application of DEA under VRS assumption and SFA confirmed the outcome of DEA under CRS assumption that cost and natural resource savings could mainly be achieved by increasing technical efficiency, rather than increasing allocative efficiency.

Regarding the identification of farm-specific win-wins and trade-off situations, Table 4.7 compares whether the farm-specific diagnosis was similar according to the different approaches. Comparing DEA under CRS and VRS assumption, a total number of 45 farms (44% of the farms in the data sample) were similarly identified. DEA under VRS assumption generated slightly more optimistic results than DEA under CRS assumption: the number of fully efficient farms and the number of farms that could achieve a win-win by substituting inputs were higher under the VRS assumption (Table 4.7). The potential improvement margins (cost and natural resource savings) under the VRS assumption, however, were smaller, because the efficiency scores under the VRS assumption were generally higher than under CRS assumption and, thus, the efficiency gaps were smaller. On average, the possible cost reduction for real farms in the data sample decreased with 2 eurocents per euro earned (-19%) under the VRS assumption compared to the CRS assumption, while the CEENE-total reduction decreased with 4.5 MJ_{ex} per euro earned (-23%).

Table 4.7 Comparison of identified win-wins and trade-offs in terms of costs and CEENE-total when applying Data Envelopment Analysis (DEA) under constant returns to scale (CRS) assumption and variable returns to scale (VRS) assumption, and when applying Stochastic Frontier Analysis (SFA).

	DEA (CRS) vs. DEA (VRS)		DEA (CRS) vs. SFA		DEA (VRS) vs. SFA	
	DEA (CRS)	DEA (VRS)	DEA (CRS)	SFA	DEA (VRS)	SFA
Number of efficient farms in terms of both costs and CEENE-total	1	6	1	0	6	0[a]
Number (percentage) of efficient farms in terms of both costs and CEENE-total that were similar between two compared approaches	1 (100%)	1 (17%)	0 (0%)	0 (n/a)	0 (0%)	0 (n/a)
Number of farms that could achieve a win-win by substituting inputs	83	88	83	80	88	80
Number (percentage) of farms that could achieve a win-win by substituting inputs and that were similar between two compared approaches	37 (45%)	37 (42%)	29 (35%)	29 (36%)	50 (57%)	50 (63%)
Number of farms that showed a trade-off when substituting inputs	19	9	19	23	9	23
Number (percentage) of farms that showed a trade-off when substituting inputs and that were similar between two compared approaches	7 (37%)	7 (78%)	8 (42%)	8 (35%)	7 (78%)	7 (30%)
Total number (percentage) of farms that were similarly identified by the two compared approaches	45 (44%)		37 (36%)		57 (55%)	

Looking into the substitutions, it was very remarkable that, under VRS assumption, most farms (50 farms, i.e. 49% of all cases) could achieve a win-win in terms of costs and CEENE-total by substituting cost for roughages by kilograms of concentrates and by-products, while 38 farms (37% of all cases) could achieve a win-win by substituting kilograms of concentrates and by-products by costs for roughages. This is in contrast to the results of DEA under CRS assumption, where most farms (56 farms, i.e. 54% of all cases) could reach a win-win by substituting kilograms of concentrates and by-products by costs for roughages. These different results can be explained by the fact that the curvature of the constructed frontier will be different under both assumptions and therefore the substitution win-win can be different and even opposite in some cases. When performing DEA, it thus seems very important to know whether a farm is operating under constant or variable returns to scale. Further research into this aspect is required to improve the reliability of DEA for farm-specific decision support.

Comparing DEA with SFA (using a predefined Cobb-Douglas production function (Supplementary material C3 in Appendix C)), the percentage of farms that were similarly identified was much lower under CRS assumption (36%) compared to VRS assumption (55%) (Table 4.7). Even more pronounced than in the case of DEA under VRS assumption, SFA indicated that most farms (78 farms, i.e. 76% of all cases) could achieve a win-win in terms of costs and CEENE-total by substituting costs for roughages by kilograms of concentrates and by-products, while only 2 farms (2% of all cases) could realize a win-win by substituting kilograms of concentrates and by-products by costs for roughages. Comparing the potential improvement margins between DEA and SFA, the cost and natural resource savings were smaller when applying SFA, because the efficiency gaps were smaller (Supplementary material C7 in Appendix C). On average, the possible cost reduction for real farms in the data sample decreased with 8 eurocents per euro earned (-70%) when applying SFA compared to DEA under CRS assumption, while the CEENE-total reduction decreased with 12.8 MJ_{ex} per euro earned (-65%). This comparison confirms that the shape of the constructed frontier has a very large influence on the determined improvement margins and on the identified win-wins and trade-offs by substitution of inputs. The need to construct the frontier in a correct way was also stated by Van Meensel (2010b), who compared the application of DEA and SFA, two data-driven methods, with a mechanistic approach for pig finishing farms in Flanders. The major advantage of the latter is that the construction of a mechanistic frontier can be based on underlying growth, feed uptake and mortality functions. The mechanistic approach can be used as a reference for evaluating the suitability of the conventional data-driven methods, although the mechanistic approach also has disadvantages. Disadvantages are the fact that assumptions may be involved in establishing these functions, that this approach is also sensitive to outliers, and that one has to dispose of the required technical information to construct mechanistic production functions (Van Meensel et al., 2010b).

4.4.1.2 Uncertainties and limitations

In addition to the uncertainty about the results caused by the applied frontier method, some additional aspects cause uncertainty. Uncertainty related to the CEENE coefficients can be subdivided into (i) uncertainty about the life cycle inventory (LCI) data and (ii) uncertainty about the exergy values of the elementary flows (natural resources). With respect to the first type of uncertainty, we judge the uncertainty of our study, which focuses on resource consumption, similar as, and potentially lower than, studies that focus on emissions. Data inventories about resource consumption generally are established by direct data collection (primary data), while data about emissions are often obtained by modelling (secondary data) when they are not experimentally determined for the case under study. In our study that was mainly based on primary data, primary data could however not be collected about the fuel consumption during contract work and the type of machinery used during field operations (see '4.2.4 Data'). With respect to the second type of uncertainty, exergy-based resource accounting can be regarded as an advanced accounting method, which is situated along the cause-effect chain between methods that account for resources at the inventory level (mass, energy, area) and methods that assess impacts related to resource consumption at the midpoint level, and further on along the cause-effect chain at the endpoint level (Sala et al., 2016). Moving along the cause-effect chain, the level of uncertainty generally increases, with the lowest uncertainty level associated with the pure inventory methods and the highest uncertainty level linked to the endpoint impact assessment methods (Finnveden et al., 2009). The level of uncertainty involved in case of exergy-based resource accounting could be situated between the uncertainty level of the pure inventory methods and the uncertainty level of the midpoint impact assessment methods, but closer to the pure inventory methods due to the consistent scientific basis of exergy-based resource accounting. De Meester et al. (2006) performed an uncertainty analysis of the exergy value of chemical elements and mineral resources based on different literature sources. For chemical elements, De Meester et al. (2006) concluded that their exergy value is robust (exergy values differing by 1.2% on average and not differing by more than 3%), whereas the exergy values of mineral resources were more uncertain (differing by factors up to 14) due to incompleteness, inconsistencies and dated thermochemical data. Based on their analysis, De Meester et al. (2006) established a consistent dataset with exergy values of 73 minerals, which were incorporated in the CEENE method (Dewulf et al., 2007a). Exergy values of organic substances (e.g. fossil resources) are regarded to be more robust, because of the availability of a sound literature basis, according to De Meester et al. (2006).

The static character of the adopted prices and CEENE coefficients of the inputs causes additional uncertainty. Although the prices and CEENE coefficients were farm-specific, it is not certain that the value of these coefficients would remain the same when farmers are optimizing the efficiency of their farm. When a farmer changes feed rations in order to optimize the efficiency, the prices and CEENE coefficients of the feed inputs may change. Another aspect that causes uncertainty about the results is the fact that increases of internal labor (e.g. by the dairy farmer and his/her family) and investments that could be required to optimize cost efficiency were not taken into account.

The fact that frontier analysis is based on real farm data can be regarded as both an advantage and a disadvantage. Because real farms are considered instead of a normative (typical) farm, it is a major advantage that realistic performance benchmarks can be identified. However, frontier analysis depends on the group of farms that are considered, thus it might be that the real best practice farm is not included in the dataset.

Only two feed inputs were distinguished in this work because of the limited size of the dataset and in order to allow a didactical graphical presentation of the results. Especially because concentrates and by-products differ much in terms of overall resource intensity (see Table 3.3 in Chapter 3), frontier analysis with three feed inputs, after subdivision of the purchased feeds into concentrates and by-products, would be interesting to perform for a larger dataset in future research. Finally, using the costs for on-farm roughage feed production as input variable instead of the quantities of roughage feeds is a limitation in this work, because a distinction between reduced quantities of roughage feeds and reduced costs for roughage feed production could not be made. To resolve this, data from farm-specific measurements of the consumed quantities of roughage feed would be required.

4.4.2 Thematic discussion

From an environmental point of view, this work has focused on natural resource savings. Although a better resource efficiency can help to reduce the production of harmful emissions, we would like to highlight that the focus of our work should be complementary to the analysis of emissions-related impacts such as global warming. Especially in the debate about the substitution between roughages and concentrates, the analysis of enteric methane emissions cannot be omitted to ensure a more holistic farm decision support. In literature, a scientific debate is ongoing about the proportional use of roughages and concentrates in the feed ration. Several studies focusing on environmental sustainability can be found in which a lower consumption of concentrates accompanied by an increased use of on-farm produced roughage feed was recommended (Arsenault et al., 2009; Meul et al., 2012; Thomassen et al., 2008). However, it is known that a higher roughage-to-concentrate ratio results in higher enteric methane emissions (Hindrichsen et al., 2006; Lovett et al., 2003). By optimizing the production and preservation of roughages, the nutritional quality of roughages could be improved, which could allow an increased replacement of concentrates by high-quality roughages (Boadi et al., 2004; Patel, 2012). In this work, a significantly lower ratio of grassland area over area for maize production was found at the 10% significance level in the group of farms with high cost and CEENE-total efficiencies. This could imply a win-win between cost efficiency, overall resource efficiency and methane emissions, because maize generally yields less methane than grass, due to their difference in carbohydrate composition and digestibility (Knapp et al., 2014). However, grasslands as well are known to have several potential advantages compared to arable land. First, it is not always feasible to grow crops (e.g. maize) instead of grass because some lands do not allow a profitable crop production due to too wet or too dry soil conditions (Wageningen UR, 2013). Second, grasslands have several environmental advantages compared to arable land, i.e. a lower erosion sensitivity and a lower loss of nutrients (Rumpel et al., 2015). Third, compared to arable land, permanent grasslands may have higher carbon sequestration potentials and thus offset carbon dioxide emissions, although uncertainties about soil carbon stock data are high (Lugato et al., 2014a; Lugato et al., 2014b). Further research with in vivo feeding experiments integrated in whole-farm life cycle analysis is required to unravel win-wins and trade-offs between cost efficiency, resource efficiency, methane and other emissions.

In the KPI analysis, the replacement rate turned out to be less suitable as a univocal performance indicator, because it can fluctuate sharply from one year to another. High replacement rates during a particular year can be caused by large numbers of sick cows during that year. This confirms the need for analyses over longer time periods in order to see the evolution of farm performances in relation to their KPIs. Simply replacing cows earlier as a strategy to optimize cost and overall resource efficiency is certainly not a good general advice, because evidence exists that a higher replacement rate leads to a larger young stock and thus a higher replacement cost and higher methane emissions at herd level (Knapp et al., 2014).

Finally, it should be noted that the identified win-wins and trade-offs through input substitution can change in time, e.g. as a result of price changes. If concentrates become more expensive in the future, the cost efficient benchmarks would move further to a relatively lower consumption of concentrates, assuming a constant production cost for roughages. A movement in the opposite direction could occur in case of rising production costs for roughages (e.g. increasing fuel price), assuming a constant price for concentrates.

4.5 Conclusions and perspectives

The results obtained through frontier analysis showed that cost and overall natural resource savings (economic-exergetic win-wins) could simultaneously be made on dairy farms. The possible improvements could mainly be obtained by increasing the technical efficiency (proportionally minimizing both feed inputs), rather than by substituting feed inputs (kilograms of purchased concentrates and by-products versus costs for on-farm produced roughages) in cost and overall natural resource use (CEENE-total) minimizing proportions. While all farms, except the identified technically efficient farms, could achieve a win-win by increasing the technical efficiency, not all farms could achieve a win-win through input substitution. Whether a specific farm could achieve a win-win by input substitution depended on (i) the input proportion that this farm was using, (ii) the farm-specific prices and CEENE-total coefficients of the inputs, and (iii) the shape of the constructed frontier, which depended on the applied frontier method. Although frontier analysis was very suitable to analyse farm-specific win-wins and trade-offs, further research in correctly constructing the frontier is needed, because it influences the quantified improvement margins and the diagnosis of win-win and trade-off situations. The frontier methodology still has to take some substantial steps in further methodological development in order to be reliable for farm-specific decision support. While this methodological development is in progress, the reliability problem of frontier analysis could partially be overcome by KPI analysis and consulting farm advisors and other experts for validation of the results.

Combination of frontier analysis with analysis of Key Performance Indicators (KPIs) allowed identification of improvable KPIs. An example is the costs for on-farm roughage production per ha, which was significantly lower at farms with high cost and CEENE-total efficiencies. Another example is the significantly higher milk yield per cow, while the consumption of concentrates and by-products per kg produced milk was significantly lower, which implies the strategy to optimize milk yield with as little as possible consumption of concentrates and by-products. The improvable KPIs can be used as starting points in benchmarking exercises to steer farmers towards appropriate changes in their farm management.

Consulting farm advisors and other agricultural experts with the results of this work provided additional insights that were valuable to this research and future research. An important feedback for future research was the need to visualize the effects of improving KPIs on the farm performances through simulation. Feedback also included the need for analyses over longer time periods in order to see the evolution of farm performances in relation to their KPIs and to analyse the effects of strategic decisions on long-term farm performances.

It should be noted that the results of this work are not necessarily representative for dairy farms in other countries, or even the Belgian dairy farming sector, because the production technology may be different between countries and regions. The dairy farming systems in Belgium, for example, differ substantially between the northern (Flanders) and the southern region (the Walloon region).

Overall, the conclusion of this work is that the combined use of frontier analysis and KPI analysis, provided that further methodological development in frontier construction takes place, is very promising to investigate cost and resource efficiency win-wins on dairy farms and to support farmers' decision making. Further research should take into account environmental burdens such as greenhouse gas emissions, eutrophication and acidification, in order to ensure that possible trade-offs between cost efficiency, resource efficiency and other environmental issues are unraveled. Also, assessing these trade-offs between a more limited milk production, for a large part based on roughages, and higher-yielding animals that require higher amounts of concentrates would be an interesting future research topic. Furthermore, as this work mainly focused on overall natural resource use, trade-offs between different types of resource categories could be further investigated.

Figure 4.7 presents an overview of the specific objectives addressed in Chapter 4.

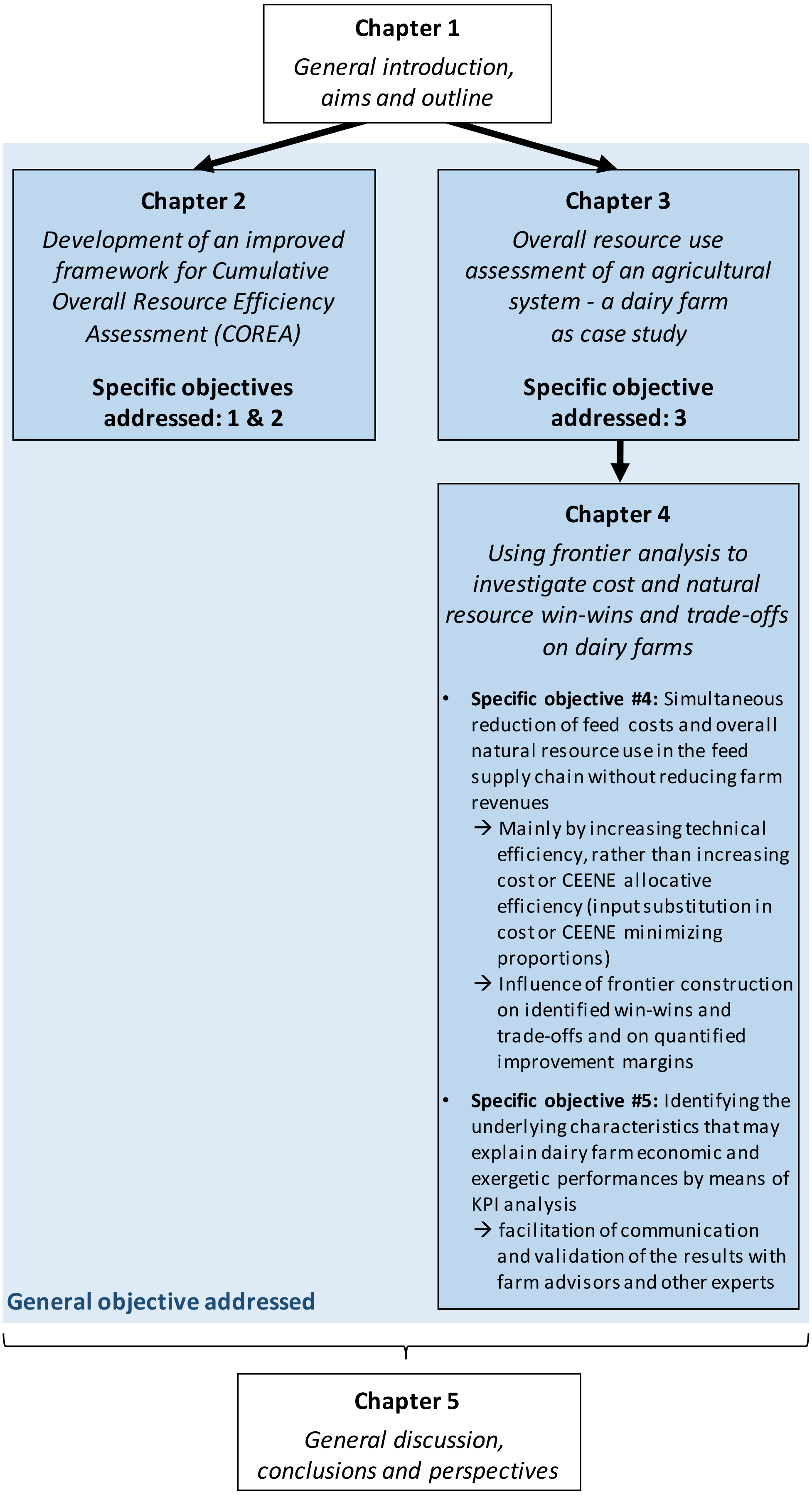


Figure 4.7 Overview of the specific objectives addressed in Chapter 4.

CHAPTER 5

GENERAL DISCUSSION, CONCLUSIONS AND PERSPECTIVES

CHAPTER 5: GENERAL DISCUSSION, CONCLUSIONS AND PERSPECTIVES

The general objective of this PhD thesis was to improve the framework of exergy-based natural resource use accounting for its application within sustainability assessment of agricultural production systems (Chapter 2), and to provide insight into its value by case study illustrations (Chapters 2 to 4). An additional methodological focus was the use of frontier analysis to investigate farm-specific economic-exergetic win-wins (Chapter 4). Thematically, this PhD thesis addressed two major challenges within the current debate on sustainable development of agriculture, i.e. (i) the bioeconomy (Chapter 2), and (ii) animal food production, which was narrowed down to dairy farms (Chapters 3 and 4) in this thesis. This final chapter, first, provides insight on the value of the exergy accounting methodology within sustainability assessment of agricultural production systems. The strengths of the exergy accounting methodology are illustrated with results from the case studies in the previous chapters. A critical view on the exergy accounting methodology follows with some suggestions for potential further development. Second, efforts that were made to translate research into practice in order to support the decision-making of farmers are discussed. Finally, concluding remarks with respect to both thematic and methodological aspects are provided.

5.1 Insights into the value of the exergy accounting methodology

5.1.1 Illustrations of the strengths of the exergy accounting methodology

The value of the exergy accounting methodology lies within two main applications: life cycle resource use accounting or *resource footprinting* and resource efficiency assessment.

Resource footprinting

The Cumulative Exergy Extraction from the Natural Environment (CEENE) method can currently be regarded as the most complete aggregated quantifier of natural resource use from a life cycle perspective (European Commission, 2011c; Liao et al., 2012; Sala et al., 2016). In addition to land resources, water, minerals, metals, fossil energy, nuclear energy, abiotic renewable energy and atmospheric resources, marine resources (biomass from natural systems (e.g. wild fish) and marine area occupation in human-made systems (e.g. artificial islands)) have been included since the last update of the method in 2014 (Taelman et al., 2014). The CEENE resource footprint provides insight into the magnitude of the overall natural resource need of a particular product system as well as into which types of natural resources that system mostly relies on. Table 5.1 gives an overview of the calculated CEENE v2013 resource footprints of the products studied in this PhD thesis.

Table 5.1 Overview of the calculated CEENE v2013 resource footprints of the products studied in this PhD thesis.

Product	**milk**	**bio-based electricity**	**fossil-based electricity**	**bio-based PVC**	**fossil-based PVC**
Unit	kg FPCM[a]	kWh	kWh	kg	kg
CEENE v2013 (MJ_{ex})	28.3[b]	8.3-13.2 [c]	10.3	102.8	53.3
Land (%)	77	84-89 [c]	<1	77	<1
Water (%)	2	1	1	4	2
Minerals (%)	<1	<1	<1	<1	<1
Metals (%)	<1	<1	<1	<1	<1
Fossil energy (%)	17	8-12 [d]	99	13	87
Nuclear energy (%)	3	2	<1	1	2
Abiotic renewable energy (%)	1	1	<1	5	9
Atmospheric resources (%)	0	0	0	0	0
Renewability (%)	78	85-89 [c]	<1	82	9
More information	Chapter 3	Chapter 2	Chapter 2	Chapter 2	Chapter 2

[a] FPCM: fat-and-protein-corrected milk; [b] This value was obtained after biological allocation (IDF, 2010) between milk (90.1%), animals culled (8.4%) and surplus calves (1.5%); [c] The highest value was calculated considering an entire year of land occupation as temporal system boundary for silage maize and silage rye, while the lowest value was calculated considering the actual cultivation period of both crops (see Chapter 2); [d] The lowest value was calculated considering an entire year of land occupation as temporal system boundary for silage maize and silage rye, while the highest value was calculated considering the actual cultivation period of both crops (see Chapter 2).

The products with an agricultural supply chain, i.e. milk, bio-based electricity and bio-based PVC, had a resource footprint dominated by land resources. The primary biomass production stage accordingly dominated the overall resource footprint of these products. In Chapter 2, the cultivation of maize and rye, which were anaerobically digested to produce bio-based electricity, contributed to more than 90% of the CEENE of bio-based electricity. The cultivation of sugarcane, which was used to produce bio-ethanol, an alternative for fossil ethylene in PVC production, had a share of 80% in the CEENE of bio-based PVC. In Chapter 3, the largest contributor to the CEENE v2013 resource footprint of one kg fat-and-protein-corrected milk (FPCM) was the feed supply with 93%. Nevertheless, all these bio-based products are not 100% renewable; their renewable resource use fraction, taking land resources and abiotic renewable energy into account, amounts between 78 and 89% in these cases. Land resources were included in the renewable fraction because we consider land occupation, which represents the potential to capture solar radiation, a renewable resource. In contrary, soil, a non-renewable resource, is not considered by the CEENE method. The resource footprint of the fossil-based products is, not surprisingly, dominated by fossil resources; their renewable resource use fraction is accordingly low. Considering only the non-renewable fraction of the CEENE v2013 resource footprint, the bio-based products in our case study have lower footprints than their fossil-based counterparts. Note that the results presented in Table 5.1 are based on case studies and, thus, not representative for the average produced milk in Flanders, nor for the average bio- or fossil-based electricity and PVC. Nevertheless, the primary conclusions are expected to be generally valid.

Mineral and metal resources have very low contributions to the overall resource footprint of all products in Table 5.1. The share of these resources in the overall resource footprint can be more substantial in case of building materials. Dewulf et al. (2007a) calculated an average share of 6 and 7% for mineral and metal resources, respectively, in the CEENE v2007 resource footprint of building materials. The contribution of atmospheric resources was equal to zero for all products in Table 5.1. When oxygen, nitrogen and carbon dioxide in the air are used by biological systems, they are assigned a zero exergy value because their concentration in the air is chosen as a reference by Morris and Szargut (1986). But, when argon, present in the air at 0.9%, is industrially produced by fractional distillation of liquid air, it is assigned an exergy value different from zero. Water resources have a small contribution to the overall resource footprint of all products in Table 5.1. Their share can be more substantial, for example in aquaculture of Pangasius fish in Vietnam water resources contributed to 31% of the CEENE v2007 resource footprint (Huysveld et al., 2013). The difference between the CEENE v2007 and CEENE v2013 resource footprints is situated in their accounting approach for land resources in human-made systems. CEENE v2007 uses the exergy content of the solar radiation that can be metabolized through photosynthesis by natural ecosystems as a proxy for land occupation, because this solar exergy is no longer available to nature. CEENE v2013 accounts for the exergy content of the potential natural net primary production (NPP) of that land, which is a better proxy for the resource value of land, because in addition to solar radiation other local conditions such as temperature, water availability and soil type are taken into account (Alvarenga et al., 2013c). Because the CEENE v2013 land use characterization factors are generally lower than those of CEENE v2007, the CEENE v2013 resource footprint is lower than the CEENE v2007 resource footprint due to a decreased land footprint.

Because feed was identified as by far the most resource-demanding input of the dairy farm studied in Chapter 3, the feed supply chain's CEENE v2013 resource footprint was calculated for a larger population of 103 dairy farms in Chapter 4. The feed supply chain's resource footprints were subsequently integrated in frontier analysis, a methodology based on economic production theory. This integration allowed investigation of economic-exergetic win-wins, i.e. whether feed costs and overall natural resource use in the feed supply chain could simultaneously be reduced without reducing farm revenues. In this analysis, revenues from milk and meat (animals awaiting slaughter) were considered as a combined output that had to be maintained. Based on the data of the dairy farm population, frontier methods construct a 'best practice' efficiency frontier, representing how feed inputs can together be used most efficiently. How efficiently they are used, compared to the frontier, is expressed by a technical efficiency score. The frontier envelops the dairy farm population and the less technical efficient a farm is, the further it is located from that frontier. Three commonly used frontier approaches were applied to the same dataset in Chapter 4. Overall, the results showed that for almost all farms cost and overall natural resource savings could simultaneously be made. These improvements could mainly be obtained by increasing the technical efficiency (proportionally minimizing both feed inputs), rather than by substituting the feed inputs (kilograms of purchased concentrates and by-products versus costs for on-farm produced roughages) in cost or CEENE minimizing proportions. The optimal allocation of feed inputs was reflected from both a cost and a CEENE allocative efficiency viewpoint. Increasing both technical and cost or CEENE allocative efficiency led to the maximum achievable savings in terms of feed costs or overall natural resource use of the feed supply chain, respectively. While increasing technical efficiency always led to an economic-exergetic win-win, not all farms could achieve an economic-exergetic win-win by input substitution. When the implied substitution to reduce costs was opposite to the implied substitution to reduce CEENE, an economic-exergetic trade-off occurred. Whether an economic-exergetic win-win could be achieved by substitution was farm-specific, and depended on (i) the input proportion that a specific farm was using, (ii) the farm-specific prices and CEENE coefficients of the inputs, and (iii) the shape of the constructed frontier, which in turn depended on the applied frontier method. Although frontier analysis was very suitable to analyse farm-specific win-wins and trade-offs, further research in correctly constructing the frontier is needed, because it influenced the quantified improvement margins and the diagnosis of win-win and trade-off situations. The frontier methodology, therefore, still has to take some substantial steps in further methodological development in order to be reliable for farm-specific decision support. While this methodological development is in progress, the reliability problem of frontier analysis could partially be overcome by KPI analysis and consulting farm advisors and other experts for validation of the results (see further on section 5.2).

Based on the calculated feed supply chain's CEENE v2013 resource footprints for a larger population of dairy farms in Chapter 4, a multiple linear regression model was built to determine the main variables that explain the variation of the annual feed supply chain's CEENE v2013 resource footprint (MJ_{ex} per year) of specialized dairy farms in Flanders (see Supplementary material D1 in Appendix D). A dataset with 31 candidate predictor variables was established for which data of 103 specialized dairy farms were retrieved from their farm accountancy files for a one-year period in 2010-2011. The dataset was randomly split in a training dataset of 75 farms and a validation dataset of the remaining 28 farms. In building the regression model, a balance was sought between model complexity, i.e. the number of predictor variables, and the accuracy and precision of the prediction. Starting from a first regression model with seven predictor variables, a model with five predictor variables was concluded to be the best balance between providing high reliability (validation $R^2 = 0.976$, n = 28) and reducing model complexity:

$$annual\ feed\ supply\ chain's\ CEENE\ v2013\ resource\ footprint\ of$$
$$specialized\ dairy\ farms\ in\ Flanders\ (MJ_{ex}\ per\ year)$$

$$= -62240.651 + 322026.050 \times L + 36.755 \times C_{s\text{-}d} + 42.435 \times C_{m\text{-}d} + 23.440 \times BP_d + 15.107 \times R$$

with

L: available on-farm land for feed production (ha)

$C_{s\text{-}d}$: total amount of concentrates based on a single ingredient (e.g. soybean meal) fed to dairy cows (kg)

$C_{m\text{-}d}$: total amount of mixed concentrates (e.g. high-protein compound concentrate) fed to dairy cows (kg)

BP_d: total amount of by-products fed to dairy cows (kg dry matter)

R: purchased quantity of roughages corrected for roughage stock changes (kg dry matter)

According to this regression model, the annual feed supply chain's CEENE v2013 resource footprint (MJ_{ex} per year) of specialized dairy farms in Flanders can be assessed without the knowledge of data about feed for young cattle and inputs for on-farm roughage production other than land use. This regression model makes data about feed for young cattle and inputs for on-farm roughage production different from land use unnecessary. Besides identifying the main variables that explain the variation of the annual feed supply chain's CEENE v2013 resource footprint of specialized dairy farms in Flanders, the regression model enables to simplify the calculation of this resource footprint in the future, for those cases in which a certain degree of simplification could be justified. Of the included predictor variables, data collection for the variable 'purchased quantity of roughages corrected for roughage stock changes (R)' required relatively extra effort compared to the other variables. A model with only four predictor variables, excluding the last variable, still has an acceptable reliability (validation R^2 = 0.964, n = 28), and could therefore also be used (see Supplementary material D1 in Appendix D). Note, however, that the representativeness of regression models for resource footprints of feed supply chains, in terms of both time and location, could be limited. For example, the model coefficient for the variable 'total amount of mixed concentrates fed to dairy cows' could change in time, because the composition of the mixed concentrates changes over time due to market effects; consequently, the resource footprint of the mixed concentrates also changes over time (see Figure 3.6 in Chapter 3). Regarding location, representativeness may be limited because feed use is region-specific; the considered feeds in building the model were specific to dairy farms in Flanders.

With respect to resource footprinting, it can be concluded that the CEENE method provides a very comprehensive view on natural resource consumption along the life cycle of a product. The CEENE method allows you to identify hotspots of resource consumption along the product life cycle for seven (eight, including atmospheric resources) different resource types (see Figures 5.1 and 5.2 for the case study of a dairy farm). Across different product groups, the CEENE method allows you to identify the most important natural resources on which a particular type of product relies (cfr. Figure 3.4 in Chapter 3). Besides being comprehensive, the CEENE method accounts for natural resources in a consistent and scientifically-sound way. In combination with the CEENE method, the application of exergy analysis of processes in the foreground system is very useful because it allows you to identify the main causes of inefficient resource transformation in the core of the studied system and hence to search for improvements in terms of resource efficiency.

	Land resources	Water resources	Metal resources	Mineral resources
On-farm roughage production	63%	12%	73%	20%
- Land area	62%	0%	0%	0%
- Fuel	<1%	1%	<1%	1%
- Seed	1%	<1%	<1%	1%
- Mineral fertilisers	<1%	3%	11%	11%
- Pesticides	<1%	<1%	<1%	<1%
- Groundwater for spraying pesticides	0%	<1%	0%	0%
- Agricultural machinery	<1%	7%	61%	7%
Feed purchased	34%	31%	20%	70%
- Maize silage	4%	<1%	2%	1%
- Concentrates	24%	13%	10%	41%
- Wet by-products	6%	13%	5%	17%
- Others	<1%	4%	4%	11%
Other inputs of dairy production	3%	58%	6%	11%
- Land area of stables, sheds, etc.	1%	0%	0%	0%
- Energy (electricity and fuel)	<1%	7%	4%	1%
- Groundwater	0%	49%	0%	0%
- Bedding material purchased	2%	<1%	<1%	2%
- Chemicals	<1%	1%	<1%	1%
- Infrastructure: milking parlour	<1%	<1%	2%	6%
Total	100%	100%	100%	100%

Figure 5.1 Representation of the share of the input flows to the dairy farm (see Chapter 3) in the CEENE categories land resources, water resources, metal resources and mineral resources. *Chemicals* include lime, disinfectants and detergents for cleaning. *Others* include milk powder, micronutrients and feed additives.

	Fossil resources	Nuclear resources	Abiotic renewable resources
On-farm roughage production	38%	21%	50%
- Land area	0%	0%	0%
- Fuel	13%	1%	1%
- Seed	<1%	1%	1%
- Mineral fertilisers	12%	3%	7%
- Pesticides	<1%	<1%	<1
- Groundwater for spraying pesticides	0%	0%	0%
- Agricultural machinery	12%	17%	41%
Feed purchased	51%	32%	43%
- Maize silage	1%	1%	2%
- Concentrates	21%	11%	19%
- Wet by-products	27%	18%	18%
- Others	2%	3%	4%
Other inputs of dairy production	11%	47%	7%
- Land area of stables, sheds, etc.	0%	0%	0%
- Energy (electricity and fuel)	10%	46%	6%
- Groundwater	0%	0%	0%
- Bedding material purchased	<1%	<1%	<1
- Chemicals	<1%	1%	1%
- Infrastructure: milking parlour	<1%	<1%	<1
Total	100%	100%	100%

Figure 5.2 Representation of the share of the input flows to the dairy farm (see Chapter 3) in the CEENE categories fossil resources, nuclear resources and abiotic renewable resources. *Chemicals* include lime, disinfectants and detergents for cleaning. *Others* include milk powder, micronutrients and feed additives.

Resource efficiency assessment

Because the exergy accounting methodology enables the quantification of both material and energy inputs and outputs of a process in a common unit, it is particularly suitable for the calculation of an overall resource efficiency. The exergy efficiency of a process reflects how efficient the process overall converts resources into the desired product(s). Losses of exergy can be caused by both process irreversibilities, as a consequence of the second law of thermodynamics, and waste flows. Note the different meaning between efficiencies quantified by frontier analysis (Chapter 4) and the exergy efficiency (Chapters 2 and 3). While the first type of efficiency reflects the distance from the optimum in an existing population, the exergy efficiency reflects the distance from the thermodynamic optimum. Frontier analysis compares efficiencies within a population of similar production systems, whereas exergy analysis can do the same in addition to comparing efficiencies of different production systems. In Chapter 3, exergy analysis of milk production was performed at the level of the cattle herd on a dairy farm. Feed consumption of both dairy cows and young cattle was taken into account, because the renewal of the dairy herd by producing young cattle guarantees continuous milk production. More than half of the resources consumed by the dairy farm's herd was irreversibly lost. The remaining went for almost two-thirds to manure and methane emissions, while only one-third went to milk and animals awaiting slaughter. Milk was produced with an exergy efficiency of 15.2% at herd level. When taking the by-products culled animals and surplus calves into account, the efficiency increased only slightly to 16.1%. When also taking manure into account (also a type of by-product because it is used as a fertilizer), the efficiency increases to 42.0%. This analysis showed that the process of milk production has a rather low efficiency in converting resources into the desired product. The reduction of exergy losses in favour of an increase in milk yield requires a further increase of animal efficiency, which is subject to a biological limit. Besides milk production, the chemical exergy in the animal feed is expended in the biological metabolism (e.g. regulating constant body temperature, excretion of waste products, etc.), movement, growth and reproduction (Blumberg, 2002). Other potential improvement from a resource efficiency viewpoint could be sought in better utilizing the exergy-rich output manure via anaerobic digestion.

Exergy analysis at process level can be extended to the life cycle level in order to calculate a cumulative overall resource efficiency. This efficiency, also called Cumulative Degree of Perfection (CDP), can be calculated by the ratio of the exergy contained in a product to the cumulative exergy consumption of its supply chain. However, because exergy analysis has mainly been elaborated in an industrial context, it was unclear how to account for bio-productive land resources as an input during the quantification of efficiencies. To address this issue, an improved framework, called Cumulative Overall Resource Efficiency Assessment (COREA), was developed in Chapter 2. This framework is based on the CEENE v2007 accounting approach for land resources, but redefines the fraction of the solar surface radiation that has to be taken into account. Although the land use accounting approach in CEENE v2013 is more appropriate than the one of CEENE v2007 for the purpose of resource footprinting, the CEENE v2013 land use accounting approach is not adequate for the purpose of CDP calculation. When using CEENE v2013 to calculate cumulative exergy consumption, CDPs higher than 100% are theoretically achievable, because the actual NPP of agricultural cultivation can be higher than the potential natural NPP at a given location. Because the CDP reflects the distance from the thermodynamic optimum, CDPs higher than 100% are not scientifically sound. In the case of CEENE v2007, which accounts roughly for the upper limit on the gross primary production (GPP) of natural ecosystems (=2% of the solar surface radiation), it was not yet clear whether or not this approach is sufficient to avoid that CDPs higher than 100% are achievable in the context of human-made systems (e.g. agriculture). In Chapter 2, we therefore appealed to photosynthesis research to define the appropriate fraction of the solar radiation that has to be taken into account. Two appropriate fractions were determined: (1) 4.8% is the theoretical maximum efficiency to convert solar surface radiation into harvestable (aboveground) biomass (resulting in the method CEENE $v2007_{TMCA}$) and (2) 2.3% is the global actually observed maximum efficiency to convert solar surface radiation into harvestable (aboveground) biomass (resulting in the method CEENE $v2007_{OMCA}$). The gap between these two references indicates ample room for improvement of crop efficiency without altering the photosynthesis mechanism. In Chapter 2, it was also concluded that, with a status quo of the currently observed maximum efficiency, CDPs higher than 100% are not achievable with the original CEENE v2007 approach (CEENE $v2007_{2\%}$).

In addition to the conversion efficiency of solar radiation, it is, from a resource efficiency point of view, appropriate to account for land use efficiency by the temporal system boundary of primary biomass production. In photosynthesis research, crop efficiencies are generally calculated by accounting for only the surface solar radiation during the growing period of the crop. A distinction, however, should be made between monoculture systems, which usually grow only during a limited period with the most favourable local conditions, and both perennial systems, which grow over several years, and multiple-cropping systems, which tend to grow several crops over a longer period thanks to a well-planned crop rotating system. In Chapter 3, we therefore suggested to account for an entire year of land occupation, which is then fully assigned to one (in case of monoculture or perennial systems) or more crop products (in case of multiple-cropping systems). When calculating the efficiency of each crop product of a multiple-cropping system separately, the seasonal variation of the surface solar radiation should be taken into account in order to reflect the lower production potential of the land during periods with less solar radiation. However, in the case of harvested catch crops, whose function is mainly the reduction of adverse effects on the soil between two main crop cultivations rather than productivity, this approach still might assign a too large proportion of the occupied land to the harvested catch crop. In the context of the efficiency-diversity dilemma, it could, therefore, be more appropriate to calculate efficiency at the level of the entire basket of crop products in case of multiple-cropping systems.

Using the newly developed COREA framework, two bio-based products (electricity and PVC) were compared with their fossil-based counterparts in terms of cumulative overall natural resource efficiency. Both fossil-based products were ranked in favour of their bio-based alternatives. Using CEENE v2007$_{TMCA}$, the bio-based electricity was between 7.7 and 4.6 times less resource efficient than its fossil-based counterpart, depending on the considered temporal boundary. These values dropped to 3.7 and 2.2 times less resource efficient when using CEENE v2007$_{OMCA}$. The bio-based PVC was 10.7 and 5.3 times less resource efficient than its fossil-derived counterpart, when using CEENE v2007$_{TMCA}$ and CEENE v2007$_{OMCA}$, respectively. These results, however, overlooked the ancient solar energy use during formation of fossil resources. The COREA framework, therefore, was extended by including this ancient solar energy consumption (resulting in the methods $CEENE_{COREA(TMCA)}$ and $CEENE_{COREA(OMCA)}$). The effect of this alternative accounting approach for fossil resources on the comparison of the bio-based products with their fossil-based counterparts was large. Using $CEENE_{COREA(TMCA)}$, the fossil-based electricity was between 18.6 and 15.6 times less resource efficient than the bio-based product, depending on the considered temporal boundary. Similar values were obtained using $CEENE_{COREA(OMCA)}$. The fossil-based PVC was about 3.5 times less resource efficient than the bio-based alternative, when using both $CEENE_{COREA(TMCA)}$ and $CEENE_{COREA(OMCA)}$. These results confirm that accounting for the ancient solar energy consumption of fossil fuels definitely reflects their non-renewability.

The newly developed COREA framework can be compared with other resource-oriented indicators that have the purpose to assess the potential benefits of bio-based products compared to their fossil-based counterparts. One example is the calculation of the *energy output/input ratio* for biofuels, which is defined as the ratio between the energy value of the biofuel and the non-renewable energy input (often only the fossil energy input) (von Blottnitz and Curran, 2007). Compared to the cumulative overall resource efficiency indicator of the COREA framework, this ratio does not take into account non-energetic resources such as land, water, metals and minerals. Another example is the metric *land use efficiency* (expressed in GJ/ha), which is defined as the ratio between the savings in non-renewable energy use and the additional land use of a bio-based product compared to its fossil-based alternative (Pawelzik et al., 2013). Although this metric addresses land use, it does not (i) spatially differentiate in terms of land use and (ii) take into account non-energetic natural resources, i.e. metals, minerals and water. Application of the COREA framework in the casestudy of bio-based PVC, for example, shows that it can be important to address a wide range of natural resources: compared to fossil-based PVC, bio-based PVC resulted into savings of fossil resources (70%) and nuclear energy (45%), however, at the expense of additional use of land resources (with a factor 450), metals (with a factor 6), minerals (with a factor 2) and water (with a factor 4). In the casestudy of bio-based electricity, savings of fossil resources (90%) occurred at the expense of additional use of land resources (with a factor 2060; taking into account the actual cultivation period of the crops), nuclear energy (with a factor 16), renewable energy (with a factor 4) and metals (with a factor 4). Furthermore, in contrary to the cumulative overall resource efficiency indicator, the metric *land use efficiency* does not reflect the actual efficiency of the conversion of natural resources into products. One of the key advantages of the cumulative overall resource efficiency indicator is its distance-to-target approach: it allows one to measure the distance reduction from the potential optimum in biomass yield that can be achieved by human intervention.

5.1.2 Critical view on the exergy accounting methodology and suggestions for further research

One of the added values of the exergy accounting methodology is situated in accounting for *overall* natural resource use and calculating *overall* natural resource efficiency, based on objective thermodynamic laws. The main advantage of having an overall value, however, has also drawbacks. Because different types of resources are quantified in a common unit and added up to a total value, detailed information is lost after this aggregation. The CEENE resource footprint of products with an agricultural supply chain is generally dominated by the category land resources. When the ancient solar energy consumption of fossil resources was taken into account in Chapter 2, the contribution of the category fossil resources even became larger than the share of the category land resources and, thus, dominated the overall resource use. Changes to other resource categories with a small contribution to the overall resource use therefore become (nearly) invisible in the overall value. Dominant resource categories imply the risk of not being aware about possible trade-offs between different resource categories. In Chapter 3, the average CEENE v2013 resource footprint of three different types of feeds (roughages, concentrates and wet by-products) were compared in terms of the different CEENE categories as well as in terms of the overall CEENE value. For example, although wet by-products were 34% less resource-intensive than roughages on the basis of the overall CEENE value, wet by-products required more resources per kg dry matter for the categories fossil, nuclear, water, mineral and abiotic renewable energy resources. In Chapter 4, trade-offs between different CEENE resource categories were identified when investigating the substitution between purchased feed and on-farm produced feed. Furthermore, an overall CEENE value does not discriminate between renewable and non-renewable resource use. These drawbacks can be addressed by reporting the results for the separate resource categories in addition to the overall resource use in order to identify possible trade-offs. And so, the constructed regression model (Supplementary material D1 in Appendix D) to predict the feed supply chain's CEENE v2013 resource footprint of specialized dairy farms in Flanders could also be built for the separate CEENE resource categories.

Another critical point is the use of exergy to account for non-energetic resources like water, minerals and metals. Natural resources are very diverse and they have certain values based on different characteristics. By using exergy, they are quantified based on their ability to perform work, or in other words, based on their disequilibrium with the reference environment. For example, the chemical exergy of liquid water is 0.05 MJ_{ex} per kg, which is determined by the concentration of water vapour in the ambient air (see Supplementary material D2 in Appendix D). For comparison, the chemical exergy of crude oil is 46.22 MJ_{ex} per kg. By using exergy, substances are quantified from a useful energy viewpoint, although it is doubtful that this properly reflects their main value in the case of non-energetic resources like water, minerals and metals. As a consequence, their contributions in the overall CEENE resource footprint are usually rather small. However, evaluations can be made separately for each resource category according to the philosophy 'less is better'. Material Flow Analysis (MFA) (Spangenberg et al., 1999) is another resource accounting method, which takes into account (only) material resources, but in that case 1 kg of sand is similar to 1 kg of water. Like the CEENE method, the Solar Energy Demand (SED) method (Rugani et al., 2011) takes into account non-energetic resources in addition to energetic resources. Whereas the contribution of water is generally also very low in the overall SED result (see for example Figure 2.2 in Chapter 2), the non-renewable resource categories minerals and metals, together with fossil resources, often dominate SED results of agricultural products.

Exergy methods are classified in the group of resource accounting methods because they sum up different types of resources in a common unit. They do not provide information on resource depletion or the local scarcity of a resource. Potential further development could be situated in developing a resource depletion variant of the CEENE method. The current CEENE characterization factors could be multiplied with use-to-availability ratios in case of the non-renewable resources. A major difficulty, however, is a good estimation of the actual amount and quality of available stocks. Furthermore, spatially-differentiated characterization factors could be implemented for those resources for which local scarcity may be an issue, e.g. water. Blue water consumption, quantified in exergy terms, could be multiplied with the water stress index (WSI) developed by Pfister et al. (2009). For land resources, it is possible to quantify the depletion of primary biotic resources delivered by the land, by accounting for the loss of net primary production (NPP) between a reference natural state and the current land use (Alvarenga et al., 2015). This result can be positive or negative. This way of accounting for biotic resource depletion has an ecocentric approach because the potential natural NPP is used as a reference. An anthropocentric approach could use attainable agricultural yields as a reference, taking into account local constraints for optimal primary biomass production such as temperature, water availability, soil conditions and terrain characteristics. Using this approach, the result should always be positive (or equal to zero). To estimate attainable agricultural yields under several local constraints and under different climate scenarios, various models were integrated in the Global Agro-Ecological Zones (GAEZ) framework, developed by IIASA/FAO (2012). This framework could be a starting point to develop spatially-differentiated characterization factors for land resources in the resource depletion variant of the CEENE method. Furthermore, when calculating a cumulative overall resource efficiency (Chapter 2), additional information about local constraints could be used to define a more practical boundary for optimal primary production.

When discussing the value of the exergy accounting methodology within environmental sustainability assessment, a critical positioning of the exergy accounting methodology against other methods is necessary. An important group of life cycle impact assessment methods focuses on the evaluation of emissions-related impacts (e.g. global warming, eutrophication, acidification, etc.). Although it is possible to quantify emissions in exergy terms or to quantify the exergy use in abatement processes of emissions, the exergy-based approach cannot properly reflect the environmental impact of emissions. Combining the application of exergy-based methods with the application of emission-oriented approaches is recommended. Furthermore, there is now a large consensus about the need to include impacts on biodiversity in environmental sustainability assessment of agricultural production systems. Impacts on biodiversity are very complex to assess, because they are linked to both resource use and emissions. The application of conventional exergy-based resource accounting should be seen especially in the area of protection (AoP) 'natural resources'. Recently, however, Taelman et al. (2016) proposed two exergy-based indicators, based on the loss of NPP due to land use, as a good starting point for determining the possible impact land use can have on biodiversity within the AoP natural environment.

The variety of environmental issues cannot be assessed using one single indicator. Trade-offs may occur between resource efficiency and other environmental impacts. For example, when comparing the environmental performance of extensive or semi-intensive systems against intensive systems, the latter generally have a higher land use efficiency, but at the same time they could perform worse in terms of eutrophication (Bava et al., 2014). Research is required that integrates different types of approaches (exergy methods, emission-oriented impact assessment methods, etc.) to achieve a more complete insight into environmental sustainability. It would therefore be interesting to extend the integration of frontier analysis and cumulative overall resource use accounting (Chapter 4) in future research by including other environmental impacts.

Besides the role of resource efficiency within environmental sustainability assessment, potential side-effects of efficiency gains on other aspects of sustainability should be taken into account in overall sustainability assessments. An example is the potential effect of intensification of livestock farms by increasing the number of animals per available area on animal health and welfare. Another present-day example, from a more industrial context, is the development of new digital and automated technologies to increase resource efficiency of firms, which may have negative consequences on employment.

When evaluating environmental sustainability of food systems, exergy methods have some context-specific limitations. The exergy content of a food product does not properly reflect its nutritional characteristics. Also the exergy content does not reflect whether a product is edible by humans or not. In order to account for the competition between food and feed, it is of crucial importance whether feed ingredients are human-edible or not. For example, grass, an important ingredient in the feed ration of ruminants, is not edible by humans, and thus, it does not imply competition between food and feed. In contrary, humans and animals compete for cereals, which have been increasingly included in the feed ration of livestock, especially poultry and pigs, to achieve a higher feed efficiency. Feeding human-edible products to livestock has become a substantial problem, because globally livestock consumes more human-edible protein than it produces (FAO, 2006). Milk production by dairy cows is an exception because they generally consume less feed that is edible by humans (Wilkinson, 2011). By calculating human-edible protein and energy conversion ratios, the environmental sustainability of different animal-derived foods can be compared. These conversion ratios equal the amount of human-edible protein or energy present in the animal feed over the amount of human-edible protein or energy that is present in the animal-derived food. Conversion ratios larger than one are regarded as not sustainable, because it is more efficient to directly consume the human-edible portion of the animal feed. In the context of evaluating different livestock systems, human-edible protein and energy conversion ratios provide valuable information, which cannot be acquired by calculating exergy efficiency.

To evaluate different livestock systems, Van Zanten et al. (2015) went a step further by accounting for differences in land suitability for the cultivation of food crops, because feed that is not edible by humans can still be produced on land that could otherwise be used for food crop cultivation. While feed production on land that is also suitable for the cultivation of food crops implies a competition between food and feed, feed production on land that is not suitable for the cultivation of food crops because of unfavourable soil and climatic conditions does not imply a competition between food and feed. An example of the latter, given by Van Zanten et al. (2015), is peat soil. This type of soil is too wet for competitive food crop production (Van Kernebeek et al., 2015), but it can be used for the cultivation of grass. Based on this distinction, Van Zanten et al. (2015) developed the land use ratio (LUR), which is defined as 'the maximum amount of human-digestible protein (HDP) derived from food crops on all land used to cultivate feed required to produce one kilogram of animal-derived food over the amount of HDP in that one kg animal-derived food'. To calculate the numerator of this ratio, first, the amount of land used to cultivate all the feed that is required to produce one kilogram of animal-derived food has to be quantified. Second, for all that land, the suitability for food crop cultivation is determined based on the suitability index (0-100) of the GAEZ database (IIASA/FAO, 2012). Land with a suitability index lower than 55 was considered not suitable for food crop cultivation. Third, for each area of land suitable for food crop cultivation, the maximum amount of HDP derived from food crops is determined. A LUR smaller than one was considered to be efficient in terms of land use because then animals produce more HDP per unit area than food crops. Van Zanten et al. (2015) concluded that a better LUR is obtained the more livestock systems produce their feed on land unsuitable for food production and the more human-inedible by-products from industries are included in their feed. When evaluating resource use of livestock supply chains, the CEENE method currently does not differentiate for the suitability of land to produce food crops, neither for the fact whether feed ingredients are human-edible or not. To address this limitation in future research, the CEENE v2013 resource footprint of livestock supply chains could be calculated in an additional way. First, in the case of human-inedible feed ingredients that are by-products from food or energy industries, their overall resource use could be excluded. In this way, the CEENE method could better reflect that it is more favourable to utilize these by-products as feed ingredients than wasting them. Second, in the case of feed crop cultivation, only use of land that is suitable for food crop cultivation could be included.

5.2 From research to practice

When performing sustainability assessments, the final purpose is generally to support decision-makers in their decisions towards a more sustainable process, product or society in general. Translating research into practical knowledge that can be used to set up concrete improvement actions, however, should be a final step for researchers. Such a step is not straightforward and is often lacking. In Chapter 4, frontier analysis was combined with analysis of Key Performance Indicators (KPIs) for specialized dairy farms in Flanders in order to reduce the gap between scientific knowledge about potential economic-exergetic win-wins, acquired with frontier analysis, and practical knowledge, based on improvable KPIs that are traditionally used by farmers and their advisors. Integration of frontier analysis and KPI analysis enables to benefit from both approaches. While frontier analysis is particularly suitable to identify farm-specific benchmarks and improvement paths, KPI analysis allows the identification of suboptimal KPIs that can be starting points for exploring possible improvement actions. The added value of combining KPI analysis with frontier analysis is avoiding the direct use of KPIs as benchmarks, because KPIs are only partial benchmarks (e.g. concentrate use in kg per cow, milk production in kg FPCM per cow), which together may form an unrealistic situation; the 10% best farms for one KPI may not be similar to the 10% best farms for another KPI. By using frontier analysis, this limitation is addressed because frontier analysis uses (a linear combination of) actual farms as benchmarks (Van Meensel et al., 2012).

The outcomes of the integration of frontier analysis and KPI analysis (Chapter 4) were presented at meetings with agricultural experts and advisors from the farm advisory company that supplied data of a population of specialized dairy farms for a one-year period in 2010-2011. Important lessons could be learned from these meetings. First, the involvement of practical experts provides additional insights which may have an important influence on the conclusions of the analysis. For example, the KPI replacement rate was significantly higher for farms with high cost and overall natural resource efficiencies. This could suggest that a high replacement rate is required to be cost and resource efficient. Consulting an agricultural expert, however, revealed that the replacement rate is a very complex indicator to grasp and, therefore, less suitable as a univocal performance indicator. Within one farm, the replacement rate can fluctuate sharply from one year to another. Some reasons can be an expansion of the dairy herd or a large number of sick cows. Further analysis of the data confirmed the warning of the agricultural expert; the significantly higher replacement rate was caused by farms with significantly higher percentages of forcedly disposed dairy cows (because of death, health problems or infertility). Excluding these farms, the replacement rate was no longer significantly higher for farms with high cost and overall natural resource efficiencies. During another meeting with farm advisors, however, this information was not acquired, although these advisors could have the best knowledge of the farms under study. Several sessions for discussion and reflection with different types of experts are highly recommended because they lead to the highest knowledge acquisition. The added value of involving practical experts in the validation of decision support systems was also emphasized by other authors (e.g. de Olde et al. (2016), Meul et al. (2009), Van Meensel et al. (2012)). Some authors (e.g. Cain et al. (2003), Van Meensel et al. (2012), Vayssières et al. (2011)) even went a step further by involving stakeholders already from the development phase of a decision support system (DSS), which is called a participatory approach. A more intense cooperation between researchers and intended users of agricultural DSSs could increase their adoption rate, which is currently limited (de Olde et al., 2016; Van Meensel et al., 2012). Success factors for adoption of agricultural DSSs have been identified by multiple authors. Van Meensel et al. (2012) reported flexibility, perceived usefulness, accessibility, credibility, intended users, and maintenance and adaptability as critical success factors. Context specificity, user-friendliness, complexity, language use, and correspondence between value judgements of DSS developers and farmers, were perceived as very important aspects by de Olde et al. (2016). Also, de Olde et al. (2016) emphasized the need for additional efforts to support farmers in using outcomes from research in their decision making. Farm advisors are well suited as intermediaries between researchers and farmers. It is more realistic to make farm advisors familiar with DSSs than farmers themselves, because farm advisors can acquire experience by applying DSSs for multiple farms (Van Meensel et al., 2012).

A second important lesson learned is that additional research efforts are needed to know the effects of improving KPIs on the farm performances (e.g. economic, exergetic, etc.). Knowing which KPIs are suboptimal for a specific farm should not be an endpoint; simulation of the effects of possible actions on the farm performances was perceived as a necessary following research step in knowing how to achieve improvement. Realizing a better farm performance is not as simple as just changing a suboptimal KPI to the required level, because several KPIs are interlinked in a complex way and not all changes can be performed for each farm. The need for simulation was also reported by Van Meensel et al. (2012), who highlighted the important role of farm advisors in this analysis. Farm advisors are expected to be the best qualified persons to have the required knowledge about indirect linkages between KPIs and limiting factors, which both are farm-specific and depend on the simulated improvement action (Van Meensel et al., 2012). In addition to the need for simulation, feedback received during the meetings included the need for analyses over longer time periods in combination with more background information about the farms (e.g. grazing management, breeding type of dairy cows, soil type, etc.) and the acquirement of a higher confidence in the established frontier. When farms could be monitored and analysed over several years, valuable insights could be gained about the evolution of their farm performances in relation to their KPIs. Furthermore, effects of strategic decisions on long-term farm performances could then be analysed.

Finally, it is important to note that, in order to ensure that improvements at the level of individual farms also contribute to overall improvement of the entire sector, sustainability assessments should be carried out at different levels (farm, sector, country, etc.) and by different actors (farm advisors, policy makers, etc.) (Van Passel and Meul, 2012).

5.3 Concluding remarks

Given the outline of this PhD thesis with both a methodological and a thematic focus, some concluding remarks can be made on both aspects. We start with the thematic focus areas of this PhD thesis. In the context of the transition towards a sustainable bioeconomy, the first thematic focus, there is an increasing demand for biomass that has to be produced in a sustainable way. Because bio-productive land is globally limited and because of the competition for land between food, feed, biomaterials and bioenergy, meeting the rising demand for sustainably produced biomass is a major challenge. Furthermore, the increasing world population and rising living standards in developing countries also demand more and more area for infrastructure, industry and housing, which will put extra pressure on the globally available land area. To meet the increasing biomass demand when agricultural expansion into natural habitats is to be avoided, increasing biomass yields should be done with caution to prevent damage to the natural environment by the production of pollutant emissions and to safeguard long-term productivity of the soil. Hereto, context-specificity is very important: while in some regions sustainable intensification could close yield gaps, in other regions the sustainable threshold has already been reached or even exceeded.

Given the preferred cascading order of biomass use (food-feed-biomaterial-bioenergy), more research efforts are needed to improve the potential of the full range of abiotic renewable energy sources to meet the demand for energy. With respect to solar energy, this renewable energy source can be utilized much more efficiently, compared to photosynthesis, using photovoltaics (Williams et al., 2015). Due to the intermittent and variable nature of renewable energy sources like solar and wind energy, however, there is a need to develop cost efficient storage technologies in order to ensure a reliable energy supply. Besides storage solutions, management of energy use at the demand side could provide part of the solution, by better aligning energy consumption and production (European Commission, 2011b). Both topics can be addressed by smart grids. In contrast to solar and wind energy, geothermal resources could provide constant power and heat, and, therefore, they have potential to supply base-load electricity, when technical and economic barriers would be overcome (Sigfússon and Uihlein, 2015).

Additionally, the potential of biomass sources different from terrestrial primary biomass should be further investigated in order to meet the rising demand for biomass. Aquatic biomass production is promising, because aquatic plants generally are more efficient in converting solar energy into biomass, because of a less complex cellular structure (Taelman et al., 2015a). The process of drying aquatic biomass, which is performed to conserve the biomass for a longer time period, however, appears to be a major bottleneck in achieving a resource efficient production process, because it has a high energy consumption (Taelman et al., 2015b).

In the context of livestock production, the second thematic focus, a concluding remark is that the environmental sustainability highly depends on the production and conversion of feed. Intrinsically, it is obviously less resource efficient to consume animal-derived food products instead of directly consuming plant-based food products, because an extra trophic level is included in the food production chain. Exceptions may exist when livestock production relies on the conversion of human-inedible plants, which were not produced on land suitable for competitive food crop production or which are by-products from food and energy industries (van Zanten et al., 2015). A wide-scale adoption of this resource efficient way of livestock production, however, would require a major reversal of how livestock production is nowadays performed on a large scale in the world. The major yield increases achieved during the last century were partially built upon feeding ingredients of increasing quality. The main purpose of livestock production has evolved from managing human-inedible flows, and thereby creating the benefit of producing nutritious animal food products, into producing animal food products while continuously striving for higher feed efficiencies for which the quality of feed has been increased substantially. This way of modern demand-driven production, however, contrasts sharply with smallholders in developing regions who still perform livestock production in a traditional supply-driven way, by feeding mainly waste and other low-value biomass sources, and who depend on livestock to survive (Gerber et al., 2013). This contradiction between developed and developing regions again shows that sustainable development is highly context-specific. While increasing food security is the key priority in many developing regions, changing consumer behaviour towards consuming less animal food products may be the sole way towards a sustainable food system in developed regions.

Another concluding remark is that resource efficiency improvements in animal food production could also be sought in breeding animal species with higher feed efficiencies. Mammals and birds are endothermic species, which regulate their body temperature through metabolic regulations, such as respiration. In contrary, fish and insects, are mostly ectothermic species, which rely on external heat sources for regulating their body temperature. As a consequence, endothermic species have lower feed efficiencies than ectothermic species (Blumberg, 2002).

With respect to the main methodological focus of this PhD thesis, it can be concluded that the added value of exergy methods lies in (i) showing how efficient a system utilizes useful energy by exergy efficiency assessment and (ii) providing insight into the overall resource requirements of this system by resource footprinting, rather than properly evaluating the impact of a system on the environment. To do the latter, a large and diverse number of impact assessment methods were developed in the last decade. Although many required methodological improvements could be discussed, two methodological aspects that need further attention and development are highlighted here. First, because of the diversity of environmental issues, and the trade-offs that may occur between them, it remains a major challenge to identify concrete improvement paths. More research efforts are needed to develop new and improve existing methods like frontier analysis in order to support decision-making and, so, reduce the gap between research and practice. Second, environmental sustainability assessments typically are relative: they compare the environmental performance of contrasting production systems that produce the same product (e.g. organic versus conventional dairy farms) or different products with similar functions (e.g. beef versus pork) in order to identify the best alternative. Although these comparisons are valuable towards better environmental performances of individual systems or even an entire sector, these achieved improvements may be insufficient to avoid exceeding the carrying capacity of the Earth, which can be regarded as a precondition for sustainability. Human exploitation of natural resources should not occur at a rate beyond the Earth's carrying capacity and human activities should not produce pollutant emissions beyond the ability of the Earth to absorb their harmful effects. The large and still increasing world population and the already high living standard in developed regions and its projected increase in developing regions calls to bring the attention more and more to the planetary boundaries. There is a need for LCA approaches that focus on targets that should be achieved or should not be exceeded in order to stay within the planetary boundaries. In this context, the work of Van Kernebeek et al. (2015) is interesting, because it determined the optimal proportion, from a land-use perspective, of dietary animal protein in the human diet. This was done by developing a land use optimization model for the Netherlands as a case study, taking into account population size, land availability and quality. In order to assess exceedance of planetary boundaries in terms of land use, this work could be extended to the global level. Other developments to assess exceedance of planetary boundaries are situated in the normalization step of LCA. Bjorn and Hauschild (2015) developed carrying capacity-based normalization references in order to assess, per impact category, whether current per capita emissions or resource uses exceed the global or regional carrying capacity.

Figure 5.3 presents an overview of the general discussion in Chapter 5.

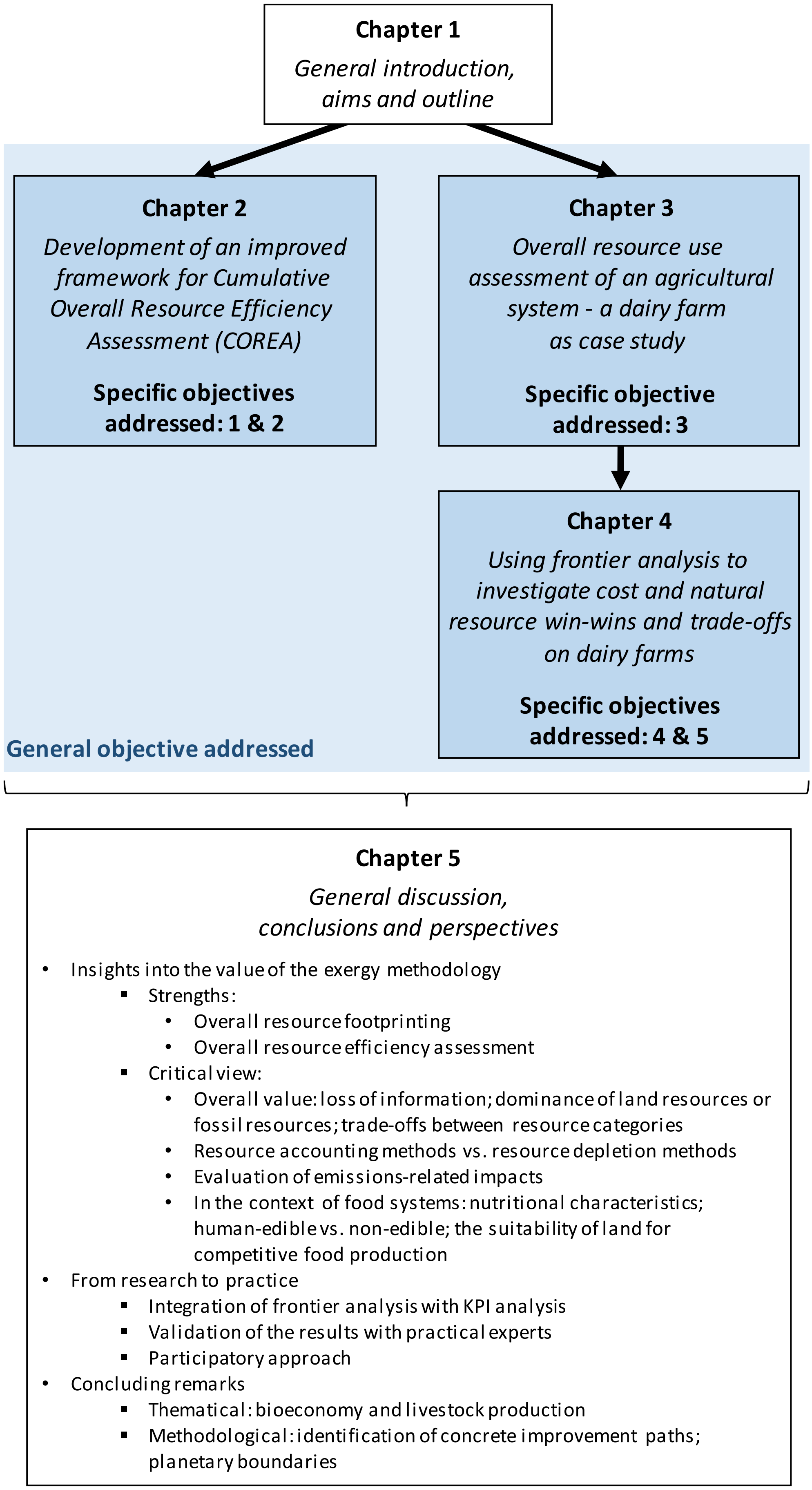


Figure 5.3 Overview of the general discussion in Chapter 5

APPENDIX A: SUPPLEMENTARY MATERIAL CHAPTER 2

Supplementary material A1: Accounting for the ancient solar energy consumption of fossil resources: characterization factors

Supplementary material A2: Data inventory silage maize, silage rye and sugarcane

Supplementary material A3: Relative contributions of the different resource categories to the five available thermodynamics-based resource indicators (CED, CExD, CEENE v2007$_{2\%}$, CEENE v2013 and SED) for the crops in both case studies, when considering the actual cultivation period as temporal system boundary

Supplementary material A4: The share of the foreground primary biomass production stage in the resource consumption of the entire supply chain of the bio-based products

Supplementary material A5: Accounting for the ancient solar energy consumption of fossil resources: results

Supplementary material A1: Accounting for the ancient solar energy consumption of fossil resources: characterization factors

Table A.1 Recovery factors of different fuels (Source: Dukes (2003)).

Fuel type	Recovery factor
Gas	0.000084
Oil	0.000093
Hard coal	0.074
Brown coal	0.103
Peat	0.156

Table A.2 Carbon content (%) of different fuels.

Fuel type	Carbon content (%)
Gas (density 0.84 kg/m^3)	75
Oil	85
Hard coal	85
Brown coal	68
Peat	55

Table A.3 Calculation of characterization factors (CFs) for each fuel type (Based on: Dukes (2003)).

Fuel type	Gas	Oil	Hard coal	Brown coal	Peat
kg C biomass per kg C fuel	11904.8	10752.7	13.5	9.7	6.4
kg plant matter per kg C fuel	26455.0	23894.9	29.9	21.6	14.2
MJ plant matter per kg C fuel	529100.5	477897.3	598.4	432.4	284.9
MJ PAR per kg C fuel	31123560.5	28111603.1	35200.3	25432.6	16758.8
MJ total solar energy per kg C fuel	62247121.1	56223206.1	70400.5	50865.3	33517.7
MJ solar energy per kg C fuel (TMCA/ OMCA)	3174603.2 (TMCA) 1493930.9 (OMCA)	2867383.5 (TMCA) 1349356.9 (OMCA)	3590.4 (TMCA) 1689.6 (OMCA)	2594.1 (TMCA) 1220.8 (OMCA)	1709.4 (TMCA) 804.4 (OMCA)
MJ solar energy per kg fuel	2380952.4 (TMCA) 1120448.2 (OMCA)	2437276.0 (TMCA) 1146953.4 (OCMA)	3033.9 (TMCA) 1427.7 (OMCA)	1751.0 (TMCA) 824.0 (OMCA)	940.2 (TMCA) 442.4 (OMCA)
MJ solar exergy per kg fuel	2220714.3 (TMCA) 1045042.0 (OMCA)	2273247.3 (TMCA) 1069763.4 (OMCA)	2829.7 (TMCA) 1331.6 (OMCA)	1633.2 (TMCA) 768.6 (OMCA)	876.9 (TMCA) 412.7 (OMCA)
MJ solar exergy per m³ fuel	1865400.0 (TMCA) 877835.3 (OMCA)	n/a	n/a	n/a	n/a

The example of gas (TMCA):

$$\frac{1\ kg\ C\ biomass}{0.000084\ kg\ C\ fuel} \times \frac{1\ kg\ plant\ matter}{0.45\ kg\ C\ biomass} \times \frac{20\ MJ\ plant\ matter}{1\ kg\ plant\ matter}$$

$$\times \frac{1\ MJ\ PAR\ radiation}{0.017\ MJ\ plant\ matter} \times \frac{2\ MJ\ total\ solar\ energy}{1\ MJ\ PAR\ radiation}$$

$$\times \frac{4.8\%\ (\varepsilon_{total(TMCA)})}{95.0\%\ (\varepsilon_{total(TOT)})} \times \frac{0.75\ kg\ C\ fuel}{1\ kg\ fuel} \times \frac{0.9327\ MJ\ solar\ exergy}{1\ MJ\ solar\ energy}$$

$$\times \frac{0.84\ kg\ gas}{1\ m^3\ gas} = \mathbf{1865400.0\ MJ\ solar\ exergy\ per\ m^3\ gas}$$

Supplementary material A2: Data inventory silage maize, silage rye and sugarcane

Inventory silage maize production, Germany

Table A.4 Life cycle inventory data (LCI) of 1000 kg of silage maize produced in Germany.

	Data	Unit	Source
Land occupation (considering an entire year of cultivation)	202.19	m^2a	farmer and experts (Personal communication with Steven De Meester (De Meester et al., 2012))
Diesel consumption	1.64	kg	farmer and experts (Personal communication with Steven De Meester (De Meester et al., 2012))
Seed	0.52	kg	farmer and experts (Personal communication with Steven De Meester (De Meester et al., 2012))
Pesticide (nitrile-compounds)	1.52	g	farmer and experts (Personal communication with Steven De Meester (De Meester et al., 2012))
Pesticide (metolachlor)	16.68	g	farmer and experts (Personal communication with Steven De Meester (De Meester et al., 2012))
Pesticide (triazine-compounds)	10.01	g	farmer and experts (Personal communication with Steven De Meester (De Meester et al., 2012))
Pesticide (diuron)	0.81	g	farmer and experts (Personal communication with Steven De Meester (De Meester et al., 2012))
Pesticide (others)	30.36	g	farmer and experts (Personal communication with Steven De Meester (De Meester et al., 2012))
Water (for spraying pesticides)	9.03	kg	farmer and experts (Personal communication with Steven De Meester (De Meester et al., 2012))
Mineral fertilizer (diammonium phosphate)	2.53	kg	farmer and experts (Personal communication with Steven De Meester (De Meester et al., 2012))
Digestate (15% dry matter)	619.90	kg	farmer and experts (Personal communication with Steven De Meester (De Meester et al., 2012))
Machinery for sowing	2.02E-02	ha	calculation based on personal communication with Steven De Meester (De Meester et al., 2012) and *ecoinvent* v2.2 (*silage maize IP, at farm* (CH))
Machinery for application of plant protection products, by field sprayer	2.02E-02	ha	calculation based on personal communication with Steven De Meester (De Meester et al., 2012) and *ecoinvent* v2.2 (*silage maize IP, at farm* (CH))
Machinery for fertilising, by broadcaster	4.04E-02	ha	calculation based on personal communication with Steven De Meester (De Meester et al., 2012) and *ecoinvent* v2.2 (*silage maize IP, at farm* (CH))
Machinery for tillage, harrowing, by spring tine harrow	2.02E-02	ha	calculation based on personal communication with Steven De Meester (De Meester et al., 2012) and *ecoinvent* v2.2 (*silage maize IP, at farm* (CH))

Machinery for tillage, ploughing	2.04E-02	ha	calculation based on personal communication with Steven De Meester (De Meester et al., 2012) and *ecoinvent* v2.2 (*silage maize IP, at farm* (CH))
Machinery for chopping	2.02E-02	ha	calculation based on personal communication with Steven De Meester (De Meester et al., 2012) and *ecoinvent* v2.2 (*silage maize IP, at farm* (CH))
Silage maize (35.9% dry matter)	1000.00	kg	farmer and experts (Personal communication with Steven De Meester (De Meester et al., 2012))

Inventory silage rye production, Germany

De Meester et al. (2012) retrieved the inventory data of rye from the *ecoinvent v2.2* database (‘*rye IP, at farm (CH)’*). As this *ecoinvent* process deals with rye cultivation for the purpose of grains, we modified these data in order to better reflect the production of rye silage. An overview of our main modifications:

- we assigned all input flows to the multi-output *ecoinvent* process of rye cultivation (‘*rye grains IP, at farm (CH)’ and ‘rye straw IP, at farm (CH)’*) to the production of rye silage. The dry matter yield of rye silage was set equal to the total dry matter yield of rye grains and rye straw, which amounted to 10545 kg per ha (6334 kg grains and 4211 kg straw). Considering a dry matter content of 40.3% for rye silage, we calculated a fresh matter yield of 26186 kg rye silage per ha.
- the cultivation period of rye for the purpose of grains (84% dry matter) is longer than the cultivation period of rye for the purpose of silage (40.3% dry matter). We reduced the period of land occupation from 314 days (from September 25 until August 5) in the case of rye grains to 264 days (from September 25 until June 15) in the case of rye silage.
- the input flow ‘energy, gross calorific value, in biomass’ was set equal to 7.23 MJ per kg fresh rye silage. This input flow is used by the CED and CExD methods.
- the input flow ‘grain drying, low temperature’ was excluded.
- the input flow ‘combine harvesting’ was substituted by ‘chopping, maize’.
- the input flows for transportation (of inputs) were excluded for the purpose of consistency because these flows were also not taken into account in the LCI of silage maize production.

Table A.5 Life cycle inventory data (LCI) of 1000 kg of silage rye produced in Germany.

	Data	Unit	Source
land occupation, arable, non-irrigated (considering an entire year of cultivation)	381.89	m^2a	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
rye seed IP, at regional storehouse	5.35	kg	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
[sulfonyl]urea-compounds, at regional storehouse	51.18	g	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
benzimidazole-compounds, at regional storehouse	13.37	g	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
cyclic N-compounds, at regional storehouse	14.67	g	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
dinitroaniline-compounds, at regional storehouse	16.04	g	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
organophosphorus-compounds, at regional storehouse	12.22	g	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
pesticide unspecified, at regional storehouse	1.53	g	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
ammonium nitrate, as N, at regional storehouse	1.45	kg	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
ammonium sulphate, as N, at regional storehouse	0.11	kg	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
calcium ammonium nitrate, as N, at regional storehouse	0.73	kg	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
diammonium phosphate, as N, at regional storehouse	0.27	kg	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
diammonium phosphate, as P2O5, at regional storehouse	0.69	kg	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
potassium chloride, as K2O, at regional storehouse	1.63	kg	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
potassium sulphate, as K2O, at regional storehouse	0.11	kg	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
single superphosphate, as P2O5, at regional storehouse	0.04	kg	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
phosphate rock, as P2O5, beneficiated, dry, at plant	0.59	kg	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
thomas meal, as P2O5, at regional storehouse	0.13	kg	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))

triple superphosphate, as P2O5, at regional storehouse	1.01	kg	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
urea, as N, at regional storehouse	0.51	kg	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
machinery and fuel for sowing	3.82E-02	ha	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
machinery and fuel for application of plant protection products, by field sprayer	0.10	ha	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
machinery and fuel for fertilising, by broadcaster	0.11	ha	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
machinery and fuel for slurry spreading, by vacuum tanker	0.26	m^3	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
machinery and fuel for solid manure loading and spreading, by hydraulic loader and spreader	70.09	kg	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
machinery and fuel for tillage, cultivating, chiselling	0.04	ha	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
machinery and fuel for tillage, currying, by weeder	0.04	ha	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
machinery and fuel for tillage, harrowing, by spring tine harrow	0.08	ha	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
machinery and fuel for tillage, ploughing	0.04	ha	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
machinery and fuel for chopping	0.04	ha	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))
silage rye (40.3% dry matter)	1000.00	kg	*ecoinvent v2.2* (we modified the process *rye grains IP, at farm* (CH))

Inventory sugarcane production, Brazil (Sao Paulo region) (Alvarenga et al., 2013a)

Table A.6 Life cycle inventory data (LCI) of 1000 kg of sugarcane produced in Brazil.

	Data	Unit	Source
Land occupation	143.39	m^2a	IGBE (2011) and based on Macedo et al. (2008)
Diesel consumption	1.79	kg	Based on: Macedo et al. (2004); Macedo et al. (2008)
N – nitrogen (Urea)	0.71	kg	Based on: Macedo et al. (2004); Macedo et al. (2008)
P_2O_5	0.16	kg	Based on: Macedo et al. (2004); Macedo et al. (2008)
K_2O	0.60	kg	Based on: Macedo et al. (2004); Macedo et al. (2008)
Filter cake	12.00	kg	Based on: Macedo (2005)
Vinasse	1.08	m^3	Based on: Macedo (2005)
Limestone	4.54	kg	Macedo et al. (2008)
Pesticide (atrazine/gliphosate)	31.46	g	Macedo (2005)
Pesticide (carbofuran)	1.61	g	Macedo (2005)
Pesticide (others)	0.57	g	Macedo (2005)
Tractors, harvesters	0.10	kg	Based on: Macedo et al. (2008)
Implements	0.03	kg	Based on: Macedo et al. (2008)
Trucks	0.20	kg	Based on: Macedo et al. (2008)
Cane (taking out "seeds") (32.5% dry matter)	1000.00	kg	Based on: IBGE (2011); Macedo et al. (2004); Macedo et al. (2008)
Trash	129.07	kg	Ronquim 2007
Trash burnt	80.02	kg	Based on: Ronquim (2007)
Trash left of the field	49.05	kg	Based on: Ronquim (2007)
Trash taken to the mill	0.00	kg	Based on: Ronquim (2007); Macedo et al. (2008)

Supplementary material A3: Relative contributions of the different resource categories to the five available thermodynamics-based resource indicators (CED, CExD, CEENE v2007*2%*, CEENE v2013 and SED) for the crops in both case studies, when considering the actual cultivation period as temporal system boundary.

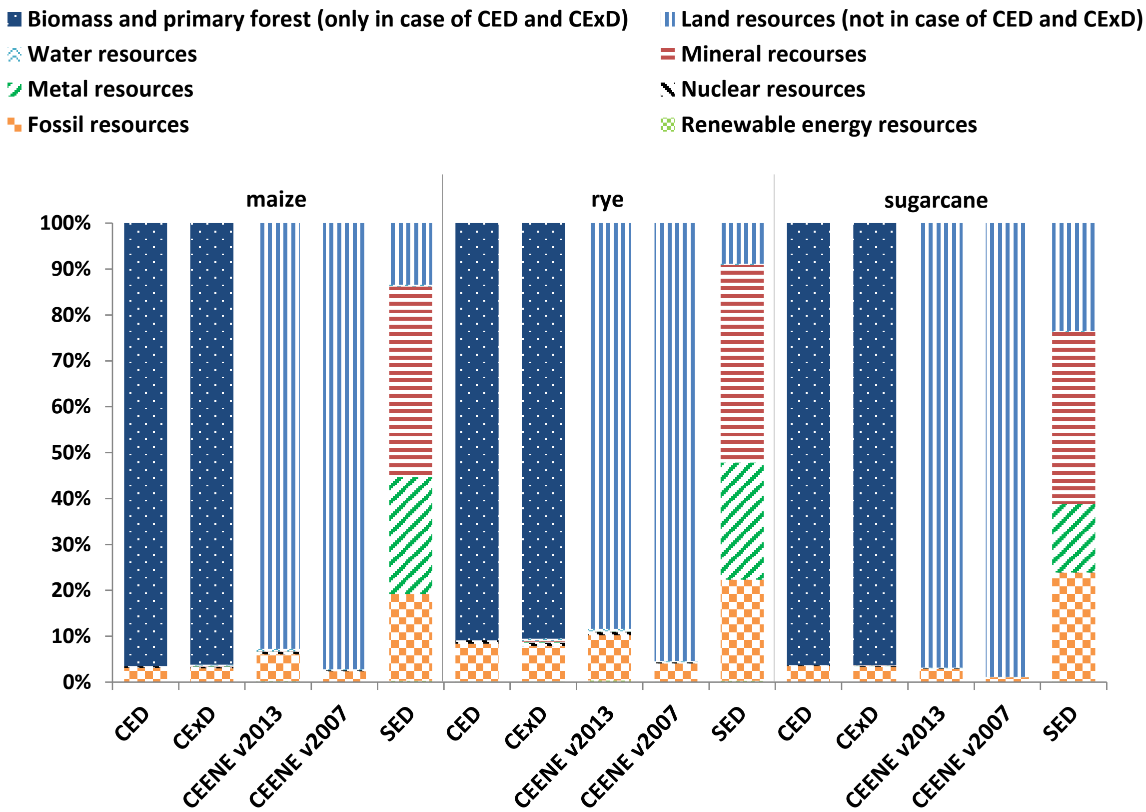


Figure A.1 Relative contributions of the different resource categories to the five available thermodynamics-based resource indicators (CED, CExD, CEENE v2007*2%*, CEENE v2013 and SED) for the crops in both case studies, when considering the actual cultivation period as temporal system boundary. *Renewable energy resources* include hydropower and wind energy in the case of CEENE v2007*2%* and CEENE v2013. In the case of SED, *Renewable energy resources* include hydropower, wind energy and geothermal energy. In the case of CExD and CED, *Renewable energy resources* include hydropower, wind energy and solar energy.

Supplementary material A4: The share of the foreground primary biomass production stage in the resource consumption of the entire supply chain of the bio-based products

Table A.7 shows the share of the foreground primary biomass production stage (silage maize and silage rye in case study 1; sugarcane in case study 2) in the resource consumption of the entire supply chain of the bio-based products. The shares are depicted for each resource category separately and for the total resource consumption. We explain the most remarkable observed similarities and differences between all applied approaches.

Table A.7 visualises that there exist no differences in all resource categories except the land resources category between all CEENE-based approaches (CEENE v2007$_{2\%}$, CEENE v2013, CEENE v2007$_{TOT}$, CEENE v2007$_{PAR}$, CEENE v2007$_{TMC}$, CEENE v2007$_{TMCA}$ and CEENE v2007$_{OMCA}$). Excluding CEENE v2013, we can also see that the share of the foreground primary biomass production stage in the land resources category is similar for all other CEENE-based approaches. This can be explained by the fact that CEENE v2007$_{2\%}$, CEENE v2007$_{TOT}$, CEENE v2007$_{PAR}$, CEENE v2007$_{TMC}$, CEENE v2007$_{TMCA}$ and CEENE v2007$_{OMCA}$ have a similar conceptual approach, i.e. they are all multiplying the surface solar radiation with a different total efficiency (ε_{tot}), while CEENE v2013 is based on the potential natural NPP of the occupied land. Because the share of the land resources category in the total resource consumption is different for each of the CEENE-based approaches, the share of the foreground primary biomass production stage in the total resource consumption is different among these approaches.

Table A.7 also shows that there exist no differences for the nuclear resources category between all approaches. This is due to the fact that all approaches account for one single *ecoinvent* reference flow (*Uranium, in ground*).

Regarding the water resources category, only for the SED method different values can be seen in Table A.7, because the SED method assigns different characterization factors to the five *ecoinvent* water resource reference flows (*Water, cooling, unspecified natural origin*; *Water, lake*; *Water, river*; *Water, unspecified natural origin*; *Water, well, in* *ground*) compared to the other approaches that all assign one single characterization factor of 50 MJ_{ex}/m^3 to these flows.

For case study 2, the share of the foreground primary biomass production stage in the biomass (and primary forest) category is similar for CED and CExD. The reason for this is that the exergy value of the harvested sugarcane biomass was calculated by following the approach of the *ecoinvent v2.2* database (i.e. multiplying the gross calorific value of the biomass by a constant factor of 1.05), because we did not have data on the macronutrient composition of the biomass. In contrast for case study 1, the exergy values of the harvested maize and rye silage were calculated based on their macronutrient composition, and therefore the share of the foreground primary biomass production stage in the biomass (and primary forest) category is different for CED and CExD.

Table A.7 Overview of the share (%) of the foreground primary biomass production stage (silage maize and silage rye in case study 1; sugarcane in case study 2) in the resource consumption of the entire supply chain of the bio-based products (electricity in case study 1; PVC in case study 2). Shares are depicted for each resource category separately and for the total resource consumption. For case study 1, the first values were calculated considering an entire year of land occupation as temporal system boundary for silage maize and silage rye, while the second values between parentheses were calculated considering the actual cultivation period of both crops.

Case study 1	Renew-able energy	Fossil	Nu-clear	Metal	Mine-ral	Water	Land	Biomass and primary forest	Total
CED	46.4 (43.1)	55.9 (50.7)	26.2[h] (23.5[i]	n.a.[a]	n.a.[a]	n.a.[a]	n.a.[b]	99.9[z, ab] (99.9[aa,	95.2 (95.1)
CExD	43.9 (40.5)	55.4 (50.2)	26.2[h] (23.5[i]	79.8 (76.8)	98.5 (98.4)	39.9[n] (38.4[o])		99.9[z, ac] (99.9[aa,	94.8[ad] (94.8[ad]
SED	57.7 (54.3)	42.5 (37.7)	26.2[h] (23.5[i]	85.0 (82.8)	86.0 (84.6)	40.4 (38.9)	99.7[af] (99.5)	n.a.d.c.[c]	72.6 (68.3)
CEENE v2007$_{2\%}$	29.7[d] (26.9[e])	55.7[f] (50.4[g])	26.2[h] (23.5[i]	77.1[j] (73.8[k]	82.9[l] (80.1[m])	39.9[n] (38.4[o])	99.7[p] (99.6[q])		97.5 (95.9)
CEENE v2013	29.7[d] (26.9[e])	55.7[f] (50.4[g])	26.2[h] (23.5[i]	77.1[j] (73.8[k]	82.9[l] (80.1[m])	39.9[n] (38.4[o])	99.8 (99.6[ag])		94.1 (90.6)
CEENE v2007$_{TOT}$	29.7[d] (26.9[e])	55.7[f] (50.4[g])	26.2[h] (23.5[i]	77.1[j] (73.8[k]	82.9[l] (80.1[m])	39.9[n] (38.4[o])	99.7[p] (99.6[q])		99.7 (99.5)
CEENE v2007$_{PAR}$	29.7[d] (26.9[e])	55.7[f] (50.4[g])	26.2[h] (23.5[i]	77.1[j] (73.8[k]	82.9[l] (80.1[m])	39.9[n] (38.4[o])	99.7[p] (99.6[q])		99.6 (99.4)
CEENE v2007$_{TMC}$	29.7[d] (26.9[e])	55.7[f] (50.4[g])	26.2[h] (23.5[i]	77.1[j] (73.8[k]	82.9[l] (80.1[m])	39.9[n] (38.4[o])	99.7[p] (99.6[q])		98.9 (98.2)
CEENE v2007$_{TMCA}$	29.7[d] (26.9[e])	55.7[f] (50.4[g])	26.2[h] (23.5[i]	77.1[j] (73.8[k]	82.9[l] (80.1[m])	39.9[n] (38.4[o])	99.7[p] (99.6[q])		98.8 (98.0)
CEENE v2007$_{OMCA}$	29.7[d] (26.9[e])	55.7[f] (50.4[g])	26.2[h] (23.5[i]	77.1[j] (73.8[k]	82.9[l] (80.1[m])	39.9[n] (38.4[o])	99.7[p] (99.6[q])		97.8 (96.3)

Case study 2	Renew-able energy	Fossil	Nu-clear	Metal	Mine-ral	Water	Land	Biomass and primary forest	Total
CED	0.7[ae]	16.7	20.0[t]	n.a.[a]	n.a.[a]	n.a.[a]	n.a.[b]	99.6[y]	80.4
CExD	0.7[ae]	17.4	20.0[t]	32.7	72.8	2.0[w]		99.6[y]	78.0
SED	0.6	19.5	20.0[t]	34.7	8.7	1.3	99.5	n.a.d.c.[c]	15.3
CEENE v2007$_{2\%}$	0.8[r]	16.8[s]	20.0[t]	35.6[u]	15.0[v]	2.0[w]	99.7[x]		91.4
CEENE v2013	0.8[r]	16.8[s]	20.0[t]	35.6[u]	15.0[v]	2.0[w]	99.7[ah]		79.3

CEENE v2007$_{TOT}$	0.8[r]	16.8[s]	20.0[t]	35.6[u]	15.0[v]	2.0[w]	99.7[x]		99.5
CEENE v2007$_{PAR}$	0.8[r]	16.8[s]	20.0[t]	35.6[u]	15.0[v]	2.0[w]	99.7[x]		99.3
CEENE v2007$_{TMC}$	0.8[r]	16.8[s]	20.0[t]	35.6[u]	15.0[v]	2.0[w]	99.7[x]		96.6
CEENE v2007$_{TMCA}$	0.8[r]	16.8[s]	20.0[t]	35.6[u]	15.0[v]	2.0[w]	99.7[x]		96.1
CEENE v2007$_{OMCA}$	0.8[r]	16.8[s]	20.0[t]	35.6[u]	15.0[v]	2.0[w]	99.7[x]		92.4

[a] n.a.: not accounted for by the CED method; [b] n.a.: not accounted for by the CED and CExD method; [c] n.a.d.c.: not accounted for to avoid double counting with the land resources category; [d-y] Values are identical; [z-ae] Values are different but differences are smaller than 0.5%; [af] Value is slightly lower than the 99.7% values indicated with [p] but this difference is smaller than 0.5%; [ag] Value is slightly lower than the 99.6% values indicated with [q] but this difference is smaller than 0.5%; [ah] Value is slightly higher than the 99.7% values indicated with [x] but this difference is smaller than 0.5%.

Supplementary material A5: Accounting for the ancient solar energy consumption of fossil resources: results

Table A.8 Overview of the share of the resource categories in the overall resource footprint using the applied approaches with and without accounting for the ancient solar energy consumption of fossil resources. In case of the bio-based electricity, the first values were calculated considering an entire year of land occupation as temporal system boundary for silage maize and silage rye, while the second values between parentheses were calculated considering the actual cultivation period of both crops.

Solar system boundary level	**Accounting for ancient solar energy consumption of fossil resources?**	**Product**	**Land resources (%)**	**Fossil resources (%)**	**Other (%)**
TMCA	yes	**bio-based electricity**	<1 (<1)	100 (100)	<1 (<1)
		fossil-based electricity	<1	100	<1
	no	**bio-based electricity**	98 (97)	1 (2)	1 (1)
		fossil-based electricity	<1	99	1

Solar system boundary level	**Accounting for ancient solar energy consumption of fossil resources?**	**Product**	**Land resources (%)**	**Fossil resources (%)**	**Other (%)**
OMCA	yes	**bio-based electricity**	<1 (<1)	100 (100)	<1 (<1)
		fossil-based electricity	<1	100	<1
	no	**bio-based electricity**	96 (94)	3 (4)	1 (2)
		fossil-based electricity	<1	99	1

Solar system boundary level	**Accounting for ancient solar energy consumption of fossil resources?**	**Product**	**Land resources (%)**	**Fossil resources (%)**	**Other (%)**
TMCA	yes	**bio-based PVC**	<1	100	<1
		fossil-based PVC	<1	100	<1
	no	**bio-based PVC**	96	2	2
		fossil-based PVC	2	85	13

Solar system boundary level	**Accounting for ancient solar energy consumption of fossil resources?**	**Product**	**Land resources (%)**	**Fossil resources (%)**	**Other (%)**
OMCA	yes	**bio-based PVC**	<1	100	<1
		fossil-based PVC	<1	100	<1
	no	**bio-based PVC**	92	5	3
		fossil-based PVC	1	86	13

APPENDIX B: SUPPLEMENTARY MATERIAL CHAPTER 3

Groundwater reduction at the farm under study could be achieved in two ways: 1) by reusing the rinsing effluent for other applications and 2) by investing in a water-saving milking installation.

Considering the first way: **5.0%** reduction of the total on-farm groundwater consumption could be possible based on reusing rinsing effluent for other applications:

- groundwater consumption for cleaning the milking parlour accounted for 8.3% of total on-farm groundwater consumption
- groundwater consumption for cleaning milking installation accounted for 6.3% of total on-farm groundwater consumption
- groundwater consumption for cleaning milking tank for 0.8% of total on-farm groundwater consumption

The second and third rinse-water flow of the milking installation can be used to clean the milking parlour: this equals two-thirds of the water use of the milking installation (VMM, 2006).

→ 6.3% * 2/3 = 4.2%

Also the water consumed to clean the milking tank can be reused to clean the milking parlour (VMM, 2006).

→ 4.2% + 0.8% = 5.0% of the consumed water can be reused to clean the milking parlour. Only 3.3% (8.3% - 5.0%) fresh water is used to clean the milking parlour.

Considering the second way: **5.0%** reduction of the total on-farm groundwater consumption could be possible based on a water-saving installation (and reusing rinsing effluent of milking tank for other applications)

According to VMM (2006), "doorschuifreiniging" is a water-saving installation:

in this case only fresh water is used for the third rinse. After the third rinse, this water is used for the second rinse in the next run, and afterwards it is again used for the first rinse. Two-thirds of the water consumption could be reduced in this way (VMM, 2006).

→ 6.3% * 2/3 = 4.2%

And again the water consumed to clean the milking tank can be reused to clean the milking parlour (VMM, 2006).

→ 4.2% + 0.8% = 5.0% of the total on-farm groundwater consumption can be saved. Only 7.5% (8.3% - 0.8%) fresh water is used to clean the milking parlour. Only 2.1% (6.3% - 4.2%) fresh water is used to clean the milking installation.

APPENDIX C: SUPPLEMENTARY MATERIAL CHAPTER 4

Supplementary material C1: Constant and variable returns to scale

Supplementary material C2: Data Envelopment Analysis (DEA)

Supplementary material C3: Stochastic Frontier Analysis (SFA)

Supplementary material C4: CEENE input coefficients per resource category

Supplementary material C5: CEENE efficiency scores per resource category when applying DEA under CRS assumption

Supplementary material C6: other tested Key Performance Indicators (KPIs) that were not significantly different

Supplementary material C7: efficiency scores when applying DEA under VRS assumption and SFA

Supplementary material C1: Constant and variable returns to scale

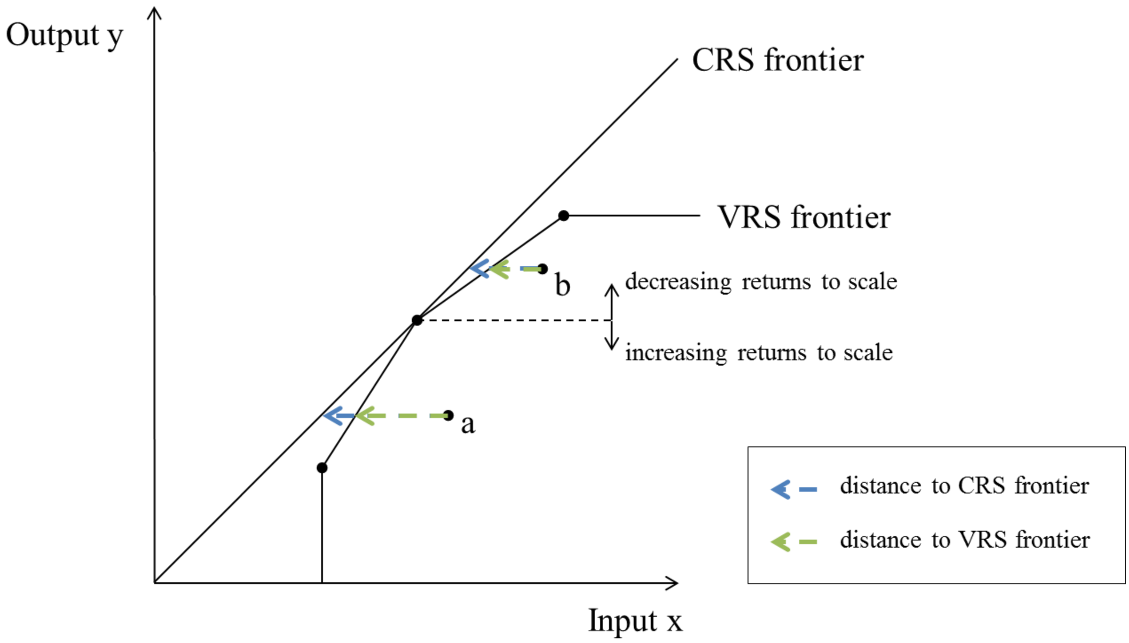


Figure C.1 Illustration of the difference in calculating technical efficiency between DEA under constant returns to scale (CRS) and variable returns to scale (VRS). Technical efficiencies are higher under VRS assumption because the distance to the frontier is smaller (this figure is based on Coelli et al. (2005)).

Supplementary material C2: Data Envelopment analysis (DEA)

DEA involves the use of linear programming to construct a non-parametric frontier that envelops the data points by piecewise connecting the best-performing farms in the dataset. The technical efficiency score of the ith farm (TE_i) is calculated by solving the following linear program for each farm (Coelli et al., 2005), thus 103 times for the farms in our dataset:

$$\min_{\theta,\lambda} \theta$$

$$subject\ to$$

$$(y_1\lambda_1 + y_2\lambda_2 + \cdots + y_{103}\lambda_{103}) \geq y_i$$
$$(x_{1,1}\lambda_1 + x_{1,2}\lambda_2 + \cdots + x_{1,103}\lambda_{103}) \leq \theta x_{1,i}$$
$$(x_{2,1}\lambda_1 + x_{2,2}\lambda_2 + \cdots + x_{2,103}\lambda_{103}) \leq \theta x_{2,i}$$
$$\lambda_1 + \lambda_2 + \cdots + \lambda_{103} = 1\ (only\ necessary\ under\ VRS\ assumption)$$
$$\lambda \geq 0$$

with:

θ: technical efficiency score for the ith farm (=TE_i)

y_i: milk and meat production (euro/year)

$x_{1,i}$: roughages (euro/year)

$x_{2,i}$: concentrates and by-products (kg/year)

$\lambda = (\lambda_1, \lambda_2, \ldots, \lambda_{103})$: vector of constants (-)

i: farm index (1-103)

Calculations are performed using the DEAP version 2.1 computer program (Coelli, 1996a). For each farm in the sample, this involves finding values for θ and $\lambda = (\lambda_1, \lambda_2, \ldots, \lambda_{103})$ that minimize technical efficiency score for the ith farm, subject to the constraints that all efficiency scores must be less than or equal to one (Coelli et al., 2005). For the ith farm, values for λ are different from zero when their index number (1-103) corresponds to technically efficient farms that form the endpoint(s) of the line on which the technically efficient benchmark of the ith farm is located. In other words, the technically efficient benchmark of the ith farm is a linear combination of the technically efficient farms on the same line (=*peers*), where the weights in this linear combination are represented by the λs. To better grasp this approach, a simple theoretical example can be found in Coelli et al. (2005). The additional constraint under VRS assumption ($\lambda_1 + \lambda_2 + \cdots + \lambda_{103} = 1$) ensures that the technically efficient benchmark of the ith farm is a convex combination of the peers instead of a linear combination (Supplementary material S1) (Coelli et al., 2005).

To calculate the cost and environmental (CEENE) efficiency scores, the following two linear programs have to be solved for each farm:

$$\min_{\lambda, x_i^{ce}} p_{1,i} x_{1,i}^{ce} + p_{2,i} x_{2,i}^{ce}$$

$subject\ to$

$$(y_1\lambda_1 + y_2\lambda_2 + \cdots + y_{103}\lambda_{103}) \geq y_i$$
$$\left(x_{1,1}\lambda_1 + x_{1,2}\lambda_2 + \cdots + x_{1,103}\lambda_{103}\right) \leq x_{1,i}^{ce}$$
$$\left(x_{2,1}\lambda_1 + x_{2,2}\lambda_2 + \cdots + x_{2,103}\lambda_{103}\right) \leq x_{2,i}^{ce}$$
$$\lambda_1 + \lambda_2 + \cdots + \lambda_{103} = 1\ (only\ necessary\ under\ VRS\ assumption\ )$$
$$\lambda \geq 0$$

with:

$p_{1,i}$: price roughages (euro/euro); this always equals 1.

$p_{2,i}$: price concentrates and by-products (euro/kg)

$x_{1,i}^{ce}$: cost efficient roughage use (euro/year)

$x_{2,i}^{ce}$: cost efficient concentrates and by-products use (kg/year)

For each farm in the sample, this involves finding values for $\lambda = (\lambda_1, \lambda_2, \ldots, \lambda_{103})$ and $x_i^{ce} = \left(x_{1,i}^{ce}\ and\ x_{2,i}^{ce}\right)$ in order that the total costs for the ith farm are minimized, subject to the constraints that all cost efficiency scores must be less than or equal to one.

$$\min_{\lambda, x_i^{ee}} c_{1,i,j} x_{1,i}^{ee} + c_{2,i,j} x_{2,i}^{ee}$$

$subject\ to$

$$(y_1\lambda_1 + y_2\lambda_2 + \cdots + y_{103}\lambda_{103}) \geq y_i$$
$$\left(x_{1,1}\lambda_1 + x_{1,2}\lambda_2 + \cdots + x_{1,103}\lambda_{103}\right) \leq x_{1,i}^{ee}$$
$$\left(x_{2,1}\lambda_1 + x_{2,2}\lambda_2 + \cdots + x_{2,103}\lambda_{103}\right) \leq x_{2,i}^{ee}$$
$$\lambda_1 + \lambda_2 + \cdots + \lambda_{103} = 1\ (only\ necessary\ under\ VRS\ assumption\ )$$
$$\lambda \geq 0$$

with:

$c_{1,i,j}$: environmental (CEENE) coefficient roughages (MJ_{ex}/euro);

$c_{2,i,j}$: environmental (CEENE) coefficient concentrates and by-products (MJ_{ex} /kg)

$x_{1,i,j}^{ee}$: environmentally (CEENE) efficient roughage use (euro/year)

$x_{2,i,j}^{ee}$: environmentally (CEENE) efficient concentrates and by-products use (kg/year)

j: index for CEENE-total or one of the CEENE categories (land (LAN), water (WAT), minerals (MIN), metals (MET), fossil energy (FOS), nuclear energy (NUC) and abiotic renewable energy (REN))

For each farm in the sample, this involves finding values for $\lambda = (\lambda_1, \lambda_2, \ldots, \lambda_{103})$ and $x_i^{ee} = \left(x_{1,i}^{ee} \ and \ x_{2,i}^{ee}\right)$ in order that the total CEENE for the ith farm is minimized, subject to the constraints that all environmental (CEENE) efficiency scores must be less than or equal to one.

From the cost ($x_{1,i}^{ce}$ and $x_{2,i}^{ce}$) and environmentally (CEENE) ($x_{1,i,j}^{ee}$ and $x_{2,i,j}^{ee}$) efficient input targets obtained by solving the linear programs, cost and environmental (CEENE) efficiencies are calculated, respectively, as:

$$CE_i = \frac{p_{1,i}x_{1,i}^{ce} + p_{2,i}x_{2,i}^{ce}}{p_{1,i}x_{1,i} + p_{2,i}x_{2,i}} \quad (1)$$

$$EE_{i,j} = \frac{c_{1,i,j}x_{1,i,j}^{ee} + c_{2,i,j}x_{2,i,j}^{ee}}{c_{1,i,j}x_{1,i} + c_{2,i,j}x_{2,i}} \quad (2)$$

Knowing the technical efficiency score (TE_i), cost allocative and environmental (CEENE) allocative efficiencies are calculated, respectively, as:

$$CAE_i = \frac{CE_i}{TE_i} \quad (3)$$

$$EAE_{i,j} = \frac{EE_{i,j}}{TE_i} \quad (4)$$

Supplementary material C3: Stochastic Frontier Analysis (SFA)

SFA fits a parametric continuous production frontier to given data, and specifies a two-part error term to account for both random errors and the degree of technical inefficiency. The functional form of the frontier has to be chosen by the researcher. We estimated a Cobb-Douglas production function, using the same data that are used in the non-parametric DEA. Parameters and error terms were specified using maximum likelihood estimation with the FRONTIER 4.1 computer program (Coelli, 1996).

$$y_i = A \times x_{1,i}^a \times x_{2,i}^b \times e^{v_i} \times e^{-u_i}$$

with:

y_i: milk and meat production (euro/year)

$x_{1,i}$: roughage production (euro/year)

$x_{2,i}$: concentrates and by-products (kg/year)

v_i: random error

u_i: technical inefficiency

A, a, b: parameters

i: farm index

The technical efficient input targets ($x_{1,i}^{te}$ and $x_{2,i}^{te}$) can be calculated by simultaneously solving the following two equations (e^{-u_i} = 1 when technical efficient):

$$\frac{y_i}{e^{v_i}} = A \times \left(x_{1,i}^{te}\right)^a \times \left(x_{2,i}^{te}\right)^b$$

$$\frac{x_{1,i}^{te}}{x_{2,i}^{te}} = k_i$$

with k_i: farm-specific constant

The technical efficiency score is then calculated as:

$$TE_i = \frac{x_{1,i}^{te} + x_{2,i}^{te}}{x_{1,i} + x_{2,i}}$$

To obtain cost and environmental (CEENE) efficient benchmarks, the following cost and environmental function are established, using vector notations:

$$CE_i = f\left(P_i, \frac{Y_i}{e^{v_i}}, \alpha\right)$$

$$EE_{i,j} = f\left(C_{i,j}, \frac{Y_i}{e^{v_i}}, \alpha\right)$$

with:

CE_i: vector of minimum costs (euro)

P_i: vector of input prices (euro)

$EE_{i,j}$: vector of minimum CEENE (MJ_{ex})

$C_{i,j}$: vector of CEENE coefficients of inputs (MJ_{ex})

α: parameters

The cost and environmental function represent minimum costs of inputs as a function of output and prices of inputs, and minimum CEENE of inputs as a function of output and CEENE coefficients of inputs, respectively.

To obtain the cost ($x_{1,i}^{ce}$ and $x_{2,i}^{ce}$) and environmental (CEENE) ($x_{1,i,j}^{ee}$ and $x_{2,i,j}^{ee}$) efficient input targets, Shephard's Lemma (Coelli et al., 2005), which is the first partial derivative with respect to each of the input prices and CEENE coefficients, is applied to the cost and environmental function, respectively (Van Meensel et al., 2010b). Subsequently, the cost and environmental efficiency can be calculated as the ratio of minimum costs to observed costs and minimum CEENE to observed CEENE, respectively.

Supplementary material C4: CEENE input coefficients per resource category

Table C.1 Descriptive statistics of the CEENE coefficients for the different resource categories for the sample of 103 specialized dairy farms in Flanders during a one-year period in 2010-2011.

	Symbol	Description	Mean	Min.	Max.	Median	Interquartile range[a]
CEENE-land coefficients	$c - lan_1$	roughages (MJ_{ex}/euro)	200.1	117.8	391.2	191.0	59.1
	$c - lan_2$	concentrates and by-products (MJ_{ex}/kg)	28.0	19.8	41.0	27.6	5.7
CEENE-water coefficients	$c - wat_1$	roughages (MJ_{ex}/euro)	0.5	0.3	1.2	0.5	0.2
	$c - wat_2$	concentrates and by-products (MJ_{ex}/kg)	0.4	0.3	0.6	0.4	0.1
CEENE-minerals coefficients	$c - min_1$	roughages (MJ_{ex}/euro)	0.033	0.011	0.062	0.033	0.010
	$c - min_2$	concentrates and by-products (MJ_{ex}/kg)	0.026	0.015	0.036	0.026	0.004
CEENE-metals coefficients	$c - met_1$	roughages (MJ_{ex}/euro)	0.064	0.028	0.160	0.061	0.021
	$c - met_2$	concentrates and by-products (MJ_{ex}/kg)	0.009	0.007	0.014	0.009	0.002
CEENE-fossil energy coefficients	$c - fos_1$	roughages (MJ_{ex}/euro)	16.8	9.4	34.2	16.4	3.9
	$c - fos_2$	concentrates and by-products (MJ_{ex}/kg)	6.4	4.9	9.3	6.4	0.9
CEENE-nuclear energy coefficients	$c - nuc_1$	roughages (MJ_{ex}/euro)	1.3	0.6	3.3	1.2	0.4
	$c - nuc_2$	concentrates and by-products (MJ_{ex}/kg)	0.7	0.5	1.0	0.6	0.1
CEENE-abiotic renewable energy coefficients	$c - ren_1$	roughages (MJ_{ex}/euro)	0.6	0.2	1.5	0.6	0.2
	$c - ren_2$	concentrates and by-products (MJ_{ex}/kg)	0.2	0.1	0.3	0.2	<0.1

[a] The interquartile range is a measure of dispersion and equals the difference between the upper quartile (third quartile) and lower quartile (first quartile). The first quartile splits off the lowest 25% of data from the highest 75%. The third quartile splits off the highest 25% of data from the lowest 75%.

Supplementary material C5: CEENE efficiency scores per resource category when applying DEA under CRS assumption

Table C.2 CEENE efficiency scores per resource category calculated with Data Envelopment Analysis (DEA) under constant returns to scale (CRS) assumption for the sample of 103 specialized dairy farms in Flanders during a one-year period in 2010-2011.

Efficiency type	Average	Min.	Max.	Median	Interquartile range[a]
CEENE LAN	0.752	0.522	1.000	0.741	0.127
CEENE FOS	0.728	0.523	1.000	0.718	0.108
CEENE WAT	0.704	0.523	0.990	0.691	0.112
CEENE MIN	0.701	0.498	1.000	0.689	0.124
CEENE MET	0.749	0.503	1.000	0.737	0.127
CEENE NUC	0.721	0.523	1.000	0.712	0.108
CEENE REN	0.729	0.523	1.000	0.717	0.116
CEENE allocative LAN	0.980	0.819	1.000	0.991	0.017
CEENE allocative FOS	0.950	0.761	1.000	0.962	0.057
CEENE allocative WAT	0.918	0.676	1.000	0.931	0.090
CEENE allocative MIN	0.914	0.633	1.000	0.925	0.100
CEENE allocative MET	0.975	0.815	1.000	0.989	0.026
CEENE allocative NUC	0.940	0.687	1.000	0.952	0.078
CEENE allocative REN	0.951	0.717	1.000	0.963	0.058

[a] The interquartile range is a measure of dispersion and equals the difference between the upper quartile (third quartile) and lower quartile (first quartile). The first quartile splits off the lowest 25% of data from the highest 75%. The third quartile splits off the highest 25% of data from the lowest 75%.

Supplementary material C6: other tested Key Performance Indicators (KPIs) that were not significantly different

Table C.3 Comparison of Key Performance Indicators (KPIs) between the 10% of the farms closest to the average farm (group 1) and the 10% of the farms closest to the coinciding cost and CEENE-total efficient benchmarks for the average farm (group 2), identified with DEA under CRS assumption. A comparison between group 1 and group 2 excluding two farms with high replacement rates is also presented. The average value for each group is presented and the nonparametric Wilcoxon two sample test was used to check whether KPI values significantly differed between both groups.

Key Performance Indicators	Group 1	Group 2	Group 2 excluding two farms with high replacement rates
kg concentrates and by-products per euro roughages	3.84	3.78	3.85
kg concentrates and by-products per dairy cow	2713	2431	2504
kg concentrates per dairy cow	1791	1771	1803
kg by-products per dairy cow	923	660	701
FPCM[a] produced per available area (kg/ha)	18928	18313	18357
average milk price (euro/100L)	32.78	32.76	32.76
average price concentrates and by-products (euro/kg)	0.24	0.25	0.25
kg concentrates and by-products per ha	5895	4691	4762
kg concentrates per ha	3853	3387	3411
kg by-products per ha	2042	1304	1351
kg by-products per kg total concentrates and by-products	0.33	0.26	0.27
kg soybean meal per kg total concentrates and by-products	0.11	0.13	0.14
young cattle/dairy cows (%)	79.09	82.55	77.00
average age dairy cows (days)	1521	1516	1524
births per 100 dairy cows	106	103	104
number of dairy cows per ha	2.17	1.93	1.90

[a] FPCM: fat-and-protein-corrected milk (IDF, 2010)

Table C.4 Comparison of Key Performance Indicators (KPIs) between the '*green*' farms (reference group) and the '*purple*' and '*blue*' farms, presented in Figure 4.4, and identified with DEA under CRS assumption. The average value for each group is presented and the nonparametric Wilcoxon two sample test was used to check whether KPI values significantly differed between the reference group and the other two groups.

Key Performance Indicators	'*Green*' farms	'*Purple*' farms	'*Blue*' farms
area grass per total area (%)	52.11	53.82	52.62
area maize per total area (%)	47.59	45.33	47.38
area grass per area maize	1.22	1.32	1.97
contract work (euro/ha)	411	390	349
euro milk and meat per dairy cow	3347	3187	3162
euro milk per dairy cow	3075	2910	2928
euro meat per dairy cow	273	277	235
average milk price (euro/100L)	32.68	33.16	32.85
replacement rate	33.06	33.01	32.25
farm size (ha)	49.54	52.92	60.02
farm size (number of dairy cows)	107	96	107
labor income per kg FPCM[a] produced	0.12	0.11	0.13
kg soybean meal per kg total concentrates and by-products	0.09	0.08	0.05
young cattle/dairy cows (%)	82.43	87.33	83.32
average age dairy cows (days)	1512	1570	1517
births per 100 dairy cows	106	104	102

[a] FPCM: fat-and-protein-corrected milk (IDF, 2010)

Supplementary material C7: efficiency scores when applying DEA under VRS assumption and SFA

Data Envelopment Analysis (DEA) - VRS assumption

Table C.5 Efficiency scores calculated with Data Envelopment Analysis (DEA) under variable returns to scale (VRS) assumption for the sample of 103 specialized dairy farms in Flanders during a one-year period in 2010-2011.

Efficiency type	Average	Min.	Max.	Median	Interquartile range[a]
Technical (TE)	0.823	0.557	1.000	0.803	0.153
Cost (CE)	0.792	0.555	1.000	0.783	0.136
Exergetic – CEENE-total (EE)	0.796	0.554	1.000	0.786	0.136
Cost allocative (CAE)	0.964	0.849	1.000	0.973	0.043
Exergetic allocative - CEENE-total (EAE)	0.968	0.848	1.000	0.986	0.045

[a] The interquartile range is a measure of dispersion and equals the difference between the upper quartile (third quartile) and lower quartile (first quartile). The first quartile splits off the lowest 25% of data from the highest 75%. The third quartile splits off the highest 25% of data from the lowest 75%.

Table C.6 CEENE efficiency scores per resource category calculated with Data Envelopment Analysis (DEA) under variable returns to scale (VRS) assumption for the sample of 103 specialized dairy farms in Flanders during a one-year period in 2010-2011.

Efficiency type	Average	Min.	Max.	Median	Interquartile range[a]
CEENE LAN	0.795	0.554	1.000	0.784	0.144
CEENE FOS	0.787	0.557	1.000	0.774	0.140
CEENE WAT	0.770	0.556	1.000	0.747	0.174
CEENE MIN	0.767	0.554	1.000	0.742	0.176
CEENE MET	0.792	0.530	1.000	0.782	0.134
CEENE NUC	0.781	0.557	1.000	0.757	0.152
CEENE REN	0.788	0.557	1.000	0.770	0.143
CEENE allocative LAN	0.967	0.840	1.000	0.986	0.046
CEENE allocative FOS	0.957	0.789	1.000	0.974	0.053
CEENE allocative WAT	0.936	0.686	1.000	0.962	0.094
CEENE allocative MIN	0.932	0.652	1.000	0.961	0.097
CEENE allocative MET	0.963	0.818	1.000	0.982	0.050
CEENE allocative NUC	0.950	0.697	1.000	0.973	0.072
CEENE allocative REN	0.959	0.727	1.000	0.975	0.054

[a] The interquartile range is a measure of dispersion and equals the difference between the upper quartile (third quartile) and lower quartile (first quartile). The first quartile splits off the lowest 25% of data from the highest 75%. The third quartile splits off the highest 25% of data from the lowest 75%.

Stochastic Frontier Analysis (SFA)

Table C.7 Efficiency scores calculated with Stochastic Frontier Analysis (SFA) for the sample of 103 specialized dairy farms in Flanders during a one-year period in 2010-2011.

Efficiency type	Average	Min.	Max.	Median	Interquartile range[a]
Technical (TE)	0.927	0.841	0.965	0.932	0.029
Cost (CE)	0.918	0.814	0.965	0.925	0.032
Exergetic – CEENE-total (EE)	0.892	0.759	0.950	0.897	0.053
Cost allocative (CAE)	0.989	0.928	1.000	0.995	0.015
Exergetic allocative - CEENE-total (EAE)	0.962	0.845	1.000	0.968	0.041

[a] The interquartile range is a measure of dispersion and equals the difference between the upper quartile (third quartile) and lower quartile (first quartile). The first quartile splits off the lowest 25% of data from the highest 75%. The third quartile splits off the highest 25% of data from the lowest 75%.

Table C.8 CEENE efficiency scores per resource category calculated with Stochastic Frontier Analysis (SFA) for the sample of 103 specialized dairy farms in Flanders during a one-year period in 2010-2011.

Efficiency type	Average	Min.	Max.	Median	Interquartile range[a]
CEENE LAN	0.873	0.713	0.955	0.881	0.058
CEENE FOS	0.903	0.805	0.958	0.907	0.046
CEENE WAT	0.814	0.641	0.955	0.813	0.075
CEENE MIN	0.804	0.496	0.938	0.807	0.078
CEENE MET	0.874	0.704	0.945	0.882	0.061
CEENE NUC	0.873	0.673	0.957	0.885	0.077
CEENE REN	0.900	0.717	0.962	0.909	0.046
CEENE allocative LAN	0.941	0.793	1.000	0.947	0.053
CEENE allocative FOS	0.974	0.842	1.000	0.985	0.032
CEENE allocative WAT	0.877	0.683	0.997	0.884	0.079
CEENE allocative MIN	0.867	0.528	0.991	0.873	0.083
CEENE allocative MET	0.943	0.770	1.000	0.957	0.058
CEENE allocative NUC	0.941	0.703	1.000	0.955	0.063
CEENE allocative REN	0.971	0.750	1.000	0.984	0.034

[a] The interquartile range is a measure of dispersion and equals the difference between the upper quartile (third quartile) and lower quartile (first quartile). The first quartile splits off the lowest 25% of data from the highest 75%. The third quartile splits off the highest 25% of data from the lowest 75%.

APPENDIX D: SUPPLEMENTARY MATERIAL CHAPTER 5

Supplementary material D1: Multiple linear regression

Supplementary material D2: Chemical exergy value of liquid water

Supplementary material D1: Multiple linear regression

Multiple linear regression was performed to determine the main variables that explain the variation between the feed supply chain's CEENE v2013 resource footprint of specialized dairy farms in Flanders. A dataset with 31 candidate predictor variables was established for which data of 103 specialized dairy farms were retrieved from their farm accountancy files for a one-year period in 2010-2011. The selection of the candidate predictor variables was based on how they were reported in the farm accountancy files, which were organized in the same format because all the considered dairy farms were affiliated with the same farm advisory company. Also the candidate predictor variables were all variables that were used in the calculation of the feed supply chain's CEENE v2013 resource footprint. The dataset was randomly split in a training dataset of 75 farms and a validation dataset of the remaining 28 farms. The annual feed supply chain's CEENE v2013 resource footprint (expressed in MJ_{ex} per year) was set as the dependent variable. The considered independent candidate predictor variables are presented in Table D.1.

Table D.1 The considered independent candidate predictor variables.

No.	Independent variables	Unit
1	available on-farm land for feed production	ha
2	available on-farm land for grass production	ha
3	available on-farm land for maize production	ha
4	fuel used by farmer	L / year
5	cost for contract work	euro / year
6	quantity of mineral N fertilizers	kg N / year
7	quantity of mineral P fertilizers	kg P_2O_5 / year
8	quantity of mineral K fertilizers	kg K_2O / year
9	quantity of pesticides	kg A.I. / year
10	consumed quantity of purchased concentrates based one ingredient by dairy cows	kg / year
11	consumed quantity of purchased soybean meal by dairy cows	kg / year
12	consumed quantity of dry beat pulp by dairy cows	kg / year
13	consumed quantity of maize gluten by dairy cows	kg / year
14	consumed quantity of purchased rapeseed by dairy cows	kg / year
15	consumed quantity of purchased grains by dairy cows	kg / year
16	consumed quantity of other purchased concentrates based one ingredient by dairy cows	kg / year
17	consumed quantity of purchased mixed concentrates by dairy cows	kg / year
18	consumed quantity of purchased high-protein mixed concentrates by dairy cows	kg / year
19	consumed quantity of purchased low-protein mixed concentrates by dairy cows	kg / year
20	consumed quantity of purchased by-products by dairy cows	kg DM / year
21	consumed quantity of purchased beet pressed pulp by dairy cows	kg DM / year
22	consumed quantity of purchased brewers grains by dairy cows	kg DM / year
23	consumed quantity of purchased fodder beets by dairy cows	kg DM / year
24	consumed quantity of purchased potatoes by dairy cows	kg DM / year
25	consumed quantity of purchased CCM by dairy cows	kg DM / year
26	consumed quantity of purchased CCS by dairy cows	kg DM / year
27	consumed quantity of purchased other by-products by dairy cows	kg DM / year
28	consumed quantity of purchased feed by young cattle	kg / year
29	consumed quantity of purchased dry concentrates by young cattle	kg / year
30	consumed quantity of purchased by-products by young cattle	kg / year
31	purchased quantity of roughages corrected for roughage stock changes	kg DM / year

Pearson correlation was quantified between all independent candidate predictor variables. Variables for which Pearson correlation was higher than 0.6 were not included together in the regression models. A choice between these highly correlated variables was made based on their significance ($p < 0.05$) and their determination coefficient (R^2). Only significant variables ($p < 0.05$) were included in the regression models. SPSS was used as a software package.

Based on this analysis, seven candidate predictor variables could be retained in the first regression model A (adjusted R^2=0.992 based on training dataset n = 75): 1) available on-farm land for feed production (No. 1), 2) consumed quantity of purchased concentrates based on one ingredient by dairy cows (No. 10), 3) consumed quantity of purchased mixed concentrates by dairy cows (No. 17), 4) consumed quantity of purchased low-protein mixed concentrates by dairy cows (No. 19), 5) consumed quantity of purchased by-products by dairy cows (No. 20), 6) consumed quantity of purchased by-products by young cattle (No. 30) and 7) purchased quantity of roughages corrected for roughage stock changes (No. 31).

$$\begin{aligned} &feed\ supply\ chain\text{'}s\ CEENE\ v2013\ resource\ footprint\ (MJ_{ex}\ per\ year) \\ &\qquad - Model\ A \\ &\qquad = -24563.329 + 319957.920 \times L + 37.721 \times C_{s\text{-}d} + 46.319 \times C_{m\text{-}d} \\ &\qquad - 7.825 \times C_{m\text{-}l\text{-}d} + 20.051 \times BP_d + 96.797 \times BP_y + 15.584 \times R \end{aligned}$$

with

L: available on-farm land for feed production (ha)

$C_{s\text{-}d}$: total amount of concentrates based on one ingredient (e.g. soybean meal) fed to dairy cows (kg)

$C_{m\text{-}d}$: total amount of mixed concentrates fed to dairy cows (kg)

$C_{m\text{-}l\text{-}d}$: total amount of mixed low-protein concentrates fed to dairy cows (kg)

BP_d: total amount of by-products fed to dairy cows (kg dry matter)

BP_y: total amount of by-products fed to young cattle (kg)

R: purchased quantity of roughages corrected for roughage stock changes (kg dry matter)

Validation of regression model A with seven predictor variables showed a high coefficient of determination equal to 0.9802 (n = 28) (Figure D.1).

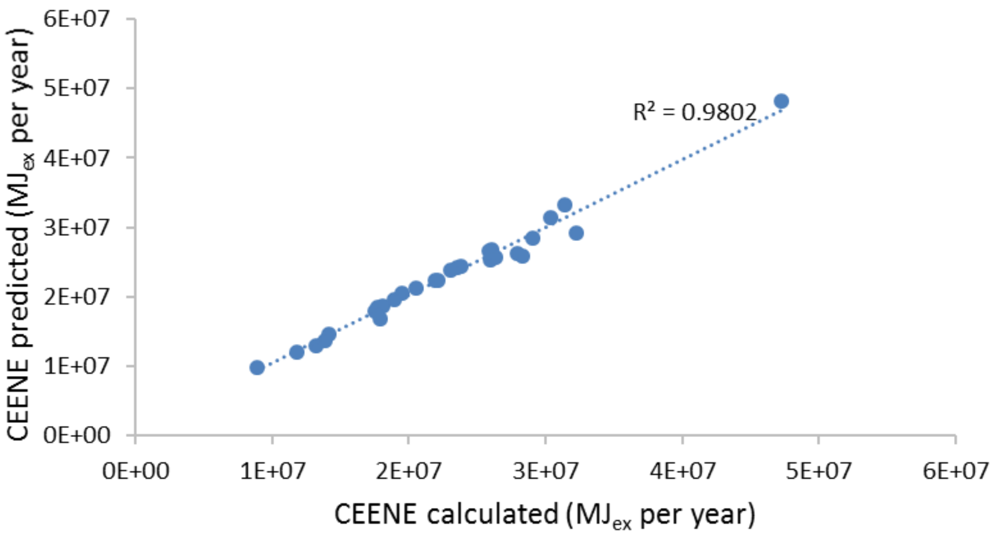


Figure D.1 Validation of regression model A with seven variables to predict the annual feed supply chain's CEENE v2013 resource footprint of specialized dairy farms in Flanders. The training dataset to build to regression model was based on 75 farms; the validation dataset was based on 28 farms.

After building this first regression model A, a balance was sought between model complexity, i.e. the number of predictor variables, and the accuracy and precision of the prediction. Predictor variables were first removed based on their level of significance ($p < 0.001$ vs. $p < 0.01$ vs $p < 0.05$). Second, predictor variables were removed based on their standardized regression coefficients ('beta coefficients'). These coefficients can be used to compare the relative strength of predictor variables within the regression model, because they are measured in standard deviations, instead of the variables' units. Model B (adjusted R^2=0.991 based on training dataset n = 75) includes six predictor variables, after exclusion of the variable consumed quantity of purchased low-protein mixed concentrates by dairy cows (No. 19), because this variable was only significant at the 5% significance level, while the other variables were significant at the 0.1% significance level. The excluded variable also had the lowest beta coefficient (-0.035).

$$feed\ supply\ chain's\ CEENE\ v2013\ resource\ footprint\ (MJ_{ex}\ per\ year) - Model\ B = -102327.905 + 329109.657 \times L + 35.760 \times C_{s\text{-}d} + 41.065 \times C_{m\text{-}d} + 19.242 \times BP_d + 96.196 \times BP_y + 15.164 \times R$$

Model C (adjusted R^2=0.989 based on training dataset n = 75) includes five predictor variables, after exclusion of the variable consumed quantity of purchased by-products by young cattle (No. 30), because this variable had the lowest beta coefficient (0.053).

$$
\begin{aligned}
&feed\ supply\ chain's\ CEENE\ v2013\ resource\ footprint\ (MJ_{ex}\ per\ year) \\
&\qquad - Model\ C \\
&\qquad = -62240.651 + 322026.050 \times L + 36.755 \times C_{s\text{-}d} + 42.435 \times C_{m\text{-}d} \\
&\qquad + 23.440 \times BP_d + 15.107 \times R
\end{aligned}
$$

The remaining five predictor variables had following beta coefficients: 1) available on-farm land for feed production (No. 1): 0.629, 2) consumed quantity of purchased concentrates based on one ingredient by dairy cows (No. 10): 0.179, 3) consumed quantity of purchased mixed concentrates by dairy cows (No. 17): 0.240, 4) consumed quantity of purchased by-products by dairy cows (No. 20): 0.180 and 5) purchased quantity of roughages corrected for roughage stock changes (No. 31): 0.219. Because the first variable, available on-farm land for feed production, has a remarkably high beta coefficient compared to the others, this variable should not be removed. Also, data about this variable is very easy to collect, because it often stays constant at a particular farm over many years. Because the other four variables had rather similar beta coefficients, four models with four predictor variables (D-G), in each of which a different variable was removed, were constructed. In model D (adjusted R^2=0.973 based on training dataset n = 75) the variable consumed quantity of purchased concentrates based on one ingredient by dairy cows (No. 10) was excluded.

$$
\begin{aligned}
&feed\ supply\ chain's\ CEENE\ v2013\ resource\ footprint\ (MJ_{ex}\ per\ year) \\
&\qquad - Model\ D \\
&\qquad = 746413.187 + 386769.734 \times L + 22.171 \times C_{m\text{-}d} + 24.390 \times BP_d \\
&\qquad + 15.412 \times R
\end{aligned}
$$

In model E (adjusted R^2=0.960 based on training dataset n = 75) the variable consumed quantity of purchased mixed concentrates by dairy cows (No. 17) was excluded.

$$feed\ supply\ chain's\ CEENE\ v2013\ resource\ footprint\ (MJ_{ex}\ per\ year) - Model\ E = 1553286.839 + 412736.411 \times L + 5.477 \times C_{s\text{-}d} + 22.996 \times BP_d + 15.847 \times R$$

In model F (adjusted R^2=0.970 based on training dataset n = 75) the variable consumed quantity of purchased by-products by dairy cows (No. 20) was excluded.

$$feed\ supply\ chain's\ CEENE\ v2013\ resource\ footprint\ (MJ_{ex}\ per\ year) - Model\ F = -186707.476 + 372684.975 \times L + 38.496 \times C_{s\text{-}d} + 41.908 \times C_{m\text{-}d} + 19.389 \times R$$

In model G (adjusted R^2=0.948 based on training dataset n = 75) the variable purchased quantity of roughages corrected for roughage stock changes (No. 31) was excluded.

$$feed\ supply\ chain's\ CEENE\ v2013\ resource\ footprint\ (MJ_{ex}\ per\ year) - Model\ G = 436471.892 + 286694.700 \times L + 38.664 \times C_{s\text{-}d} + 45.434 \times C_{m\text{-}d} + 38.056 \times BP_d$$

For model A until G, Table D.2 compares the determination coefficient of the validation, and the average, median, minimum and maximum of ($CEENE_{predicted}$-$CEENE_{calculated}$)/$CEENE_{calculated}$. Based on this analysis, we conclude that model C can provide high reliability, while reducing model complexity to five predictor variables. Nevertheless, of all models with only four predictor variables, model G is preferred, because this model has the highest validation R^2, and collection of data about the excluded variable from this model, i.e. purchased quantity of roughages corrected for roughage stock changes, requires relatively extra effort compared to the other variables.

Table D.2 Comparison of the complexity and the reliability of the seven regression models.

	Model A	Model B	Model C	Model D	Model E	Model F	Model G
Number of predictor variables	7	6	5	4	4	4	4
Validation R^2 (n = 28)	0.9802	0.9731	0.9762	0.9215	0.9318	0.9342	0.9640
Average $(CEENE_{predicted}-CEENE_{calculated})/CEENE_{calculated}$	+0.9%	+1.5%	+1.6%	+1.7%	+3.6%	+2.0%	+4.9%
Median $(CEENE_{predicted}-CEENE_{calculated})/CEENE_{calculated}$	+2.3%	+3.0%	+3.6%	+1.8%	+2.7%	+3.7%	+4.3%
Minimum $(CEENE_{predicted}-CEENE_{calculated})/CEENE_{calculated}$	-9%	-11%	-9%	-16%	-16%	-17%	-8%
Maximum $(CEENE_{predicted}-CEENE_{calculated})/CEENE_{calculated}$	+11%	+12%	+13%	+31%	+38%	+15%	+20%

Supplementary material D2: Chemical exergy value of liquid water

For the chemical exergy of liquid water, we follow the approach of Szargut et al. (1988), who calculated a value of 0.05 MJ_{ex} per kg liquid water. Water vapour in the ambient air has been chosen as dead state reference (exergy = 0). Others have chosen for liquid water as dead state reference (Lems et al., 2007).

A partial pressure of 2.2 kPa (relative humidity of 0.70) has been adopted for water vapour in the ambient air at T0. Liquid water has a saturated vapour pressure of 3.169 kPa at T0. The chemical exergy of liquid water or saturated vapour can be calculated as:

$$\Delta EX_{2.2kPa \rightarrow 3.169kPa} = R * T0 * \ln(\frac{3.169}{2.2})$$

with

R: 8.31 J/mol.K

T0 : 298 K (25 °C)

→ 0.90 kJ_{ex} /mol H_2O or 0.05 MJ_{ex}/kg H_2O

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CURRICULUM VITAE

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EDUCATION

2011 – 2015	Doctoral Schools of Bioscience Engineering, Ghent University
2006 – 2011	Master of Science in Bioscience Engineering, option Environmental Technology, Faculty of Bioscience Engineering, Ghent University Master thesis: Life Cycle Assessment of Pangasius production in the Mekong Delta: a cradle-to-farm-gate study. Laureate of the Arcelor Mittal Environment Award 2011.
2000 – 2006	Secondary school, Mathematics – Sciences, Sint-Pietersinstituut, Ghent

PROFESSIONAL CAREER

2011/09/01 – 2015/12/31

Doctoral research in Applied Biological Sciences.

Ghent University, Faculty of Bioscience Engineering, Department of Sustainable Organic Chemistry and Technology, Research group Environmental Organic Chemistry and Technology (EnVOC)

Institute for Agricultural and Fisheries Research (ILVO), Technology and Food Science Unit, Agricultural Engineering

Research subject: 'Development of an exergy-based measurement and evaluation instrument towards sustainable Flemish agriculture and horticulture production.'

Financial support 2011/09/01 – 2011/12/31: Institute for Agricultural and Fisheries Research (ILVO)

Financial support 2012/01/01 – 2015/12/31: Agency for Innovation by Science and Technology in Flanders (IWT) (Grant number: 111297)

2016/04/01 – …

Scientific employee at Ghent University and at Institute for Agricultural and Fisheries Research (ILVO)

TEACHING AND TUTORING EXPERIENCE

2012-2013

tutor of 1 Master student for internship:

Exergie analyse van een anaerobe vergister (English translation: Exergy analysis of an anaerobic digester). Jan-Klaas Blomme, first year of Master of Science in Bioscience Engineering.

2013-2014

teaching exercises Chemical Analytical Methods (Part Organic)
teaching exercises Clean Technology

tutor of 2 Master students for Master thesis:

- Environmental sustainability assessment of milk production and improvement scenarios: resource-driven versus emissions-driven analysis. Carolina Andrea Jaramillo Escalante, International Master of Science in Environmental Technology and Engineering.

- Life cycle assessment of integrated farming in the Mekong Delta: a cradle-to-farm-gate study. Pieterjan Serruys, Master of Science in Bioscience Engineering.

2014-2015 teaching exercises Chemical Analytical Methods (Part Organic)

COURSES

2012

- Doctoral school course *Scientific Computing Schools*
- Doctoral school course *Effective Scientific Communication*
- Doctoral school course *Advanced Academic English: Conference Skills - Academic Posters*

2013

- Course *Practical uncertainty analysis in Life Cycle Assessment* organized by International Life Cycle Academy (ILCA), Barcelona
- Doctoral school course *Communication skills – basics*

2014

- Doctoral school course *Communication skills - module conflict handling*
- Summer School *Economics of Electricity Markets* organized by Ghent University

2015

- Advanced course *Environmental impact assessment of livestock systems* organized by Wageningen University

PUBLICATIONS

INTERNATIONAL JOURNALS WITH PEER REVIEW

Huysveld, S., Schaubroeck, T., De Meester, S., Sorgeloos, P., Van Langenhove, H., Van linden, V., and Dewulf, J. (2013). Resource use analysis of Pangasius aquaculture in the Mekong Delta in Vietnam using Exergetic Life Cycle Assessment. *Journal of Cleaner Production* 51, 225-233.

Huysveld, S., De Meester, S., Van linden, V., Muylle, H., Peiren, N., Lauwers, L. and Dewulf, J. (2015). Cumulative Overall Resource Efficiency Assessment (COREA) for comparing bio-based products with their fossil-derived counterparts. *Resources, Conservation and Recycling* 102, 113-127.

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Nhu, T.T., Dewulf, J., Serruys, P., Huysveld, S., Nguyen, C.V., Sorgeloos, P. and Schaubroeck, T. (2015). Resource usage of integrated Pig–Biogas–Fish system: Partitioning and substituudotion within attributional life cycle assessment. *Resources, Conservation and Recycling* 102, 27-38.

Huysveld, S., Van Meensel, J., Van linden, V., De Meester, S., Peiren, N., Muylle, H., Dewulf, J. & Lauwers, L. Using frontier analysis to investigate cost and natural resource win-wins and trade-offs on dairy farms. *Agricultural Systems:* under review.

CONFERENCE PROCEEDINGS (FIRST AUTHOR)

Huysveld, S., Dewulf, J. and Van linden, V. (2013). Development of an exergy-based life cycle assessment tool to evaluate resource use of dairy farms in Flanders. In Communications in Agricultural and Applied Biological Sciences, Book of Short Abstracts, 78(1), 25. Faculty of Bioscience Engineering, Ghent University.

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Huysveld, S., Van linden, V., De Meester, S., Peiren, N., Muylle, H., Lauwers, L. and Dewulf, J. (2014). Analysis of the overall resource consumption of a Flemish dairy farm using Exergetic Life Cycle Assessment. In Proceedings of the International Conference of Agricultural Engineering (AgEng2014 Zurich) - Engineering for Improving Resource Efficiency. The European Society of Agricultural Engineers (EurAgEng).

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Huysveld, S., Van Wesemael, D., Van linden, V., Dewulf, J., De Campeneere, S. and Peiren, N. (2016). Evaluation of the overall environmental life cycle sustainability of nutritional strategies to reduce emissions of methane and nitrogen at dairy farms in Flanders (Belgium). In Proceedings of the 6th Greenhouse Gases and Animal Agriculture Conference (GGAA2016), *to be published*.

PRESENTATIONS AT CONFERENCES

ORAL PRESENTATIONS AT CONFERENCES

July 8, 2014	Analysis of the overall resource consumption of a Flemish dairy farm using Exergetic Life Cycle Assessment. *International Conference of Agricultural Engineering (AgEng 2014) - Engineering for Improving Resource Efficiency, Zurich, Switzerland.*
July 13, 2015	Comparison of bio-based products and their fossil-based alternatives: an exergy-based life cycle resource efficiency analysis. *SETAC Europe 21th LCA Case Study Symposium jointly organised with 4th International Exergy, Life Cycle Assessment, and Sustainability Workshop & Symposium (ELCAS-4), Nisyros, Greece.*
October 7, 2015	Combining frontier analysis and Exergetic Life Cycle Assessment towards identification of economic-environmental win-win situations on dairy farms. *EXPO 2015 Conference "LCA for feeding the planet and energy for life", Stresa, Italy.*

POSTER PRESENTATIONS AT CONFERENCES

February 8, 2013	Development of an exergy-based life cycle assessment tool to evaluate resource use of dairy farms in Flanders. *the 18th National Symposium on Applied Biological Sciences, Ghent, Belgium*
June 24-26, 2013	Development of an exergetic life cycle assessment (ELCA) tool to evaluate environmental impact of dairy farms in Flanders (Belgium). *the 5th Greenhouse Gases and Animal Agriculture Conference (GGAA2013), Dublin, Ireland*
October 8-10, 2014	Critical analysis of resource-driven Exergetic Life Cycle Assessment (ELCA) versus emission-driven LCA of a dairy farm. Towards ecological-economic analysis. *LCA Food 2014: the 9th International Conference on Life Cycle Assessment in the Agri-Food Sector, San Francisco, United States of America.*

DANKWOORD

Ongeveer vijf jaar geleden ben ik met dit doctoraatswerk begonnen. Enerzijds lijkt het lang geleden, anderzijds is de tijd voorbijgevlogen. Vandaag ben ik blij dat ik kan afronden. Het is een verrijkende periode geweest met veel uitdagingen op persoonlijk en wetenschappelijk vlak. Ik kijk terug op een periode die veel doorzettingsvermogen heeft gevraagd. Gelukkig kon ik onderweg rekenen op de steun van heel veel mensen. Van deze laatste pagina's wil ik gebruik maken om hen te bedanken.

Jo, Veerle en Ludwig, mijn drie promotoren, bedankt voor jullie vertrouwen en de kans die jullie mij gaven om dit doctoraatsonderzoek te starten. Ik moet toegeven, het was soms verre van gemakkelijk om de neuzen in dezelfde richting te krijgen, maar jullie hulp, advies en verfrissende ideeën waren nodig om dit avontuur tot een goed einde te brengen. Jo, onze samenwerking loopt het langst, reeds sinds mijn Master thesis, en ik blijf jouw efficiënte manier van vergaderen en zoeken naar oplossingen heel erg waarderen. Veerle, bij jou kon ik ieder moment onaangekondigd binnenspringen voor zowel de wetenschappelijke als de heel praktische vragen. Ludwig, hoewel we elkaar niet altijd meteen begrepen, zijn we erin geslaagd het economisch-ecologische onderzoeksluik te volbrengen.

Nico, Hilde, Jef en Steven, ook jullie hebben mij in grote mate geholpen tijdens dit onderzoek. Bedankt voor de aangename samenwerking, jullie vernieuwende inzichten en advies. Jef, jou wil ik nog speciaal bedanken voor de uitstekende hulp bij het laatste luik van mijn doctoraatsonderzoek, dat helemaal nieuw voor mij was. Heel erg bedankt voor de heldere uitleg en de bemoedigende woorden.

Jo, Veerle en Nico, ook wil ik jullie bedanken voor het vertrouwen en de inspanningen die ervoor hebben gezorgd dat er voor mij na dit doctoraatswerk nieuwe perspectieven binnen zowel ILVO als UGent zijn.

Stephanie en Johan, jullie wil ik speciaal bedanken voor de snelle en efficiënte hulp bij plots opduikende vragen gaande van statistiek tot voederwaardering.

Karel en Sofie, ik kan niet genoeg benadrukken hoe groot jullie bijdrage tot dit werk is. Toen ik nog maar pas was begonnen, gaven jullie mij de nodige gegevens over jullie bedrijf om van start te kunnen gaan. Daarna volgden nog véle vragen. Hoewel jullie het zeer druk hadden, probeerden jullie mij zo snel mogelijk te helpen en was ik steeds heel erg welkom bij jullie thuis. Als de drukke periode rond de verdediging van dit doctoraat achter de rug is, kom ik zeker nog eens langs zoals beloofd.

Johan, Niels, Marina, Ines en andere LIBA collega's, ook jullie wil ik bedanken voor de open samenwerking. Dankzij LIBA kreeg ik toegang tot een grote hoeveelheid gegevens die nodig waren voor het uitvoeren van het laatste onderzoeksluik binnen mijn doctoraat.

Collega's van EnVOC (UGent) en T&V (ILVO), jullie waren heel belangrijk tijdens de afgelopen jaren. Zonder jullie zou het maar een eenzame tocht geweest zijn. Ik heb heel goede herinneringen aan de ontspannende babbels tijdens de koffie- en lunchpauzes, de T&V personeelsuitstappen, de leuke initiatieven door het T&V feestteam, de EnVOC zomeractiviteiten, de memorabele EnVOC kerstfeestjes en niet te vergeten de *afterwork drinks* in de "Koe". Mijn vaste bureaugenoten op T&V wil ik nog eens speciaal bedanken voor de vele deugddoende babbels en de aangename werksfeer.

De voorbije jaren werden mijn batterijen opgeladen dankzij vele momenten van ontspanning, plezier en samenzijn met vrienden en familie. Vrienden, ik heb geluk met zo'n fantastische bende. Bedankt voor jullie luisterend oor wanneer het nodig was. Mijn familie, bedankt voor de zorg en de vele schouderklopjes. Mama en papa, jullie staan altijd voor mij klaar en helpen graag waar jullie kunnen. Bedankt om al die jaren mijn grootste supporters te zijn. Olivier, jij bent mijn grote broer op wie ik kan rekenen. Veel bewondering voor wat Liesbet en jij reeds hebben verwezenlijkt. Arwen en Flinn, lieve schatttigaards, naar jullie komst heb ik mee uitgekeken. Ook nu kijk ik er telkens naar uit om jullie te zien. Adriaan's familie, ik ben blij om jullie erbij te hebben. Bedankt voor jullie steun en de gezellige momenten. Adriaan, *liefje*, je brengt me aan het lachen en maakt me gelukkig. Je bent mijn beste maatje. Je maakte de laatste loodjes lichter en coachte mij erdoorheen.

Sophie Huysveld

Mariakerke, augustus 2016