Resource efficiency evaluation of the introduction of resource recovery technologies in the bio-based economy

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Pictures from Sophie Sfez (calf in an Indian village, kitchen in rural India), Sofie Van Den Hende (MaB-flocs raceway pond) and Faezeh Mahdavi (wastewater treatment plant of Eindhoven)

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Summary

The world is facing a tremendous resource supply challenge. Securing access to the resources necessary to fulfil society's needs has become one of the priorities of nations around the world. Industry is also playing a key role in the transition toward a more efficient society as their activities are threatened by the competitive use of resources. Several actions can be taken to tackle this challenge: changing our consumption patterns, increasing the resource efficiency of production and use phases of products, and avoiding that resources are dissipated after use. In this context, new technologies are being developed to design more resource efficient products and to recover resources from waste streams. These technologies play a key role in the transition towards a bio-based economy as large amounts of resources are available in organic waste streams. However, these technologies need to be assessed to evaluate whether or not they contribute to increase resource efficiency. One major issue today is that there is no consensus on how to assess the resource efficiency of processes and products and project developers follow a wide range of approaches. This does not allow policy makers to have a clear view on which technologies should be further implemented. Chapter 2 discusses the existing approaches and methods used to evaluate resource efficiency. Resource efficiency is defined as the ratio between the benefits obtained from resources and the impact or amount of resources used. The most challenging step is the determination of this ratio's denominator because a wide range of methods to quantify resource consumption exists and is being used. They can be classified as gate-to-gate or life cycle based methods and subdivided into accounting and impact assessment methods. Each method considers different aspects of resources; thus, no single method aims to answer the same research questions. Therefore, a well-informed choice should be made about which method to use. This chapter provides recommendations to support this choice, as well as the overall evaluation and the valorisation of the resource efficiency ratio in the framework of research and innovation projects. The

implementation of three recommendations presented in Chapter 2 are then tested on case studies in Chapters 3, 4 and 5.

In Chapter 3, the need to upscale newly developed technologies to allow a fair comparison with current technologies is implemented. The resource use and impact from emissions of aquaculture wastewater treatment by microalgal bacterial flocs (MaB-flocs) in an outdoor raceway pond was analyzed using life cycle assessment (LCA). Pikeperch aquaculture wastewater treated at pilot and a hypothetical industrial scale were compared. The integration of the MaB-floc raceway pond in a broader aquaculture waste treatment system was studied, comparing the valorisation of MaB-flocs as shrimp feed and as biogas. Up-scaling improves the resource footprint of the plant (-67%) as well as its carbon footprint and eutrophication potential. At industrial scale, the valorisation of MaB-flocs as shrimp feed is overall more sustainable than as biogas. However, upscaling shows that even at industrial scale, improvements should be made to reduce the energy use of the MaB-floc raceway pond, especially by improving the energy-efficiency of the pond stirring system.

In Chapter 4, two recommendations are implemented: the need to analyze new systems at the substance level and to conduct evaluations at both gate-to-gate and life cycle level. These recommendations are applied to the prospective analysis of the implementation of community digesters co-digesting cow dung and rice straw in rural India (Chhattisgarh). Substance flow analysis on carbon, nitrogen, phosphorus and potassium as well as exergy analysis at the foreground system level are coupled with LCA assessing the impact on resource use, climate change and human health. Indicators of farmers' dependency toward synthetic fertilizers are calculated. Implementing anaerobic digestion barely impacts the dependency of the rural community to nitrogen and phosphorus from synthetic fertilizers (it increases by 0.2% and decreases by 0.9% for nitrogen and phosporus, respectively), but the dependency of farmers on potassium from synthetic fertilizers decreases by 13%. It returns more organic carbon to agricultural land and thus has a potential positive effect on soil quality. Anaerobic digestion can reduce the health impact of the local population by 48%, increase the resource efficiency of the system by 57% and lower the impact on climate change by 12%. New insights are thus provided to decision makers when coupling local and global assessments and conducting analyses at the substance level to assess the sustainability of such systems.

Chapter 5 discusses the need to review the way the resource footprint of resource recovery technologies is assessed. Today, the “zero burden” assumption, which considers that waste streams do not bear any environmental burden, is mostly applied in LCA studies. However, in the context of a paradigm shift from a linear to a circular economy which considers waste as a resource, considering that waste does not have any burden should be re-evaluated. Chapter 5 tests the effect of discarding the “zero burden” assumption on the resource footprint of products obtained from the valorisation of municipal sewage sludge in the Netherlands. A similar approach as followed in the sector of material recycling in which “end-of-life” formulas are applied is followed. These formulas allocate the impact of the different processes of a cascading use of resources among the different products of the process chain. Five formulas are tested on the case study. The formula allocating the impact degressively among the products appears to be the one that reflects best the concept of industrial ecology. The resource use of valorisation products assessed with the “zero burden” assumption is 73% higher than the benchmark products and discarding the “zero burden” assumption makes this difference even larger as the environmental burden of consumer goods production is allocated to them. Therefore, implementing this approach might discourage the implementation of these recovery technologies and further work is necessary to evaluate the added value of this approach for decision making.

Chapter 6 draws conclusions on the potential of the technologies presented in the three case studies to contribute to increase resource efficiency at macro level. The benefits obtained from implementing the four recommendations are also compared to the “efforts” that the LCA community would need to invest to implement them. Finally, perspectives on how resource efficiency evaluation in research and innovation projects could be improved are presented.

Samenvatting

De huidige maatschappij voorzien van voldoende grondstoffen is een grote uitdaging. Het veiligstellen van de toevoer van grondstoffen is dan ook een prioriteit geworden van regeringen wereldwijd. Ook de industrie speelt een belangrijk rol. Grondstoffenschaarste bedreigt de activiteiten van bedrijven en zet hen aan om mee te werken aan een efficiëntere maatschappij. Er zijn verscheidene acties mogelijk om deze problematiek aan te pakken: het veranderen van consumptiepatronen, de grondstoffenefficiëntie van processen verhogen of vermijden dat grondstoffen verloren raken na gebruik. Er worden continu nieuwe technologieën ontwikkeld die op vlak van grondstoffen meer efficiënte producten produceren of toelaten om grondstoffen te winnen uit afvalstromen. Aangezien grote hoeveelheden grondstoffen aanwezig zijn in organische afvalstromen zijn deze technologieën ook een cruciale factor in de transitie naar een bio-gebaseerde economie. Het is belangrijk om deze nieuwe technologieën adequaat te analyseren om na te gaan of ze al dan niet bijdragen aan een hogere grondstoffenefficiëntie. Op dit moment is er geen consensus welke methode gebruikt moet worden om grondstoffenefficiëntie te evalueren. Projectontwikkelaars gebruiken een wijde range aan methodes, waardoor beleidsmakers moeilijk een duidelijk beeld kunnen krijgen van het potentieel van deze technologieën.

In het tweede hoofdstuk worden de bestaande methodes voor de bepaling van grondstoffenefficiëntie besproken. Onder grondstoffenefficiëntie verstaan we de voordelen gelinkt aan het gebruik van grondstoffen gedeeld door de impact, of de hoeveelheid, van deze grondstoffen. Het bepalen van de noemer in deze breuk is de grootste uitdaging, omwille van de grote hoeveelheid methodes om grondstoffenverbruik te bepalen. De methodes kunnen verdeeld worden in gate-to-gate en levenscyclus gebaseerde methodes en in accounting en impact assessment methodes. Elke methode focust op een ander aspect van grondstoffen en formuleert een antwoord op een andere onderzoeksvraag. Het is dan ook cruciaal om bij elke vraag de juiste methode te selecteren. In dit hoofdstuk worden aanbevelingen gegeven bij het

maken van een keuze uit deze methodes. Tevens wordt de berekening van grondstoffenefficiëntie bij onderzoeks- en innovatieprojecten in zijn totaliteit geëvalueerd. In hoofdstuk 3, 4 en 5 worden drie van deze aanbevelingen getest op casestudies.

In hoofdstuk 3 wordt aangetoond dat het belangrijk is om rekening te houden met upscaling bij het vergelijken van nieuwe en bestaande technologieën. Het grondstoffenverbruik en de impact van emissies van een aquacultuur waterzuiveringsinstallatie met microalgal bacterial flocs (MaB-flocs) in een openlucht raceway pond werd geanalyseerd aan de hand van levenscyclusanalyse (LCA). Een vergelijking werd gemaakt tussen afvalwater van een snoekbaars aquacultuur die wordt behandeld in een pilootinstallatie of hypothetisch op industriële schaal. De integratie van deze MaB-floc raceway pond in een breder aquacultuur waterzuiveringssysteem werd geanalyseerd aan de hand van een vergelijking tussen de valorisatie van MaB-flocs als garnalenvoeder en als biogas. De resource footprint van de plant verbetert (-67 %) door upscaling. Er is ook een verbetering op vlak van carbon footprint en eutrofiëring. De valorisatie van MaB-vlokken als garnalenvoeder is op industriële schaal duurzamer dan de valorisatie als biogas. Er zijn echter, ook op industriële schaal, nog verbeteringen nodig op vlak van energieverbruik en vooral de energie-efficiënte van het pond stirring systeem.

In hoofdstuk 4 worden twee van de gedane aanbevelingen geïmplementeerd. Enerzijds wordt gekeken naar het uitvoeren van analyses op substance-niveau en anderzijds naar het uitvoeren van zowel een gate-to-gate als een levenscyclusanalyse. Deze aanbevelingen worden toegepast voor het analyseren van mogelijke vergassers die koeienmest en rijststro zouden vergisten in ruraal India (Chhattisgarh). Een substance flow analyse voor koolstof, stikstof, fosfor en kalium worden samen met een exergieanalyse op het foreground systeem gekoppeld aan de LCA-impact op vlak van grondstoffenverbruik, klimaatverandering en menselijke gezondheid. De afhankelijkheid van boeren van synthetische meststoffen worden tevens berekend. Het implementeren van een anaerobe vergasser heeft weinig invloed op de afhankelijkheid van lokale boeren voor stikstof en fosfor uit kunstmeststoffen (er is een stijging van 0.2% en en een daling van 0.9% voor respectievelijk stikstof en fosfor), maar de afhankelijkheid van lokale boeren voor kalium daalt wel met 13%. Er wordt meer organische koolstof gerecirculeerd naar de landbouwgrond met een gunstig effect voor de kwaliteit van de grond tot gevolg. Anaerobe vergisting kan de gezondheidsimpact op de lokale bevolking verlagen met 48%, de grondstofefficiëntie verbeteren met 57% en de impact op klimaatverandering verlagen met 12%. Het koppelen van lokale en globale analyses levert in deze casestudies nieuwe inzichten over de duurzaamheid van deze systemen.

Hoofdstuk 5 behandelt de methodes voor het bepalen van de resource footprint van technologieën die grondstoffen valoriseren uit afvalstromen. Hedendaags wordt in LCA's vaak geen milieu-impact toebedeelt aan afvalstromen. Bij de transitie van een lineaire naar een circulaire economie wordt afval echter steeds meer gezien als een grondstof, waardoor de aanname dat afval geen impact heeft herzien moet worden. In dit hoofdstuk wordt getest welk effect het weglaten van deze assumptie heeft op de resource footprint van producten gevaloriseerd uit stedelijk afvalwater in Nederland. Een gelijkaardige aanpak als bij de recyclage van materialen is gebruikt. Hierbij worden "end-of-life" formules toegepast. Deze formules alloceren een deel van de impact van de processen over de verschillende producten in een productieketen. In de casestudie in dit hoofdstuk werden vijf formules getest. De formule die het best aansluit bij het concept van industriële ecologie verdeelt de impact degressief over de producten. Bij de veronderstelling dat afval geen impact heeft is het grondstoffenverbruik van de valorisatieproducten 73% hoger dan het gebenchmarkte product. Door het wegvallen van deze veronderstelling wordt het verschil in grondstoffenverbruik nog groter. Deze aanpak zou het valoriseren van grondstoffen uit afvalketens kunnen ontmoedigen. Er is verder onderzoek nodig om de meerwaarde van deze aanpak voor beleidsmakers te evalueren.

In hoofdstuk 6 wordt een conclusie geformuleerd over de potentiele meerwaarde van de technologieën uit de drie case studies om bij te dragen tot een hogere grondstoffenefficiëntie op macroniveau. De voordelen die gehaald kunnen worden uit de vier aanbevelingen worden getoetst ten opzichte van de moeilijkheid om ze te implementeren. Ten slotte worden perspectieven geformuleerd over hoe de analyse van grondstoffenefficiëntie in onderzoek en innovatieprojecten verder verbeterd kan worden.

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List of abbreviations

AADP	Anthropogenic stock extended Abiotic Depletion Potential
AD	Anaerobic Digestion
ADP	Abiotic Depletion Potential
AoP	Area of Protection
APRSCP	Asia-Pacific Roundtable for Sustainable Consumption and Production
BOD	Biological Oxygen Demand
CED	Cumulative Energy Demand
CEENE	Cumulative Exergy Extracted from the Natural Environment
CexD	Cumulative Exergy Demand
CHP	Combined Heat and Power
COD	Chemical Oxygen Demand
DALY	Disability-Adjusted Life Year
DF	Damage Factor
DF*i*	Dependency Factor of substance *i*
DM	Dry Matter
DMC	Domestic Material Consumption
EC	European Commission
EEE	Electrical and Electronic Equipment
EEF	Exergy Efficiency at the level of the Foreground system
EELC	Exergy Efficiency at the level of the Life Cycle
EF	Effect Factor
ELCD	European Life Cycle Database
EPS	Environmental Priority Strategies
EU	European Union
Ex	Exergy
FF	Fate Factor
FoF	Factories of the Future
GDP	Gross Domestic Product
GHG	Greenhouse Gases
GWP	Global Warming Potential
ha	Hectare
HHI	Human Health Impact

HRT	Hydraulic Retention Time
IMP	Impact
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organization for Standardization
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LPG	Liquid Petroleum Gas
MaB-floc	Microalgal Bacterial Floc
MFA	Material Flow Analysis
NGO	Non-Governmental Organization
NMVOC	Non-Methane Volatile Organic Compound
NOx	Nitrogen Oxides
OECD	Organisation for Economic Co-operation and Development
OEF	Orgnaizational Environmental Footprint
P	Phosphorus
PE	Person Equivalent
PED	Primary Energy Demand
PEF	Product Environmental Footprint
PGM	Platinum Group Metals
PM	Particulate Matter
PPP	Public-Private Partnership
Prep-HPLC	Preparative High Performance Liquid Chromatography
Prep-SFC	Preparative Supercritical Fluid Chromatography
RAS	Recirculating Aquaculture Systems
RBR	Recyclability Benefit Rate
RE	Resource Efficiency
REE	Rare Earth Elements
RMC	Raw Material Consumption
RME	Raw Material Equivalent
SBR	Sequencing Batch Reactor
SFA	Substance Flow Analysis
SPIRE	Sustainable Process Industry through Resource and Energy Efficiency
SRT	Solids Retention Time

THP	Thermal Hydrolysis Process
TN	Total Nitrogen
TOC	Total Organic Carbon
TP	Total Phosphorus
TRL	Technology Readiness Level
TSS	Total Suspended Solids
VOC	Volatile Organic Compound
VS	Volatile Solids
VSS	Volatile Suspended Solids
WEEE	Waste Electrical and Electronic Equipment
WWTP	Wastewater Treatment Plant
y	Year

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Chapter 1: General introduction, objectives and outline

1. The challenges of resource consumption

Resources are the basis of our life on Earth. In the context of this thesis, resources are defined as land, energy, primary and secondary raw materials and water (see Chapter 2 for a more detailed discussion on the definition of resources). While some resources such as water are vital for all forms of life, others became essential to our way of living. This is the case of fossil resources used for transportation, in many industrial processes or for the production of heat and electricity. Other examples are metals and minerals which are used in a wide range of applications, from the production of our phones and computers, to the growing of our food. The growing of the worldwide population, which is expected to reach 9.5 billion by 2050, results in an increase of the consumption of resources. The global consumption of materials increased by 60% from 1980 to 2008, and is expected to increase by almost 40% by 2030 compared to 2010, reaching 100 Gt per year (OECD, 2015). The increase of resource consumption is also accentuated by the increase of the world average income induced by economic growth in non-OECD countries such as China and India (WID World, 2017).

The worldwide consequences of this evolution are manifold. First, because most natural resources are exhaustible, it threatens the availability of resources which are essential to human activities in the short and medium terms: by definition, natural resources cannot be "produced" by humans and once depleted, they are difficult to replace or restore (OECD, 2015). For example, some estimations show that the reserves of phosphate rock, a mineral which is 90% used for food production, could be depleted in 50 to 100 years (Cordell et al., 2009), causing a risk of price increase and supply shortage of phosphorus fertilizers in the coming decades which could threaten food security around the world. Moreover, the increase of resource consumption results in a higher demand for key resources which are only abundant at specific places on Earth. The abundance of natural resources in some parts of the world has been shown to be a determinant of conflicts, which can be induced by both the poverty of the population ("grievance") and the lure of profit ("greed") in countries with unstable or corrupt governance (Welsch, 2008). A historical example is oil, which 90% of proven reserves are located in 15 countries and which have been shown to increase the likelihood of conflict in the areas where they are located (Lujala & Rustad, 2011). The aforementioned example of phosphate rock, which reserves are mainly located in three countries (Morocco, China and the US) shows that there could be new geopolitical tensions associated with the access to other resources in the near future.

Another consequence of an increase of natural resource consumption is the increase of the environmental impacts associated with resource extraction and consumption. First, it results in larger amounts of emissions from resource use. Higher consumption of fossil resources causes larger amounts of direct greenhouse gas emissions (GHG) from industry and transportation. GHG emissions are the main cause of climate change, which consequences are e.g., desertification, sea level rise and extreme climatic events. These consequences threaten the survival of some populations and create socio-political tensions to access vital resources such as water and arable land. Moreover, larger resource consumption induces larger amounts of hazardous substances that need to be disposed, e.g., heavy metals that are landfilled and introduce a risk of soil contamination via leaching. The higher consumption of resources has also other environmental impacts. Some resource extraction processes have impacts on biodiversity (e.g., induced by deforestation to access arable land (Vieira et al., 2008)), and the quality of air and groundwater due to airborne emissions and aquifer contamination, respectively. The latter impacts can be caused by some fossil and mineral resource extraction processes (Ernst, 2012).

2. The way to a sustainable use of resources

Despite this pessimistic picture, there are ways to overcome this challenge. The first measure that we can think of when talking about overpopulation threatening our access to resources is to reduce the world population. This was already stressed by the biologist Paul Ehrlich in 1968 in its book “The population Bomb”, in which he proposes several measures to reduce the world population’s growth rate. However, the issue of resource shortage is not only the consequence of the size of the population, but also of its way of life. Wiedmann et al. (2015) showed that the material footprint per capita in the USA is twice as high as in China and more than four times higher than in India. Therefore, extending the high quality of life of all around earth won’t be possible if the resources consumed to achieve these living standards are not drastically reduced by implementing sustainable technologies. One expected consequence of shifting towards a more sustainable society is that people’s welfare and education would increase and result in a stabilization or decrease of the population, as it can be observed in European countries. Therefore, efforts related to both population planning and the sustainable use of resources should be pursued. To focus the effort, studies have been conducted to identify the “hotspots” of our way of life. This is for example the case of the WBCSD studies on the lifestyle material footprint of different countries. They show that the lifestyle “hotspots” differ from one country to another. While in the USA, most of the material footprint is due to the transport sector (24% of the footprint, especially from personal transport in individual cars), housing (22% of the footprint, especially from electricity and heat consumption) and services (21% of the footprint, especially from restaurant, catering and education), the material footprint of Brazil is mainly due to food (36% of the footprint, especially from meat consumption) and housing (23% of the footprint) (WBCSD, 2015a, 2015b). However, housing and food both always highly contribute to the material footprint of the four countries analysed by the WBCSD Sustainable Lifestyles reports (USA, India, China and Brazil). The impact from food is related to the consumption of meat and the losses in food waste while the impact from housing is related to the type of fuels used (e.g., inefficient biomass fuel used in rural India) and the size of the houses, typically in the USA. These figures show that there is still room for improving the resource use of our activities and move towards a sustainable use of resources. The concept of sustainable development was first introduced in 1987 by the World commission on Environment and Development, also called the Brundtland commission and was defined as a "development which meets the needs of current generations without compromising the ability of future generations to meet their own needs". Several complementary ways can be followed to reach this sustainability and are discussed below. They are illustrated by examples around the world and with a specific attention on the initiatives undertaken in the EU.

2.1 Changing consumption patterns

In the last decade, it has been stressed that behavioural change at the level of individuals will be essential to reach the sustainability targets defined worldwide (Baum & Gross, 2017; Roy & Pal, 2009). This is particularly the case in the richest regions of the world such as Europe, where for example in 2014, households were responsible for 24.8% of final energy consumption of the EU-28 (Eurostat, 2017). The focus of awareness campaigns towards households around the globe was so far mainly made on the energy and water savings. The experience from these campaigns, which have been running for several decades, has shown that measures applied and perceived by households as contributing to save energy and water are still more symbolic (e.g., taking shorter showers and turning off the light in unoccupied rooms) than significant (e.g., cancelling holidays at the other side of the globe and taking more public transportation) (Jensen, 2008). Despite this fact, these campaigns have the merit to contribute to make people aware of the link between energy and water use and their impact on the environment. This is less the case for other types of consumption behaviours such as material good and food consumption, which link with natural resource consumption are less understood or being ignored. This is accentuated by the constant exposure of people to advertisements displayed in the streets, on TV or via other means. This encourages (over-)

consumption and makes conspicuous consumption a symbol of wealth and higher social status (Roy & Pal, 2009). Pro-environmental behaviours are not being integrated in our lifestyles and even if symbolic actions do contribute to save resources, more radical changes of habits which are not only related to direct water and energy use are necessary to make the society sustainable. This has to go hand in hand with raising people's awareness on what do or do not contribute to increase well-being. While the goal of the society is to increase the well-being of its citizen, the fact that higher consumption patterns result in higher well-being is being questioned in literature. Based on Tukker et al. (2014), Fig. 1 shows that the Human Development Index (which takes three dimensions into account, i.e., long and healthy life, knowledge and decent standard of living) and the Happy Life Years (which focuses on experienced well-being and its duration) level off at a certain level of material use.

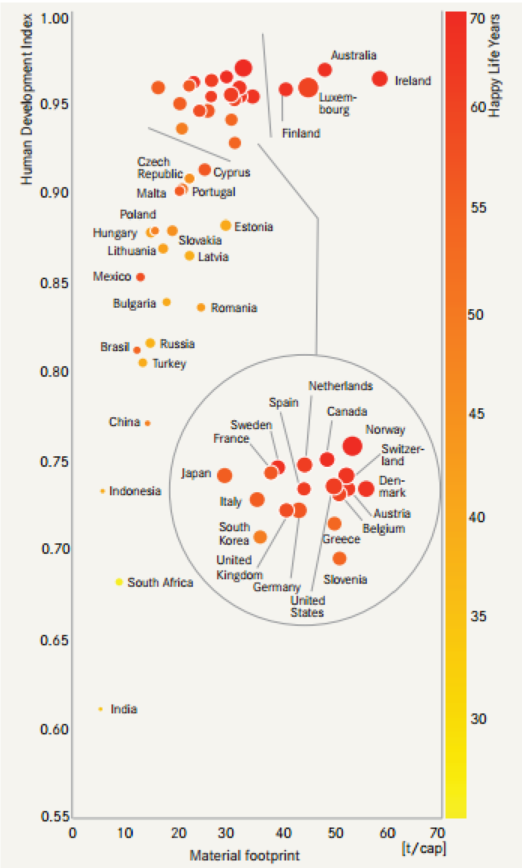

Figure 1: *Dependence of human development index (y axis) and happy life years (colour) on per capita material consumption. The dots are sized according to the purchasing power parity GDP per capita of the country. Retrieved from Tukker et al. (2014).*

This graph suggests that at a certain level, consumption does not contribute anymore to increase people's well-being.

This is widely acknowledged by organisations around the world, especially since the International Expert Meeting on the 10-Year Framework of Programmes for Sustainable Consumption hold in Morocco in 2003 during which it was noted that the real challenge is now to move from "the more generic to the specific and focus on implementation." Sustainable consumption is part of the 17 sustainable development goals of the United Nations, which launched several related programs such as the "Sustainable lifestyles and education programme" and the "Consumer information" programme (UN, 2017). Measures to orientate consumers towards sustainable consumption are more and more integrated in regional and national policies. In the last decades, many studies have been published to help policy makers in this way, e.g., by reviewing past initiatives (BIO Intelligence Service, 2012; OECD, 2008) and trying to explain the link between households socio-economic characteristics and attitude towards consumption (Jensen (2008); Martinsson et al. (2011); OECD EPIC project). Several different policy instruments are used worldwide, e.g., regulatory (e.g., standards and bans), economic (e.g., subsidies) and communicative (e.g., awareness campaigns) instruments.

The challenge of unsustainable consumption patterns has become a major aspect of the EU policy since the publication of the 2008 Action Plan for Sustainable Consumption and Production (EC, 2008). One major initiative following the conclusions of this Action Plan was the launch of the Product Environmental Footprint (PEF) and Organizational Environmental Footprint (OEF), which aim is to develop a "common methodological approach to enable Member States and the private sector to assess, display and benchmark the environmental performance of products, services and companies" with the final goal to orientate consumers towards sustainable products (EC, 2013). Product labelling to orientate consumers towards more resource efficient products has been developed for many years worldwide, especially for household appliances (e.g., see the Energy Star label in the USA and the Energy labelling in the EU). It is also one of the projects of the Asia-Pacific Roundtable for Sustainable

Consumption and Production (APRSCP), which is evaluating the feasibility of developing cooperative eco-labelling between China, Japan and South Korea. The attitude of consumers towards repair should also be changed by providing information on product repair possibilities. In the framework of the EU Action Plan for the Circular Economy, the EC will evaluate the possibility to propose requirements on repair information provision (Brunner & Rechberger, 2015).

2.2 Increasing the resource efficiency of the production and use phases

The production step of products and services is a key step to be optimized to save resources. There are several ways to do so. One is to change the feedstock used in the production. A typical example is the use of biomass to replace fossil resources to produce plastics or energy. The process used to produce the desired product can also be changed while keeping the same feedstock. For example for a chemical process, more efficient catalysts can be used or solvents can be replaced by another separation process, e.g., by using supercritical CO_2. The conditions under which a process is conducted can be changed to increase its efficiency, e.g., by changing the temperature, the pressure and the mixing rate. The technology can also be changed through process integration, i.e., by using flows released by processes to provide the conditions or materials necessary for other processes. For example, energy integration is widely implemented in industry. It consists in using the heat of flows that need to be cooled to heat flows that need their temperature raised.

The resource efficiency of production is the focus of many initiatives worldwide. The private sector itself is an active actor in the transition towards a more resource efficient industrial sector for several reasons. With the fluctuating price of resources and the increasing international competition, the increase of resource efficiency is a mean to reduce production costs and thus increase competitiveness (Brunke et al., 2014; CEFIC, 2015). It is also a way to improve the brand and customer reputation and foster innovation (IDEA, 2014). However, initiatives from the public sector are still necessary to foster the implementation of more resource efficient production routes. Industrial resource efficiency is one of the main focuses of the United Nations Industrial Development Organization (UNIDO) which, together with UNEP, developed a Resource Efficient and Cleaner Production programme which aims to raise awareness and train experts in resource efficient methods and technologies. The APRSCP is also focusing on this challenge by pushing the upscaling of Resource Efficiency and Cleaner Production practices in Asia through public-private partnerships and training sessions. Increasing the resource efficiency of production is also a major focus of the EC, which built several Public-Private Partnerships (PPP) implemented though research and innovation calls under the Horizon 2020 funding program. Two major PPPs aiming at increasing resource efficiency of the manufacturing sector are the Sustainable Process Industry through Resource and Energy Efficiency (SPIRE) and the Factories of the Future (FoF) PPPs.

The increase of the resource efficiency of a product use phase is also key. This can be achieved at the design step, e.g., by producing products with longer lifetimes or appliances which require less energy for their functioning. The EC Ecodesign Directive sets rules to improve the resource efficiency of products such as appliances. It lists up ecodesign requirement parameters that should be selected in product-specific regulations, when appropriate (e.g., consumption of energy, minimum guaranteed lifetime and reparability) (EC, 2009a). In the EU Action Plan for the Circular Economy, the EC also plans to evaluate the possibility of an independent testing programme on planned obsolescence (EC, 2015a). The resource efficiency of the use phase can also be increased by re-designing the product in function of the service it provides rather than for the product itself. For example, new systems based on the payment of a service instead of the purchase of a tangible product are arising, e.g., the payment of washing cycles instead of the purchase of a washing machine, or the payment of kilometres instead of tires for cars. Such systems encourage the production of products with a longer lifetime, which potentially increases their resource efficiency. These latter systems are also driven by consumption patterns and the demand of some consumers for longer lasting products.

2.3 Avoiding resource dissipation by implementing the concept of "industrial ecology"

One major challenge our society is facing is the dissipation of non-renewable resources in the anthroposphere and the natural environment. Large amounts of resources are lost at different stages of the products' life cycle. This can be illustrated by the substance flow analysis of phosphorus in the EU-15 conducted by Ott and Rechberger (2012). The study shows that the EU is essentially dependent on imports of phosphorus to fulfil its needs. Most of the phosphorus is used in agriculture to produce fodder and food but only 26% of the consumed phosphorus (4.7 kg P capita^{-1} year^{-1}) reaches the consumer. The remaining fraction is accumulated in agricultural fields (2.9 kg P capita^{-1} year^{-1}), lost in landfills (1.4 kg P capita^{-1} year^{-1}) and in the hydrosphere via e.g., landfill leaching and runoff from agricultural land (0.6 kg P capita^{-1} year^{-1}). Therefore, while the EU is still largely importing phosphorus, its self-sufficiency in phosphorus could be increased by developing more resource efficient agriculture, recycling and recovery systems at the waste and wastewater management steps to reduce phosphorus dissipation. Similar studies have been conducted around the world for other resources such as copper (Tanimoto et al., 2010), iron (Yan & Wang, 2014), chromium (Timmermans & Van Holderbeke, 2004) and forest resources (primarily wood and wood by-products; Cheng et al. (2010)). These studies highlight the need to develop more integrated industrial systems, where the waste or by-product of one process or industry is used as an input in another process or industry, as it is done with resources in natural ecosystems. This is the so-called concept of industrial ecology. Industrial ecology is based on the analysis of materials and energy flows within the anthroposphere and aims to avoid that these flows leave the anthropogenic system. Many initiatives are being launched to apply these principles around the world. One example is industrial symbiosis implemented in eco-industrial parks. They are based on inter-organizational networks and consists in exchanges of waste, by-products and energy flows and share of resources and information between industries and enterprises located in a defined area (Lambert & Boons, 2002). One of the first implementation of the principle of industrial symbiosis is the eco-industrial park of Kalundborg in Denmark, presented in Fig. 2. The park is built around a power plant and six main production plants that exchange up to 29 different energy, water and material flows (Fig. 2). Several programmes have been launched to promote the implementation of parks, e.g., the eco-industrial park demonstration programme in China and the national strategies for eco-industrial parks in Thailand and the Philippines (Lehtoranta et al., 2011). In the EU, industrial symbiosis is one of the focuses of the SPIRE PPP and is presented as one of the means to improve the re-use of raw materials in the "Roadmap to a Resource Efficient Europe" (EC, 2011b). The concept of industrial symbiosis has been extended to urban areas with the concept of urban symbiosis in which urban metabolism is analyzed to look for ways to optimize exchanges of energy and materials between industrial and urban areas (e.g., see Geng et al. (2010)).

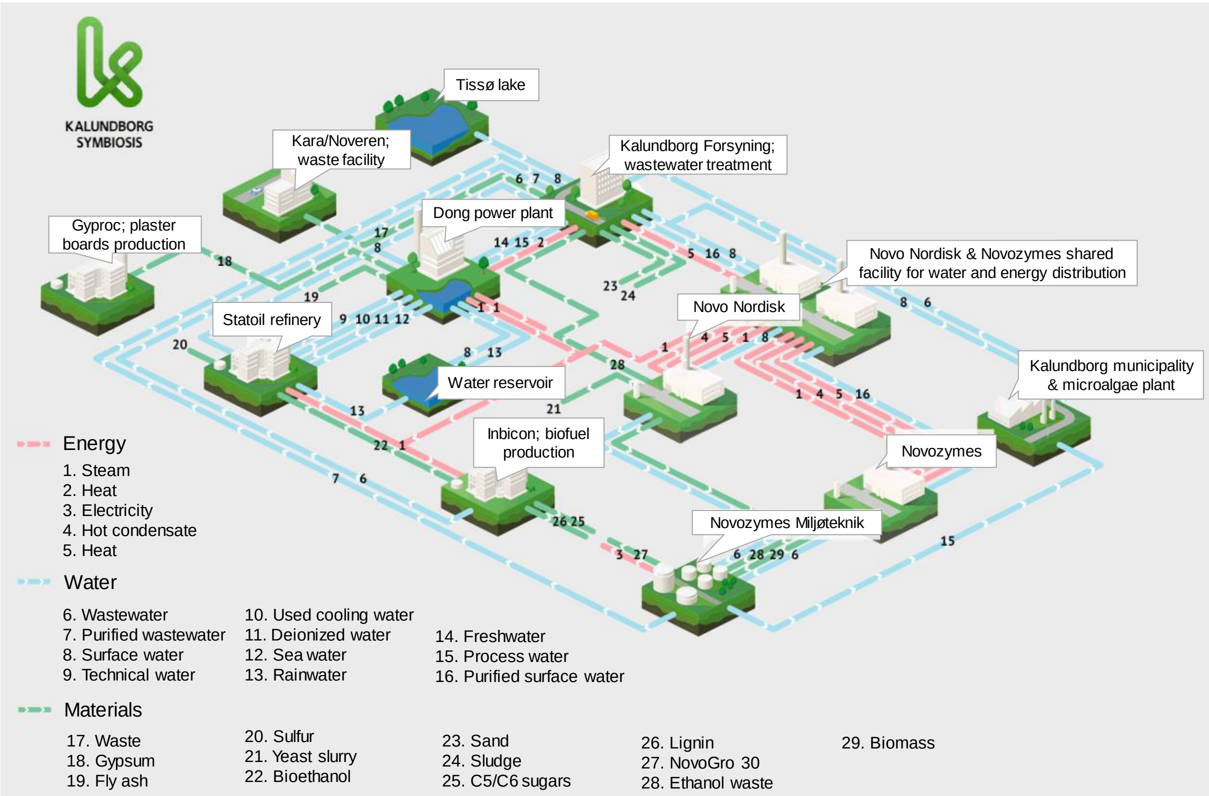

Figure 2: *Eco-industrial park of Kalundborg, Denmark. Adapted from Kalundborg Symbiosis (2017).*

Industrial ecology also embeds all the initiatives related to resource recovery and recycling from waste flows generated by households and industry. The first technologies now considered as resource recovery and recycling technologies were not implemented with the first aim to recover resources but to avoid the emissions of harmful emissions in the environment. Therefore, they were implemented as “end-of-pipe” solutions as they were implemented as the last stage before the disposal of the waste flows. In the 90s, to reduce the volume of solid waste treated and because some waste streams were identified as containing large amounts of recyclable materials (e.g., paper and plastics), sorting of waste at the source and specific recycling technologies started to be implemented. From the second half of the 2000s, the development of recycling technologies started to be driven by the value that could be obtained from the waste rather that the volume of waste. Very specific technologies to recover resources from targeted waste streams such as Waste Electrical & Electronic Equipment (WEEE) and end-of-life vehicles started to be developed. The development of the wastewater treatment sector did not follow the same path and until recently mainly stayed as an end-of-pipe solution: wastewater treatment plants are implemented just before releasing the clean water in water bodies and after mixing both household and wastewater streams with very different compositions. It results in the dilution of some valuable substances in the mixed stream, which makes them more difficult to recover, and of pollutants, which makes them more difficult to remove. Decentralised approaches to treat selected wastewater streams in a more effective way are being implemented, e.g., the separation of urine and faeces at households using dry toilets or small scale mechanical-biological treatment plants allowing the on-site reuse of clean water (Libralato et al., 2012).

Several policy instruments exist to foster recovery and recycling, especially in the field of solid waste management. Worldwide, the Extended Producer Responsibility, a policy approach which gives producers a large responsibility for waste treatment and disposal has proven to be an efficient tool to increase the recycling rate of targeted products (OECD, 2006). Today, worldwide policies to promote recycling are being implemented under constraints for resource availability and, in countries where waste management is still at its infancy, by environmental sanitation. In China, the Circular Economy Promotion Law was promulgated in 2008 and encourages the implementation of reuse and recycling technologies, e.g., the implementation of water reuse systems, the building of grid-connected power generation projects with waste heat and steam and the development of biogas production from agricultural waste (Republic of China, 2008). In the USA, the Sustainable Material Management Program sets a strong preference for resource conservation rather than disposal by aiming to decrease the disposal rate of materials recycling (US EPA, 2015). Moreover, the USA developed a specific strategy on critical raw materials in the Critical Materials Strategy in which research and development for developing recycling technologies specific for waste containing critical raw materials plays a key role (US Department of Energy, 2010). In India, the Indian Energy Policy promotes the valorization of agricultural waste as biogas to contribute to the energy self-sufficiency of India (Government of India, 2006). In the EU, the EC published the Waste Framework Directive in 2008, which defines the so-called “waste hierarchy”. It gives the priority order of measures and waste treatments that should be implemented by member states, i.e., waste prevention, preparing for reuse, recycling, recovery and disposal. Many initiatives are being undertaken by the EC to foster the implementation of this hierarchy and increase the rate of re-used and recycled materials in the EU, several of which are introduced in the 2015 EU Action Plan for the Circular Economy. Examples are the measures to foster the development of a market for secondary raw materials with the introduction of quality standards for by-products and regulations for “end-of-waste” criteria, and the development of guidance documents for a better integration of water reuse in water planning and management (EC, 2015a). Moreover, as part of the action plan, a revision of the Waste Framework Directive was proposed in 2015 and defines new targets related to recycling and landfilling to be reached by 2030, i.e., a common EU target for recycling 65 and 75% of municipal and plastic waste, respectively, and for reducing landfilling to a maximum of 10% of municipal waste (EC, 2015b). The Ecodesign Directive also encourages reuse and recycling by including eco-design requirement parameters such as the incorporation of used components and the use of materials issued from recycling activities (EC, 2009a).

3. The need for the assessment of innovative technologies and products

To implement the different measures presented in the previous section, new processes, products and services are being developed. It is especially the case in the sectors of secondary resources management such as the waste and wastewater management sectors. This is because systems and technologies aiming to use secondary material and energy flows to produce new products or services play a key role in the three measures presented in section 2. Developing more goods that are reparable and which end-of-life can be delayed and systems that encourage households to buy second hand products can contribute to change consumption patterns (section 2.1). Moreover, the resource efficiency of the production steps can be highly increased by material and energy integration resulting in a re-use of resources in processes (section 2.2.). Finally, the principle of industrial ecology is fully based on the concept of secondary resources valorisation (section 2.3). The transition towards a bio-based economy especially relies on the development of such technologies in the waste and wastewater treatment sectors. Bio-based economy is based on "production paradigms that rely on biological processes and, as with natural ecosystems, use natural inputs, expend minimum amounts of energy and do not produce waste as all materials discarded by one process are inputs for another process and are reused in the ecosystem" (EC, 2011a). Therefore, to make a transition towards a more bio-based economy, both technologies using organic waste streams as feedstock to produce resources and technologies based on biological processes should be developed. Such technologies are already implemented but there is still an untapped potential. One example of organic waste streams that could be better valorized is food chain waste (household and slaughterhouse waste) and sewage streams, from which 21% of the phosphorus is recovered today in the EU-27. The unrecovered phosphorus from these streams has the potential to replace 40% of the mineral P fertilizer used in crop production (Buckwell & Nadeu, 2016). More specifically, unrecovered phosphorus from sewage streams could cover 16% of the demand for P fertilizer in Europe (Buckwell & Nadeu, 2016). The same goes at a global scale: if collected globally through the implementation of innovative technologies, phosphorus from urine and faeces could cover 22% of the global demand for phosphorous (Mihelcic et al., 2011). The potential for nitrogen recovery in the EU-27 is lower as today only 17 to 23% of nitrogen contained in sewage and household waste is recovered but the unrecovered nitrogen has the potential to replace 14% of the mineral N fertilizer used in crop production (Buckwell & Nadeu, 2016). The worldwide potential of energy production from organic waste is also untapped. Today, bio-energy covers 10% of the global energy consumption, mainly through the burning of firewood, dung and charcoal (Haberl et al., 2010). Haberl et al. (2010) estimated that by 2050, the energy available in biomass to produce bio-energy could almost be multiplied by 4, mostly thanks to the valorization of organic waste streams which would then contribute 61% to the total bio-energy produced. Several waste streams have been identified globally as partially untapped for the production of bio-energy, especially livestock waste in Asia to produce biogas. Today, cow manure digestion only represents 27% of its potential in India, only 4% in Nepal, 19% in rural China and in Bangladesh, 80% of cow manure could be made available for the production of biogas (Bond & Templeton, 2011). Globally, Surendra et al. (2014) estimated that 5818 PJ year^{-1} of biogas could be produced from animal waste and human excreta, which could cover 1% of the global energy consumption. In the EU, the total additional feedstock that could be made available by 2030 (manure, agricultural residues, organic waste and sewage sludge) could contribute to produce 470 to 890 PJ of biogas per year, mostly from the valorization of liquid manure and organic waste. It represents 0.6 to 1.3% of the European energy consumption while today around 0.3% of energy needs are covered by the production of biogas from these streams (EC, 2016).

The figures above show that first estimates point out the potential of technologies processing organic waste streams to contribute tackling the global resource supply challenge. However, there is a need to assess if the introduction of such technologies really reduce resource consumption, as new processes that might seem more resource efficient might actually consume more resources. A past example in the bio-based economy is the worldwide development of the first generation biofuels in the 90s, which were driven by several objectives such as reducing oil price volatility, fostering energy self-sufficiency and reducing the GHG emissions from the transport sector (Bourguignon, 2015). Several policies were implemented around the world to foster the development of biofuel production. However, in the mid-2000s critics regarding their sustainability started to arise from NGOs and the scientific community. In addition to the consequences on food price volatility, the issues related to the competitive use of resources to produce biofuels and food were highlighted, especially land and water (Bourguignon, 2015). Moreover, the scientific community started to question the energy efficiency of the production of first generation biofuels and several assessment studies showed that their sustainability depends on many criteria and cannot be always proven (de Vries et al., 2010; Ponton, 2009). These concerns lead to the revision of policies around the world and new rules are now being set to support and develop biofuels from other feedstock, especially secondary resources (Sorda et al., 2010). This example highlights the need to measure the sustainability of new systems before their full implementation, especially before deciding of policy measures to encourage their development. Based on the Brundtland definition, sustainability is defined in terms of three pillars: environment, social and economic. Because these three pillars are equally important, many intents are being made to develop an assessment method able to cover them all. One example is Life Cycle Sustainability Assessment, which intends to combine environmental Life Cycle Assessment (LCA), life cycle costing and social LCA to obtain one sustainability indicator (Klöpffer, 2008). However, there is still no consensus on how these three pillars should be integrated and they are still commonly assessed separately. In this thesis, the focus is put on the assessment of environmental sustainability, which evaluates the impact on four areas of protection (i.e., entities that we want to protect): human health, natural resources, natural environment and man-made environment (De Haes Udo et al., 1999). Note that these areas of protection are subject to debate today as some overlap with social and economic aspects: human health is not only an environmental problem but also a social issue while resource consumption can be seen as a purely economic issue.

The need for evaluating the environmental sustainability of new systems and technologies is more and more included in the EU legislation and several EU Directives already stress the need for assessment studies. The Waste Framework Directive stipulates that some waste streams can depart from the waste hierarchy if it is justified by "life-cycle thinking on the overall impacts of the generation and management of such waste". Moreover, the Directive stresses the fact that more link should be made between environmental impacts and economic valuation of waste. The Directive on Waste Electrical and Electronic Equipment (WEEE) stresses the fact that the environmental performance of all the operators involved in the life cycle of EEE should be improved, and that the whole life cycle of the product should be taken into account when optimizing reuse and recovery through product design (EC, 2012). Similarly, the Directive on Packaging and Packaging Waste requires Member States to conduct life cycle assessment studies to justify the hierarchy applied among reuse, recycling and recovery (EC, 1994). Several methods exist to assess the impact of new systems and technologies on resource consumption and emissions and are described in the following section.

4. Overview of evaluation methods

In this section, environmental sustainability assessment methods are divided into two types of methods: the ones focusing on the system or process under study (called here methods at the process level) and the ones evaluating the performance of the whole product life cycle (methods at the life-cycle level). Examples of applications are given in the sector of waste management, which is, as mentioned earlier, a sector that plays a key role in the strategies to increase resource efficiency worldwide.

4.1 Evaluation methods at the process level

These evaluation methods study the flows of energy and/or substances and materials within the studied system or process, also called the foreground process. They can be referred to as gate-to-gate analyses. In this section, four accounting methods (material and substance flow analyses, energy, exergy and emergy analyses) and one impact assessment method (risk assessment) are presented.

4.1.1. Material and substance flow analyses

Material and substance flow analyses (MFA and SFA, respectively) are preliminary steps to impact assessments, but are also used to conduct process and system efficiency studies on their own. MFA and SFA consist in a thorough analysis of the fate of materials (structure made of a large number of combined substances, e.g., wood and plastic) and substances (elements and small molecules, e.g., CO_2, Pb, Zn), respectively, within the studied system and are used to calculate performance indicators. Note that MFA and SFA are presented here as evaluation methods at the process level as they are mostly conducted at this level, but they can also go beyond processes and be conducted at life cycle level (see 4.2.).

In the waste management sector, MFA is mainly conducted to have a macro-scale vision of waste management and mainly used in waste management planning. Examples of MFA indicators are recovery or recycling rates of specific materials, volume of waste to landfill (Arena & Di Gregorio, 2014) or stock of material in landfill. Similar indicators, called "resource efficiency indicators", were used in the revision of the targets set by the EU Waste Framework Directive (EC, 2015b). SFA is used in the waste management sector to reach two goals (Brunner & Rechberger, 2003): 1) ensure that a limited amount of hazardous substances is emitted to the environment during the final disposal of waste; 2) ensure that hazardous substances do not accumulate in recycled materials or that recycling or reuse processes are not associated with harmful emissions to the environment. When considering waste as a resource, a third goal can be defined: identify where valuable substances accumulate in order to optimize their recovery. SFA is mainly used at process level, e.g., to track precious “trace elements” from a specific type of waste (Chancerel et al., 2009) and to compare possible treatment technologies for specific waste streams (Arena & Di Gregorio, 2013; Cascarosa et al., 2013). However, SFA has also been used to track substances at regional or sectorial level (e.g., in Arena and Di Gregorio (2014) and Vyzinkarova and Brunner (2013)). Examples of indicators are the amount of a specific substance landfilled or in recycled products (Arena and Di Gregorio (2014); Vyzinkarova and Brunner (2013)), the velocity of the consumer stock evolution (Vyzinkarova & Brunner, 2013) or the carbon conversion efficiency (Arena et al., 2011).

One advantage of MFA and SFA is that they are relatively easy analyses to understand. Moreover, trace elements are often the focus of the analysis whereas they are often neglected when other methods are applied. Another advantage is that MFA/SFA studies are easily comparable with one another. Most of the limitations associated with MFA/SFA rely on their practical application (e.g. when studying a complex system, conducting a MFA/SFA in an excel file can be a challenge and source of many errors), data availability and the interpretation of the results as it requires a thorough understanding of the chemical and physical processes occurring within the studied system or process. For example, the recovery potential of metals after thermal treatments depends on which form they remain after the treatment: gasification allows recovering iron and copper under metallic form but not combustion after which metals are available in their oxidized form (Arena & Di Gregorio, 2013). A simple mass balance without any further understanding of the process would result in considering oxidized metals as recoverable as non-oxidized metals.

4.1.2. Energy analysis

An energy analysis is the analysis of all the energy flows going through and stocked within a system. There is no clear methodology defined to conduct energy analysis. Different ways of accounting for energy consumption and generation can be found in literature, and most of them are gathered behind the common term “energy balance”. For example in the waste management sector, some studies only evaluate the balance between the chemical energy embedded in the input flows (e.g., “feedstock energy” in Arena et al. (2011)) and the output products, others calculate a ratio based on input energy from transportation and processing and output energy from the waste-treatment by-product (e.g., Comparetti et al. (2014)) and some mix both (Cascarosa et al., 2013). Another approach also considered as an energy analysis converts all input sources of energy (electricity, gas, fuel etc) into primary energy and compares them to the energy embedded in the output products (Cimpan & Wenzel, 2013; Wallmann et al., 2008). However, the fact that all input sources of energy are converted into primary energy carriers goes beyond the process level as it also accounts for the primary resources necessary to produce these energy flows. Many indicators based on energy balance can be found in literature: lost and available feedstock energy (Arena & Di Gregorio, 2014), Primary Energy Input to Output (Pöschl et al., 2010), electricity efficiency (De Meester et al., 2012), etc.

One advantage of energy analysis is that it is easy to understand and accessible to non-experts. However, the lack of harmonization of the methodology does not always allow comparing one study to another. Moreover, energy analyses based on the conversion of energy flows in terms of embedded energy (or feedstock energy) and primary energy require the use of conversion factors or specific formulas which can have high impacts on the results of the study. Finally, energy analysis is not suitable for comparing a technology which delivers energy to the market (e.g., anaerobic digestion) to one which does not (e.g., composting).

4.1.3. Exergy analysis

Exergy is the maximum theoretical work that can be obtained from a system brought to equilibrium with the surrounding environment. It is based on the second law of thermodynamics, which states that the entropy of an isolated system can only increase over time or remain the same in ideal cases. While energy is never destroyed, exergy is always destroyed in irreversible processes (Fig. 3). Exergy informs on the quantity but also on the quality of the energy embedded in process flows. The first step of an exergy analysis is to conduct a thorough material, substance and energy accounting. Each flow is then expressed in terms of exergy based on databases as provided by Szargut (2005) or on calculations using the composition of materials. Two main types of exergy efficiencies can be calculated: the functional and universal exergy efficiencies. The functional exergy efficiency is the ratio between the exergy of the product of interest and the exergy inputs of the system. The universal exergy efficiency is the ratio between the output and input exergy flows.

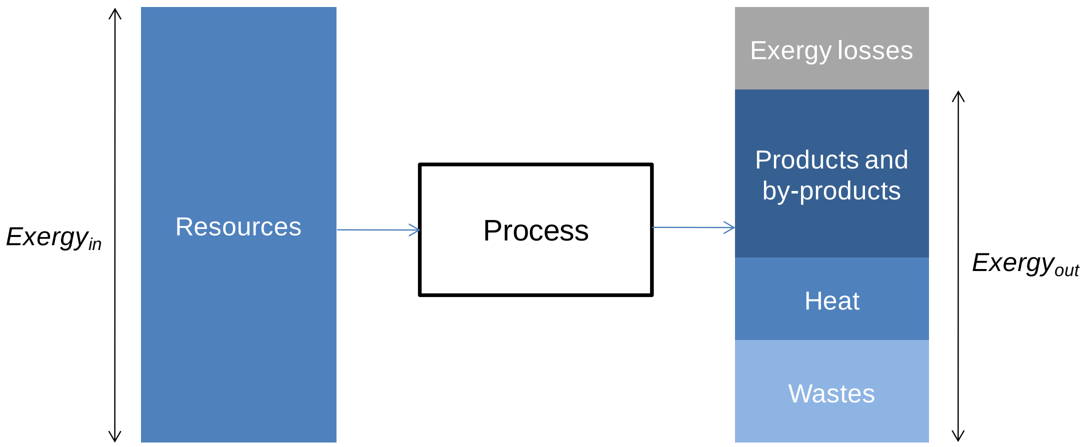

Figure 3: *Schematic representation of a process input and output exergy flows (based on Dewulf et al. (2008)).*

The suitability of exergy analysis to assess the efficiency of waste management systems has already been shown in the early 2000s (Dewulf & Van Langenhove, 2002) and regularly highlighted by the scientific community (Brunner & Rechberger, 2015; Hiraki & Akiyama, 2009; Zhou et al., 2011) but few practitioners are using this method. Exergy analyses found in literature are applied to compare different treatment scenarios for a wide range of waste types, e.g., food waste (Vandermeersch et al., 2014), aluminium waste (Hiraki & Akiyama, 2009), municipal organic waste (De Meester et al., 2012) and municipal solid waste (Xydis et al., 2013; Zhou et al., 2011).

The usefulness of exergy analysis compared to energy analysis has been pointed out in the BREF document on energy efficiency where it is stated that "exergy analysis, although less used and more complex, is more useful because it points directly to where energy can be saved" (EC, 2009b). The main advantage of exergy compared to energy is its ability to translate both quantity and quality of energy. Moreover, it expresses all inventory flows (i.e. mass and energy flows) in the same unit, i.e. MJ_{exergy}. The limited use of exergy analysis in the industry seems to be related to its seeming complexity and to the fact that additional data have to be collected (i.e. exergy content of inputs and outputs). In practice, exergy analysis is not more complex than converting the flows in term of primary energy. Tables on exergy content are however less accessible due to the limited use of exergy analysis by industry. To facilitate the use of exergy analysis, some tools such as an online converter and a software tool (ExerCom) have been developed. Another limitation to the use of exergy analysis by industry is the lack of benchmark data that can be used to compare their own efficiency (EC, 2009b).

4.1.4. Risk assessment

Risk assessment is a term which gathers several types of assessments. Finnveden et al. (2007) define two types of risk assessments: chemical risk assessment and accident risk assessment. Accident risk assessment evaluates the potential impacts associated with accidents (e.g., due to explosions, extreme natural conditions etc) on the studied site and is more related to safety measures. The aim of chemical risk assessment is to quantify the exposure of (magnitude and duration) and the effect on the environment surrounding an emission source to emitted substances. It is divided into two main assessments methods: human health risk assessment and ecological risk assessment, which assess the impact of emitted substances on humans and ecosystems, respectively. Note that some studies only evaluate the fate of emitted substances, without assessing their impact on receptors. When assessed, the impact of a substance on receptors is calculated following equation 1.

$$IMP = FF_i \times EF_i \times DF \tag{1}$$

Where i is a substance, IMP is the impact on the studied receptor, FF_i is the fate factor of i in the studied receptors (e.g., average ingested daily dose), EF_i is the effect factor of i and DF is the damage factor of the effect considered.

For example in the waste management sector, risk assessment studies are applied for assessing the risk of exposure to harmful substances in actual or planned conditions of a site management or plant operation at a steady state (e.g., Cangialosi et al. (2008); Davoli et al. (2010)) or for assessing the risk of pollution in case of the modification of the actual or planned conditions of a site management or plant operation (e.g., Ollson et al. (2014a); Rapti-Caputo et al. (2006)). Some studies focus on few specific substances while others focus on specific environmental compartments such as the aquifer or the surrounding atmosphere. Most studies follow a conservative approach, i.e., they use maximum estimations or values from measurement campaigns. Some other studies choose average data reflecting the real situation rather than a risk of pollution. However, studies assessing the impact of substances follow a conservative approach by considering a maximum exposure to assess the impact of emissions on the receptors. Examples of indicators of impacts on human health are the hazard index for non-carcinogenic pollutants (also called hazard quotient or hazard ratio) (e.g., Davoli et al. (2010)) and the cancer risk for carcinogenic pollutants (e.g., Cangialosi et al. (2008)). The receptors studied in ecological risk assessment studies are diverse, e.g., aquifers (Rapti-Caputo et al., 2006), wildlife (Ollson et al., 2014b) or soils and vegetation (Wang et al., 2011). One main advantage of risk assessment studies is that they evaluate the risk of impact under local specific conditions. One intrinsic limitation is that it cannot evaluate global scale issues such as climate change. Similarly, it focuses on emissions and does not evaluate the risks that a site or plant consumes specific resources from the environment (Benetto et al., 2007). Another limitation is related to the fact that risk assessment is hardly accessible to non-experts and requires involving experts having specific knowledge on pollutant dispersion in the aquifer, lithosphere and/or atmosphere.

4.2 Evaluation methods at the life cycle level

The previous methods evaluate processes at the level of the process itself and follow a so-called “gate-to-gate” approach. The common limitation to all these methods is their inability to identify displacement of environmental burdens upstream and downstream the studied system. The life cycle approach aims to consider other steps of the product life cycle than the production process itself, e.g., from the extraction of the raw materials to the end-of-life of the product ("cradle-to-grave") or to the end of the production step ("cradle-to-gate"), and thus allows identifying the displacement of environmental burdens. As aforementioned, some approaches applying energy analysis convert all energy flows in terms of primary energy. This approach is a life cycle-based approach as the amount of raw energy carriers are accounted for. Similarly, the emergy concept described below is also an evaluation method at the life cycle level. However, the main method that applies such an approach is Life Cycle Assessment (LCA).

4.2.1. Emergy analysis

Emergy accounts for all the original energy, i.e., solar energy, tidal energy and geothermal energy, which has been consumed in the earlier steps of product or service making. Emergy was introduced by Odum (1995) based on the principle that the value of a resource depends on the amount of the three aforementioned energy types which were consumed to produce it. Emergy analysis is not often used to assess the efficiency of processes. However, it is subject to a growing interest in the USA, where a pilot project is running on its application in industry. The concept of emergy is rarely applied in the waste management sector. Examples can be found in Asia, e.g., on waste exchanges within a sulfuric acid production system and a titanium dioxide production system in China (Zhang et al., 2011), to compare four treatment technologies for urban solid waste (Liu et al., 2013), on an e-waste treatment process (Song et al., 2012). Indicators calculated out of these analyses are both typical emergy indicators (e.g., the Emergy Yield Ratio defined as the total emergy input by the total emergy purchased on the market; Song et al. (2012)) and indicators specific to the waste management sector (e.g., the Landfill to Recycle Ratio defined as the ratio of emergy required for landfilling a material to the emergy required for recycling (Agostinho et al., 2013)).

One advantage of emergy analysis is that it aims at accounting for the impact of a system on ecosystems services. It considers that emissions to air and water will be diluted by ecosystems services to reach an acceptable concentration. For example, emissions to air will be diluted by the action of wind, and emissions to water by the action of water flow. Therefore, impacts on ecosystems services are calculated based on the amount of emergy from nature necessary to dilute the pollutants. However, this approach is highly based on transformities values, i.e. the values used to convert flows in terms of original energy (geothermal, solar and tidal) consumed by the studied system, which have often been criticized by the scientific community for their lack of uncertainty quantification.

4.2.2. Life Cycle Assessment

Life Cycle Assessment (LCA) is a recognized methodology to assess the environmental burdens of a system and follows the framework of International Standards Organization (ISO) 14040 and 14044 (ISO, 2006b, 2006c). As it does not only consider the process under study but also processes upstream and dowstream, LCA allows comparing the environmental impact of different steps of a studied process, identifying the steps which could be improved and avoiding environmental impact shifting from one step to another. The ISO standards define four steps to conduct an LCA: 1) Definition of the goal and scope; 2) Inventory analysis; 3) Impact assessment; 4) Interpretation (Fig. 4).

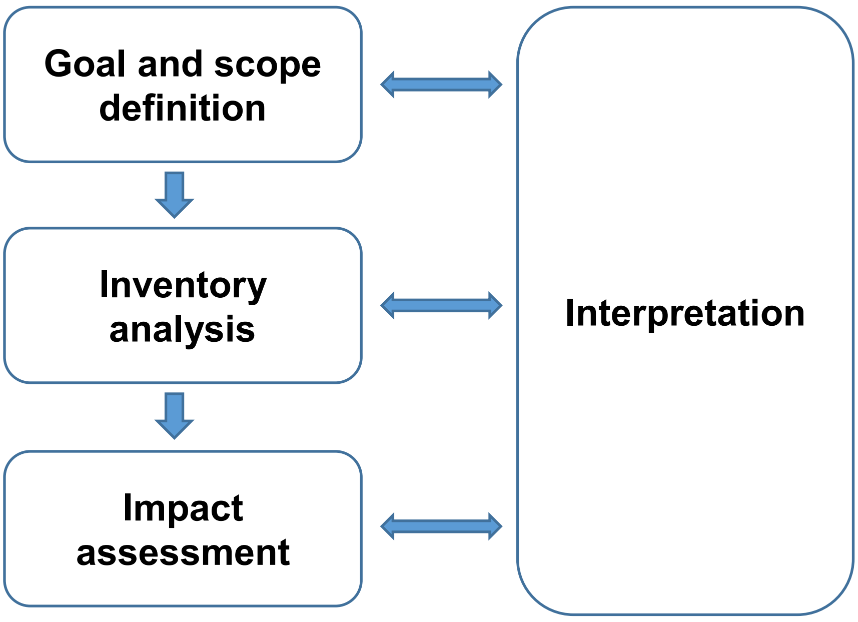

Figure 4: *The four steps of an LCA (ISO, 2006b)*

During the definition of the goal and scope, the process under study is clearly described, as well as the objective of the study. Elements of the process description are the geographical location of the process, the timeframe for which the results are valid and the function of the product. The function of the product is defined by the functional unit, which provides the reference to which all data in the product systems are normalized. Methodological choices such as the list of the processes included in the analysis (i.e., definition of the system boundaries), the choice of the impact assessment method, the allocation approach and any specific assumptions are also reported. The allocation approach applies in case of systems that produce several co-products and aims to partition flows among these co-products. Several partitioning approaches can be followed. The most common ones are system expansion, cut-off and allocation based on a parameter that reflects best the relationship between the environmental burden and the function of the co-products (e.g., mass, energy, exergy and monetary value). According to the ISO 14041 standard, system expansion should be prioritized over allocation. It consists in including the production of the co-products in the system boundaries by defining a “basket of products” as a functional unit or by keeping the same functional unit while considering products avoided by the co-products.

In the inventory analysis, all data necessary for the impact assessment are gathered for the process under study and for the upstream and downstream processes, i.e., material, substance and energy flows. Data for downstream processes are obtained from databases such as ecoinvent (Frischknecht & Rebitzer, 2005), Gabi (PE International, 2013) and ELCD (JRC, 2014).

The third step is impact assessment, during which emissions and resources consumed are multiplied with characterization factors for each impact category (i.e., sustainability “theme”) studied. For example, when the impact category Climate change is analyzed, each emission contributing to global warming is multiplied by a characterization factor, which converts it to a common unit, i.e., kg of CO_2 equivalent. Characterization factors are gathered in the so-called “impact assessment methods” which all follow their own methodology and assumptions to define the characterization factors. There are two types of impact assessment methods: the emission-based and the resource-based methods. The emission-based methods convert all the emissions from the product life cycle into impacts. Several methods have been developed by different actors for a same impact category. This is the case for most impact categories. However, some methods reach a larger consensus than others. This is the case for Climate change, which characterization factors are very similar from one method to the other. This is not the case for other methods such as aquatic ecotoxicity and human toxicity for which using different impact assessment methods can result in different conclusions regarding the sustainability of the studied product (Pant et al., 2004; Renou et al., 2008). The resource-based methods do not characterize the impact of emission but focus on the amount of resources consumed by a product life cycle and their impacts. Some resource-based impact assessment methods only account for resources based on a physical property (typically mass/volume, energy, exergy and area). They are called resource accounting methods. Other methods account for resources but also characterize their impact based on different aspects. These methods are further explained in Chapter 2.

The last step of an LCA is interpretation, which aim is to check the validity of the data and methodological choices made to conduct the study and draw the conclusions regarding the sustainability of the product. Several additional tools can be used. Sensitivity analysis consists in modifying one parameter or assumption of the model and analyzing its impact on the LCA results. It can be a way to identify the assumptions that need to be refined to obtain more accurate results. Uncertainty analysis consists in taking into account the uncertainty of input data in the calculation of the LCA results. This can be done by scenario analysis, based on the Pedigree matrix or on statistical analyses. The uncertainty of the input data can then be propagated to the LCA results using methods such as the Monte-Carlo analysis. Uncertainty analysis informs on how significant the conclusions of the LCA study are and supports decision making.

The main advantage of LCA is related to its life cycle thinking approach. It allows identifying the causes of the most impactful environmental burdens within the system or technology of primary interest but also those occurring in the upstream and downstream systems. It also allows identifying displacement of environmental burdens to other sectors. Moreover, LCA allows evaluating the impacts of a wide range of hazardous substances. It also allow analyzing both emissions into air, soil and water, and the consumption of resources. One major limitation of LCA today is that it does not allow characterizing the impacts geographically. Indeed, some local conditions have a direct effect on the impact of a specific compound released in the atmosphere. They can affect pollution dispersion (e.g., wind, rainfall) or the reaction of the emitted pollutant with compounds already present in the atmosphere (e.g., the concentration of ammonia, which reacts with NO_x to form nitric acid). Another limitation of LCA is that even if it is framed by international standards (ISO, 2006a, 2006b), several methodological choices should still be made by the person in charge of the study, which does not allow a direct comparability of LCA studies made by different people.

5. The lack of consistency in the use of evaluation methods

The evaluation of the environmental sustainability of newly developed technologies can be conducted in the context of research and innovation projects funded by public organizations, for communication purposes or for internal use in the company that undertakes the research, e.g., as an element for process improvement. Today, many different approaches are followed to conduct this evaluation. To illustrate this point, a short comparison of the approaches followed to assess resource use and impact from emissions from research and innovation projects from the FP7 Energy and FP7 Environment European funding programs and aiming to develop new technologies was made and presented in Table 1. They were chosen randomly from these two programs at the condition that they include elements to evaluate the environmental sustainability of the technology and belong to 15 research topics.

Table 1: *Methods used to evaluate the environmental sustainability of newly developed technologies in 17 FP7 projects. This table is based on the information available online. In some cases no information on the method could be found (e.g., Abiotic Resource Use) so the names of the methods appear as indicated in the projects outcomes.*

Project name	**Gate-to-gate analysis**		**Life cycle-based analysis**	
	Emission-based	***Resource-based***	***Emission-based impact categories***	***Resource-based impact categories***
P-REX	Risk assessment for PCDD/F, dl-PCB, PAH, As, Cd, Cr, Cu, Hg, Ni, Pb and Zn		GWP, TA, FEU, ME, FEC, HT	CED, Metal Depletion Potential
HEROMAT			GWP, TA, FEU, HT, ODP, POC	Energy needs, ADP
WASTE2GO			GWP, TA, POC	PED
END-O-SLUDG	Risk assessment for phosphorus, heavy metals and polyaromatic hydrocarbons		GWP, TA, FEU	PED, Abiotic Resource Use
RECOPHOS		Partial SFA on phosphorus, Energy balance		
LIGHT2CAT	NOx emissions accounting			
FFW		Energy balance	GWP, TA, FEU, FEC	CED, ADP, Water Footprint
GreenHP		Energy balance	GWP	PER
NXTHPG		Energy balance	GWP	
SECTOR			GWP	
Green-CC		Energy balance		
CYANOFACTORY		Light conversion efficiency		
ITAKA			GWP	
SORT-IT			GWP, TA, FEU, POC, HT	ADP

Project name	Gate-to-gate analysis		Life cycle-based analysis	
	Emission-based	***Resource-based***	***Emission-based impact categories***	***Resource-based impact categories***
BIOCORE		Local land and water use	GWP, ODP, RI, POF, TA, FEU	CED (non-renewable)
INNWIND.EU		Energy balance		
NANOSUSTAIN	Risk assessment		GWP, TA, FEU, POC, ODP	CED, ADP

PCDD: polychlorobenzodioxines; PCDDF: polychlorodibenzofuranes; dl-PCB: dioxin-like polychlorinated biphenyl; PAH: Polycyclic aromatic hydrocarbon; As: arsenic; Cd: cadmium; Cr: chromium; Cu: copper; Hg: mercury; Ni: nickel; Pb: lead; Zn: zinc; GWP: Global Warming Potential; TA: Terrestrial Acidification; FEU: Freshwater Eutrophication; ME: Marine Eutrophication; FEC: Freshwater Ecotoxicity; HT: Human Toxicity; ODP: Ozone Depletion Potential; POC: Photochemical Oxidation Potential; RI: Respiratory inorganics; CED: Cumulative Energy Demand; ADP: Abiotic Depletion Potential; PED: Primary Energy Demand; PER: Primary Energy Ratio

Table 1 shows that there is a wide range of approaches followed by project developers, from a simple energy balance at gate-to-gate level to a combination of emission- and resource-based analyses at gate-to-gate and/or life cycle level. For emission-based analyses, most are conducted at the life cycle level and the impact from emissions on the local environment is rarely discussed. At the life cycle level, some impact categories are analyzed by almost all the projects conducting such analysis (e.g., GWP, TA and FEU) and sometimes completed by other impact categories. There is more discrepancy among resource-based analyses. Both gate-to-gate and life cycle analyses are conducted. At gate-to-gate level, basic process efficiency indicators based on energy balance are often calculated, often without any life cycle consideration. At the life cycle level, there is a wide range of methods followed; the ones used the most are the ADP and CED methods.

This random overview shows that there are large variations in interpretations and approaches followed in individual projects, especially to quantify resource use. This confusion is a major bottleneck to know and benchmark how projects can effectively contribute to increase resource efficiency at macro-scale. In the context of innovation funded by public funds, project evaluation is a key step to help public authorities and PPPs to better evaluate and define resource efficiency targets, and outline a related strategic agenda. A proper evaluation of project outcomes would help orientating the focus of future calls towards the most resource efficient fields of research. In a more general context, when assessment studies are conducted with the aim to use the results for communication to the consumers such as marketing, there is a risk of “green washing”, as companies have the freedom to choose which approach to follow. Therefore, there is a need for a framework to assess the resource efficiency of new resource efficient technologies.

6. Objectives and outline of the PhD

This PhD has two main goals:

- With the development of new systems and technologies aiming at reducing resource consumption, metrics are necessary to inform decision makers about their actual contribution to save resources. The first goal of this work is to propose ways to improve the evaluation of resource efficiency of newly developed processes to allow a better comparability.
- The second objective of this work is to test the implementation of recommendations to improve the evaluation of resource efficiency of newly developed processes in three case studies in the bio-based economy. Even though these technologies do not all have the same potential to contribute to increase the resource efficiency of territories, they are all examples of resource recovery technologies aiming at reducing resource use at a wider scale. The first investigated technology is a pilot MaB-flocs raceway pond treating aquaculture wastewater in Belgium, which aims to produce biomass that could be used to substitute conventional energy sources or agricultural products. The second technology is the anaerobic digestion of cow dung and rice straw in rural India to produce biogas to substitute conventional cooking fuels, pointed out by the WBCSD as highly contributing to the material footprint of the country (WBCSD, 2015c). The third

technology is the valorization of sewage sludge from a Dutch wastewater treatment plant as different chemicals, biogas and building material which could replace conventional fertilizers, fuels and materials.

These objectives are addressed into Chapters 2 to 5. **Chapter 2** presents the challenges related to the evaluation of the resource efficiency of newly developed services and technologies and which result in their limited comparability, thus limiting the information necessary to orientate policies. Chapter 2 presents recommendations to improve this evaluation. In Chapters 3, 4 and 5, selected recommendations are tested in case studies. **Chapter 3** presents the implementation of a first recommendation: the need to upscale newly developed technologies to allow a fair comparison with current technologies. It is applied to the pilot MaB-flocs raceway pond treating aquaculture wastewater in Belgium. **Chapter 4** implements two other recommendations: the need to analyze new systems at the substance level and to conduct evaluations at both gate-to-gate and life cycle levels. These are applied to the evaluation of the implementation of anaerobic digestion at the level of a state in Central India. **Chapter 5** presents an attempt to improve the way resources are accounted for in LCA studies of circular systems, as discussed in Chapter 2. The need for an allocation of the impacts of upstream and downstream processes along a chain with a multiple use of resources that better consider the principles of industrial ecology is discussed by applying different allocation methods. This is applied to the case of a wastewater treatment chain in the Netherlands. Finally, **Chapter 6** discusses the lessons learnt from these three case studies. Conclusions are drawn and perspectives for further research are provided. The structure of the PhD is presented in Fig. 5.

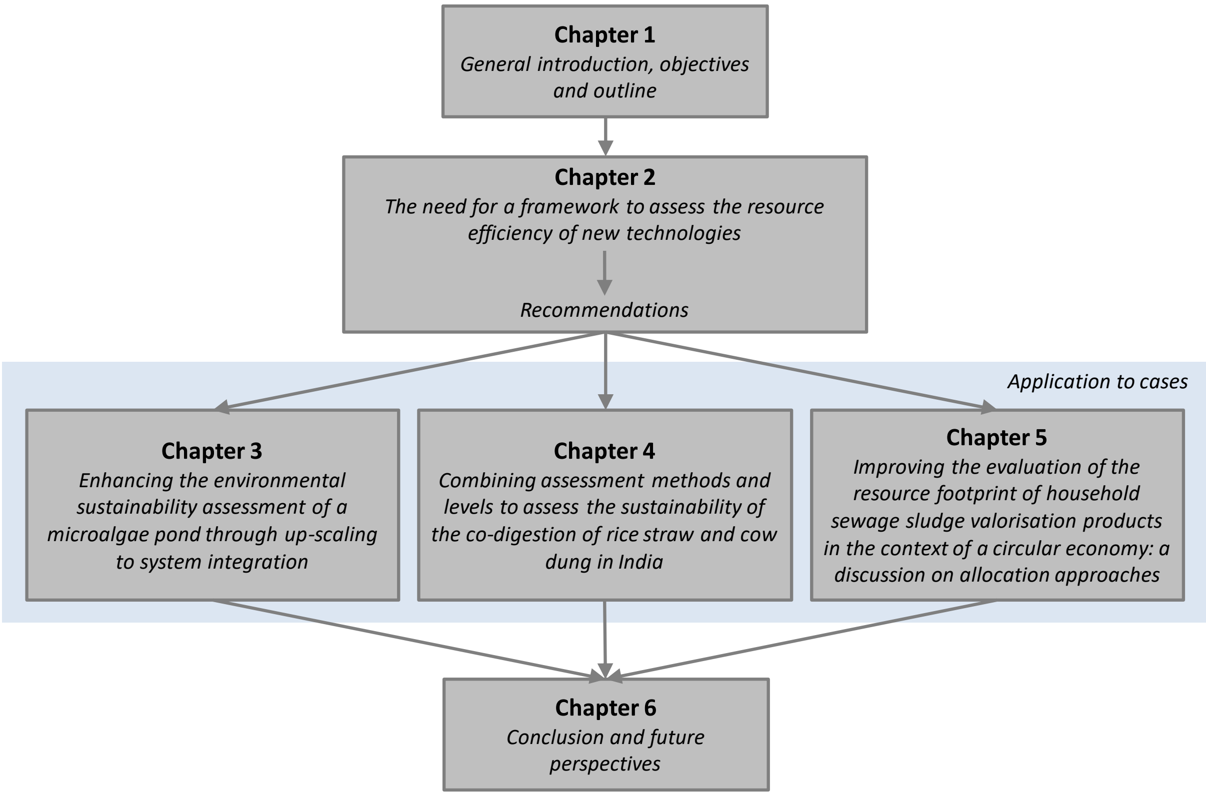

Figure 5: *Structure of the PhD and its chapters*

Chapter 2: The need for a framework to assess the resource efficiency of new technologies

Redrafted from:

Sfez, S., Dewulf, J., De Soete, W., Schaubroeck, T., Mathieux, F., Kralisch, D. And De Meester, S. (2017). *Toward a Framework for Resource Efficiency Evaluation in Industry: Recommendations for Research and Innovation Projects.* Resources. 6(1), 5.

1. Introduction

As discussed in Chapter 1, many programs are being launched worldwide to increase the resource efficiency (RE) of our economy. To measure progress of the different programs at the macro-level, different indicators are being defined by regions and nations. In the EU, a system of indicators called the Resource Efficiency Scoreboard was developed, beginning with "Resource Productivity", which was defined as the ratio between Gross Domestic Product (GDP) and Domestic Material Consumption (DMC). This indicator was further disaggregated into macro- and thematic indicators, including water productivity and energy intensity, among others. Research and innovation aiming to develop new technologies at the micro-level play a key role in increasing the resource efficiency in existing programs worldwide. In the EU, support for research and innovation is one of the four pillars identified as part of the 2011 "Roadmap to a Resource Efficient Europe" (EC, 2011b) to help transform the economy, while RE was identified as one of the five societal challenges to be addressed by innovation partnerships in the "Innovation Union" strategy (EC, 2010b). Consequently, the EC introduced several calls in its Horizon 2020 funding program on RE, including one focusing on the Processing Industry via the Public Private Partnership (PPP) Sustainable Process Industry through Resource Efficiency (SPIRE). This PPP focuses on eight process industry sectors. It is designed to contribute to the Roadmap and defines its own RE targets. Primarily, it aims to reduce non-renewable primary raw materials and fossil energy intensities by 30 and 20% by 2030, respectively (SPIRE, 2013). The PPP Factories of the Future (FoF) sets similar targets: increase of energy from renewable by 20% and in energy efficiency by 20% (EFFRA, 2013). These goals are translated into objectives in individual innovation projects. Examples of objectives listed in the Horizon 2020 project calls are the “energy consumption […] reduction for the product of at least 30% from cradle to grave” (FoF-3-2014), “gains in productivity, in material and energy efficiency” (WASTE-1-2014) and an increase in “the resource and energy efficiency for the process industries by at least 20%” (SPIRE-3-2014).

Whereas the RE indicators are clearly defined at the macro-level, the measurement of the RE of research and innovation actions, often conducted at the process level, is difficult for the broader community to understand and results in a lack of consistency that could allow the comparability of projects outcomes.

This chapter aims to present the main hurdles that limit the comparison of projects’ outcomes on resource efficiency and propose a path to move toward an improved evaluation of projects. After a presentation of the methodology followed to identify these hurdles and propose recommendations, the terms resources and resource efficiency are defined. Then, the different choices to be made by project developers when evaluating the outcomes of their project are presented. In a last section, recommendations are proposed.

2. Methodology

The first part of the chapter aims to present the existing understandings of “resource efficiency” concepts and the approaches available in the scientific literature. This section is based on a review of the scientific literature on resource consumption and management and a state-of-the -art of resource accounting and impact assessment methods in the field of sustainability evaluation. The literature review was based on web search tools such as Web of Science and

Google Scholar using keywords such as “resource efficiency”, “resource management”, “resource consumption” coupled with “gate-to-gate”, “life cycle assessment” and “process level”. In section 4, the drawbacks and limitations of these approaches are discussed. The discussion is based on a review of the outcomes of case studies found in literature and aiming to evaluate the resource efficiency of industrial processes as well as discussions held during three workshops. The SPIRE Workshop on Resource Efficiency Monitoring, Assessment and Optimization was organized by A.SPIRE and gathered SPIRE project developers and representatives of the European process industry and academia. The two other workshops were organized in the framework of the Horizon 2020 project MEASURE (“Metrics for Sustainability Assessment in European Process Industries”) in Kortrijk (Belgium) and in London (UK). During these two workshops, the current state of resource efficiency evaluation was presented to representatives of the European process industry and policy makers and the current concepts and understandings were discussed. First recommendations were derived from these discussions and the analysis of the literature. The recommendations were presented during the final workshop of the MEASURE project in Berlin, during which representatives of the European process industry and stakeholders involved in the management of research and innovation projects provided feedback. This feedback was incorporated in this study. Moreover, a parallel task conducted in the framework of the MEASURE project was to write an overview of the practice of impact assessment in the waste management sector (Sfez et al., 2016). This report allowed identifying additional recommendations to improve the resource efficiency evaluation of resource recovery processes.

The combination of a literature review of methodological papers and case studies and expert involvement during several workshops allowed summarizing the challenges and potential ways to improve resource efficiency evaluation in research and innovation projects. The role of different stakeholders (e.g., project developers, who are in charge of conducting the research and innovation projects, and stakeholders in charge of writing the calls) could also be suggested.

3. The need for a common understanding of resources and resource efficiency

3.1 Definition of resources

In the context of the resource-efficient initiatives around the world, resource consumption is limited to the "environmental" context and thus labour, capital, time, etc. are not considered as resources in the RE evaluation here. Within the environmental dimension, a general distinction can be made between resources in the broad sense and the strict sense (Berger & Finkbeiner, 2010). The former considers resources as "inputs" into a system and the environment itself as a sink and accounts for its role in absorbing emissions. Resources defined in the strict sense only consider "inputs" entering an anthropogenic system. While the former definition of resources is primarily used in a policy context (macro-level), the second definition is mainly used in industry and engineering, as resource consumption is the starting point for all economic production and consumption activities (Huysman et al., 2015b). Moreover, the impacts of process emissions are covered by other specific policy actions, (e.g., see the EU Directive on industrial emissions (EC, 2010a)), and by separate monitoring processes to evaluate their impact (e.g., see the report on the impact of policy measures on Europe's air quality (EEA, 2010)). Therefore, the latter viewpoint on resource definition is used as a basis in this chapter. Even when considering this viewpoint, several definitions of resources in the strict sense exist, differing primarily in the number and types of resources considered. For example, SPIRE defines resources as "energy, raw materials and water" (SPIRE, 2013). Another definition defines resources as "objects of nature which are extracted by man from nature and taken as useful input to man-controlled processes, mostly economic processes" (Udo de Haes et al., 1999). Similarly, the OECD defines natural resources as "natural assets (raw materials) occurring in nature that can be used for economic production or consumption". Because "objects of nature" and "natural assets" are very broad terms and because the process industry also uses waste (i.e., "substances or objects which the holder intends or is required to dispose of" ISO (2006c)) as a resource, we focus on the SPIRE definition, which allows considering waste energy, raw materials and water as resources. Moreover, water is a key resource in the process industry (EC, 2014a), and this definition explicitly considers it as a resource. Atmospheric resources and elements present in water bodies are also considered as resources in the scientific literature (Dewulf et al., 2007). However, they are abundant in their media and do not necessarily represent a major challenge for the industry today. Land, on the other hand, is generally considered a key natural resource in literature (see for example the classification of natural resources in Klinglmair et al. (2014) and Giljum et al. (2011)) and work is ongoing on how to better account for this resource in sustainability evaluations and especially in LCA (e.g., see Taelman et al. (2016)). However, land is missing in the SPIRE definition. In conclusion, land, energy, primary and secondary raw materials and water are considered relevant resources within the scope of this chapter.

3.2 Definition of resource efficiency

The resource efficiency platform of the EC defines RE as "using the Earth's limited resources in a sustainable manner while minimizing impacts on the environment" (EC-OREP, 2014). This definition does not reflect a concrete formula but does contain two essential ingredients: the use of resources and their impact. When focusing on the calculation procedure, efficiency is defined as the ratio between the benefits obtained from a process or system, i.e., all the indirect benefits to mankind obtained out of resources or their derived products, and the "efforts" put into this process or system:

$$\textit{Resource efficiency (RE)} = \frac{\textit{Benefits from resources}}{\textit{(Impact from) Resources used}} \qquad (1)$$

As an example, the aforementioned "Resource Productivity" is defined by the EC as an RE indicator at the EU level. In this indicator, the benefits from resource use are expressed in monetary terms (GDP). However, the benefits from resource use can also be expressed in terms of the function provided by the product or the quantity of product produced (e.g., health benefits of one medical treatment (Debaveye et al., 2016) and of one nutritional value (Stylianou et al., 2016)). Often confused with this, but actually the inverse of RE is resource intensity:

$$\textit{Resource intensity} = \frac{\textit{(Impact from) Resources used}}{\textit{Benefits from resources}} \qquad (2)$$

The fact that RE and any (production) efficiency in general are ratios can be easily agreed upon by the policy, industry and scientific communities.

3.3 Level of RE evaluation

RE calculations also depend on the type of system under study. RE can be calculated at different levels. The foreground system can be defined as a single process unit, a production plant, an industrial sector or a country/region. It consumes both resources directly extracted from the natural environment and processed natural resources, and it delivers products and services to end users (ISO, 2006a) (Fig. 1).

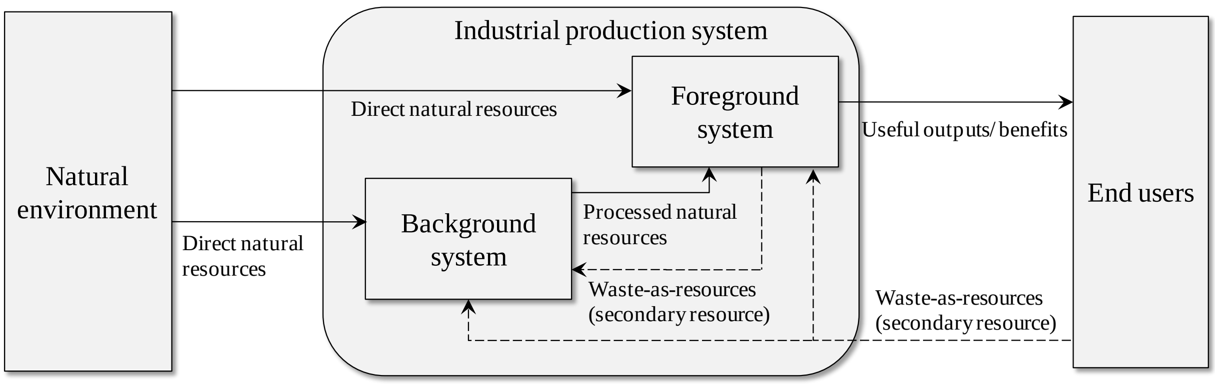

Figure 1: *Simplified system diagram of resource use in processing industries (emissions of waste from end users and the industrial production system into the natural environment are not depicted).*

At the different levels of analysis, one can choose to focus on the foreground system itself, thus only considering the resources directly entering and leaving the system in the denominator, without taking the resources used in the upstream and downstream steps into account. This is the case with the EC indicator "Resource Productivity", which only considers inputs of materials entering the EU but does not consider upstream or downstream resource inputs in the denominator. Such analyses are classified among the "gate-to-gate" analyses presented in Chapter 1. Another option calculates the denominator at the industrial production system level by following a life cycle perspective. As explained in Chapter 1, all direct and indirect resources consumed upstream and downstream along with, in some cases, the resource consumption avoided by the delivery of services or products to the market are considered. Depending of the chosen scope and level of analysis, different databases can be used to quantify/obtain the consumed resources. These databases include input-output tables at the country or sector level (macro level) (e.g., Exiobase (Tukker et al., 2009); the World Input-Output database (Dietzenbacher et al., 2013)), and LCA databases at the product or process level (micro-level) such as ecoinvent (Frischknecht & Rebitzer, 2005), ELCD (JRC, 2014) or Gabi (PE International, 2013). Such analyses encompass the so-called cradle-to-gate (from resource extraction to production) and cradle-to-grave (from resource extraction to disposal) analyses.

Gate-to-gate and LC-based analyses do not consider resources in the same way. In a gate-to-gate analysis, processed natural resources, direct natural resources and waste-as-resources are considered equally:

$$\textit{Resource efficiency (gate-to-gate base)} = \frac{\textit{Benefits from resources}}{\textit{Processed natural resources} + \textit{Direct natural resources} + \textit{Waste-as-resources}} \quad (3)$$

This means that their use is considered in the foreground system only and that resource consumption that occurs elsewhere is not included. Gate-to-gate analysis methods account for consumed resources but typically do not characterize the impact of resource consumption.

When calculating life cycle-based RE, waste-as-a-resource used as an input in a process is generally not accounted for and is seen as gratuitous (approach also called the "zero burden assumption"; Ekvall et al. (2007)). Waste is indirectly accounted for by a decrease in natural resource consumption, but is not taken into account on its own. The inclusion of waste production in LCA is subject to debate within the LCA community. Thus, RE is calculated in LC-based analyses using the following ratio:

$$\textit{Resource efficiency (life cycle base)} = \frac{\textit{Benefits from resources}}{\textit{(Impact from) Indirect and direct natural resources} + \textit{((Impact from) Waste-as-resources)}} \tag{4}$$

In this case, resources consumed by the foreground and background systems are included. The resources consumed by the foreground system can be directly and/or indirectly extracted from the natural environment. In equation 4, the indirect natural resources refer to the direct natural resources processed in the background system and then consumed in the foreground system. The brackets in equation 4 indicate the choices that can be made to account for resources at the life cycle level: account or not for waste-as-resources (see previous paragraph) and consider resources in terms of their quantity or their impact.

3.4 Methods available to quantify resources

The numerator of the RE equation, i.e., the benefits obtained from resources, is often easier to quantify than the denominator, as benefits are generally delivered to end users and can often be expressed in tangible units: kg, MJ, money, etc. However, this is not always the case, especially when benefits have a social function. The denominator requires additional calculations and discussion. It has been subject to debate since the mid-nineties (Heijungs et al., 1997) and Zhong et al. (2016) showed that the interest of the scientific community for natural resource accounting has grown rapidly during the last fifteen years. Recently, Klinglmair et al. (2014) and Swart et al. (2015) proposed a classification of methods to evaluate resource use in LCA. As a basis for a better understanding of the next sections, this section summarizes the outcome of these two studies on existing methods to evaluate the denominator of the resource efficiency ratio. It can be calculated according to two principles:

- *A physical accounting of resources:* the quantity of resources consumed by the studied system is systematically accounted for based on a physical property (mass or volume, energy, exergy or area).
- *An assessment of the impact from resource use:* this is done by considering one of the following elements: the amount of resources available in the Earth's crust, predefined targets, future consequences of resource extraction, or willingness-to-pay (WTP).

Resources can be classified as renewable or non-renewable and as biotic or abiotic (Table 1). Renewable resources are able to regenerate within a human lifetime but can be exhausted if consumed beyond their regeneration capacity (Dewulf et al., 2015b). They can be biotic (i.e., "derived from presently living organisms"; e.g., wood) or abiotic (i.e., a "product of past biological or physical/chemical processes"; e.g., wind energy) (Swart et al., 2015). On the contrary, non-renewable resources cannot be renewed in the natural environment or can be renewed but not within a human lifetime (e.g., metals or natural gas, respectively). The methods used to quantify resources do not all consider these resource sub-categories in the same way.

3.4.1. Resource accounting methods

Resource accounting methods can be used in both gate-to-gate and LC-based analyses. Each method accounts for resources based on a specific physical property. Four main properties are considered by existing methods: mass/volume, energy, exergy and area. Because all resources do not have the same properties, resource accounting methods do not necessarily account for the same resources. For example, energy-based methods do not account for water and land, whereas exergy-based methods do account for these resources (Alvarenga et al., 2013). Similarly, area-based methods neither account for non-renewable material resources nor for abiotic renewable energy resources. However, some area-based methods, such as the Ecological footprint, account for bio-productive land necessary to absorb CO_2 emissions, as well as for the amount of consumed nuclear energy carrier (Huijbregts et al., 2008). Moreover, some methods are only able to account for a fraction of a resource “category”. For example, mass accounting methods are not able to account for all energy carriers, typically wind energy and electricity. Current exergy-based methods account for the largest number of resources.

3.4.2. Impact assessment methods

Impact assessment methods are only applicable in LC-based analyses (Table 1). Similarly to gate-to-gate analyses, they do not all cover the same resources (e.g., some cover nuclear energy whereas other do not). Following the classifications from Klinglmair et al. (2014) and Swart et al. (2015), most developed methods can be classified as based on the quantity/quality of reserves, distance-to-target, future consequences and willingness-to-pay.

- *Methods based on the quantity/quality of reserves:* these methods consider that the quantity and/or quality of resources available in the natural environment is decreasing and thus that the consumption of resources has an impact on resource availability. Some methods such as the Ore Requirement Indicator (Swart & Dewulf, 2013) or the Ore Grade Decrease methods (Vieira et al., 2012) consider the decrease of ore grade as an indicator of resource availability in the natural environment, while other methods such as the ADP method (Guinée & Heijungs, 1995) put the amount of resources consumed in perspective with the reserves remaining in the natural environment relative to those of a reference species (e.g., antimony in the ADP method). The last approach is most common in the literature because most associated methods were developed prior to other approaches and are available in most LCA software tools. Methods based on the quantity/quality of reserves are only able to account for non-renewable resources and are heavily discussed by the scientific community and the industry sector (Drielsma et al., 2016).
- *Methods based on distance-to-target:* these methods compare the quantity of resources consumed to previously defined targets. The most used distance-to-target LCA method is the Ecological Scarcity method (Frischknecht & Büsser Knöpfel, 2013), which

puts the quantity of consumed resources in perspective with a critical flow of resources based on political targets or international policy.

- *Methods based on willingness-to-pay:* these methods estimate the amount of money people are ready to invest to restore damages caused to natural resources. The main LCA method that follows this approach in its weighting step is the EPS 2000 method (Steen, 1999).
- *Methods based on future consequences:* these methods consider the impact of current resource consumption on future parameters as a result of a decrease in the quality of ore in the natural environment. The most used parameters are the surplus energy (e.g., Impact 2002+ (Jolliet et al., 2003)) or surplus costs (e.g., ReCiPe Endpoint (Goedkoop et al., 2013); further developed by Vieira et al. (2016)) necessary to extract the same amount of resources in the future as today.

Table 1: *Existing methods to quantify resource consumption and examples (based on Swart et al. (2015) and Klinglmair et al. (2014); empty cells: resources not covered by the method; "biotic resources" are repeated for "Materials and substances" and for "Energy" as they can be materials or energy carriers; (X): Indirectly accounted for).*

			Scope		Resource classification									
					Water	Land	*Materials and substances*			*Energy*				
							Non-renewable		Biotic renewable	Non-renewable		Abiotic renewable	Biotic renewable
Methods based on...		**Examples of methods**	**Gate-to-gate**	**Life cycle**			**Atmospheric resources**	**Metals and minerals**	**Biomass**	**Fossil energy**	**Nuclear energy**	**Flow energy resources**	**Biomass**
Accounting methods	**Mass or volume**	Material flow analysis [a]	X		X			X	X	X	X		X
		ReCiPe Midpoint - Water depletion [b]		X	X								
		EDIP 97/2003 - renewable resources [c]		X					X[1]				X[1]
		Material Input Per Service Unit [d]		X				X	X[2]	X	X		X[2]
	Energy	Energy analysis [e]	X						X	X	X	X	X
		CED/PED [f][g]		X					X[3]	X	X	X	X[3]
		ADP - fossil fuels [h][i]		X						X			
		Impact 2002+ - non-renewable energy [j]		X					X[4]	X	X		X[4]
		ReCiPe Midpoint - Fossil depletion [b]		X						X			
	Exergy	Exergy analysis [k]	X		X		X	X	X	X	X	X	X
		CEENE [l]		X	X	(X)	X	X	X[4]	X	X	X	X[4]
		CexD [m]		X	X		X	X	X[4]	X	X	X	X[4]
	Area	Direct land accounting	X			X							
		Ecological Footprint [n]		X		X			(X)	(X)	(X)		(X)
Impact assessment methods	**Resource reserves quality/quantity**	ADP [h][i]		X				X			X		
		EDIP 97/2003 - non-renewable resources [c]		X				X		X	X		
	Distance to target	Ecological Scarcity [o]		X	X	(X)		X	X[4]	X	X	X	X[4]
	Willingness-to-pay	EPS2000 - land occupation and abiotic stock resources [p]		X		X		X		X	X		
	Future consequences	Impact 2002+ [j]		X				X					
		Eco-Indicator 99 [q]		X				X		X			
		ReCiPe Endpoint – resources [b]		X				X		X	X		

[1] Wood; [2] Plant biomass from cultivation and biomass from uncultivated areas; [3] Energy from wood and biomass from primary forest, and wood, food products, and biomass from agriculture; [4] Energy from biomass and biomass from primary forest;
[a]Brunner and Rechberger (2003); (b) Goedkoop et al. (2013); [c]Hauschild and Wenzel (1998); [d]Ritthoff et al. (2002); [e]Bullard et al. (1976); [f]Hischier et al. (2009); [g]PE International (2013); [h]Guinée and Heijungs (1995); [i]van Oers et al. (2002); [j]Jolliet et al. (2003); [k]Szargut et al. (1987); [l]Dewulf et al. (2007); [m] Bösch et al. (2007); [n]Global Footprint Network (2009); [o]Frischknecht and Büsser Knöpfel (2013); [p]Steen (1999); [q]Goedkoop and Spriensma (2000)

4. Points to consider when determining the resource efficiency ratio's numerator and denominator

In the previous section, we described and clarified the possible ways to calculate RE and the choices to be made by project developers to calculate this ratio. In the next section, some typical bottlenecks and drawbacks related to specific choices made during the evaluation of RE of innovation projects in industry are discussed.

4.1 Gate-to-gate versus life cycle analysis

As aforementioned, a gate-to-gate analysis provides information on the conversion efficiency of a process, but is not able to identify the displacement of resource consumption within the larger industrial production system. This can be a particular issue in the analysis of bio-feedstock processing, as biomass production often contributes to the upstream consumption of high amounts of natural resources such as fossil fuels, land and water (UNEP, 2010) and the replacement of fossil-based material by bio-based material can introduce a competitive use of resources already consumed by other sectors (e.g., agriculture and energy) (Geldermann et al., 2016). Even if increasing RE at the gate-to-gate level will most probably induce the same effect at the life cycle level, there is no guarantee that RE can be improved without an increase in resource consumption upstream and downstream. Moreover, when comparing one process or plant with a benchmark system, it may have a higher RE at the gate-to-gate level but a lower efficiency at the life cycle level (see De Soete et al. (2013) and the example in Box). Therefore, because gate-to-gate analyses have the advantage of being less time and data intensive (and thus also less costly), project developers could use such approach as a screening tool prior to further analysis or during the design of innovations. Gate-to-gate material- and energy-based indicators are particularly relevant for resource recovery technologies or nutrients extraction processes from raw materials. For example, the carbon conversion efficiency applied to one gasification process (Arena et al., 2011) and the phosphorus utilization efficiency calculated for the production of phosphorus-based chemicals (Ma et al., 2015) have proven to be valuable indicators to identify losses of resources along the process chain, and to compare the strengths and weaknesses of process alternatives. Moreover, as illustrated in Ma et al. (2015), gate-to-gate indicators based on resources at the substance level allow characterizing the "metabolism" of a process and identifying optimization measures aiming to decrease the dependency of an industry towards this resources. This approach is also highly relevant at macro-scale. However, this requires conducting an MFA or SFA to model a realistic and consistent system in which the systems outputs (e.g., phosphorus content of digested sewage sludge) are linked to the inputs (e.g., composition of the wastewater to be treated). Sfez et al. (2016) reported that this is rarely done in the waste management sector which encompasses resource recovery processes. Nevertheless, the completeness of an analysis based on life cycle thinking to evaluate and compare RE should also be pursued.

Another difference between these approaches is that the methods applied in gate-to-gate analyses account for waste-as-a-resource but do not evaluate the impacts and benefits of delivering secondary products to the economy. On the contrary, methods applied in LC-based analyses do not generally account for waste-as-a-resource, but they are able to consider the impacts and benefits of delivering secondary products to the economy when the system expansion approach is followed.

Resource efficiency of two chiral separation techniques

Van der Vorst et al. (2009) compared the resource consumption of two chiral separation techniques in the field of fine chemicals and the pharmaceutical industry: the preparative supercritical fluid chromatography (Prep-SFC) and the preparative high performance liquid chromatography (Prep-HPLC). Resource consumption by these processes was evaluated at three levels: process, plant and life cycle. At the process and plant levels, a gate-to-gate exergy analysis was conducted, while at the life cycle level, the Cumulative Exergy Extracted from the Natural Environment (CEENE) was estimated.

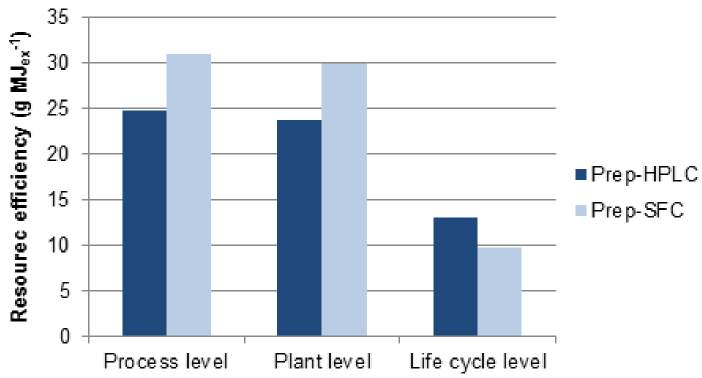

Figure 2: *Comparison of exergy efficiency in the prep-HPLC and prep-SFC methods at the process, plant and life cycle levels, expressed in grams of isolated enantiomer per MJ_{ex} consumed (based on (Van der Vorst et al., 2009)).*

At the process and plant levels, the resource efficiency of prep-HPLC is lower than prep-SFC (20% and 21% lower, respectively) (Fig. 2). However, at the life cycle level, prep-HPLC becomes more favourable and its resource efficiency becomes 34% higher than the prep-SFC method. The authors explain this difference primarily by the large amount of energy required to produce supercritical CO_2, which is not taken into account at the process or plant levels.

Box 1: *Comparison of the resource efficiency of two chiral separation techniques at the gate-to-gate and life cycle levels.*

4.2 Accounting versus impact assessment

At the life cycle level, resource accounting and impact assessment methods have both advantages and limitations. One of the main advantages of resource accounting methods in the context of RE is their ability to allow the expression of RE as a dimensionless value; in many cases, the amount of a product can be expressed in terms of mass/volume, energy or exergy. However, this can be difficult when analyzing systems that produce services or when the method chosen is based on area. Moreover, there is greater consensus about the different resource accounting methods than the LCIA methods in the scientific community. A main disadvantage of these methods is that they do not assess the indirect impacts of resource extraction. The main advantage of LCIA methods is their ability to evaluate the impact of resource consumption. However, methods based on future consequences and willingness-to-pay, even if relevant from a business perspective, do not always reflect the quantity of resources consumed and are associated with high uncertainty. Both types of methods allow the aggregation of results from different impact categories into a single score, which leaves project developers the choice to analyze aggregated or disaggregated results.

4.3 Resource coverage of life cycle-based methods

Recently, Finnveden et al. (2016) showed that at the life cycle level, choosing different methods to evaluate abiotic resource use leads to different results. One reason is that resource accounting and impact assessment methods do not all cover the same resources. The resource categories covered by the chosen LCIA method should always be listed by project developers in order to avoid the exclusion of sub-categories. Indeed, lowering the consumption of one specific natural resource can induce higher consumption of another one (see example in Box). Therefore, project developers should choose the method that covers the widest number of resource categories to avoid burden shifting.

Case study: resource efficiency of two valorisation pathways for algae grown in wastewater

In the framework of the Seventh Framework Programme (FP7) project EnAlgae, Sfez et al. (2015) compared the potential environmental burdens of two valorisation pathways for algae grown on aquaculture wastewater: valorisation as shrimp feed (scenario 1) and valorisation as biogas via anaerobic digestion (scenario 2). Sfez et al. (2015) used the CEENE method to calculate the resource footprint of the two scenarios. Based on data available in this paper and its supporting information, the resource efficiency of the two scenarios studied in Sfez et al. (2015) were calculated using two other methods: ADP and Eco-indicator 99 (end-point indicator "Resources"). While the CEENE method accounts for land use, the two other methods do not.

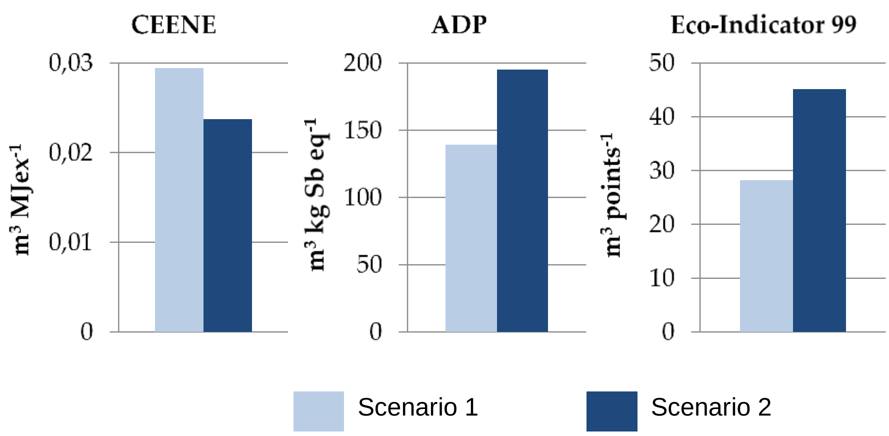

Figure 3: *Resource efficiency of two valorisation pathways for algae grown in wastewater using different life cycle level resource accounting and impact assessment methods, expressed in m^3 of treated water per (impact from) resource used.*

Different results are obtained when using these methods and lead to different conclusions regarding the most favourable valorisation pathway for algae (Fig. 3). Because valorizing algae into shrimp feed replaces the consumption of wheat (Sfez et al., 2015), it also avoids the use of land associated with wheat production. This benefit cannot be shown by methods that do not consider land as a resource, here the ADP and Eco-indicator 99 methods.

Box 2: *Comparison of the resource efficiency of algae cultivation in wastewater followed by two valorisation pathways using different LCIA methods.*

Even if several methods cover a same resource category, not all methods consider the same set of resources within a given category. For example, some methods include peat within fossil fuels, whereas others do not. This might be an issue when evaluating energy systems in countries such as Finland, where peat represents a significant share of the country's energy mix. Thus, special attention should be paid to the coverage of all resource categories considered by the chosen LCIA method in order to identify possible trade-offs between resource consumption and avoid the involuntary exclusion of one resource and its associated potential impacts (Vadenbo et al., 2014).

Considering the broadest number of resources in the denominator also means considering abiotic renewable resources. Including such resources in an RE evaluation will likely decrease the RE of the studied process, although these resources can be considered inexhaustible. Thus, projects in which fossil fuels are replaced by abiotic renewable resources might show lower RE than fossil-based projects if such resources are taken into account (Sfez et al., 2015). In these cases, results should differentiate between biotic and abiotic renewable resources. Moreover, technologies that consume abiotic renewable resources might require specialty metals and should therefore always be included in the analysis.

Special attention should be paid to the differences in categorization if two different LCIA methods are used to quantify fossil fuels and metals/minerals because some methods may account for resources differently, e.g., uranium as abiotic non-renewable energy (e.g., Impact 2002+ (Jolliet et al., 2003)) or as a metal (e.g., ReCiPe Midpoint (Goedkoop et al., 2013)).

Another important aspect concerns the coverage of metals and minerals by LCIA methods. Metals and minerals provide services to society (e.g., wastewater treatment plants, power grids) and may still be usable when these structures reach the end of their lifetime. This anthropogenic resource stock is currently not covered by LC-based methods. A first attempt was made by the Anthropogenic stock extended Abiotic Depletion Potential (AADP) method, which tries to include this stock in the evaluation of RE (Schneider et al., 2011), but data are still largely unavailable, and the method is not yet fully operational. Another approach has recently been proposed by van Oers and Guinee (2016). The authors propose to consider resource depletion as a dilution problem, i.e., that the issue related to the availability of resources is more related to the dilution of resources in the environment (e.g., via leaking from landfill) than to a transfer of resources from the natural stock to the anthropogenic stock. The work from van Oers and Guinee (2016) shows that the issues related to resource availability and the impact from resource use are still under discussion. Part of these discussions is related to the definition of the so-called Areas of Protection.

4.4 Entities impacted by resource consumption

In the field of LCA, Areas of Protection (AoPs) are defined as “entities that we want to protect” (Finnveden et al., 2009). Therefore, LCIA methods aim to evaluate the impact of life cycle inventories on these entities. The three main AoPs in LCA are “Human Health”, “Ecosystem Quality” and “Natural Resources” (Dewulf et al., 2015a). Several LCA methods consider the impact of resources on an AoP other than “Natural Resources”. For example, some LCA methods account for land use but consider its impact on biodiversity, which is considered in the AoP “Ecosystem Quality” today. Discussions are ongoing about whether the AoP “Natural Resource” should be maintained as such or rethought. This new debate can be illustrated by the recently published work from Dewulf et al. (2015a) who proposes to divide the AoP “Natural Resource” into five safeguard subjects including environmental, economic and social aspects, and the presentations of the 55th Discussion Forum on LCA in Zürich, during which the definition of the AoP “Natural Resource” was addressed as a key question (Vadenbo et al., 2014). Indeed, there is no agreement in the scientific community on the nature of the impact caused by resource consumption: while the AoP “Natural Resources” has been defined in the framework of environmental LCA and thus assumes that natural resource consumption is an environmental issue, the idea that resource consumption also considers other (provisioning) capacity of resources to fulfil humans needs is emerging (Dewulf et al., 2015a). The unclear definition of this AoP can partly explain the wide range of approaches followed by LCIA methods concerning the evaluation of resource consumption and the lack of consensus around which method to use. Therefore, a clearer definition of the AoP “Natural Resources” is a key step to improve the consideration of resources in LCA and thus to improve the calculation of RE. In the meantime, project developers should be aware that some methods consider the impact of resource use on AoPs other than Natural Resources and thus reflect different sustainability issues.

4.5 Functionality of the output products (benefits) – how to account for recycling?

The functionality of a process’s output is more often discussed within LC-based analyses (when choosing the functional unit) than within gate-to-gate analyses. However, the functionality of output products should be well defined for both types of analyses and can be done by taking into account the quality and the lifetime of the products. Such aspects can be defined based on an analysis of the physical, chemical and mechanical properties of the materials and their resistance to environmental conditions (Al-Oqla et al., 2015). This is in line with the EU’s Action Plan for the Circular Economy, which aims to more systematically introduce circular economy requirements, e.g., on product durability and quality, among others (EC, 2015b).

Defining functionality is not a straightforward task, especially when evaluating processes using waste as a resource, which aims to contribute to the switch from a linear to a circular economy. On the other hand, waste treatment projects also aim to protect the environment by safely treating waste and therefore have a double function. For example, the benefits obtained from a recycling process can be defined as the recycled product itself, or as the environmental savings achieved from recycling waste. Thus, the quantification of the benefits obtained from waste valorisation is complex. The choice of benefits has a significant impact on the results of the calculation and should be communicated to allow for a comparison between processes. Considering the benefits of recycling in LCA is particularly complex and is the subject of a wide range of approaches. The differences between these approaches are typically reflected in the substitution ratio and the stakeholders to which the benefits are allocated. Given the extent to which LCA results depend on these choices, specific attention should be paid to end-of-life modelling. From a project developer's perspective, a sensitivity analysis of the end-of-life parameters is a key way to strengthen the conclusions of the RE evaluation. In addition to the strictly defined RE ratio, other metrics highlighting the environmental savings associated with waste valorisation should be considered. For example, the Recyclability Benefit Rate (RBR), defined by Ardente and Mathieux (2014) as the ratio of the potential environmental savings achieved from recycling over the environmental burdens of virgin production followed by disposal better identifies these benefits than does the RE ratio. This indicator was further developed by Huysman et al. (2015a) to account for the potential substitution of different materials. From the viewpoint of program developers, it is important to stress this fact within project calls and to provide insights to help select the most suitable approach, e.g., as done by Allacker et al. (2014) in the framework of product policies support.

A portion of the innovation projects funded aim to develop new applications (e.g., materials with new functional properties). Given this, the choice of a benchmark process or product can be challenging, particularly because such a process or product might not yet exist. However, project developers may find existing applications replaced by newly developed processes or products, and several functionalities may need to be considered. In such a case, a “basket” of products or services should be considered. This is also the case for animal feed based on a new feedstock, the composition of which should be detailed to define the functional unit (e.g., to provide certain amounts of fat, fibre, and minerals).

4.6 Criticality in the evaluation of resource efficiency

Today, LC-based methods for evaluating resource availability only consider availability issues resulting from the physical extraction of resources. However, it has been shown that resource availability highly depends on socio-economic parameters such as geopolitical issues, market stability and international regulations (Dewulf et al., 2015b). These considerations can only be accounted for in a criticality assessment, which is typically conducted outside of the LCA framework. The criticality of a resource is defined by its importance in the economy and the risk of a resource supply disruption (EC, 2014b). A criticality assessment can thus be conducted for non-renewable as well as biotic renewable resources. The EC conducted such an assessment for six platinum group metals (PGMs), seventeen rare earth elements (REEs) and three biotic resources (EC, 2014b). Such information should be considered by project developers when evaluating the RE of process or product design alternatives under development. However, the current state of method development does not yet allow this assessment to be considered in the RE ratio, and criticality indicators can only be considered, in our opinion, as additional indicators. The main drawback of criticality is that it depends significantly on socio-economic parameters, which vary over time and therefore should not be considered as a standalone aspect. Although recent attempts have been made (Sonnemann et al., 2015; VDI, 2016), a framework made available to a large public to assess criticality at the life cycle level is still lacking.

4.7 Dealing with data availability and representativeness

The availability of data at the stage of research and innovation is often a limiting factor to conduct an LCA. For example, chemical processes with a Technology Readiness Level (TRL) between 0 and 3 are most probably at a small lab scale without continuous equipment operating and without sensoring. Primary data gathered at this scale can be those of oversized or non-adapted equipment with process conditions still to be optimized and thus not representative of the final eventual processes. In those cases, a complete LCA study is difficult to conduct, and simple process efficiency indicators (e.g., atom efficiency of the chemical reaction) up to gate-to-gate indicators might be used. However, a life cycle thinking approach (which does not necessarily implies exact quantification) is still possible, for example by estimating the potential effects of the sourcing of the materials and the energy requirements (e.g., heating of the reaction) on resource efficiency, or quantitatively by already checking the Life Cycle Impact of the utility. For higher TRLs, data on the use phase of the product can still be lacking, e.g., data on the shelf life and consumers' behaviour. Moreover, when conducting an LCA, the location where the technology is assumed to be implemented can have a large effect on the RE ratio, as some key processes such as the electricity mix and the waste management scheme are spatially dependent. To deal with the different reasons for the lack of data while the research and innovation process is progressing, scenarios analysis should be conducted, for example by considering a worst case (e.g., landfilling) and a best case scenario (e.g., recycling as end-of-life stage). Those scenario analyses provide a better, more holistic understanding of hotspots of the current process under development and key drivers for improvement. All in all, they can provide valuable decision support.

Often, those studied processes are compared with benchmark products or processes that are themselves already implemented at industrial scale. In those cases, it is even more important to carefully ensure representativeness and comparability of the scale. As highlighted by Shibasaki et al. (2006) and Gavankar et al. (2015), one way to deal with this issue and account for the potential economy of scale is to model an upscaled system. Assumptions on upscaled data need to be made, for example based on experts and manufacturers consultation, process simulation or the review of literature and databases (e.g., see Gavankar et al. (2015), Taelman et al. (2013) and Kralisch et al. (2013)). Moreover, the future resource efficiency of a process can be estimated based on learning curves, as already done for house appliances (Weiss et al., 2008), to estimate future energy savings when implementing energy-efficient technologies in the US iron and steel industry (Karali et al., 2015) and to estimate the scaling effect of heat pump and biomass furnace technologies on environmental impacts (Caduff et al., 2014).

5. Paths forward

The RE of a process is the ratio of the benefits obtained from this process divided by the amount or the impact of the resources consumed. While this concept is well accepted, it is not consistently applied. Furthermore, numerous approaches are followed to calculate the numerator and denominator of this ratio. Other projects have proposed various sets of indicators to evaluate the resource efficiency of a process. However, their flexibility to the specificities of research and innovation projects is limited, e.g., because they are data intensive. Moreover, they can lead to neglect resources that are not considered in the indicators set but are key resources for some technologies (e.g., land for biomass processing). This chapter stresses the need to conduct RE evaluation based on an informed choice of the evaluation method from call managers and project developers.

Several recommendations can be drawn to harmonize and improve the approach followed to evaluate the resource efficiency of innovation projects. In the case of project funding by public authorities, these recommendations can be implemented either in the project calls or by project developers. Some calls are specific to a sector or to an application. In these cases, most of the recommendations mentioned below can be implemented as requirements in the call itself (e.g., choice of the method and resources considered). Other project calls are more general, and specific RE evaluation requirements may not be as easy to provide. In these cases, the call should require project developers to follow the recommendations given below and to clearly define and justify the choices made in the proposal to evaluate RE within their project. Fig. 4 summarizes the main steps to be followed when evaluating the RE of research and innovation projects. These steps are developed hereafter.

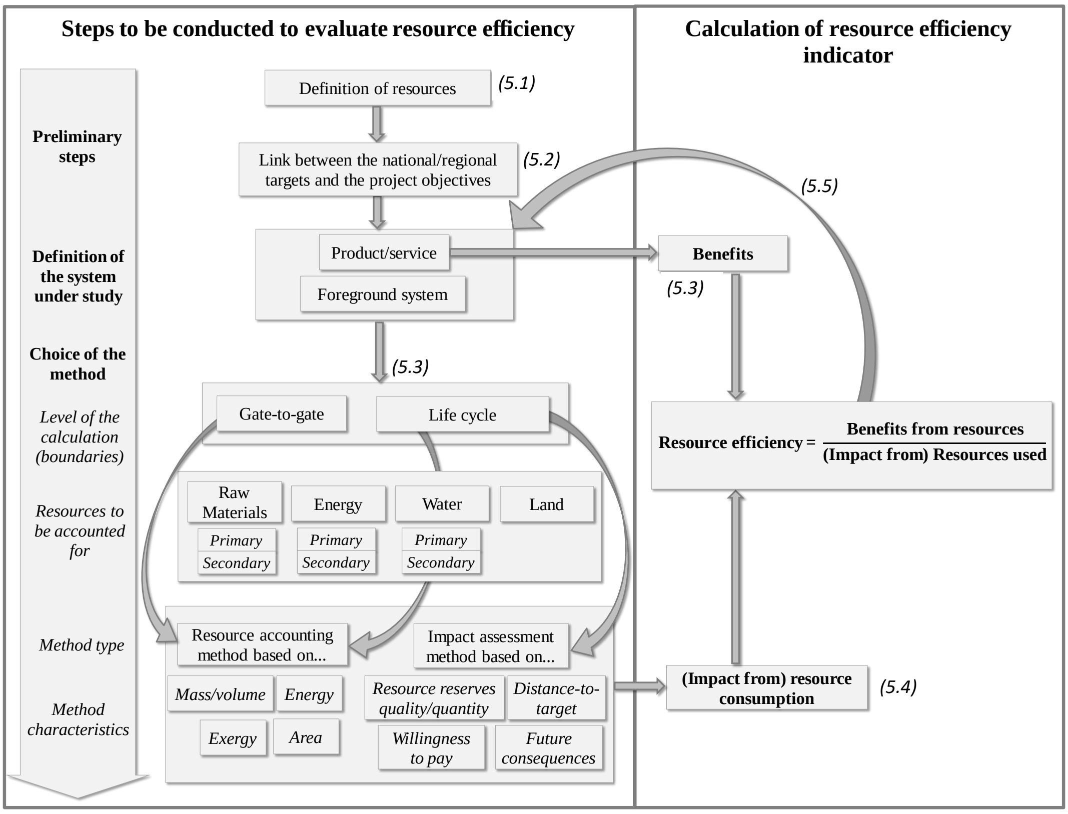

Figure 4: *Steps required to advance the evaluation of resource efficiency of innovative products and processes (the numbers on the diagram refer to the paragraphs of this part).*

5.1 Toward a more consistent vocabulary and definitions

The vocabulary used to discuss resources in research and innovation program documents is not always consistent and can be confusing for project developers. For example, the lead indicator defined by the EC in the framework of the resource-efficient Europe Flagship Initiative, GDP/DMC, includes both fossil fuels and non-energy carriers; however, programs such as SPIRE define targets for raw materials and energy intensities separately. Similar confusion can be found when comparing policy documents from national or regional programs. Thus, the terms used to set targets or define project goals in research and innovation program documents and their associated calls should be clarified. Definitions and arguments in support of these terms are provided in section 3.2.

While separate targets are defined for materials and energy, the way fossils fuels should be included in the calculation of these efficiencies is still unclear as fossils fuels can be used both as materials and as energy carriers. Today, the way LCI databases are built does not allow making the distinction between fossil fuels used as materials or as energy carrier: when one wants to assess material efficiency, fossil fuels included in the evaluation will also include fossil fuels used as energy carriers. The same goes for material efficiency calculation. This makes it difficult for project developers to compare their results with policy targets. To be conservative, the category “fossil fuels” should still be considered in the calculation of both efficiencies. Work on LCI databases should also be conducted to allow making the distinction between fossil fuels used as materials and those used as energy carriers.

5.2 Linking resource efficiencies at the micro- and macro-levels

Innovation programs should contribute to overall policy goals. However, these projects are generally conducted at the micro-level (process, factory, product). Today, there is no direct link between RE indicators calculated at the micro-level and RE targets such as those defined at the EU level. If targets are set, level and scope at which these targets are valid should be noted. This would allow for more systematic calculations and more readily link these goals to macro-level policies. As mentioned above, this link can be defined in the call itself or by the project developers in project proposals when calls are more general. This information would help evaluating the most promising projects contributing to the increase of the resource efficiency of the country or region. For example, the outcomes of research and innovation projects with a low Technology Readiness Level (TRL) could be compared, the most promising technologies in terms of RE could be identified and defined as the focus of the next calls aiming to implement technologies at higher TRLs.

One key aspect to define the link between resource efficiencies at micro- and macro-levels is the market share of the new product or service. Indeed, a small increase in a process's RE when that process is associated with large markets can contribute more to an increase in the overall national or regional RE than a large increase in RE in processes associated with niche markets. Different scenarios regarding the substitution of the current product/service by the alternative can be analyzed, as done by Rohn et al. (2014) who evaluated the resource saving potential of several alternative products by extrapolating the resource consumption of products at micro-level to the national level. For technologies with a low TRL, project developers should model an upscaled system to allow a fair comparison between the new and the benchmark technologies.

5.3 Toward a more informed choice of the numerator and denominator of the resource efficiency ratio

The benefits of resource use (numerator) should be defined based on the function of the output product or service and thus should also account for their lifetime. The definition of these benefits is key to identifying the benchmark product(s) or service(s) to which the studied product/service can be compared (see 4.5).

A wide range of methods exists to quantify resource consumption (denominator), and several choices are necessary to select the most appropriate method(s). Based on the discussion above (see 4.1 and 4.3), we propose the following recommendations as a basis for future RE evaluations in research and innovation projects:

- To calculate the denominator, an LCA should be performed. If constraints concerning money, time or data availability are too high, a life cycle approach (i.e., not necessarily including quantification) should be followed, at least based on gate-to-gate data. A gate-to-gate analysis is a limited approach but can be very useful in the calculation of intermediary indicators in order to promote continuous process improvements and provide details about the studied process. Conducting an MFA/SFA allows calculating useful gate-to-gate indicators on the process “metabolism” and allow modelling a consistent system as a basis for the LC-based analysis;

- One or several methods covering all resource categories should be selected. When dealing with abiotic renewable resources, a method that allows the presentation of results without considering these resources should also be considered;
- Only a gate-to-gate approach is able to consider waste as an input in the denominator. If an LC-based method is used to calculate the RE of a recycling process, the use of metrics other than those defined in the RE ratio here should be considered (e.g., the RBR) or the study should be completed by calculating gate-to-gate indicators;
- An ideal assessment of RE and the impact of resource consumption that could be universally used in all research and innovation projects does not yet exist and further research is needed (Geldermann et al., 2016). Various methods exist and address different aspects of RE, such as specific resource properties or specific issues related to the impact of resource consumption. The limitations of each method should be kept in mind and accounted for as much as possible via a sensitivity analysis on key methodological choices. The challenges related to resource efficiency evaluation for specific process types or sectors could be discussed in each sector, as done by Ardashkin et al. (2014), who reviewed the approaches for RE evaluation followed in the foundry sector and discussed potential ways to improve this evaluation.

These recommendations can be used as a unified basis for RE evaluation. Then, the project developers will have to adapt this evaluation to the specificities of their project. For example, for technologies with low TRLs, a full LCA is difficult to conduct because of low data availability, and gate-to-gate indicators will have to be coupled to a qualitative analysis of the life cycle impact of the product.

5.4 Aggregation versus a set of individual scores

Most methods used to quantify resource consumption allow project developers to obtain a single result as the denominator of the RE ratio. The benefit of this is that the provision of a single number is easy to communicate. However, single scores do not inform project developers of the amount of each resource consumed or the ability of a project to reach specific targets, such as those set by the SPIRE roadmap for raw materials and energy intensities. Therefore, these resources should be accounted for separately, ideally with the option to aggregate them at a later time. This may require additional work, as some LCA software tools do not provide disaggregated data for certain methods' characterization factors into different categories. Project targets should specify which resources require an increase in efficiency.

5.5 Toward the integration of resource efficiency considerations during the project lifetime

RE indicators are calculated to evaluate the impact of research and innovation projects. They offer a major opportunity for policy makers to measure progress within innovation programs and can be indirectly used to calculate a return on investment. Currently, there is too much confusion to allow such a systematic approach. Furthermore, these calculations are usually conducted at the end of a given innovation project and are often considered a constraining and subsidiary step to fulfil the call's requirements. However, a more systematic integration of RE considerations during the course of these projects could help project developers achieve higher RE goals. As with the integration of LCA in product development projects debated by Hauschild et al. (2004) and Millet et al. (2007), and successfully applied by Kralisch et al. (2013) and Ott et al. (2014), the integration of RE assessments early in a project would be useful to exclude bad options (Kralisch et al., 2016). Integration of RE evaluations at later stages may be easier but can only contribute to the optimization of the already chosen solution. Thus, the enhancement of an iterative RE evaluation throughout a given innovation project, beginning with a preliminary index that leads to more elaborate indicators at the end of the project, is recommended. Gate-to-gate analyses are easier to conduct at the early stages of process development as they require less time and data but life cycle thinking is also required at the early stages of process design, especially to account for the potential impacts of use phase and end-of-life on resource consumption. Other types of indicators and indices beyond the overall RE indicator are encouraged during process design, including the Reusability/Recyclability/Recoverability rates (Ardente & Mathieux, 2014). These may be less time consuming and data intensive as a preliminary analysis than the overall RE calculation. Moreover, RE ratios simultaneously represent the benefits and (impacts from) resource use but a closer look at the denominator - (impact from) resource use - to identify hotspots, especially for process developers, may be useful.

Further work is necessary to make the framework more operational for project developers, especially to harmonize the way methodological choices are made. Two examples of aspects to be further tailored are the calculation of the market share and the upscaling of the studied system to estimate the potential contribution of a new technology to the RE of the nation or region as a whole.

5.6 Conducting more methodological research to improve resource efficiency evaluation

With the development of new so-called “circular systems” which are supposed to contribute to a more resource efficient economy, the scientific community should invest efforts in improving the way resources are accounted for in sustainability assessment studies. As aforementioned, this can be done by better defining the Areas of Protection in LCA, by defining a framework for criticality assessment and by more clearly defining the issues related to resource use. This later point is key and research is ongoing to change the perspective under which resource use is seen today. One approach has recently been proposed by van Oers and Guinee (2016) who propose to consider resource depletion as a dilution problem, i.e., that the issue related to the availability of resources is more related to the dilution of resources in the environment (e.g., via leaking from landfill) than to a transfer of resources from the natural stock to the anthropogenic stock. In the same line, Frischknecht (2014) proposes another approach which consists in estimating the amount of resource consumed as the amount of resource lost during the production of the product, whether than the amount of resource extracted as done today by the LCA community.

Moreover, the shift from a linear to a circular economy implies to rethink the way the amount of resources consumed along the products life cycle are allocated to the different products produced during this life cycle. When calculating life cycle-based RE, waste-as-a-resource used as an input in a process is generally not accounted for and is seen as gratuitous. As waste streams are increasingly seen as resources with economic values, some studies argue that the environmental burdens from waste production should be included in LC-based analyses and some approaches attribute part of the environmental impact of the production of waste to this product as well. This methodological approach should be further tested to identify its relevance and its contribution to move towards a more sustainable society.

6. Conclusion

The challenges related to increasing the resource efficiency of our society are numerous. By developing innovative technologies, the process industry can participate in tackling the issue of decreasing resource availability. This should be monitored, benchmarked and encouraged by setting targets at the process as well as national or regional levels and by providing methods and tools to measure potential improvements and induce the integration of resource efficiency considerations into each process's design. The discussion presented in this chapter highlights the need for a framework and proposes basic recommendations to improve the evaluation of resource efficiency of newly developed technologies. Today, project developers follow a wide range of methods to evaluate sustainability and do not always follow a life cycle perspective while the aim of circular products – especially those developed in the framework of public funding programs – is to contribute to increase the overall resource efficiency at the regional or national level. One major recommendation is thus to follow a life cycle perspective, when possible by conducting an LCA study. Moreover, the approach followed to choose the resource efficiency evaluation method should be transparent and based on a deep understanding of the concepts behind each method. The potential of an innovative product or technology to contribute to the environmental sustainability of the nation/region as a whole should be estimated, e.g., based on a market analysis and/or the upscaling of the technology or product under development.

Work is still needed to further develop the framework and allow its implementation in research and innovation projects. Guidelines on specific methodological choices such as the ones made when upscaling should be provided to the project developers. Some recommendations can already be implemented as new requirements in calls (e.g., in the proposal, project developers should describe the method to evaluate resource efficiency and estimate the market potential at the proposal level) and tested during the launch of a future call to see how consortia deal with these requirements and if the outcomes are more valuable than without considering the recommendations. This testing step could help better identify specific harmonization needs. To give first insights on the challenges related to the implementation of these recommendations and the potential additional information they could provide, four recommendations are tested in three case studies in the next chapters:

- Upscaling technologies so far only developed at lab or pilot level to allow a fair comparison with benchmark products *(applied in Chapter 3 on a newly developed MaB-flocs technology in the aquaculture sector)*;

- Conducting a consistent assessment based on a material or substance mass balance *(applied in Chapter 4 on the implementation of anaerobic digestion of rice straw and cow dung in rural India)*;
- Coupling gate-to-gate and life cycle indicators *(applied in Chapter 4)*;
- Reviewing the way resources consumed by circular systems are accounted for today *(applied in Chapter 5 on the valorisation of municipal sewage sludge in the Netherlands)*.

This testing step will provide first outcomes on the balance between the efforts that need to be put to apply these recommendations and the benefits from implementing them, i.e., facilitate the decision making process.

Chapter 3: Enhancing the environmental sustainability assessment of a microalgae pond through up-scaling and system integration

Redrafted from

Sfez, S., Van Den Hende, S., Taelman, S.E., De Meester, S., Dewulf, J. (2015). *Environmental sustainability assessment of a microalgae raceway pond treating aquaculture wastewater: From up-scaling to system integration.* Bioresource Technology. 190, 321-331.

1. Introduction

From 2006 to 2011, the world aquaculture production increased by 34% (FAO, 2014), leading to an increasing production of nutrient-rich waste and wastewater that need to be treated. To enhance the sustainability of intensive aquaculture systems, new waste and wastewater treatment technologies are being developed in this sector. This is the case of recirculating aquaculture systems (RASs) including a water treatment system. These RASs offer advantages in terms of reduced water consumption, and improved opportunities for waste management and nutrient recycling compared to conventional flow through aquaculture systems (Martins et al., 2010). In most RASs, effluent rich in nutrients and sludge, e.g., microscreen backwash water, is produced (Fig. 1). This backwash water needs further treatment before its discharge into surface waters. In line with the current paradigm shift towards resource recovery in wastewater technology, the sludge and the dissolved organic matter and nutrients in aquaculture backwash wastewater should be valorized via the implementation of innovative technologies.

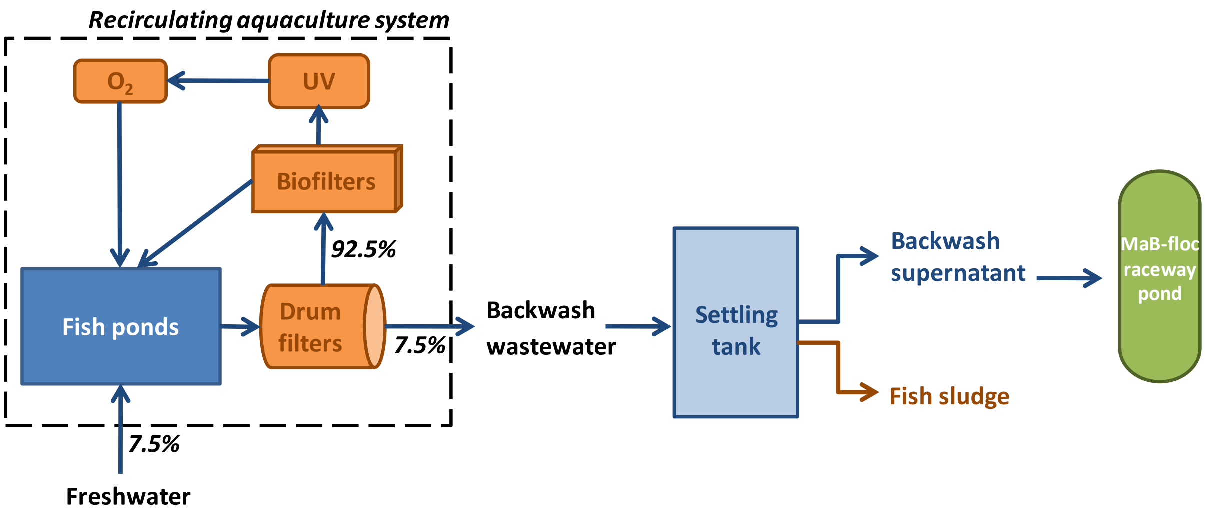

Figure 1: *Operation principle of the RAS raising pikeperch in Belgium (Aquaculture Practice Centre of Inagro, Roeselare, Belgium) releasing backwash supernatant treated in a MaB-floc raceway pond (Van Den Hende et al., 2014a).*

Because of increasingly strict regulations on discharged organic matter, the aquaculture sector is integrating fish sludge treatment in its core activities (Mirzoyan et al., 2010). Despite the presence of free ammonia, anaerobic digestion of fish sludge to produce biogas is a promising approach to reduce the environmental impacts of fish sludge treatment by recovering energy through biogas production while removing COD and BOD (Mirzoyan et al., 2010). The level of free ammonia in fish sludge can be inhibitory to anaerobic digestion but aquaculture effluents can be diluted (Mirzoyan et al., 2010) or mixed with another substrate (Nges et al., 2012) to enhance digester performance.

After removal of sludge from the wastewater, the remaining backwash supernatant needs further treatment, especially to remove dissolved organic matter and nutrients. To address the high costs of mechanical aeration in conventional activated sludge systems for treatment of backwash wastewater and to aim at a high nutrient recovery in microbial biomass, sunlight-based microalgal bacterial floc (MaB-floc, a bioflocculating consortium of bacteria and microalgae) technology was developed (Van Den Hende, 2014). In this system, costly mechanical aeration is replaced by photosynthetic aeration by the microalgae present in the MaB-flocs. In situ bioflocculation of MaB-flocs is obtained via operation as sequencing batch reactor (SBR). This enables the easy separation of MaB-flocs from the treated wastewater. Recently, in the framework of the INTERREG IVB NWE EnAlgae project, promising results were obtained for the treatment of backwash supernatant of pikeperch (*Sander lucioperca* L.) aquaculture in a pilot-scale MaB-floc raceway pond in Belgium (Fig. 1), showing a possible production of 33 ton MaB-floc TSS ha^{-1} y^{-1} (Van Den Hende et al., 2014a).

As MaB-flocs grow, they need to be harvested from the ponds and the harvested MaB-flocs need further valorisation. A possible pathway is the use as feedstock for anaerobic digestion to produce biogas. Anaerobic digestion of wastewater treatment by-products and of wastewater-grown algae has been shown to be a valuable pathway (Collet et al., 2011). Nevertheless, the anaerobic digestion conversion efficiency of MaB-flocs grown on pikeperch backwash supernatant is below 40 % (Van Den Hende et al., 2014b); a common problem in anaerobic digestion of several microalgal species (Ward et al., 2014). This is also a low-value valorisation pathway, in the order of 30-60 € per ton MaB-floc VSS (Van Den Hende, 2014).

An alternative MaB-floc valorisation pathway is using MaB-flocs as pigment-enhancing feed for herbivorous aquaculture species. Recently, it was shown that dried MaB-flocs can replace 8% of the ingredients (mainly wheat) of diets of Pacific white shrimp *Litopenaeus vannamei* (Boone, 1931) while enhancing their pigmentation (Van Den Hende et al., 2016).

Switching from linear fish aquaculture and separated aquaculture sludge and wastewater treatment to an integrated MaB-floc-based aquaculture waste treatment system could be a key strategy to mitigate the environmental footprint of the aquaculture sector; e.g., by valorizing fish sludge into biogas and recovering nutrients through MaB-floc cultivation. So far, the MaB-floc-based aquaculture waste treatment system has only been implemented at pilot scale. To know the real potential of such a technology to lower the resource use of the aquaculture sector, not only the technical potential but also the resource efficiency of such a system at industrial scale needs to be known before full implementation. Moreover, to avoid any trade-offs between resources and emissions, the impact of such a system on emissions should be conducted as well. Some studies analyzing the environmental sustainability of wastewater-based algal biofuels (Mu et al., 2014; Sander & Murthy, 2010) and biogas (Collet et al., 2011) have been performed. Few of these studies extrapolated lab or pilot scale data to model a hypothetical industrial scale system (Collet et al., 2011; Passell et al., 2013). However, the environmental sustainability of alternatives to such energy carriers has not been studied in the case of aquaculture wastewater-based microalgae production. This chapter proposes to apply an upscaling approach to a MaB-floc SBR system treating pikeperch aquaculture wastewater to provide additional insights to decision makers. It aims to answer the two following questions: (1) how can the environmental impact of a MaB-floc SBR system treating pikeperch culture wastewater be improved? (2) how should MaB-floc technology be implemented in an integrated industrial aquaculture waste treatment system to enhance its environmental performance?

To assess the environmental efficiency of an integrated MaB-floc system, this study first evaluates the resource use and impact of emissions from a pilot MaB-floc SBR raceway pond treating backwash supernatant from a pikeperch RAS in Belgium (Van Den Hende et al., 2014a). The environmental impact based on an LCA of this pilot plant was evaluated. The pilot plant was then compared to an up-scaled plant modeled as a linear projection of the pilot plant (called linearly up-scaled plant) and to three improved up-scaled plants in which some parameters were modified. To determine the potential of impact reduction associated with system integration, the improved plants were implemented into two industrial scale scenarios in which MaB-flocs were valorized into biogas or into shrimp feed. These two scenarios were compared to the baseline scenario, in which aquaculture backwash supernatant is released in the sewage system without any treatment by MaB-flocs.

2. Material and methods

2.1 Description of the MaB-floc-based raceway ponds

2.1.1. Backwash supernatant characteristics

The analyzed pilot plant treated real aquaculture backwash supernatant produced by the pikeperch RAS of the Aquaculture Practice Centre of Inagro (Roeselare, Belgium) with 174 m^3 of water in recirculation (Van Den Hende et al., 2014a). Every day, 5-10% of this water was renewed. The influent water quality (COD, BOD_5, TP, TN, pH, TOC and turbidity) was earlier presented by Van Den Hende et al. (2014a).

2.1.2. Pilot plant description

The studied system was a 28 m^2 MaB-floc raceway pond treating backwash supernatant from pikeperch culture (Van Den Hende et al., 2014a). To study the impact of several parameters on the system, Van Den Hende et al. (2014a) divided the studied period into 8 periods during which two operation parameters were modified: the hydraulic retention time (HRT) (4 or 8 days) and flue gas sparging. Thus, the average data (weighted by their respective duration) from period 4 to period 8 was used for this study, as harvesting of MaB-flocs started in period 4. On average, 2.59 m^3 day^{-1} of backwash supernatant was pumped into the raceway pond stirred by 2 propeller pumps (0.75 kW each, 14.1 h day^{-1}; see appendix A2). Bottled flue gas was sparged into the pond to regulate the pH. To study the role of flue gas and its impact on the system, no flue gas was sparged during two periods and flue gas with a lower CO_2 concentration (5% versus 12%) was sparged during the last period. Below the pond, copper tubes were used to conduct hot water to maintain a minimum pond temperature of 12°C. To maintain a concentration of 0.5 g TSS L^{-1} in the pond, MaB-flocs were harvested as previously described (Van Den Hende et al., 2014a) (Fig. 2A). On average 387 g MaB-floc TSS was daily harvested.

2.1.3. Description of the improved MaB-floc raceway pond systems

As discussed in Chapter 2, up-scaling is an important step to evaluate the potential environmental impact of the process when applied to industry, especially for MaB-floc-based systems which is a rather new field of research. Thus, a linearly up-scaled MaB-floc-based wastewater treatment plant was modeled (Fig. 2B; Table 1). Starting from this plant, three other plants were modeled, taking into account three possible improvements: improvement of stirring pumps efficiency (plant S), change of the electricity mix (plant E) and increase of MaB-floc productivity (plant M).

Up-scaling of the system – The up-scaled plant was designed to treat 1000 m^3 of pikeperch backwash supernatant per day. This volume of released wastewater corresponds to a relatively large aquaculture farm compared to what already exists in Europe for pikeperch culture. However, pikeperch culture is still a developing market in Europe, and the chosen size is a common size for other more commercial fishes, such as salmon. Thus, the chosen RAS size (13.3×10^3 m^3 of water in recirculation) allows providing realistic insights on the implementation of the MaB-floc technology in the RAS aquaculture sector. A pond area of 1 ha is necessary to treat 1000 m^3 of wastewater per day (41 raceway ponds of 245 m^2 each, with 2 meters between each pond). The HRT is set to 4 days based on Van Den Hende et al. (2014a). Therefore, for each pond of 98 m^3, every day 24.5 m^3 of the effluent is discharged and every day 24.5 m^3 of backwash supernatant is added to the pond. Because the HRT has a significant impact on MaB-floc productivity, only productivity data associated with a HRT of 4 days from the pilot plant was used for up-scaling (see appendix A7). One pond consists of a pond dug in the ground covered with a HDPE foil. One influent pump supplies each of the ponds with backwash supernatant (2.2 kW). Each pond has its own effluent pump (2.2 kW) which pumps the effluent water directly to the sewage system, and is stirred by 6 propeller pumps (0.75 kW, 14.5 h day^{-1}) similar to pilot scale (Van Den Hende et al., 2014a). The heating tubes made of copper used in the pilot plant are replaced by steel tubes. A blower is used to sparge flue gas (0.05 kW; Bosa blower, The Netherlands) with a CO_2 concentration of 5% to maintain the raceway pond pH (Van Den Hende et al., 2014a).

The harvesting steps are modified to better fit industrial conditions. One harvesting pump (2.2 kW) pumps MaB-floc liquor for 10 minutes per pond per night to one individual settling tank for each MaB-floc raceway pond. The settling tank consists of a pond dug in the ground and covered with a HDPE foil. It is only used to maintain a concentration of 0.5 g TSS L^{-1} in the pond by harvesting MaB-flocs from the raceway pond (Fig. 2). Per day, 7.4 m^3 of water from the raceway pond need to be pumped to the settling tank (effective volume: 8 m^3). On average, 155 kg MaB-floc TSS is pumped in the settling tank per day for the entire plant.

Plant S: improving stirring efficiency – Paddle wheels are the most used stirring systems in open-ponds and electricity consumption to stir microalgal raceway ponds with paddle wheels found in literature vary from 0.22 W m^{-2} (Rogers et al. (2014); velocity of 0.3 m s^{-1}) to 2.3 W m^{-2} (Passell et al. (2013), unknown velocity), which are 10 to 100 times lower than the studied linearly up-scaled plant (22 W m^{-2}). However, this data has to be used with caution as the energy required to stir a pond highly depends on its size, shape and lining and therefore electricity consumptions are hardly comparable with each other. As this technology is already widely used, it seems realistic to consider the use of paddle wheels in this study. Passell et al. (2013) extrapolated the electricity consumption used by paddle wheels in raceway ponds based on measurements made in 4 different sizes of ponds. Applying the same extrapolation to the studied up-scaled cultivation pond, an electricity consumption of 5.1 W m^{-2} was calculated. The use of paddlewheels instead of propeller pumps should of course be subject to feasibility tests and electricity consumption measured.

Plant E: changing the electricity mix – The Belgian supply electricity mix is mainly based on non-renewable energies: Belgium consumes around 80% of its electricity production, based

50% on nuclear energy and 21% on natural gas (IEA, 2012). As some electricity distributors propose to supply electricity mainly based on renewable energy, choosing such an electricity supply mix is a choice the plant managers can make. Therefore, the use of 100% of wind energy was studied to evaluate the potential benefits of using renewable energy sources. Considering this improvement option might seem utopian, as today the amount of renewable energy produced in Belgium could not supply all the Belgian plants if they would decide to switch from fossil to renewable energy. However, it is still interesting to see the potential impact reduction it could bring to processes and contribute to not forget that renewable energy sources can contribute to a more sustainable society.

Table 1: *Main differences between the designs of the MaB-floc-based pilot plants and up-scaled plants*

	Pilot scale[a]	**Linearly up-scaled plant**[b]	**Unit**
Incoming water	2.6	24.5	$m^3\ day^{-1}\ pond^{-1}$
Ponds			
Number	1	41	ponds
Pond area	12	244.6	m^2
Pond volume	28	97.9	m^3
Length	11.7	50	m
Width	2.5	5	m
Distance between each pond	-	2	m
Material	Steel + polyuretane	Pit in the ground + HDPE foil	-
HRT[c]	4 to 8 (average)	4	days
Number of stirring pumps	2	6	$pumps\ pond^{-1}$
Flue gas injection			
Concentration	89 to 214	89	$g\ CO_2\ Nm^{-3}$
Injection method	Gas valve and bottle pressure	Electrical blower	-
Power of appliance	-	0.05[c]	kW
Heating system	Copper tubes	Steel tubes	-
Settling tank	1 m^3 cubitainer	8 m^3 settling tank per raceway pond - covered with HDPE foil	-

[a] Van Den Hende et al., 2014a; [b] Collaboration with experts; [c] Bosa Ventilatoren bv - SER-8, The Netherlands; [c] At pilote scale, the HRT varied between 4 and 8 days for experimental reasons

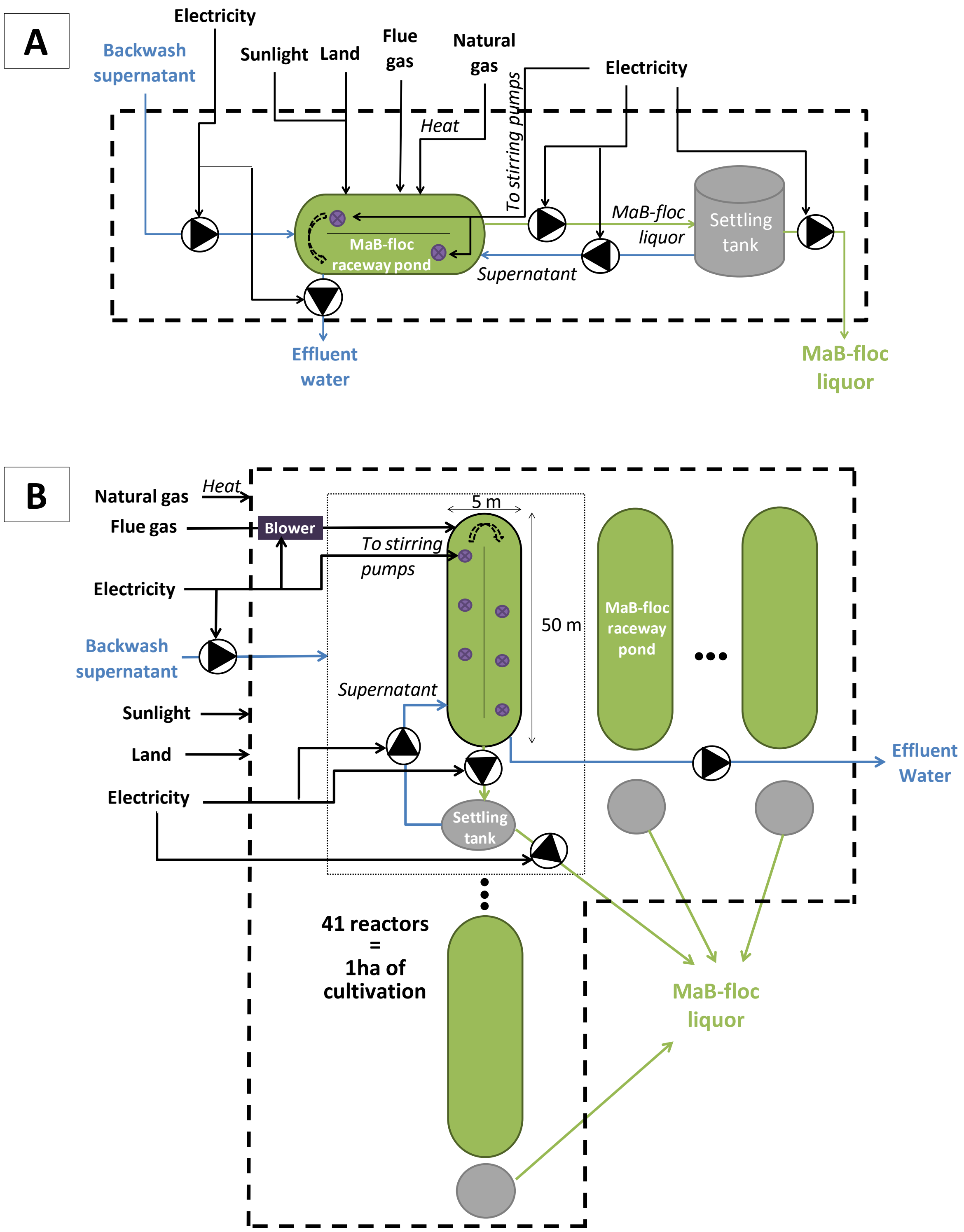

Figure 2: *Process description of the studied pilot (A) and up-scaled (B) MaB-floc-based wastewater treatment plants. The plants produce two liquid outputs: the MaB-flocs raceway pond effluent and the MaB-floc liquor.*

Plant M: improving MaB-floc productivity – At pilot scale during the treatment of pikeperch culture backwash supernatant in Belgium, 25% of the biweekly measured MaB-floc biomass productivities were negative, at least partly due to the presence of predators in the raceway pond (Van Den Hende et al., 2014a). Without the negative MaB-floc productivities, the average MaB-floc TSS productivity would have been increased by 43% (for the operation periods considered in this study). Therefore, in the presented case study, a realistic net MaB-floc productivity increase of maximum 30% was assumed.

2.2 Description of the integrated aquaculture systems

2.2.1. Overview of the main scenarios

Integrating the described MaB-floc-based wastewater treatment plant in a broader aquaculture waste treatment system can be an option to reduce the environmental impact of the aquaculture systems. In this study, three integrated scenarios with their respective valorisation scenario are considered:

- Baseline scenario: the pikeperch aquaculture system releases backwash supernatant in the sewage system. To reduce inhibition by free ammonia, fish sludge is co-digested with maize silage to produce biogas. A Combined Heat and Power system (CHP) converts the biogas into heat (used to heat the digester) and electricity which is delivered to the grid.
- Scenario 1: the pikeperch aquaculture system releases backwash supernatant treated by a MaB-floc pond. The fish sludge is co-digested with silage to produce biogas and MaB-flocs are dried to add in shrimp feed (Van Den Hende et al., 2016). Biogas is converted to heat and electricity through a CHP. Electricity is delivered to the grid and heat is used to dry MaB-flocs and to heat the raceway pond and the digester.
- Scenario 2: the pikeperch aquaculture system produces backwash supernatant treated by a MaB-floc pond but fish sludge, MaB-flocs and silage maize are co-digested to

produce biogas which is converted into heat and electricity through a CHP. Electricity is delivered to the grid and heat is used to heat the raceway pond and the digester.

For the three scenarios, the remaining heat can be used to complete the heating of the indoor pikeperch culture tanks maintained at a temperature of 24°C, without providing any surplus (Aquaculture Practice Centre of Inagro) (Fig. 3). The digestate is used as soil conditioner.

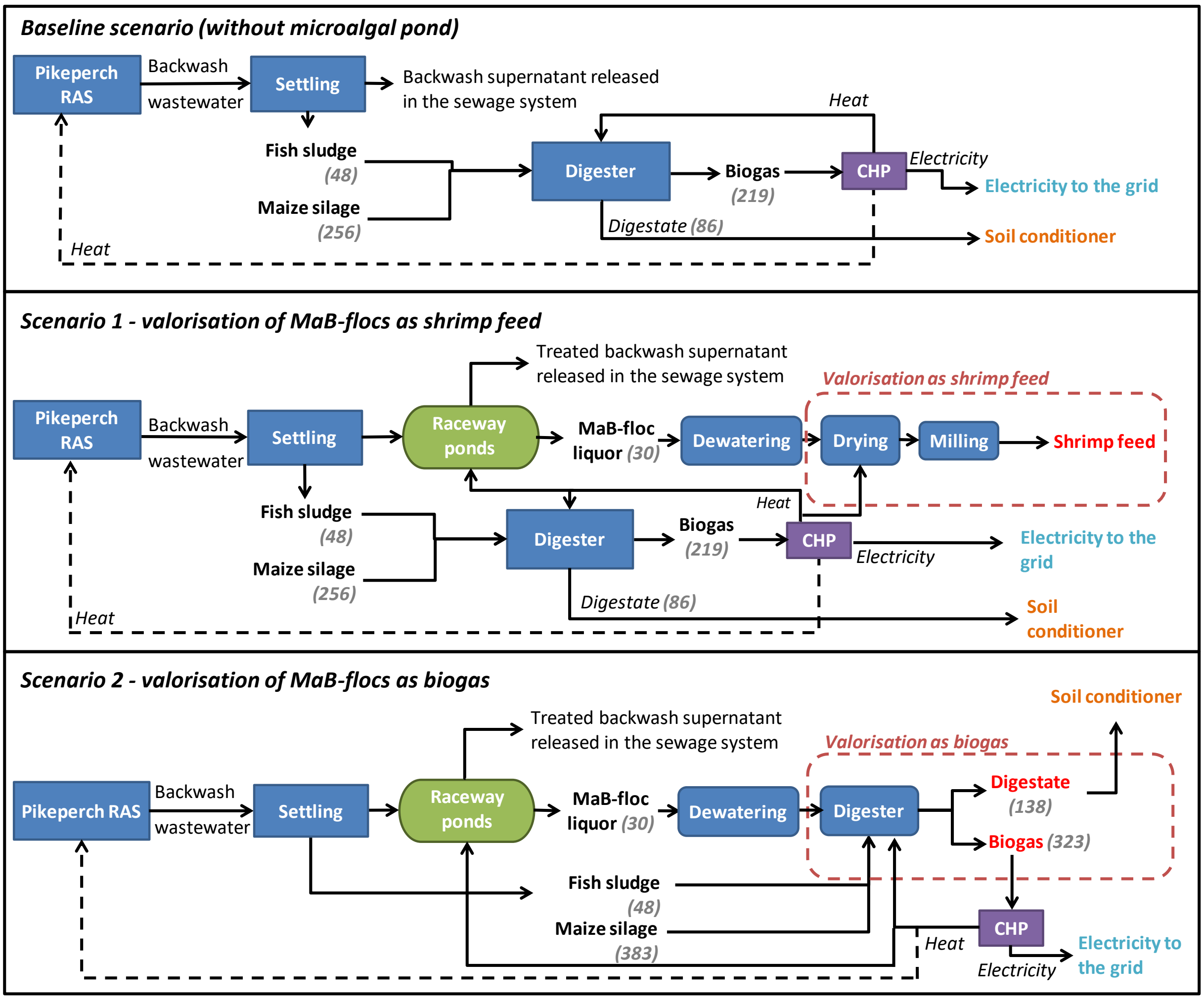

Figure 3: *Description of the three studied scenarios. The black dotted line represents the potential remaining heat produced from biogas and used to heat the fish tanks. The grey numbers between brackets represent the carbon flows within the system (in g m^{-3} of treated water).*

2.2.2. General description of the anaerobic digestion step

When silage maize and fish sludge are digested without MaB-flocs, a ratio of 1:1 for this mixture is assumed. When silage maize and fish sludge are digested together with MaB-flocs, it is assumed that the quantity of silage maize (fresh weight) added to the digester equals the total weight of the two other feedstock (wet fish sludge, which is a fixed value, and dewatered MaB-flocs; see appendix A9). The biochemical methane potentials are estimated to be 210 Nm^3 CH_4 ton^{-1} dry silage maize (33% dry matter; UK Biomass Energy Center (2014)) and 14.8 Nm^3 CH_4 ton^{-1} wet fish sludge (7.9% dry matter). The digester is fed with feedstock with the minimum moisture content of 85% required for wet anaerobic digestion (Braun et al., 2009). Co-digestion can lead to either synergetic or inhibitory interactions. As aquaculture waste, microalgae can release inhibitory ammonia during anaerobic digestion due to the low carbon to nitrogen ratio of microalgae biomass (Ward et al., 2014). However, the response of anaerobic microbes to ammonia release is a source of debate in the literature (Ward et al., 2014) and synergetic and inhibitory interactions have been reported in literature when co-digesting fish sludge or fish waste with another feedstock (Nges et al., 2012; Serrano et al., 2014). Therefore, it was assumed in this study that no synergetic or inhibitory interactions occur between feedstock. The Solid Retention Time (SRT) was set on 30 days, which is the recommended time for anaerobic continuous stirred tank reactors (Gebauer, 2004) and which is in line with the SRT for anaerobic digestion of microalgae (28 days in Sialve et al. (2009)) and fish sludge (22 to 38 days in Lanari and Franci (1998); 33 days in Nges et al. (2012)). The digester was assumed to be operated under mesophilic conditions, consuming 0.14 kWh_{elec} Nm^{-3} biogas and 4.9 MJ_{heat} Nm^{-3} biogas (ecoinvent v.2.2; Frischknecht and Rebitzer (2005)). A CHP producing electricity and heat with an efficiency of 32% and 55%, respectively (ecoinvent v.2.2) was considered. Electricity was assumed to be delivered to the grid and heat used on site and to complete heating of the fish tanks (Fig. 3). The fraction of organic carbon remaining in the digestate after anaerobic digestion was calculated as the difference between the carbon content of the feedstock and the carbon content in the biogas (see the calculation in appendix A9). It was multiplied by a humus factor, giving the total organic carbon in the digestate contributing to humus formation (Hermann et al., 2011). All processes use electricity supplied by the Belgian grid. Silage maize was assumed to be produced in Belgium and the assumption was made that maize is brought to the site and the silage is prepared there.

2.2.3. Scenario 1: MaB-flocs as shrimp feed

The use of MaB-flocs harvested from the outdoor pilot plant in Roeselare as an ingredient to substitute wheat in shrimps' diet was recently studied at pilot scale (Van Den Hende et al., 2016). In this pilot-scale study, dewatered MaB-flocs were dried in an oven at 105°C and manually milled. At industrial scale, the use of a drum dryer (3556 kJ kg^{-1} of water removed; Sander and Murthy (2010)) and of an industrial mill (4.9 kW; 150 to 160 kg h^{-1}) was assumed. After removing the supernatant from the settling tanks, the remaining settled MaB-floc liquor were dewatered through a belt filter press (0.55 kWh kg^{-1} DM; Van Den Hende et al. (2016)).

2.2.4. Scenario 2: MaB-flocs as feedstock for anaerobic digestion

The biochemical methane potential of MaB-flocs was determined in batch experiments using MaB-floc samples from the outdoor pilot plant in Roeselare (Belgium) and is 0.169 Nm^3 CH_4 kg^{-1} VS (Van Den Hende et al., 2014b). The harvesting steps (pumping and dewatering) are similar to those for scenario 1.

2.3 Life cycle assessment

The life cycle assessment methodology, as described in Chapter 1, is conducted.

2.3.1. Goal and scope

Goal 1: evaluation of the improvement potential of the MaB-floc-based wastewater treatment plant for treatment of aquaculture backwash supernatant

A cradle-to-gate LCA was conducted, from the entrance of water in the raceway ponds to the release of water in the natural environment (with the zero burden assumption). The system boundaries include materials used to build the ponds, energy consumption to operate, land occupation of the plant and treated water released into the environment (TN and TP released). The MaB-floc-based wastewater treatment plant releases water in the sewage system and produces MaB-floc liquor (Fig. 2). Water entering the sewage system is then treated in municipal wastewater treatment plants and released in the environment. Emissions of TN in the natural environment are calculated as the difference between the amount of TN released in the sewage system and the amount of TN removed in typical municipal wastewater plants. The same calculation is performed for TP released in the natural environment.

To compare the pilot and up-scaled plants, the production of 1 kg MaB-floc TSS was chosen as functional unit for the system, as the goal is to analyze how the products can be produced efficiently in the context of the treatment of aquaculture wastewater. Note that the VSS/TSS ratio of the MaB-flocs is rather low (around 30% during the last phase of the functioning of the pilot plant), which requires caution when applying the input data of this study to other types of algae that could have a higher VSS/TSS ratio. The results of the pilot plant were compared with the results of the four aforementioned up-scaled plants.

Goal 2: evaluation of the sustainability of integrating MaB-floc-based wastewater treatment systems into the aquaculture systems

A cradle-to-gate LCA was conducted, from the entrance of water in the raceway ponds to the release of water in the natural environment and the biomass valorisation as (1) production of shrimp feed (scenario 1) and (2) heat and electricity from biogas (scenario 2). The system boundaries include materials used to build the ponds, the filter press, the digester, the mill and the drum dryer, energy requirement for operation, land occupation of the plant and the digester and the release of treated water in the environment (TN and TP released).

The functional unit of the studied system is the treatment of 1 m^3 of aquaculture backwash supernatant as the goal is to analyze how this water can be used as bioresource in the most sustainable way. For the integration of the MaB-floc-based wastewater treatment plant in the integrated aquaculture system, the environmental impact was calculated for four modeled systems: $Up_{L,shrimp\ feed}$ and $Up_{L,AD}$ in which the linearly up-scaled plant is integrated in the system and the MaB-flocs are valorized into shrimp feed or into biogas, and $Up_{SEM,shrimp\ feed}$ and $Up_{SEM,AD}$ in which the up-scaled plant integrates the three improvement options. Because the amount of products delivered to the market directly depends on the MaB-floc productivity, the effect of this parameter on the environmental results of the scenarios was analyzed by comparing $Up_{L,shrimp\ feed}$, $Up_{L,AD}$ and the system integrating the up-scaled plant with a MaB-floc productivity increased by 30%.

The studied scenarios generate several other products such as biogas and shrimp feed and all the impacts cannot only be allocated to the outgoing treated water. Two approaches are possible to handle multi-outputs systems: allocation or system expansion. Following the ISO guidelines, a system expansion is used to avoid allocating the impact of the studied system to the different by-products. System expansion consists in including in the system boundaries the environmental impact of processes affected by the studied system. In many cases, these processes are avoided by the production of a substitute product. They represent the benefits of the studied system and their impact is withdrawn from the gross impact of the system. This approach was followed in this chapter (Table 2).

Table 2: *Products produced by the system and substituted equivalent products for each scenario. For scenarios 1 and 2, '-' means that values used for $Up_{L,shrimp\ feed}$ and $Up_{SEM,\ shrimp\ feed}$, or $Up_{L,AD}$ and $Up_{SEM,AD}$ are the same. Quantities are expressed per m^3 of treated water. The difference between the quantity produced and delivered to the market represents the on-site consumption.*

	Process	Products produced			Substituted equivalent products	
		Name	***System***	***Quantity***	***Name***	***Quantity***
Baseline scenario	Co-digestion of silage maize and fish sludge	Electricity from CHP engine	-	0.71 kWh	Electricity from Belgian grid	0.71 kWh
		Heat from CHP engine	-	4.38 MJ	Heat from natural gas burning (boiler)	2.37 MJ
		Digestate	-	0.49 kg compost eq.	Compost	0.49 kg
Scenario 1	Drying and milling of MaB-flocs	MaB-floc powder as shrimp feed	$Up_{L,shrimp\ feed}$	0.15 kg	Wheat-based shrimp feed	0.15 kg
			$Up_{SEM,shrimp\ feed}$	0.2 kg		0.2 kg
	Co-digestion of silage maize and fish sludge	Electricity from CHP engine	-	0.71 kWh	Electricity from Belgian grid	0.71 kWh
		Heat from CHP engine	-	4.39 MJ	No benefits (100% on-site consumption)	0 MJ
		Digestate	-	0.49 kg compost eq.	Compost	0.49 kg
Scenario 2	Co-digestion of MaB-flocs, silage maize and fish sludge	Electricity from CHP engine	$Up_{L,AD}$	1.06 kWh	Electricity from Belgian grid	1.06 kWh
			$Up_{SEM,AD}$	1.16 kWh		1.16 kWh
		Heat from CHP engine	$Up_{L,AD}$	6.53 MJ	Heat from natural gas burning (boiler)	3.47 MJ
			$Up_{SEM,AD}$	7.17 MJ		3.82 MJ
		Digestate	$Up_{L,AD}$	0.79 kg compost eq.	Compost	0.79 kg
			$Up_{SEM,AD}$	0.88 kg compost eq.		0.88 kg

The LCA community often assumes that digestate is a substitute to peat. However, the amount of organic carbon contributing to humus formation of the digestate (52.5 g kg^{-1}) is closer to the one of compost (61.2 g kg^{-1}) than the one of peat (77.7 g kg^{-1}; Hermann et al. (2011)). Thus, in practice it is assumed that the studied digestate will replace compost.

On top of the delivered products, there is also a difference in the treatment of the aquaculture backwash supernatant, i.e., in the nutrients removal rate leading to different amounts of nutrients released in the natural environment. In the baseline scenario, 1.9 g and 0.13 g of TN and TP respectively, are released per m^3 of water treated. In scenario 1 and 2, 1.3 g and 0.032 g of TN and TP, respectively, are released per m^3 of water treated.

For both goals, processes not included in the study are the construction work (excavation work and transport of material to the construction site), the release of MaB-flocs losses and press filtrate in the environment, the end-of-life of buildings, material and appliances and the transport and application of the digestate to the field.

2.3.2. Data inventory

Data for the foreground processes of the pilot plant operated in Roeselare was collected from Van Den Hende et al. (2014a), site visits and direct discussion with the author. Data for the foreground processes of the up-scaled plants and the integrated scenarios was estimated in collaboration with experts in the field and collected in literature. Data from ecoinvent version 2.2 was used to model the background systems. To model the avoided production of wheat included in shrimp feed, the ecoinvent (v.2.2) processes ‘wheat grains conventional, Barrois, at farm‘, ‘wheat grains conventional, Castilla-y-Leon, at farm’ and ‘wheat grains conventional, Saxony-Anhalt, at farm’ were used (one third of each). Electricity and heat production avoided by the valorisation of biomass into biogas were modeled by the processes ‘electricity mix BE’ and ‘heat, natural gas, at boiler condensing modulating <100kW’.

2.3.3. Life cycle impact assessment

The impact of the scenarios on 10 impact categories was studied, based on three methods. To assess the resource use of the system, the Cumulative Exergy Extraction from the Natural Environment (CEENE) method is used. It separately evaluates the consumption of abiotic renewable resources, fossil fuels, nuclear energy, metal ores, minerals, water resources and land resources (Alvarenga et al., 2013; Dewulf et al., 2007). To identify potential trade-offs between resource use and emissions, three emission-based impact categories were also investigated. The impact categories freshwater eutrophication (quantifying the emissions of phosphorus equivalents) and marine eutrophication (quantifying the emissions of nitrogen equivalents) were studied based on the ReCiPe method v1.10 (Goedkoop et al., 2013). MaB-floc productivity is expected to have an impact on nutrients removal but the exact relation between the two parameters is not known (Van Den Hende, 2014). Therefore, the effect of this parameter on the marine and freshwater eutrophication potential was not studied and only systems integrating the up-scaled plant implementing improvements S and E (called $Up_{SE,shrimp\ feed}$ and $Up_{SE,AD}$) are studied. The impact category climate change was studied based on the IPCC 2007 method (IPCC, 2007). Direct emissions of CO_2, CH_4 and N_2O are known to have a significant contribution to the global warming potential (GWP) of aerobic wastewater treatment plants (Schaubroeck et al., 2015). In the case of MaB-floc-based wastewater treatment, certain microalgal and bacterial species produce and/or remove CH_4, N_2O and CO_2 (Van Den Hende, 2014). However, no realistic data is currently available to estimate this. Therefore, the production and removal of greenhouse gases (GHG) were not taken into account in this case study.

3. Results and discussion

3.1 Sustainability and improvement potential of the MaB-floc-based aquaculture wastewater treatment plant

This section presents the results of the cradle-to-gate LCA of the pilot and the four improved up-scaled MaB-floc-based wastewater treatment plants.

3.1.1. Pilot scale

At pilot scale, 848 $MJ_{ex,CEENE}$ was required from the natural environment to produce 1 kg MaB-floc TSS (Fig. 4B). Electricity consumption to stir the raceway pond contributes the most (93%) to resource consumption from the natural environment for all CEENE impact categories except for metal ores (Fig. 4A). For the latter, the production of steel used to build the pond contributes most. Note that at pilot scale, steel was chosen to facilitate the mobility of the raceway pond to conduct experiments on different sites. As the electricity needed from the Belgian grid for stirring contributes the most to the total resource consumption of the pilot plant, mostly nuclear (50%) and fossil fuels (40%) are consumed. Infrastructure and electricity supplied to the pumps contribute each to 2.5% of the total resource footprint.

Raceway pond stirring is also the main contributor of freshwater eutrophication (75%). Infrastructure has a significant contribution to the freshwater eutrophication potential of the plant (21%), due to the use of copper as material for the heating tubes below the pond (Table 1). During copper production, phosphate is mainly released in the environment during the disposal of sulfidic tailings, a by-product of copper beneficiation (ecoinvent v.2.2.). The main contributor to marine eutrophication is the impact of direct nitrogen emissions in the natural environment (70%), followed by electricity production used to stir the pond (27%). The carbon footprint of the pilot plant is 26 kg CO_2 eq kg^{-1} MaB-floc TSS. The stirring of the pond contributes most to climate change (93%), followed by infrastructure (4%) and pumping of water (3%) (Fig. 4E).

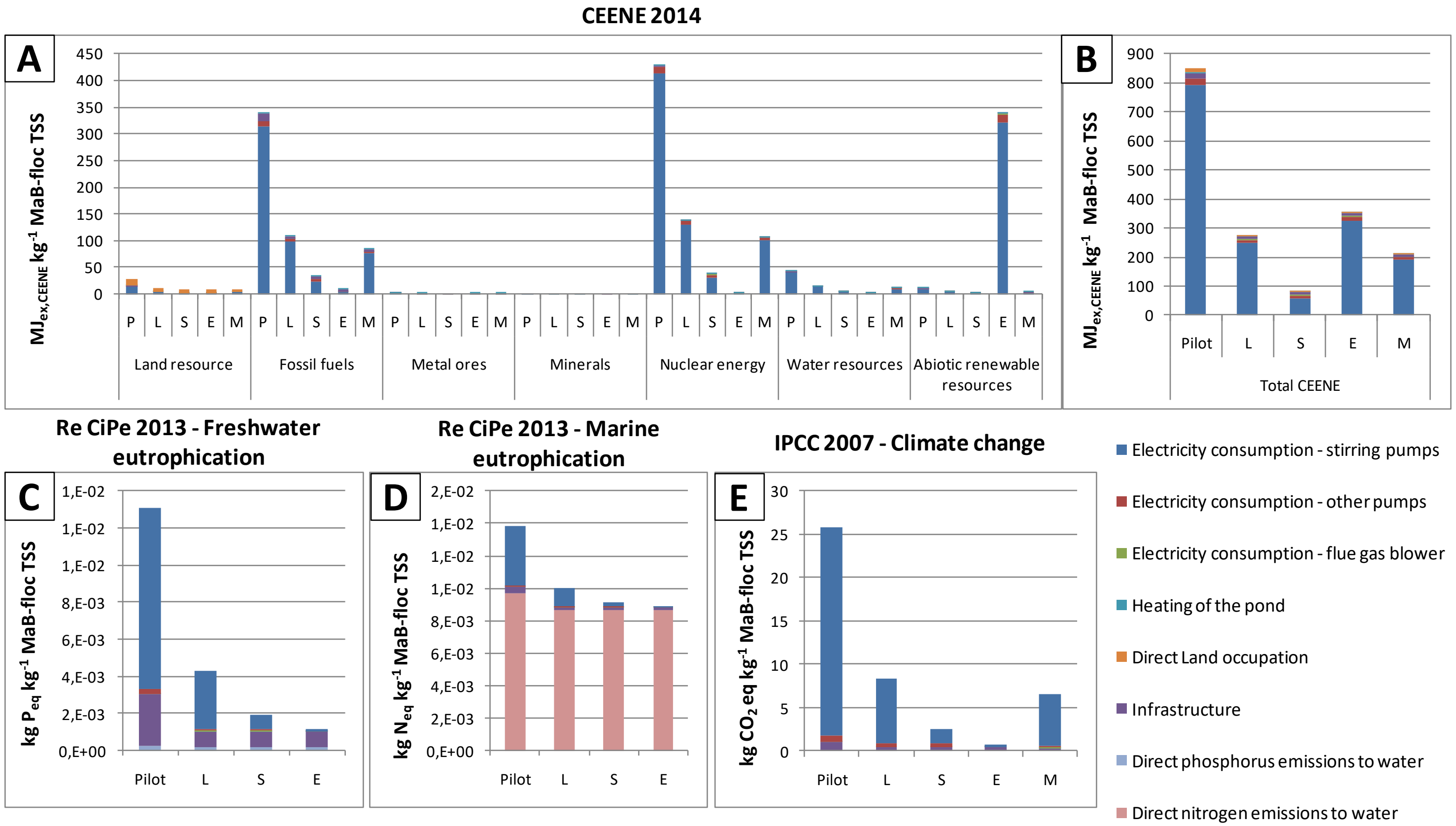

Figure 4: *Resource footprint, Freshwater eutrophication, Marine eutrophication and Carbon footprint of the pilot (P), linearly up-scaled algae-based wastewater treatment plant (L), up-scaled with paddle wheels (S), up-scaled with 100% wind power supply (E) and up-scaled with a MaB-floc productivity increased by 30% (M). Note for freshwater and marine eutrophication, the plant M was not studied.*

3.1.2. Improvement potential of up-scaling

The linearly up-scaled plant consumes 278 $MJ_{ex,CEENE}$ kg^{-1} MaB-floc TSS, which is 3 times less than the pilot plant (Fig. 4B). Up-scaling decreases land resources consumption by 57% and water resources, fossil fuels and nuclear energy by 68%. Similarly, the freshwater eutrophication and the carbon footprint of the plant both decrease by 67%. The marine eutrophication of the plant only decreases by 28% due to the high contribution of direct emissions of nitrogen and low improvement of nutrient removal rates associated with up-scaling. Up-scaling is especially beneficial for stirring and infrastructure from an environmental point of view. Due to the economy of scale, the impact of stirring decreases by 69% for all impact categories. In the up-scaled plant, the number of pumps per volume of pond is lower than at pilot scale, as the length-to-width ratio of the raceway pond is more beneficial to stirring in the up-scaled ponds (0.02 pumps per m^2 of pond compared to 0.07 pumps per m^2 at pilot scale).

Infrastructure of the up-scaled plant consumes 64% less resources compared to pilot scale (Fig. 4A). The freshwater eutrophication potential of infrastructure decreases by 70% compared to pilot scale (Fig. 4C), its marine eutrophication potential by 67% (Fig. 4D) and its carbon footprint by 62% (Fig. 4E). This is explained by the replacement of the steel tank by a dug pond and the copper heating tubes by steel tubes. Heat consumption is assumed to be proportional to the size of the pond. The 10% decrease of its contribution to all impact categories is due to a higher MaB-floc productivity per m^3 of treated water.

The high contribution of stirring in microalgal raceway ponds is in line with results found in literature (Li et al., 2014; Passell et al., 2013). Moreover, Passell et al. (2013) also highlighted the potential of impact reduction associated with up-scaling, showing a reduction of more than 90% of GHG emissions when up-scaling a 1000 m^2 microalgae production area to 101 000 m^2. However, comparison with results found in literature should be done with caution. First, unlike the studied MaB-floc raceway pond which is stirred by propeller pumps, most algae raceway ponds studied in literature are stirred by paddle wheels. Second, the scope and the functional unit are specific to each study, as well as the studied microalgae species.

3.1.3. Improvement potential of the linearly up-scaled plant

Among the different improvement options (S: replacing propeller pumps by stirring pumps; E: changing the electricity supply mix; M: increasing MaB-floc productivity), E has the lowest impact for freshwater eutrophication, marine eutrophication and climate change (Fig. 4C; 4D; 4E). The total CEENE of scenario E increases by 29% compared to the linearly up-scaled plant (Fig. 4B), mainly because of the consumption of abiotic renewable resources such as wind energy. However, abiotic renewable resources are freely available in the environment and can be withdrawn from the total CEENE. Without these renewable resources, improvement E is the option which reduces most the amount of resources consumed (plant E consumes 93% less resources than the linearly up-scaled plant, Fig. 4B). By replacing propeller pumps by paddle wheels to stir the raceway pond (plant S), the freshwater eutrophication potential of the plant decreases by 55%. It decreases by 72% by replacing the actual electricity supply by a supply mix based on wind power (plant E). The decrease of the marine eutrophication potential of the plant associated with these improvements is limited (-9% and -11% for plant S and E, respectively) as the emission of nitrogen equivalents mainly depends on direct nitrogen emissions in the environment. Improvements made in plants S and E decrease the carbon footprint of the plant, i.e., the GHG emissions decrease by 70% and 92% for plants S and E respectively (Fig. 4E).

These results show that up-scaling improves the sustainability of the studied MaB-floc raceway pond, especially its resource and carbon footprint, mainly by increasing the energy efficiency of the system.

The only parameter modified when MaB-floc productivity increases (plant M) is the volume of raceway pond water pumped from the raceway pond to the settling pond (and the volume of supernatant pumped back into the raceway pond), as more MaB-flocs have to be removed every day from the raceway pond to maintain a MaB-floc concentration of 0.5 g TSS L^{-1} in the pond. Expressed in kWh per MaB-floc TSS, the electricity consumed per day by the harvesting pump is similar than for the current productivity. When the MaB-floc productivity increases by 30%, the total CEENE and the carbon footprint of the plant decrease by 23% each.

3.1 Sustainability of integrating the MaB-floc-based wastewater treatment plant in an aquaculture system and comparison of two MaB-floc valorisation pathways

This section presents the results of the cradle-to-gate LCA of the three integrated scenarios based on the previously described up-scaled wastewater treatment plants.

3.1.1. Resource footprint

The total CEENE of the baseline scenario is -1.2 $MJ_{ex,CEENE}$ m^{-3} water treated. Even when the studied improvements are implemented in the MaB-floc-based wastewater treatment plant, the integrated systems have a higher resource footprint than the baseline scenario (Fig. 5A): the resources consumed by $Up_{SEM,shrimp\ feed}$ and $Up_{SEM,AD}$ are 65% and 70%, respectively, higher than the baseline scenario which extracts 9.8 $MJ_{ex,CEENE}$ m^{-3}. However, the two scenarios avoid more resource consumption. Scenario 1 avoids the consumption of 20.2 $MJ_{ex,CEENE}$ m^{-3}, mostly by avoiding land resources required for the production of wheat which is replaced by MaB-flocs (12 $MJ_{ex,CEENE}$ m^{-3} and 16 $MJ_{ex,CEENE}$ m^{-3} for $Up_{SEM,shrimp\ feed}$ and $Up_{L,shrimp\ feed}$, respectively). Scenario 2 avoids the consumption of 16.4 $MJ_{ex,CEENE}$ m^{-3}, mainly by avoiding electricity consumption from the Belgian grid (13 $MJ_{ex,CEENE}$ m^{-3} and 12 $MJ_{ex,CEENE}$ m^{-3} for $Up_{L,AD}$ and $Up_{SEM,AD}$, respectively). Moreover, abiotic renewable resources are freely available in the natural environment and they can thus be excluded from the total CEENE. When excluding these resources of the total CEENE, the resource footprint of $Up_{SEM,shrimp\ feed}$ and $Up_{SEM,AD}$ decrease to -10.9 $MJ_{ex,CEENE}$ m^{-3} and -0.5 $MJ_{ex,CEENE}$ m^{-3}, respectively (Fig. 5A). This then becomes competitive with the baseline scenario.

Scenario 1 has a lower resource footprint than scenario 2 ($Up_{L,shrimp\ feed}$: 34.0 $MJ_{ex,CEENE}$ m^{-3} and $Up_{L,AD}$: 42.2 $MJ_{ex,CEENE}$ m^{-3}; $Up_{SEM,shrimp\ feed}$: 4.2 $MJ_{ex,CEENE}$ m^{-3} and $Up_{SEM,AD}$: 15.1 $MJ_{ex,CEENE}$ m^{-3}), because scenario 1 consumes less resource types and a lower amount of each resource types compared to scenario 2 (Fig. 5A). As in the pilot and up-scaled non-integrated scenarios, the MaB-floc-based wastewater treatment contributes the most to the total resource consumption for both scenarios due to the high electricity consumption required to stir the pond and thus the high amount of nuclear energy and fossil fuels (for linearly up-scaled plant) or abiotic renewable resources consumed (for the up-scaled plant implementing all improvements). Anaerobic digestion is the second contributor, e.g., accounting for 36% and 50% of the consumed resources for $Up_{SEM,shrimp\ feed}$ and $Up_{SEM,AD}$ respectively (Fig. 5A). As less feedstock is digested in scenario 1, the impact of anaerobic digestion is lower than for scenario 2. Moreover, the consumption of energy from an external source to dry the MaB-flocs is low due to the on-site consumption of heat produced from biogas, fulfilling 71% of the energy requirements for drying ($Up_{SEM,shrimp\ feed}$). Thus, the gross resource consumption of scenario 1 is lower than scenario 2 (Fig. 5A).

In the current state (linearly up-scaled plant), integration options are not competitive with the baseline scenario in terms of resource efficiency mainly because the MaB-floc-based wastewater treatment plant is too energy intensive. On the contrary, when the improvement options are implemented, the benefit of delivering products to the market (shrimp feed for scenario 1 and heat and electricity for scenario 2) outweighs the gross impact of the plant itself for this impact category. In all scenarios, valorizing MaB-flocs into shrimp feed consumes less resources than using MaB-flocs to produce biogas.

3.1.2. Freshwater and marine eutrophication

Even if the direct emissions of phosphorus in the baseline scenario are 3.5 times higher than in scenarios 1 and 2, the contribution of the baseline scenario to freshwater eutrophication is lower than the two other scenarios (Fig. 5B). When the system improvements are not implemented, the freshwater eutrophication potential is 7.4 times lower (0.08 g P eq m^{-3}) as the benefits of implementing the MaB-floc-based plant (higher phosphorus removal efficiency, production of energy from anaerobic digestion) do not compensate the high impact of the inputs necessary for the functioning of the plant (e.g., electricity). However, it becomes similar when the improvements are implemented (only 1.05 and 1.02 times higher for scenario 1 and 2, respectively) as the result mainly depends on the energy efficiency of the MaB-floc-based plant. On the contrary, most of the emissions contributing to marine eutrophication are due to anaerobic digestion and direct nitrogen emissions (Fig. 5C). Therefore, the marine eutrophication potentials of scenarios 1 and 2 are lower ($Up_{L,shrimp\ feed}$: 1.9×10^{-3} kg N eq m^{-3}; $Up_{L,AD}$: 3.9×10^{-3} kg N eq m^{-3}) than the baseline scenario which releases higher amounts of nitrogen in the natural environment.

Due to a lower volume of silage maize consumed and a higher amount of avoided emissions associated with avoiding the production of wheat-based shrimp feed, scenario 1 has a lower marine eutrophication potential than scenario 2. The freshwater eutrophication potentials of both scenarios are similar (0.6 g P eq m^{-3}).

The impact of the studied systems on freshwater eutrophication is mainly reduced when improving the energy efficiency of the MaB-floc-based wastewater treatment plant whereas the impact on marine eutrophication mainly depends on direct nitrogen emissions and on the MaB-floc valorisation steps, i.e., amount of silage maize consumed and amount of replaced wheat-based shrimp feed. When all the improvements are implemented, valorizing MaB-flocs as shrimp feed has a lower marine eutrophication potential than using MaB-flocs to produce biogas, whereas the freshwater eutrophication potentials of the two options remain similar.

3.1.3. Carbon footprint

The net carbon footprint of $Up_{L,shrimp\ feed}$ and $Up_{L,AD}$ are similar (0.9 kg eq CO_2 m^{-3} and 0.6 kg eq CO_2 m^{-3}, respectively). Implementing the improvements reduces the GHG emissions by 84% and 85% for scenario 1 and 2, respectively and increases the amount of avoided emissions by 8% and 10% (Fig. 5D). When integrating the linearly up-scaled plant, the main contributor to GHG emissions is the cultivation of MaB-flocs (1.3 kg eq CO_2 m^{-3}) whereas anaerobic digestion becomes the main contributor when implementing the improvements. The production of silage maize contributes to more than 90% of the emissions associated with anaerobic digestion. The production of electricity from biogas contributes most to avoid GHG emissions. It is followed by the production of soil conditioner and by the production of MaB-floc powder, which replaces the production of wheat for scenario 1 and the production of heat from anaerobic digestion replacing the production of heat from natural gas for scenario 2. When implementing all the improvement options, the carbon footprints of scenarios 1 and 2 become negative as the emissions avoided by delivering products to the market are higher than the emissions from the processes themselves. For the same reason, the carbon footprint of the baseline scenario is negative (-0.5 kg eq CO_2 m^{-3}). The carbon footprint of $Up_{SEM,shrimp\ feed}$ is 30% higher than the baseline scenario whereas it is 60% lower for $Up_{SEM,AD}$.

The resource footprint of scenario 1 is lower than scenario 2 whereas its carbon footprint is higher, because the CEENE method applied to estimate the consumed resources takes into account land resources consumption whereas the IPCC 2007 method does not. The potential of reducing GHG emissions associated with the use of renewable energy and a more efficient stirring system is high and can result in a negative carbon footprint for both scenarios. Therefore, considering the results for all studied impact categories, valorizing MaB-flocs into shrimp feed appears to be more sustainable than using MaB-flocs to produce biogas.

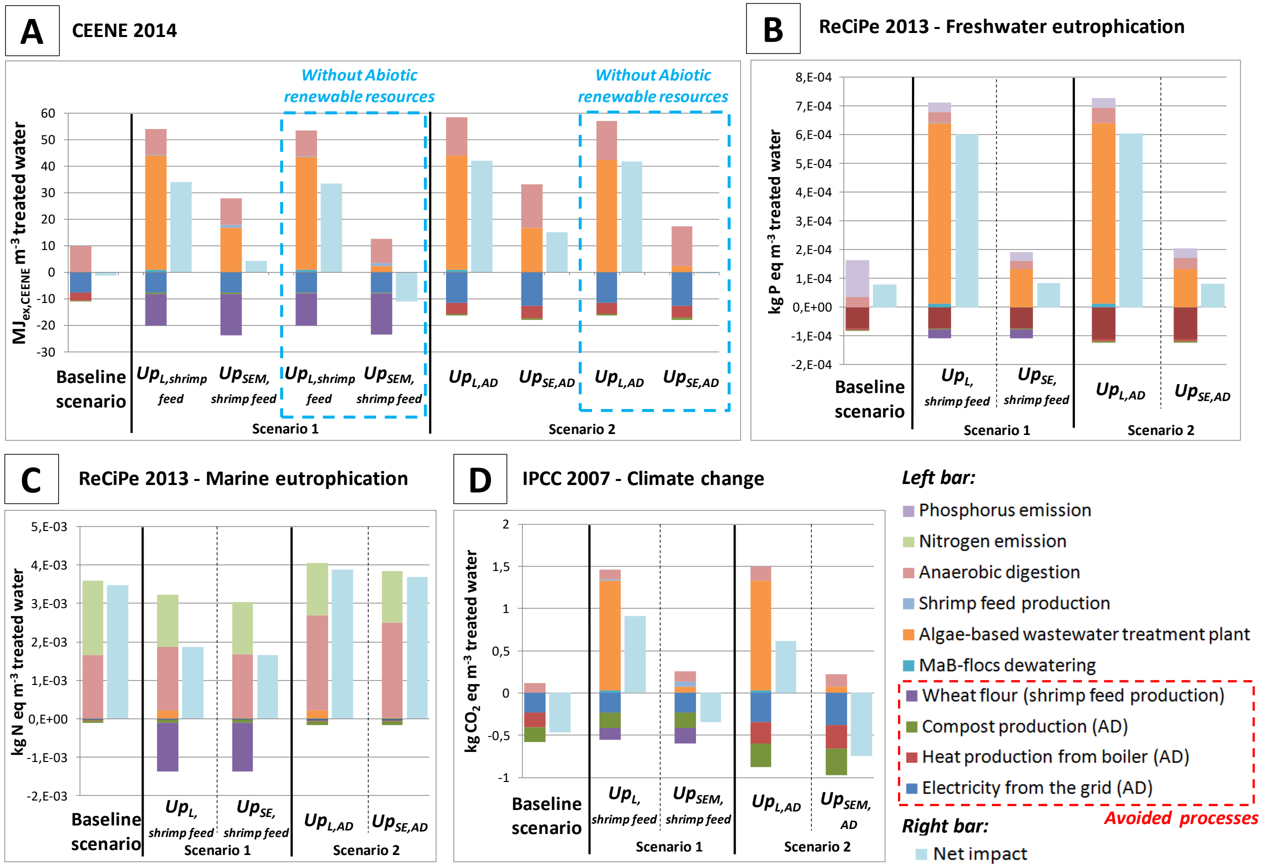

***Figure 5:** Resource footprint, freshwater and marine eutrophication and carbon footprint of the three integrated scenarios. For freshwater and marine eutrophication, only improvements S (use of a paddle wheel) and E (use of 100% wind power) are implemented. Note that the process “Algae-based wastewater treatment plant” corresponds to the results presented in Fig. 4, expressed in a different unit.*

3.1.4. Future outlooks

The energy efficiency of the MaB-floc raceway pond should be improved and a possible solution is to use paddle wheels instead of propeller pumps. This stirring system should be tested to know if it fits with Belgian conditions (possible freezing temperatures, especially during night when the pond is not stirred) as well as the conditions required for MaB-floc cultivation (necessity of a deep stirring due to the high settling speed of the MaB-flocs and possible need for high shear stress to induce bioflocculation). For both stirring systems, other improvements are possible to reduce electricity consumption. For example, changing the blade shapes of a paddle-wheel can reduce its shaft power consumption up to 50% (Li et al., 2014). Changing the design of the pond, such as adding baffle boards in the channel, can also participate to decrease the energy consumed for stirring (Chiaramonti et al., 2013).

The results show that the benefits of increasing MaB-floc productivity on the resource and carbon footprints are low compared to the efforts needed to increase this productivity. Indeed, increasing MaB-floc productivity by 30% has a negligible impact on the resource and carbon footprints of the systems, e.g., the total CEENE decreases by 7.5% for scenario 1 and increases by 0.4% for scenario 2 and the carbon footprint increases by 1.7 % for scenario 1 and decreases by 12% for scenario 2.

When MaB-flocs are valorized as shrimp feed, a more efficient drying system could allow delivering additional heat to the market and increasing the associated environmental benefits. One bottleneck when using MaB-flocs as shrimp feed in Europe is that algae grown on wastewater are not allowed to enter the European feed market (Van Den Hende et al., 2016). This restricts the use of MaB-flocs as shrimp feed ingredient at the industrial sites where they are produced.

The environmental sustainability assessment may be improved in several ways. First, data on the direct GHG emissions from the raceway pond is needed as they may have a significant contribution to the total GWP of the system. Second, comparing the characteristics of the digestate with those of the compost could help refining the substitution ratio. Indeed, in the study, only the carbon content of the digestate was considered to estimate the substitution rate of compost by digestate. However, each of the two soil conditioners have their own properties that can affect crop yield, i.e., their nutrient content (e.g., nitrogen in the digestate is more available for plants) and their C/N ratio. This should also be investigated, e.g., to assess the potential of digestate to replace synthetic fertilizers or identify in what extend replacing compost by digestate could have an impact on crop yield. A gate-to-gate analysis as recommended in Chapter 2 could be conducted to better understand the fate of the nutrients, from their emissions from the aquaculture ponds to their release in wastewater, recovery in the digestate and absorption by the plants. Third, the estimation of the amount of organic carbon available in the digestate could also be refined as some carbon can be present in the digestate in the form of CH_4 or CO_2. Fourth, the technology could be compared to other innovative technologies aiming to treat aquaculture wastewater in a decentralized manner. Fifth, in this study, the remaining heat produced by the CHP is assumed to heat the fish tanks, which will be the case most of the year in Belgium, except during hot summers. Nevertheless, valorizing MaB-flocs as shrimp feed is still expected to be the most sustainable pathway as in this case even during summer time, heat can be valorized to dry the MaB-flocs.

4. Conclusion

This chapter shows that up-scaling the MaB-floc raceway pond is essential to provide valuable insights on its environmental sustainability when comparing it with the status quo situation. The comparison of the pilot MaB-floc raceway pond with an up-scaled scenario shows a high potential of impact reduction associated with up-scaling but also a need for technology improvement to reach the level of environmental sustainability of the baseline scenario. At both scales, pond stirring has the highest contribution in all studied impact categories. When the up-scaled system is integrated, valorizing MaB-flocs into shrimp feed had a lower resource footprint, a lower marine eutrophication potential and a similar freshwater eutrophication potential than when using MaB-flocs for biogas production. In the near future, efforts should be made in priority on improving the energy efficiency of the MaB-floc raceway pond rather than on increasing MaB-floc productivity.

Chapter 4: Combining assessment methods and levels to assess the sustainability of the co-digestion of rice straw and cow dung in India

Redrafted from

Sfez, S., De Meester, S., Dewulf, J. (2017). *Co-digestion of rice straw and cow dung to supply cooking fuel and fertilizers in rural India: impact on human health, resource flows and climate change.* Science of the Total Environment. 609, 1600-1615.

1. Introduction

This chapter aims to apply a second and third recommendation presented in Chapter 2: conducting a consistent assessment based on a material or substance mass balance and coupling gate-to-gate and life cycle indicators. These recommendations are applied on the implementation of anaerobic digestion of rice straw and cow dung to provide cooking fuel in rural India, which could help the country tackle several challenges related to the access to resources but also to climate change and human health.

In India, 86% of rural households use biomass as cooking fuels (Census of India, 2011). Firewood is mainly used (73% of rural households), followed by crop residues (14%) and cow dung “cakes” (13%), which are made of a mixture of dried cow dung and crop residues (Census of India, 2011). Several surveys and measurement campaigns have shown and quantified the emissions from indoor biomass burning (e.g., NOx, N_2O, particulate matter, arsenic etc) and their impact on human health. In India, it is estimated that 400000 people prematurely die every year because of the emissions from biomass combustion in households (Wilkinson et al., 2009). Therefore, India is facing an urgent need to reduce this public health issue. In 2009, the

Indian government launched the National Biomass Cookstoves Initiative to introduce cook stoves with lower emissions. These cook stoves are mainly based on gasification technology and have shown to reduce the amount of indoor particulate matter (PM) but they are not yet as efficient in terms of indoor emissions as some other fossil fuel-based cook stoves such as LPG stoves (Venkataraman et al., 2010). Moreover, due to their low energy efficiency, they still require rural populations to dedicate a significant amount of time collecting large amounts of biomass fuels, which can represent up to 50 man-hours per month (Laxmi et al., 2003). Thus, in addition to health issues, the use of biomass fuels also lowers population's welfare as less time is left for leisure and other activities. Therefore, there is a need to implement a more resource efficient technology to provide cooking energy to the population.

Another alternative to cleaner cooking stoves is the use of biogas that can be produced from cow dung and biomass waste to replace biomass fuels. It has been encouraged since 1981 by the Indian government via two consecutive programs: the National Project on Biogas Development and the National Biogas and Manure Management Program (NBMMP) (Raha et al., 2014). These programs have encouraged the development of anaerobic digestion by the construction of household and community anaerobic digesters for lighting and cooking purposes. The results of these programs vary a lot among regions. Several surveys conducted in different Indian states showed that the percentage of non-functioning household digesters varies between 40 and 81%, mainly because of a lack of maintenance (Bond & Templeton, 2011). This can be partly explained by a variation of the socio-economic context and/or political involvement between regions and by the fact that communities do not always have the sufficient incomes to cover the maintenance costs of the constructed household digesters (Bond & Templeton, 2011). Community digesters could be an alternative in places where the implementation of household digesters has failed. Many community scale projects have been implemented, for example for the treatment of canteen waste, market waste and household cattle dung (Müller, 2007; Nasery, 2011; Reddy, 2004). Their success also highly depends on local socio-economic conditions but they have the advantage to be more energy efficient and require a lower work and maintenance load per farmer compared to household digesters (He et al., 2013).

Because of its growing population and the increase of cereals and vegetable consumption which requires higher productivity, India is expected to face a shortage of nitrogen and phosphorus in the near future and already faces a shortage of potassium in most states (Pathak et al., 2010). Synthetic fertilizers and manure are applied on 78 and 20% of the Indian gross cropped area, respectively (Agriculture Census Division, 2016), which shows that the nutrients applied in India are mainly supplied by purchased synthetic fertilizers. Therefore, the dependency of farmers towards synthetic fertilizers is likely to increase in the near future. Moreover, as a consequence of climate change, soil erosion is expected to increase in India within the next decades (Mondal et al., 2016). Soil management practices promoting the return of organic matter into the soil, including applying more soil conditioners such as manure or compost, will be necessary. In addition to provide fuels, anaerobic digestion allows valorizing various waste streams as digestate which is a stable fertilizer and soil conditioner. India produces large amounts of biomass waste that could be used as feedstock to produce biogas and digestate in which nutrients can be valorized. This is the case for a large fraction of agricultural residues, among which rice straw represents the largest volume (18.6% of the 7.6×10^8 tons of crop residues generated per year; Cardoen et al. (2015)). Around 20% of rice straw produced in India is available for further valorisation, i.e., 80% of rice straw is used for other purposes such as construction, animal feed, or cooking fuel (MNRE, 2010). Among those 20%, around 62% of available rice straw is burnt on the field (Gadde et al., 2009), which emits large amounts of greenhouse gases (GHG), air pollutants and PM (Chang et al., 2013; Jain et al., 2014; Kanabkaew & Kim Oanh, 2010). The anaerobic digestion of rice straw is seen as a promising option to valorize this waste stream and its technical feasibility has been the focus of several studies within the last years, highlighting some obstacles due to the characteristics of the straw (Li et al., 2015a). However, the biogas yield of anaerobic digestion of dry straws can be improved by pre-treatment (Hendriks & Zeeman, 2009) and the co-digestion with animal manure (Li et al., 2015a, 2015b; Mussoline et al., 2013). Moreover, the recent implementation of digesters co-digesting rice straw and piggery wastewater in Italy demonstrated the technical feasibility of straw digestion at larger scales (Mussoline et al., 2014). Therefore, the anaerobic digestion of rice straw in Asia is increasingly investigated and is a valid option to provide cooking biogas to households and replace biomass fuels in India while providing organic fertilizers and soil conditioners.

The aforementioned challenges faced by India stress the need for assessing the impacts of such systems on resource efficiency, farmers' dependency on synthetic fertilizers but also people's health, and climate change to support policy makers' in reaching their sustainability targets such as the GHG reduction targets (by 20 to 25% from 2005 to 2020; Pahuja et al. (2014)) and the energetic self-sufficiency objectives set by the Indian energy policy (Government of India, 2006). Assessment studies on the use of alternative cooking systems, anaerobic digestion and rice straw management are available in literature. However, they provide partial insights on ways to tackle the aforementioned challenges but only focusing on specific aspects. Wilkinson et al. (2009) quantified the amount of Disability-Adjusted Life Years (DALYs) that could be avoided by implementing low emissions cooking stoves in Indian households but to the author's knowledge, no study has been conducted on the avoided health issues of systems replacing conventional cooking fuels by biogas in India. Moreover, studies analyzing health impacts of indoor pollution from cooking fuels have only been conducted at local level, without considering the potential global effects of the whole system. Regarding the assessment of anaerobic digestion systems in India, most studies published on the subject focus on the optimization of biogas production and less attention is given to the digestate composition and its use as a fertilizer, especially when studying nutrients supply potentials at macro-scale (e.g., see Rao et al. (2010) and Rahman and Paatero (2012)). The potential consequences of implementing anaerobic digestion on nutrients and carbon supply at national or state level has not been evaluated yet. Moreover, few studies analyze alternative energy systems and rice straw management in India based on a life cycle approach. Kursun et al. (2015) followed a life cycle perspective and applied the emergy concept to compare several energy generation systems, including from biogas production. Singh and Gundimeda (2014) applied Life Cycle Energy Analysis to compare the energy consumption of different cooking fuels, among which biogas produced in household digesters. Soam et al. (2017) conducted the Life Cycle Assessment (LCA) of four rice straw valorisation practices, including the co-digestion with cow dung. However, this studied focused on emissions and did not evaluate the resource efficiency of this system.

This chapter aims to bring new and more complete insights on the impact of replacing conventional cooking fuels by biogas from the co-digestion of manure and rice straw in rural India by answering the following questions: 1) What are the potential impacts of using cow dung and rice straw to produce cooking energy in rural India on human health at local and global levels? 2) What is the impact of such a system on carbon and nutrients flows and on farmers' dependency on synthetic fertilizers? 3) What are the potential consequences of such a system on resource efficiency and climate change? 4) To which parameters are the results the most sensitive?

The study evaluates two scenarios. The first one represents the current cooking fuel use (mix of biomass and fossil fuels), surplus rice straw management (burnt or left on the field) and fertilizers use (mix of organic and synthetic fertilizers). The second one is a prospective scenario in which surplus rice straw and cow dung are collected and co-digested in community digesters to produce biogas used as cooking fuel and digestate as a fertilizer. The analysis is conducted for the state of Chhattisgarh, located in the so called "rice belt" of India.

2. Materials and methods

2.1 Overview of the state of Chhattisgarh

Chhattisgarh is located in central India and has an area of 135194 km^2. Agriculture is the main economic activity of the state with a dominant rice mono-cropping system covering 76% of its net sown area (CGPL, 2010). Seventy eight percent of the population lives in rural areas and 73% of workers were active in the agriculture sector in 2012 (compared to 49% for all India). Chhattisgarh is one of the poorest states of India, with a poverty rate of 40% (22% for all India) (World Bank, 2016).

2.2 Description of the current situation

2.2.1. Cooking fuel consumption

In India, one household uses on average 8.9 MJ per day for cooking (average of the values provided by Singh and Gundimeda (2014)). The state of Chhattisgarh requires $1.4x10^{10}$ MJ energy for cooking per year ($4.3x10^6$ cooking households; Census of India (2011)). In this state, 92.1% of the rural households use firewood as a cooking fuel, followed by cow dung cake, crop residues and fossil fuels (Census of India, 2011). Less than 1% of the population uses biogas. Based on the daily energy use for cooking, the energy content of the fuels and the total thermal efficiency of cooking stoves, the share of each cooking fuel can be obtained in terms of energy supply and quantities consumed (Table 1).

Firewood is mainly collected by households in the fields (twigs and thin branches) and in forests (Laxmi et al., 2003), which partly contributes to deforestation (Davidar et al., 2008; Kumar et al., 2014). Cow dung cake is prepared by mixing fresh cow dung with a small quantity of crop residues. This mixture is shaped into patties and sun dried. Considering that cow dung cake is made of 10% of crop residues (weight basis, authors' estimation), it is estimated that

2234 kt $year^{-1}$ of fresh cow dung is used to prepare cow dung cake in Chhattisgarh. Fossil fuels only represent 0.6% of the energy supply mix for cooking. They are regularly bought by households in the nearest shop. Biogas also represents a small share of the energy mix and is considered to be produced in household digesters (Singh & Gundimeda, 2014).

Table 1: *Annual cooking fuel consumption in rural Chhattisgarh.*

Cooking fuel	**Number of rural households using each fuel**[a]	**Total heat requirements for cooking (MJ $year^{-1}$)**	**Thermal efficiency of cook stoves**[b] **(%)**	**Calorific value (MJ kg^{-1})**[c]	**Quantity of fuel used (tons $year^{-1}$)**	**Energy supply for cooking (%)**
Firewood	4037767	$1.32x10^{10}$	18.0	13.9	5258519	94
Crop residues	39457	$1.29x10^{8}$	11.0	12.8	91347	1
Cow dung cake	192901	$6.31x10^{8}$	10.5	11.9	504865	4
Coal, lignite, charcoal	13152	$4.30x10^{7}$	15.5	31.4	8837	0.3
Kerosene	4384	$1.43x10^{7}$	47.0	42.9	712	0.1
LPG	7015	$2.29x10^{7}$	57.0	45.2	890	0.2
Biogas	8768	$2.87x10^{7}$	55.0[c]	17.7[d]	2944	0.2

[a] Census of India (2011); [b] Venkataraman et al. (2010); [c] Singh and Gundimeda (2014); [d] USEPA (2000)

2.2.2. Surplus rice straw management

Chhattisgarh produces $5.7x10^{6}$ tons $year^{-1}$ of rice (Pandey et al., 2016) and $8.5x10^{6}$ tons $year^{-1}$ of rice straw, 90% of which is used for other purposes such as construction and animal feed (residue to crop ratio of 1.5; CGPL (2010)) and 10% is surplus available for other usages.

In India, 62% of surplus rice straw is open-field burnt and the remaining surplus straw is left on the field (Gadde et al., 2009).

2.2.3. Cow dung management

Three flows of cow dung are considered in this study: cow dung used as fertilizer (1921 kt of farmyard manure per year obtained from the storage of cow dung in a pit; Agriculture Census Division (2016); Reddy et al. (2010)), cow dung used as a cooking fuel and cow dung used as feedstock to produce biogas. In India cow dung is also used for other purposes such as religious ceremonies or as a building material. However, these flows are not considered as available to produce biogas in community digesters.

2.2.4. Synthetic fertilizers consumption

In Chhattisgarh, synthetic fertilizers are applied on 85% of the net cultivated area. Table 2 presents the annual quantities of synthetic fertilizers applied by farmers in Chhattisgarh.

Table 2: *Annual quantities of synthetic fertilizers applied today (Agriculture Census Division, 2016)****.***

Net cropped area treated with fertilizers (ha)	Amount of nutrients from synthetic fertilizers applied (kt year^{-1})		
	N	P	K
4033900	331	162	56

2.3 Description of the prospective scenario

The prospective scenario is based on the real case of a community anaerobic digester implemented in a village of around 2200 inhabitants in the state of Gujrat in India and supplying biogas for cooking (Nasery, 2011). This case is extrapolated at the scale of the state of Chhattisgarh, assuming the implementation of one digester per village. Note that no rice straw was added in the digester in Gujrat. However, this chapter also investigates the potential of adding rice straw to produce biogas.

2.3.1. Cooking fuel and synthetic fertilizers consumption

Part of the mix used in the current scenario is replaced by biogas produced in community digesters. Therefore, a “new” mix of cooking fuels is used by households. Synthetic fertilizers are applied and the quantities required are estimated considering the quantities of nutrients supplied by the digestate (see section 2.4.3.5).

2.3.2. Surplus rice straw and cow dung management

The surplus rice straw is collected by farmers and transported to the community digester. It is stored for a continuous use in the digester during one year. Before digestion, rice straw is cut into pieces in a shredder. Every day, people from the village bring the cow dung produced by their cattle to the digester. Water is added to reach the moisture content of 85% in the input feedstock, which is required for wet anaerobic digestion (Braun et al., 2009). Biogas is then distributed via a pipeline network to households for cooking. The digestate is dried, composted and used as a fertilizer.

2.4 Sustainability assessment

The impact of both scenarios on human health, carbon and nutrients flows, resource efficiency and climate change are assessed. Impacts on carbon and nutrients flows are assessed at the level of the foreground system based on a substance flow analysis (SFA) for carbon (C), nitrogen (N), phosphorus (P) and potassium (K). An exergy analysis and an exergetic life cycle assessment are conducted to evaluate resource efficiency at the foreground and life cycle level, respectively. Impacts on human health and climate change are assessed at the life cycle level by conducting an LCA.

2.4.1. Goal and scope

The boundaries of the foreground and background systems are presented in Fig. 1. All the analyses are conducted considering the same functional unit. In this case, the “basket” of products is: the cooking energy supplied to the entire state for one year (1.4×10^{10} MJ.year^{-1}, see section 2.2.1) and the amount of nutrients made available for crops in the soil during the first year of application (234.4 kt of mineralized nitrogen N_m, 156.5 kt of P and 63.9 kt of K that plant can uptake) (see section 3.2). This amount depends on the nutrient inputs and their pathways within the system. Therefore, they are determined by conducting the SFA.

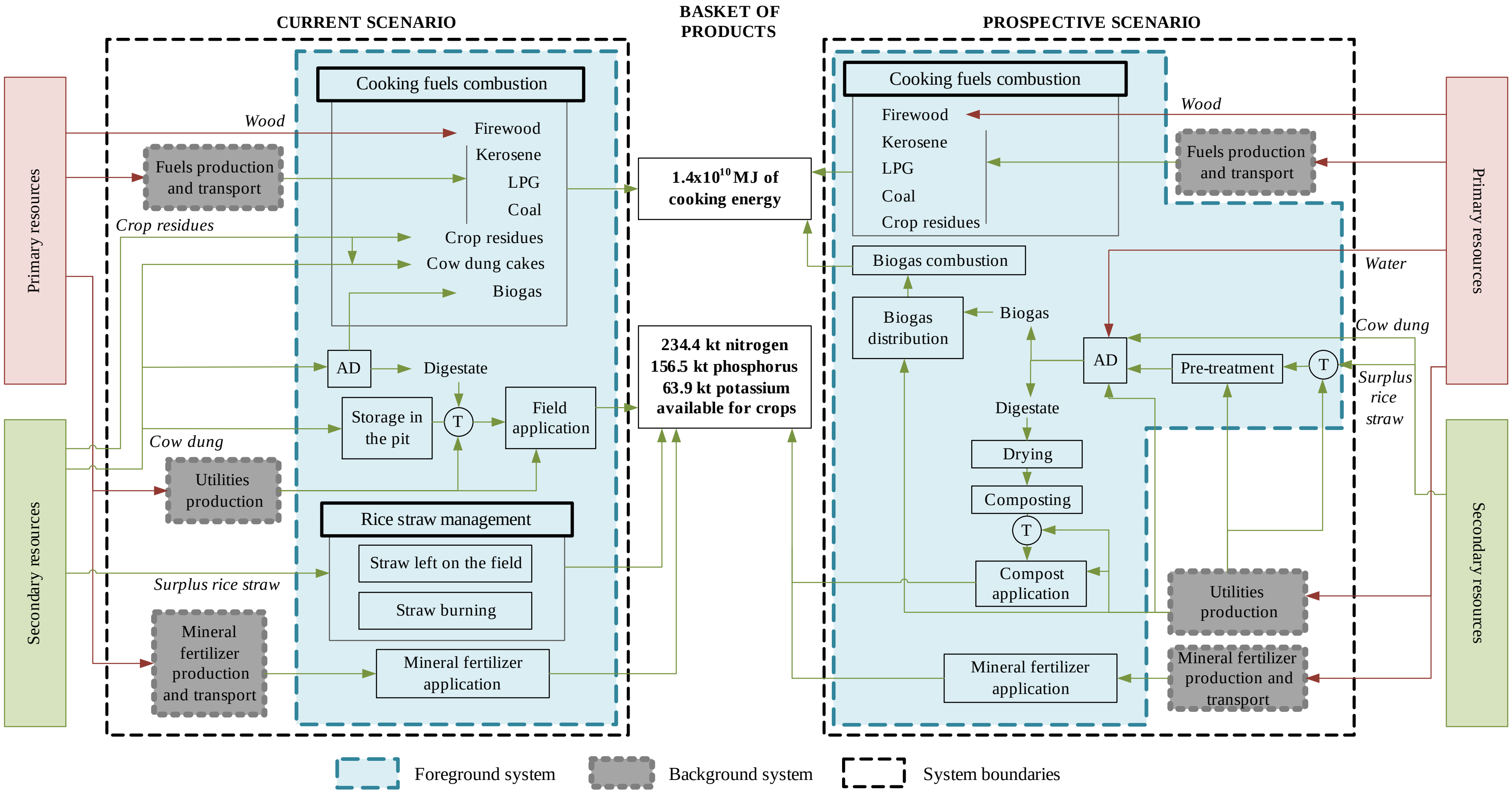

Figure 1: *System boundaries considered in the LCA analysis. Remark: the system boundaries of the SFA and the EA are those of the foreground system. AD: Anaerobic digestion; T: Transport.*

The next sections present the data inventory conducted for the assessment. Data specific for India were first looked for. If not found, data from the Asian context was investigated. As a last resort, data from other parts of the world was used.

2.4.2. Inventory of the foreground system for the current scenario

2.4.2.1. Cooking fuels consumption

Direct emissions from indoor combustion are summarized in Table 3. Ashes are assumed to be dumped without further valorisation. Fossil fuels are transported by car and motorcycle with a ratio based on Census of India (2011). The C and N content of transport fuels are based on Phyllis2 (2012). Biomass fuels are collected by foot.

Table 3: *Emissions factors of various cooking fuels*

Cooking fuel	**Emission factors (g kg fuel^{-1})**								
	CO_2[a]	**CO**	**CH_4**	**NMVOC**	**NO_x**	**N_2O**	**$PM_{2,5}$**	**PM_{10}[l]**	**SO_2**
Firewood	1553	42.8[a]	11.2[a]	9.5[a]	0.2[d]	0.1[b]	3.2[a]	10.6	0.85[f]
Crop residues	1302	65.6[b]	7.6[b]	8.5[b]	4.8[e]	0.05[b]	6.3[b]	20.9	0.56[f]
Cow dung cake	1046	39.9[b]	4.5[b]	24.2[b]	0.8[f]	0.3[b]	3.0[b]	10.7	3.5[f]
Coal, lignite, charcoal	2411	275.1[b]	7.9[b]	10.5[b]	3.3[g]	0.2[b]	17.9[b]	61.4	0.52[k]
Kerosene	2943	62[c]	1.1[c]	19.0[c]	2.7[h]	0.1[c]	n.d.	0.7[c]	0.03[k]
LPG	3085	14.9[b]	0.05[b]	18.8[b]	3.0[i]	0.15[b]	0.3[b]	1.1	0[k]
Biogas	1444	1.9[c]	1.0[c]	0.6[c]	0.9[j]	0.09[c]	n.d.	0.5[c]	0.05[j]

[a] Average value of data from Venkataraman et al. (2010) and Bhattacharya et al. (2002) ;[b] Venkataraman et al. (2010) ; [c] Total Suspended Particulates(TSP); USEPA (2000); [d] Average value of data from Venkataraman et al. (2010) and Gadi et al. (2003) ; [e] Cao et al. (2008); [f] Gadi et al. (2003); [g] Average value of data from Majumdar et al. (2013) and Sahu et al. (2012); [h] Average value of data from Pathak et al. (2009) and Sahu et al. (2012); [i] Majumdar et al. (2013); [j] Sharma and Nema (2013); [k] Grieshop et al. (2011); [l] calculated from $PM_{2,5}$ emissions, based on Ansari et al. (2010).

2.4.2.2. Surplus rice straw management

The SFA is conducted starting from the ultimate composition of rice straw (Table 4). Transfer coefficients for carbon and nutrients are then estimated for each step of rice straw management.

As the combustion is not complete, only a fraction of rice straw burnt in the field is considered to be combusted (89%; Kanabkaew and Kim Oanh (2010)). The burning of rice straw emits carbon (mainly as CO_2, CO and CH_4; see Table 5) and nutrients to the air (100%, 25% and 35% of N, P and K, respectively; Dobermann and Fairhurst (2002)).

Table 4: *Ultimate and proximate analysis of rice straw and cow dung*

		Rice straw	Cow dung
	Dry matter (%)	91.0[a]	18.7[e]
Proximate analysis (% dry matter)	Cellulose	31.0[b]	23.6[f]
	Hemicellulose	30.0[b]	13.7[f]
	Lipids	n.d.	3.2[g]
	Proteins	n.d.	18.2[h]
Ultimate analysis (% dry matter)	Carbon	39.5[c]	43.60[i]
	Nitrogen	0.64[d]	1.17[j]
	Phosphorus	0.21[d]	0.23[j]
	Potassium	1.20[d]	0.98[j]

[a] IRRI (2016); [b] Sarnklong et al. (2010), Di Blasi et al. (1999); [c] Jusoh et al. (2013); Oh and Park (2002); [d] Jusoh et al. (2013); Oh and Park (2002)[e] IAEA (2008), Ndayegamiye and Côté (1989), Chukwuma and Orakwe (2014), Liao et al. (2007), Amon et al. (2007), Chen et al. (2003); [f] Liao et al. (2007), Amon et al. (2007), Chen et al. (2003); [g] Amon et al. (2007); [h] Amon et al. (2007), Chen et al. (2003); [i] Chukwuma and Orakwe (2014); Vijayaraghavan et al. (2014); [j]Reddy et al. (2010)

Part of the carbon and nutrients remains as ashes after the combustion. The rice straw left on the field contains carbon that partly returns to the soil as organic carbon. It is estimated to be 21% of the carbon contained in straw (Hermann et al., 2011). The 79% remaining carbon is assumed to be emitted to the air.

Table 5: *Emission factors for rice straw burning*

	$PM_{2,5}$	PM_{10}	SO_2	CO_2	CO	NO_x	NH_3	CH_4	NMVOC	N_2O
Emission factors [a] (g kg DM^{-1})	5.8	6.4	0.4	1204.0	87.3	2.3	2.6[b]	9.6	7.6[c]	0.07[d]

[a] Average values from Kanabkaew and Kim Oanh (2010); [b] average values from Kanabkaew and Kim Oanh (2010) and McMeeking et al. (2009); [c] EEA (2013); [d] Chang et al. (2013)

The methane emissions from rice straw left on the field are estimated based on Liu et al. (2016) and Zhang et al. (2015) (see appendix B5). Leaving rice straw on the field can also emit dinitrogen monoxide but this quantity is not well known. Some studies show that leaving rice straw in the field emits more N_2O than when it is removed, while other studies show the contrary, depending on the cultivation and fertilization practices (Bhattacharyya et al., 2012;

Zhang et al., 2013). In this study, emissions factors from IPCC (2006) are used. It is considered that 50% of the nitrogen contained in the straws is mineralized and replaces fertilizer (Gabrielle & Gagnaire, 2008). Leached N is estimated based on IPCC (2006). The amounts of leached P and K from rice straw left on the field and ashes are based on Hokazono and Hayashi (2012) and Phong et al. (2011) (5 and 3% of initial amount, respectively).

2.4.2.3. Cow dung management

The SFA is conducted starting from the ultimate composition of cow dung (Table 4) and transfer coefficients for carbon and nutrients are estimated for each step of cow dung management. To prepare cow dung cakes, cow dung is mixed with crop residues and sun dried. Very few data is available on carbon and nitrogen losses during the drying of the cow dung cakes. The emission factors from Maeda et al. (2013) for CH_4 (2 g CH_4 kg VS^{-1}) and N_2O (20 g N_2O-N kg N^{-1}) for sun drying of feces are used. Emissions of NH_3 are taken from Laubach et al. (2013), who quantified emissions from cow dung deposited on pasture in New-Zealand. Other emissions are not taken into account. During the combustion of cow dung cakes, carbon and nutrients are emitted to the air. Carbon is lost as CO_2, CO, CH_4, non-methane volatile organic carbons (NMVOCs) and PM (Table 3). The amount of carbon in PM is calculated based on Saud et al. (2012). As for the combustion of rice straw in the field, nitrogen is considered 100% lost in the atmosphere (Dobermann & Fairhurst, 2002). The amount of phosphorus emitted to the air is estimated to 17.5% (Wang et al., 2015). For potassium, the air emission factor 18.23 mg K^+ kg^{-1} cow dung cake is used (Sen et al., 2014). Table 6 summarizes data used to estimate carbon and nutrient emissions during the storage of cow dung in the manure pit. Based on Chowdhury et al. (2014), 19% of the nitrogen in the manure after storage is estimated to be available for crops as NH_4^+. Moreover, it is assumed that 14% of the organic nitrogen in the manure is mineralized during the first year (Martínez-Blanco et al., 2013).

After storage, manure is applied on the field. In India, 95% of land preparation activities for rice are mechanized (Goyle, 2013). Fuel consumption for manure application is taken from the ecoinvent database (Frischknecht & Rebitzer, 2005). Part of the carbon in the manure remains in the soil as organic carbon after application in the field (35% of input C; Hermann et al. (2011)) and the remaining fraction is emitted to the air following biological processes. Nitrogen emissions from field application are based on IPCC (2006) (see appendix B6).

Table 6: *Data used to estimate C, N, P and K emissions from the manure pit*

Emissions		**Value**	**Unit**	**Source**
Air emissions	CH_4	6.6	mg CH_4 kg^{-1} dung day^{-1}	Gupta et al. (2007)
	CO_2-C	35.5	% of initial C content	Vu et al. (2015) and Sommer (2001)
	NH_3-N	12.5	% of initial N content	Pardo et al. (2015)
	N_2O-N	1.5	% of initial N content	
	Other N emissions (incl. leaching)	21.7	% of initial N content	
Soil emissions	N	20	% of initial N content	Reddy et al. (2010)
	P	30	% of initial P content	
	K	50	% of initial K content	

The cow dung used to produce biogas as a cooking fuel is digested in household digesters. Fugitive emissions of biogas are estimated to be 10% of the produced biogas (Bruun et al., 2014). The digestate is considered to be stored in a pit and the air and soil emissions of carbon and nutrients are considered the same as in a traditional manure pit.

2.4.2.4. Synthetic fertilizers consumption

The fuel consumption for synthetic fertilizers application is based on the percentage of mechanization for rice care in India (2%; Goyle (2013)) and on data from the ecoinvent database (Frischknecht & Rebitzer, 2005). Nitrogen emissions are based on IPCC (2006) and data to estimate P and K leaching are the same as for manure.

2.4.3. Inventory of the foreground system for the prospective scenario

2.4.3.1. Cooking fuels consumption

The cooking energy supplied by biogas replaces cooking energy supplied by other sources in the current scenario. First, it replaces all the cooking fuels that are based on cow dung in the current scenario, i.e., cow dung cakes and biogas from household digesters. Then, the cooking energy supplied by biogas is assumed to replace part of the firewood. The other assumptions (e.g., emission factors) are the same as for the current scenario.

2.4.3.2. Rice straw collection and pre-treatment

In India, respectively 75 and 25% of rice fields are harvested manually and mechanically (Goyle, 2013). When rice is harvested mechanically, it is assumed that 50% of the straw is collected by a combined harvester and 50% by a mower followed by baling. Fuel consumption for rice straw collection and transport are taken from Silalertruksa and Gheewala (2013) and Soam et al. (2017), respectively. A collection loss of 18% is considered (Mangaraj & Kulkarni, 2011). At the biogas plant, rice straws are pre-treated in a shredder (4.1 kWh kg^{-1} DW rice straw; based on data from Danagri-3S' shredders).

2.4.3.3. Biogas production and distribution

Rice straw and cow dung are co-digested to produce biogas in a floating dome digester with a retention time of 40 days (Nasery, 2011) and a capacity of 50 tons per year. The mixing electricity of the digester is taken from ecoinvent 3.1. Based on expert consultation, no heat is considered necessary in the tropical conditions of this study. The theoretical biogas potential of the mix of cow dung and rice straw and its CO_2 and CH_4 composition are estimated based on their cellulose, hemicellulose, lipid and protein content (Braun, 2007; Nzila et al., 2010) (see appendix B2). Other compounds that might be present in the biogas are neglected. The amount of biogas produced is estimated to be $6.7x10^8$ m^3 $year^{-1}$, which corresponds to a CH_4 yield of 0.255 m^3 kg VS^{-1}. This is in line with measured values found in literature for the co-digestion of rice straw with animal manure and sludge at lab scale (Lei et al., 2010; Li et al., 2015a, 2015b). The CH_4 potential calculated is higher than the one measured by Mussoline et al. (2014) for the co-digestion of pig slurry and rice straw at farm scale in Italy (0.181 m^3 kg VS^{-1}). The effect of a change of the biogas potential is tested in section 3.5. The produced biogas supplies 6.7×10^9 MJ $year^{-1}$ of cooking energy and replaces 45.6% of the firewood (weight basis). Biogas is injected into the network, which requires 0.48 MJ Nm^{-3} biogas of electricity (Evangelisti et al., 2015). Around 6.6% and 0.7% of the biogas is lost via fugitive emissions during the circulation of biogas in the inlet/outlet pipes and along the distribution network, respectively (Bruun et al., 2014; Evangelisti et al., 2015).

2.4.3.4. Digestate management

The digestate is dried and composted (windrow composting, covered). Based on Nasery (2011), the digestate is considered dried in slurry drying beds. It was assumed that the beds are made of concrete to avoid any leach in the bottom. Maurer and Müller (2010), reported that 480 m^2 are necessary to dry 60 tons of digestate during two weeks. This data is used and adapted to the amount of digestate produced. Rehl and Müller (2011) assume that 85% of the ammonium present in digestate to be dried is emitted as ammonia when reaching a dry matter content of 65% after solar drying. Based on Bernal et al. (2009) it was considered that the dry matter content of the digestate after drying would reach 45% to be compostable. Assuming that the NH_3 emissions are proportional to the amount of water which evaporates, 72% of the nitrogen is assumed to be emitted as NH_3. Moreover, based on Amon et al. (2006), 0.1% of ammonium is assumed emitted as N_2O and 0.9% and 5% of the carbon contained in the digestate to be emitted as CH_4 and CO_2, respectively.

The main difference between the manure pit (current scenario) and the compost (prospective scenario) that modifies air emissions is the fact that the compost is regularly turned and thus stored under aerobic conditions. Based on a literature review, Pardo et al. (2015) estimated that on average CO_2-C and CH_4-C emissions are 26 and 11% higher and NH_3-N and N_2O-N emissions are 68% higher and 20% lower when organic waste is under turned composting conditions than stored without turning, respectively. This data relates to input feedstock with a higher C/N ratio than in digestate but because a study on the difference of emissions between the storage of manure and the composting of manure digestate after water evaporation could not be found, these values are applied to estimate the difference of air emissions between the manure pit and the compost. Leaching losses are estimated to 2.6, 1.7 and 8.2% of initial N, P and K, respectively (Sommer, 2001). The same approach as for the current situation is followed to estimate the quantity of carbon and nutrients remaining into the soil and emitted from the field.

2.4.3.5. Synthetic fertilizers consumption

The amount of consumed synthetic fertilizers is calculated following equation 1.

$$Q_i = \frac{Q_{i,Total,CS} - Q_{i,Compost,PS} - Q_{i,Losses,PS}}{1-L} \quad (1)$$

Where Q_i is the amount of consumed nutrient *i*, $Q_{i,Total,CS}$ is the total amount of nutrient *i* made available for crops in the current scenario, $Q_{i,Compost,PS}$ is the amount of nutrient *i* from the compost made available for crops in the prospective scenario, $Q_{i,Losses,PS}$ is the amount of nutrient *i* from the rice straw remaining in the field after baling (losses) in the prospective scenario and L is the fraction of nutrient *i* from the synthetic fertilizers lost via air emissions and/or leaching in the field.

The other assumptions to evaluate nutrients emissions and fuel consumption are the same as for the current scenario.

2.4.4. Life cycle inventory

Data from ecoinvent version 3.1 (Frischknecht & Rebitzer, 2005) was used to model the background system.

2.4.5. Impact assessment

2.4.5.1. Impact on human health

The human health impact (HHI) assessment from inhalation of particulate matter (PM_{10}) is conducted at life cycle level. The HHI of PM_{10} are quantified in terms of DALYs, which is the sum of the Years of Life Lost (YLL) and Years Lost due to Disability (YLD) due to the health effects induced by the exposure to a specific substance (e.g., respiratory morbidity) (van Zelm et al., 2008). It is calculated following Equation 2.

$$HHI = FF_i \times EF_i \times DF \tag{2}$$

Where i is a substance, HHI is expressed in DALYs (years kg^{-1}), FF_i is the fate factor of i (dimensionless), EF_i is the effect factor of i (cases kg^{-1}) and DF is the damage factor of the health effect considered (years case^{-1}).

The effect and damage factors from the ReCiPe endpoint (H) method are used for cases of chronic and acute mortality, acute respiratory morbidity and acute cardiovascular morbidity (Goedkoop et al., 2013). Part of the PM_{10} emitted locally is inhaled by the local population while part contributes to the global PM_{10} pollution. Therefore, these emissions have different FF_i. The FF_i of the PM_{10} inhaled by the local population and emitted from the combustion of cooking fuels and the burning of rice straw in the field (considered as the main processes emitting PM_{10} locally) is based on the concentration of PM_{10} in the indoor environment during cooking periods (Ansari et al., 2010) and in the air during rice fields burning (Nirmalkar & Deb, 2016). An intake volume of 13 m^3 day^{-1} person^{-1} of air is used (van Zelm et al., 2008) considering that half of the household members are exposed (see appendix B11). The FF_i of the PM_{10} contributing to the global PM_{10} pollution (emitted locally or globally) is taken from the ReCiPe method. Emissions contributing to the formation of PM_{10} (e.g., NH_3) are converted into PM_{10} eq. based on the characterization factors of the impact category Particulate matter formation from the ReCiPe method. Their FF_i is taken from the ReCiPe method.

2.4.5.2. Impact on carbon and nutrient flows

An SFA is conducted for carbon, nitrogen, phosphorus and potassium at the level of the foreground system. The amount of soil organic carbon brought to the ground and nutrients from the basket of products (i.e., mineralized N, P and K) in both scenarios are accounted for. In order to evaluate the nutrient self sufficiency of the rural community, the nutrient dependency factor DF_n of the rural community for a nutrient *n* is defined as:

$$DF_n = \frac{Q_{n\ imported}}{Q_{n\ soil}} \quad (3)$$

Where $Q_{n\ imported}$ is the quantity of nutrient *n* imported from external sources into the rural community and $Q_{n\ soil}$ is the quantity of nutrient *n* contributing to the enhancement of the soil quality (see above), both expressed in kg year^{-1}.

2.4.5.3. Impact on resource efficiency

The resource efficiencies of the two scenarios are calculated at the foreground and life cycle level (Table 7). Because exergy is the physical property of resources that allow considering the widest range of resources (Sfez et al., 2017), exergy efficiency is calculated.

Table 7: *Resource efficiency ratios. Ex_{out}: exergy content of the output products; Ex_{in}: exergy content of all the inputs; CEENE: Cumulative Exergy Extracted from the Natural Environment.*

Level of evaluation	Name	Ratio
Foreground system	EE_F	$\frac{Ex_{out}}{Ex_{in}}$
Life cycle system	EE_{LC}	$\frac{Ex_{out}}{CEENE}$

At the level of the foreground system, the denominator of the resource efficiency ratio is the exergy value of all inputs entering the system while at life cycle level, the denominator is the CEENE of the system (Dewulf et al., 2007).

2.4.5.4. Impact on climate change

Impact on climate change is calculated following the ReCiPe midpoint (H) method (Goedkoop et al., 2013). Note that the characterization factor for biogenic carbon emissions is considered equal to zero.

Other impact categories such as the eutrophication and acidification potentials could be of interest to analyze as well. However, the authors chose to focus on the aforementioned impact categories because they can be directly linked to issues for which national targets have been set by the Indian government (human health, resource self-sufficiency and GHG emissions).

2.5 Perturbation analysis

The perturbation analysis aims to evaluate the level of impact that each parameter has on the results of the analysis. It is conducted by varying each parameter by a small increment (Clavreul et al., 2012). In this study, 20 parameters were modified with an increment of -10% and +10% while keeping the other parameters constant (Table 8).

Table 8: *List of the parameters considered in the perturbation analysis*

1. Biogas potential	*11. Air emissions of carbon from the pit*
2. CH_4 fugitive emissions from biogas production and distribution	*12. Air emissions of carbon from the compost*
3. Mineralization rate of nitrogen in the field	*13. Air emissions of nitrogen from the pit*
4. Humus factor of manure	*14. Air emissions of nitrogen from the compost*
5. Nutrients content of manure	*15. Air emissions of nitrogen from the application of soil conditioners*
6.Percentage of dry matter in cow dung	*16. Air emissions of nitrogen from synthetic fertilizers application*
7. Humus factor of compost	*17. Electricity consumption for digester mixing*
8. Nutrients leaching from cow dung pit	*18. Electricity consumption for biogas distribution*
9. Nutrients leaching from compost	*19. Percentage of rice straw remaining in the field after baling*
10. Nutrients leaching factor in the field	*20. Nitrogen emissions during digestate drying*

3. Results

3.1 Impact on human health

In the current scenario, the local and global impacts from PM_{10} represent 99.4 and 0.6% of the total human health impact (HHI), respectively (see Fig. 2).

Based on the assumptions presented in section 2.4.5.1, it was found that 0.03% of the locally emitted PM_{10} is inhaled by the local population. Today, the HHI of the locally emitted PM_{10} emissions on this population is estimated to $4.3x10^{6}$ DALYs per basket of product. The emissions of PM_{10} due to rice straw burning only contribute to 1.1% of the impact at local level.

The implementation of the prospective scenario reduces the HHI on the local population by 48% ($2.3x10^{6}$ DALYs per basket of product). This is mainly due to the avoided combustion of 2399 kt per year of firewood, as firewood contributes to 87 and 91% of the local health impact in the current and prospective scenarios, respectively.

The global impact due to the PM_{10} emissions and formation is $2.5x10^{4}$ and $1.5x10^{4}$ DALYs per basket of product for the current and prospective scenario, respectively. Most of the PM_{10} eq. emissions at global level are due to the local emissions from cooking fuels combustion which are not inhaled by the local population (69 and 59% of the PM_{10} eq. emissions for the current and prospective scenarios, respectively) and the field emissions from synthetic fertilizers (13% and 22% of the emissions for the current and prospective scenarios, respectively). For both scenarios, the global impact represents less than 0.7% of the total HHI (sum of the local and global impact). Therefore, most of the impact on human health is due to the local impact.

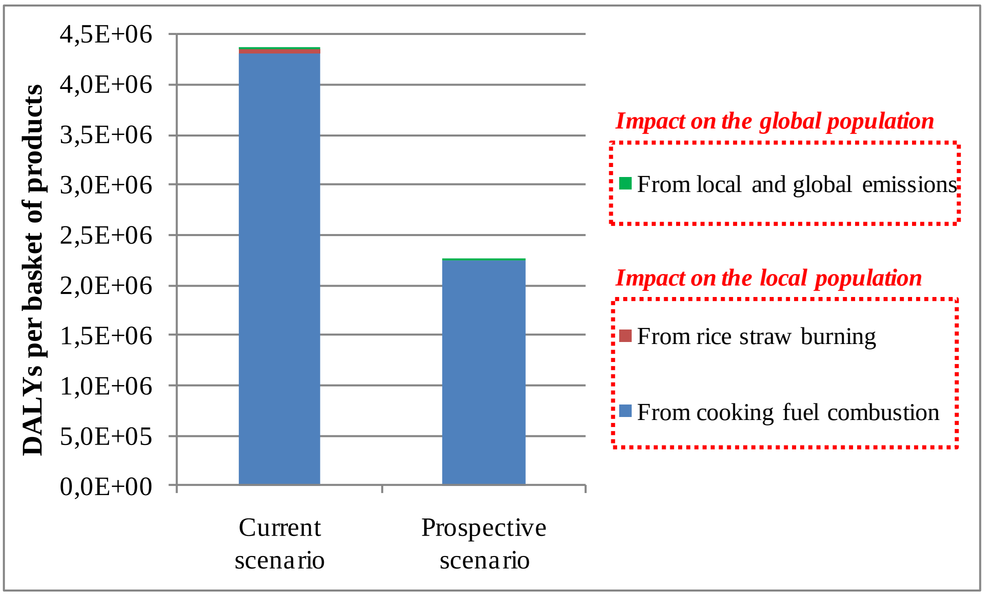

Figure 2: *Comparison of the human health impact on the global and local populations due to PM_{10} emissions and formation, expressed in DALYs per basket of products*

3.2 Impact on carbon and nutrients flows

3.2.1. Carbon flow analysis

Most of the carbon entering the system is emitted to the air (85% in both scenarios), mostly as CO_2-C (91 and 92% in the current and prospective scenarios, respectively) (Fig. 3). Direct CH_4-C, CO-C and VOCs-C emissions in the prospective scenario are 23, 55 and 57% lower than in the current scenario, respectively. This is mainly due to the replacement of cooking fuels by biogas. The same fraction of carbon in both scenarios is returned to the soil (15%) but today, 85% is in the form of ashes, against 70% in the prospective scenario. Thus, more carbon contributing to humus formation is returned to agricultural land via soil conditioner in the prospective scenario (101 kt year^{-1}) than in the current scenario (82 kt year^{-1}). The amount of organic carbon in the current and prospective scenarios represents 2.2 and 4.3% of the input carbon, respectively.

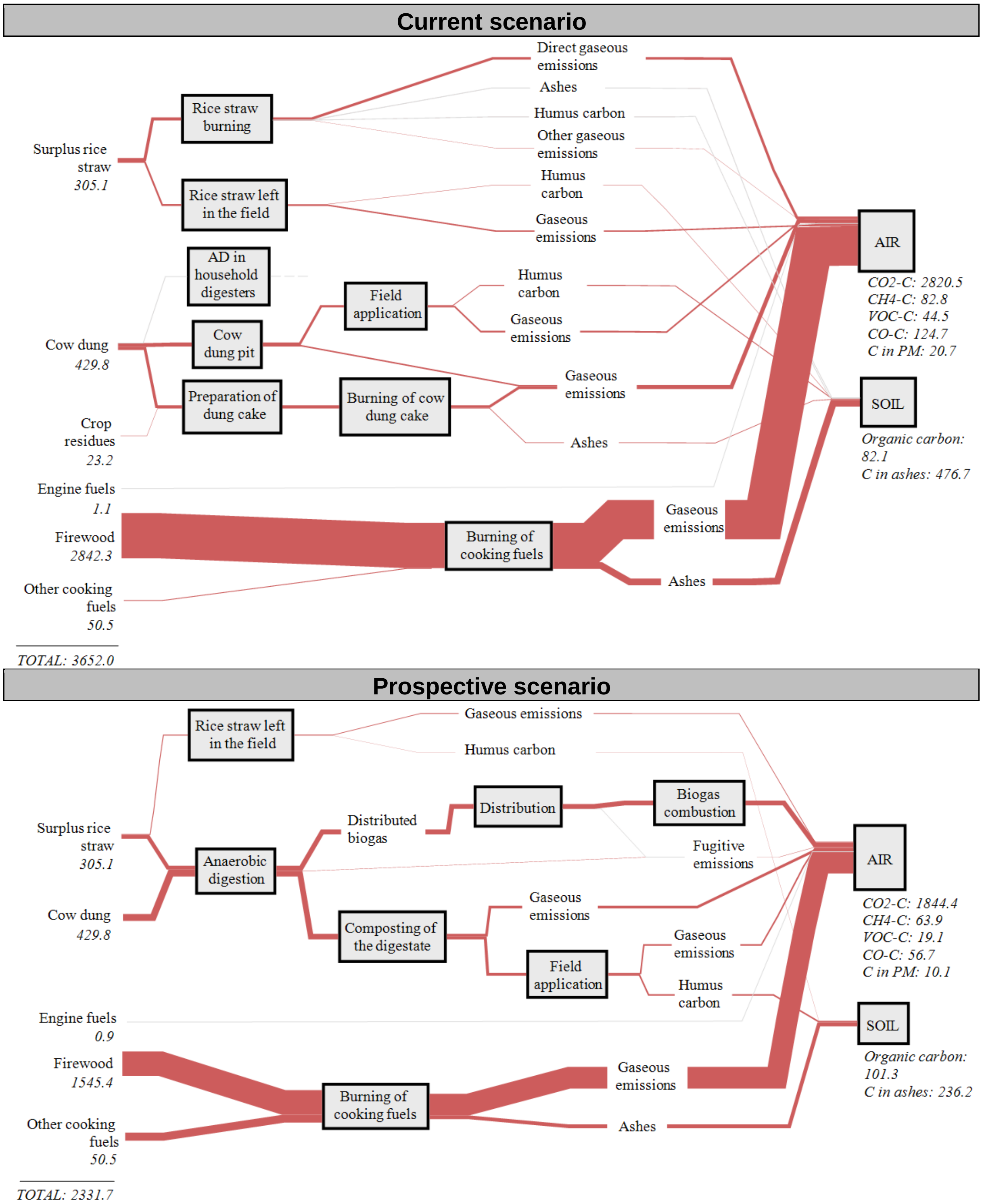

Figure 3: *Carbon flow diagram. The values are expressed in kt year^{-1}. Flows with a value lower than 10 kt are represented in grey. The flows of carbon in the sub-system providing biogas to households in the current scenario are not represented.*

3.2.2. Nutrient flow analysis

The nutrient dependency factors defined in section 2.4.5 are calculated for nitrogen, phosphorus and potassium (Table 9). In both scenarios, 235.5, 157.4 and 69.0 kt of mineralized nitrogen, phosphorus and potassium are made available for crops (basket of products).

Table 9: *Nutrients dependency factors for the current and prospective scenarios (%)*

		Current scenario	Prospective scenario
Nutrients dependency factors	DF_N	99.0%	99.1%
	DF_P	98.6%	97.7%
	DF_K	85.2%	74.3%

The DF_N of the prospective scenario is 0.2% higher than the current scenario. Today, 20% (74 kt) of the total nitrogen input is emitted to air within one year, 63% is returned to the soil (237 kt) and 17% is emitted as leachate in the water bodies (64 kt). In the prospective scenario, 17% (63 kt) of the input nitrogen is emitted to air, 65% is returned to the soil (237 kt) and 17% is emitted as leachate in the water bodies (63 kt) (Fig. 4). Air emissions are mainly NH_3 (36 and 42 kt of NH_3-N in the current and prospective scenario, respectively). The prospective scenario emits 15% more NH_3 due to higher NH_3 emissions during the drying and composting of the digestate compared to the storage of manure in the pit. All the other N-emissions from the prospective scenario are lower than in the current scenario, mostly due to the avoided combustion of firewood (-4% of N_2O-N, -19% of NO_x-N and -50% of other N-emissions). N losses under other gaseous forms have not been extensively studied in literature, especially N_2 emissions. Therefore, they are not characterized in this study. Further work is needed to identify as which gas nitrogen is emitted to air in this remaining fraction.

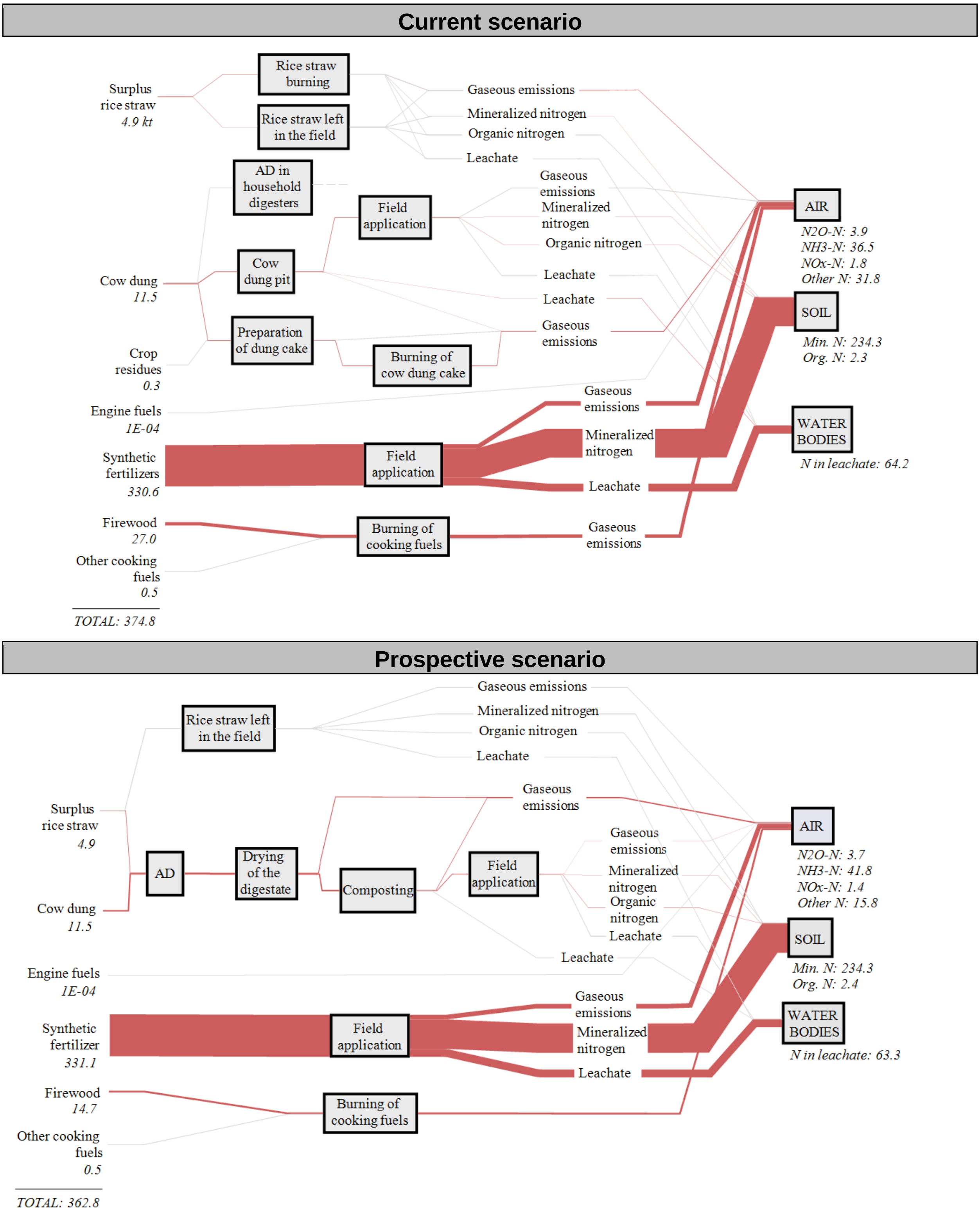

Figure 4: *Nitrogen flow diagram. The values are expressed in kt year^{-1}. Flows with a value lower than 1 kt are represented in grey. The flows of nitrogen in the sub-system providing biogas to households in the current scenario are not represented. AD: Anaerobic digestion.*

As more NH_3 is emitted in the prospective scenario and NH_3 emissions represent the main losses of nitrogen in both scenario, manure management in the prospective scenario brings less mineral nitrogen to the ground than the current scenario. However, the impact on the DF_N is low as 99% of the nitrogen made available for plants is from synthetic fertilizers.

Note that the amount of organic nitrogen in the soil is 4% higher in the prospective scenario than in the current scenario. Therefore, even though the same amount of nitrogen potentially available for plants is the same in both scenarios after one year (235 kt), there is a slightly higher potential of long term nitrogen availability in the prospective scenario as the organic nitrogen in the soil will be mineralized after the first year. Moreover, 2% less nitrogen is emitted into water bodies in the prospective scenario, mainly because of lower nitrogen leaching during storage of compost compared to storage of manure.

Today, 98% of the input phosphorus (170 kt year^{-1}) is returned to the soil, 5% is emitted to water bodies and less than 1% is emitted to the air via the emission of particulate matter (see diagram in appendix B13). The fate of input phosphorus in the prospective scenario (167 kt year^{-1}) is very similar with 99% returning to the soil, 5% emitted to water bodies and 0.2% emitted to the air. Because of the low phosphorus content of manure (0.23%) and rice straw (0.2%) and because 99% of the phosphorus applied on the fields is from synthetic fertilizers, a change in the management of organic waste does almost not affect the flows of phosphorus in the system and most of the phosphorus made available to crops is still supplied by synthetic fertilizers (98%). Therefore, the DF_P of the prospective scenario only decreases by 0.9%.

The DF_K of the prospective scenario decreases by 13% compared to the current scenario. Today, 74% of the potassium returned to the soil is made available for crops and 22% remains in dumped ashes. In the prospective scenario, 86% of the potassium returned to the soil is made available for crops and 14% remains in ashes (Fig. 5).

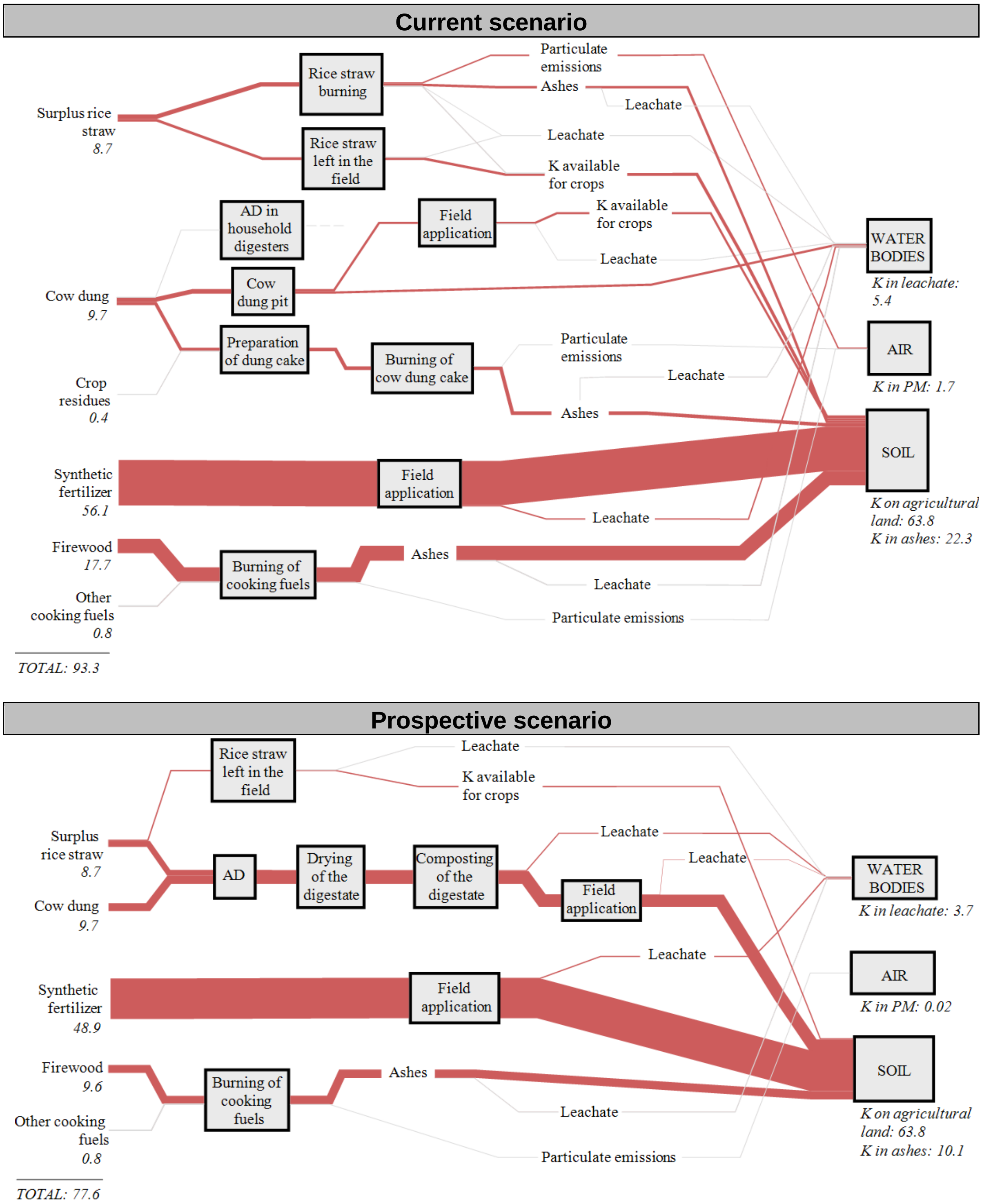

Figure 5: *Potassium flow diagram. The values are expressed in kt year^{-1}. Flows with a value lower than 1 kt are represented in grey. The flows of potassium in the sub-system providing biogas to households in the current scenario are not represented. AD: Anaerobic digestion.*

Therefore, because of the higher potassium content of cow dung compared to nitrogen and phosphorus (0.98%) and the larger fraction of potassium supplied via manure application (4% of applied potassium), a change of cow dung management, i.e., its valorisation as feedstock for anaerobic digestion rather than as cow dung cakes for cooking, has a larger effect on the potassium flows than on the nitrogen and phosphorus flows. Moreover, in the current scenario, 50% of the potassium contained in the cow dung is lost via leaching during storage whereas 8% is lost during composting in the prospective scenario. The consequence is that 13% of the potassium from synthetic fertilizers today can be replaced by potassium contained in cow dung and rice straw. Unlike nitrogen and phosphorus for which the factors do not vary much, the implementation of anaerobic digestion to produce biogas for cooking could contribute to decrease the potassium dependency of the rural population.

3.3 Impact on resource efficiency

The EE_F of the current scenario is 7.5%. By implementing the prospective scenario, the EE_F increases by 57%, thus reaching 11.7%. Today, firewood represents 68% of the exergy inputs of the foreground system (Fig. 6). Rice straw, cow dung and other cooking fuels represent 10, 9 and 8% of the total inputs, respectively. Other inputs such as synthetic fertilizers, transport fuels and electricity only contribute to 5% of the total exergy inputs. The EE_{LC} of the current and prospective scenarios are similar to the EE_F (7.4 and 11.6%, respectively). This is due to the large contribution of firewood to the CEENE (67 and 58% in the current and prospective scenarios, respectively) (Fig. 7). However, as the “zero burden” assumption is followed, the other most contributing resources are not rice straw and cow dung but the production of synthetic fertilizers (32 and 38% for the current and prospective scenarios, respectively).

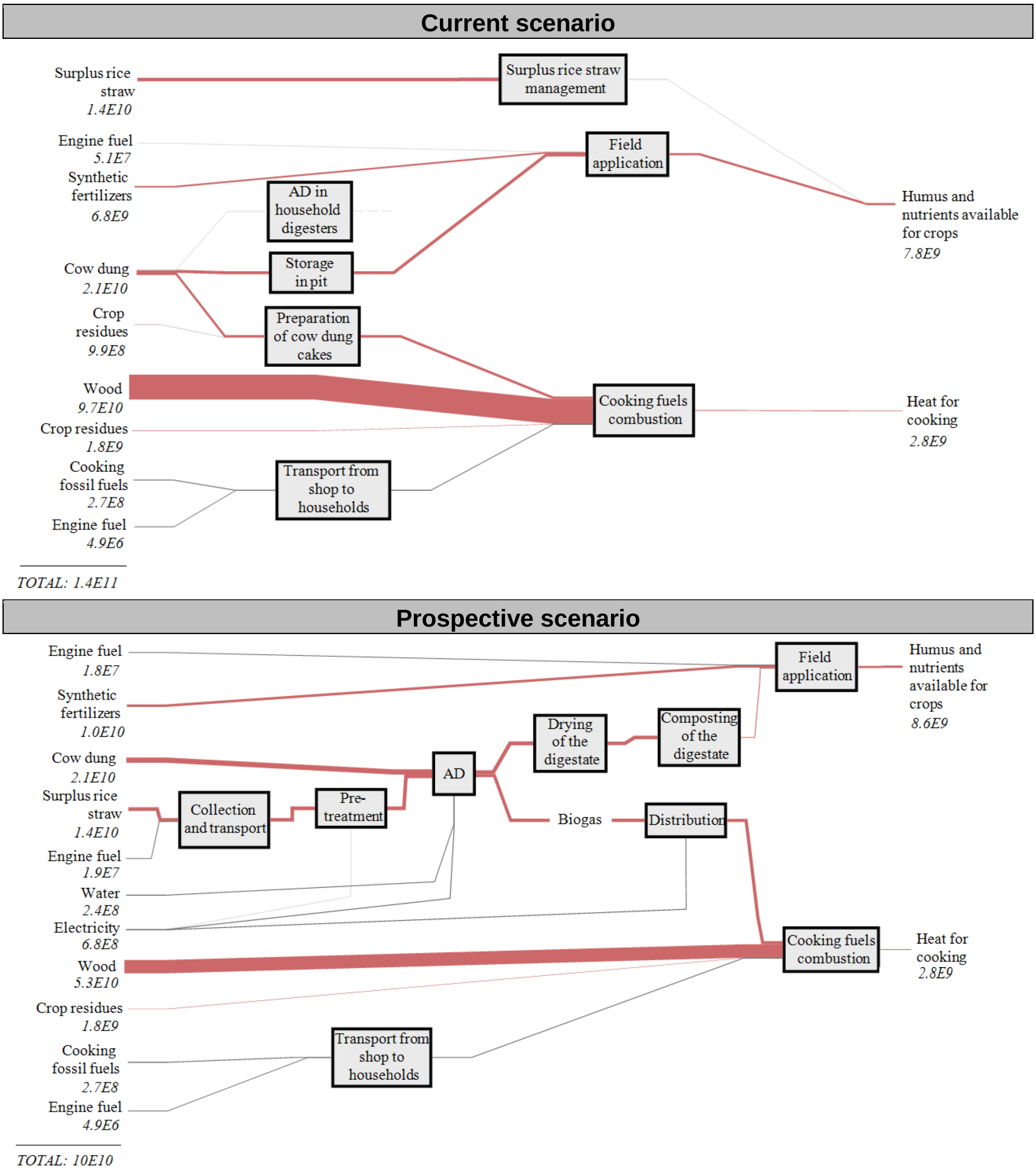

Figure 6: *Diagram of the exergy flows within the foreground system. The values are expressed in MJ_{ex} $year^{-1}$. Flows with a value lower than 1E9 MJ_{ex} are represented in grey. The flows of exergy in the sub-system providing biogas to households in the current scenario are not represented. AD: Anaerobic digestion.*

The consumption of fossil fuels contributes to 79% of the resource footprint of the synthetic fertilizers. The increase of resource efficiency factors is explained by the replacement of 46% of the firewood by biogas. Apart from cooking fuels, new inputs are required in the prospective scenarios (i.e., fuel and electricity for rice straw collection and pre-treatment and electricity for anaerobic digestion and biogas distribution) and the exergy of these resources imported in the foreground system by the rural community increases by 13%. Therefore, the rural community would rely 46% less on firewood but more on utilities bought externally.

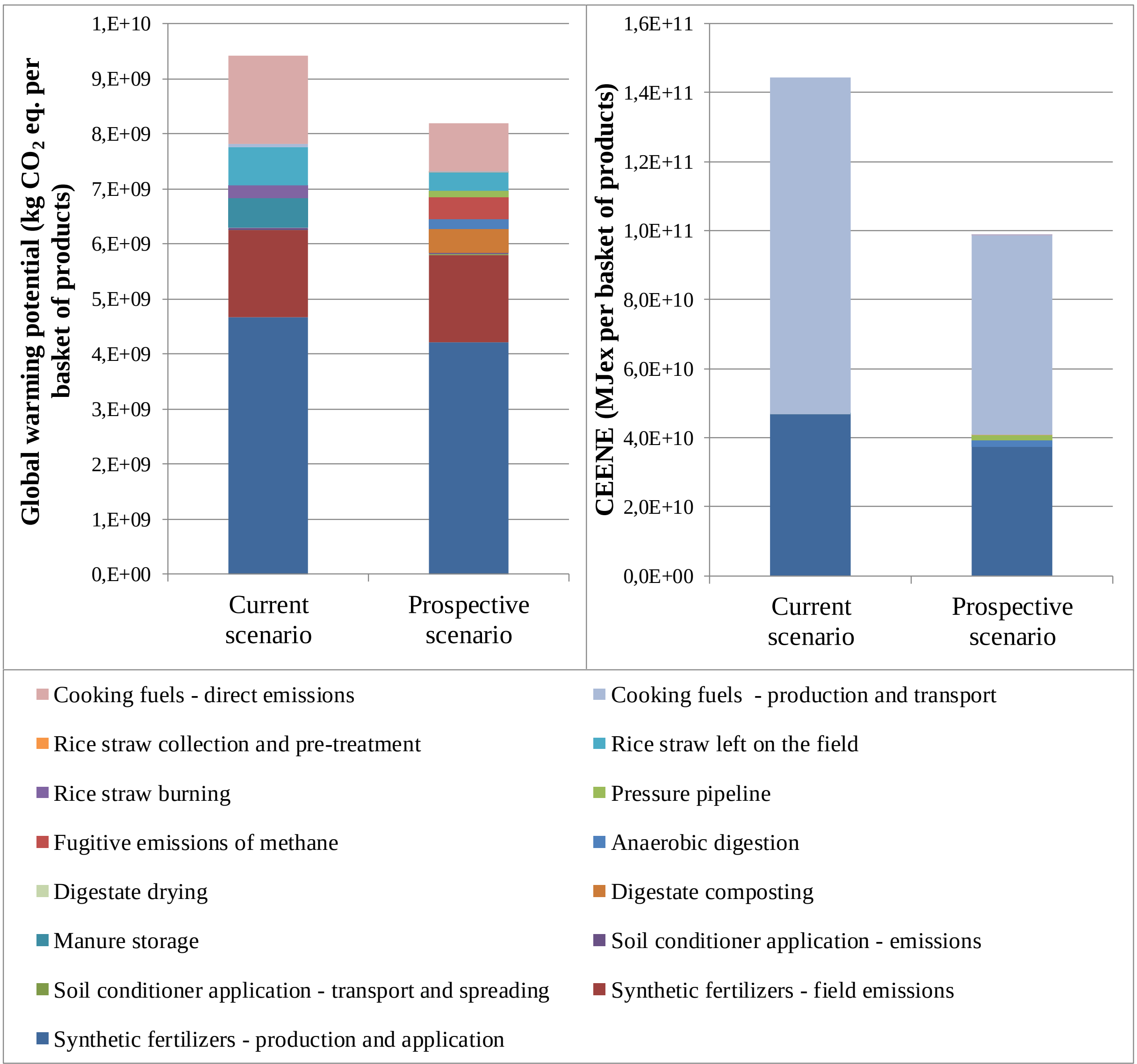

Figure 7: *Global warming potential and resource footprint (CEENE) of the current and prospective scenarios.*

3.4 Impact on climate change

The global warming potential of the prospective scenario is 12% lower than the current scenario (9.4×10^9 and 8.3×10^9 kg CO_2 eq per basket of products for the current and prospective scenarios, respectively) (Fig. 7). Today, the impact on climate change is mainly due to the production of synthetic fertilizers (50%), the emissions from cooking fuels combustion (17%), the emissions from synthetic fertilizers application (17%) and the emissions from rice straw remaining on the field (7%). In the prospective scenario, the impact on climate change is still mainly due to the production of synthetic fertilizers (51%), followed by the emissions from synthetic fertilizers application (19%) and from cooking fuels combustion (11%). Therefore, for both scenarios, more than half of the impact on climate change is due to processes located outside of the state of Chhattisgarh, where synthetic fertilizers are being produced. The GHG emissions during the composting of the digestate are lower than during the storage of manure by households. However, the total GHG emissions from manure management of the prospective scenario (anaerobic digestion and composting) are larger than the current scenario because of the fugitive emissions of CH_4 during anaerobic digestion (5% of the impact) and the emissions from the background processes to produce electricity for anaerobic digestion and the pressure lines (4% of the impact).

3.5 Perturbation analysis

A perturbation of ±10% on 20 parameters was conducted. Parameters which make the indicators variate by more than ±3% are presented in Fig. 8. Seven parameters have such an effect. The amount of carbon contributing to the formation of organic carbon in the soil is affected by 6 of these 7 parameters. The two parameters which affect the most the amount of organic carbon brought to the soil in the prospective scenario are the humus factor of compost (+8.9% for a 10% increase) and the air emissions of carbon from composting (-7.9% for a 10% increase). The exergy efficiencies of the prospective scenario are mainly affected by the humus factor of compost (+4.6% for a 10% increase for both) and the carbon emissions from composting (8.9% for a 10% increase for both). The lower these emissions are, the higher the exergy content of the compost. However, the exergy input to transport compost to the field is higher and thus the efficiency ratio decreases. The HHI of the prospective scenario is mostly affected by the biogas potential (+8.4% for a variation of -10%), as a larger amount of firewood is replaced when more biogas is produced.

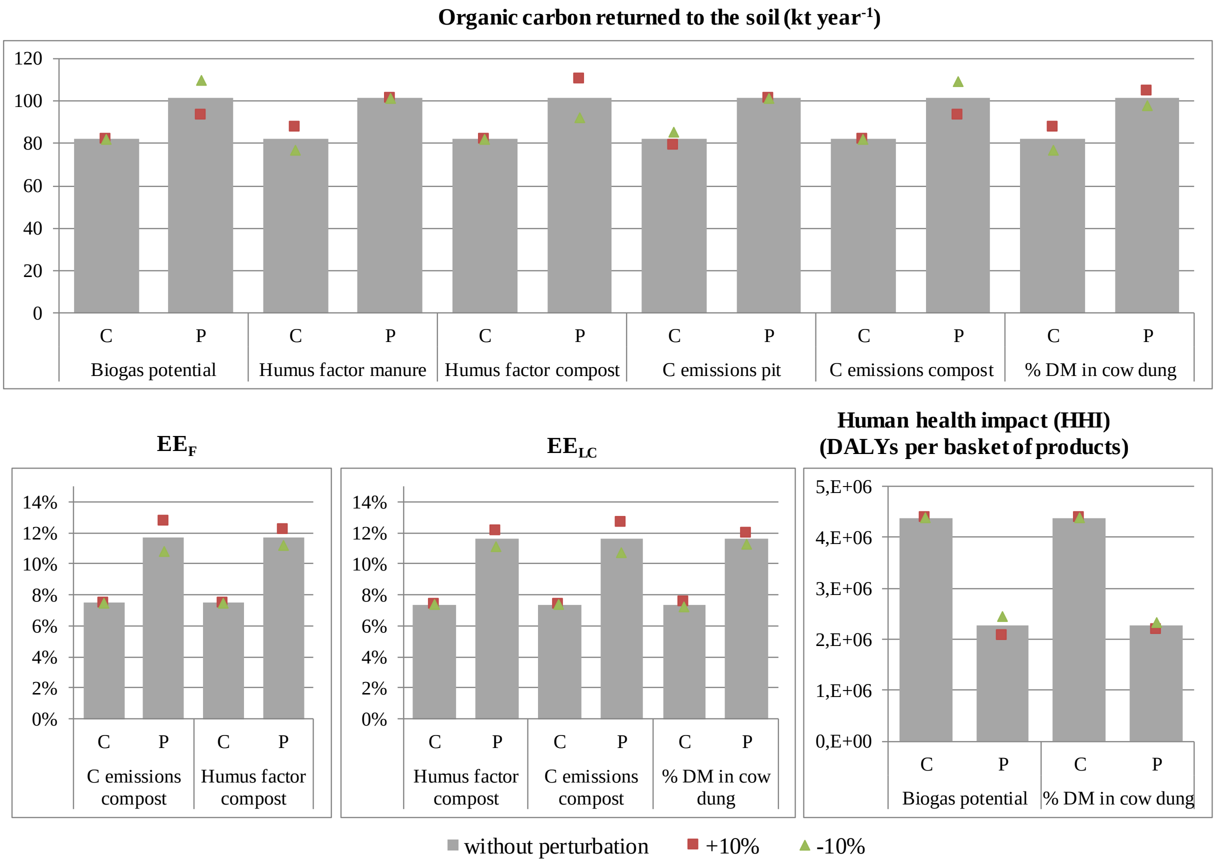

Figure 8: *Results of the perturbation analysis for the parameters affecting the most the results of the indicators (difference higher than 3% or lower than -3% compared to the results without perturbation). C: Current scenario; P: Prospective scenario.*

The parameters that characterize the feedstock under study (e.g., carbon and dry matter content) should be carefully considered when implementing such a system. Moreover, the biogas potential and the emission of carbon from composting should be carefully estimated.

However, the variation of parameters did not change the conclusions of the study, i.e., the prospective scenario has a better performance compared to the current scenario regarding the studied parameters.

4. Discussion and way forward

4.1 Towards a better cow dung management

In this study, the biogas potential of cow dung in the state of Chhattisgarh was calculated considering the amount of cow dung used as a fertilizer and as cooking fuel today. However, the full biogas potential can also be estimated based on the average cow dung availability per household (2.2; Ministry of Agriculture (2014)) and the percentage of recoverable cow dung (60% of 11 kg cow^{-1} day^{-1}; Ravindranath et al. (2005), Nasery (2011)). It is 14118 kt $year^{-1}$. Therefore, today only 37% of the cow dung produced in rural Chhattisgarh is used as fertilizer or as cooking fuel. The fate of the remaining 63% of cow dung is not known. Part might be left on the grazing field and/or used for other purposes such as religious ceremonies. When considering the total amount of cow dung produced in rural Chhattisgarh, the biogas produced can replace all the cooking fuels used today and a fraction of biogas remains available for other uses. The management of this fraction is important as poor management such as release of the surplus biogas into the atmosphere or flaring can have a large impact on climate change (Bruun et al., 2014). To consistently evaluate the real cow dung availability and thus improve the collection rate, surveys should be conducted to estimate the fate of cow dung flows in rural Chhattisgarh.

4.2 Effect of organic matter returned to the soil

The prospective scenario brings 23% more carbon contributing to the formation of organic carbon in the soil than the current scenario. This has consequences on soil quality and potentially on crop yield. Today, the effect of the carbon content on crop yield is difficult to quantify under tropical conditions such as the ones in Chhattisgarh. However, taking into account the effect of the soil carbon content would most probably strengthen the conclusions of the study. This study also highlights the need for an accurate characterization of soil conditioners in terms of nutrients content but also organic matter and their capacity to enhance soil quality in environmental sustainability studies. More data should be provided by the scientific community on the effect of different treatments of manure (e.g., storage, composting and anaerobic digestion) on the carbon and nutrients content of the final soil conditioner.

4.3 Substitution of nitrogen from synthetic fertilizers

The amount of mineralized nitrogen available for plants one year after application is considered in the study. However, the mineralization of nitrogen from soil conditioner continues during the following years and thus more nitrogen is available for the crops. In the prospective scenario, around 4% more nitrogen is returned to the soil in the form of organic matter compared to the current scenario. Therefore, the substitution of nitrogen from synthetic fertilizers by nitrogen contained in the compost could be slightly higher than as calculated in the study. However, as for carbon, considering the mineralization of nitrogen beyond the first year of application would strengthen the conclusions of the study as more nitrogen would be replaced by the nitrogen contained in the compost.

4.4 The need for a country specific database

One limitation of this study is the use of the ecoinvent database to model the background system. This database contains some specific data for India (e.g., its electricity mix) but key processes in this analysis such as the production of synthetic fertilizers are based on worldwide average data. Considering the real production mix for synthetic fertilizers consumed in India might change the contribution of the production of fertilizers as the inputs required to produce fertilizers might highly vary from one country to another (e.g., the energy mix). Having access to more complete LCA databases for India would strengthen the analysis. Efforts are being done, e.g., the Sustainable Recycling Industries (SRI) project and the work conducted by the LCA India Alliance to build capacity on LCA tools in India, including developing an Indian LCI database. Therefore, more and more country specific data should become available in the future.

4.5 Importance of the local emissions on the impact on human health

This study could compare the local and global impacts on human health due to the emissions of PM_{10} or its precursors based on the PM_{10} effect and damage factors available from the ReCiPe method and a local fate factor that could be calculated based on literature. The results show that the impact of PM_{10} on human health is mainly due to local emissions. Therefore, focusing also on the local level and not only on global levels to evaluate the impact of PM_{10} emissions is key to accurately evaluate the impact of particulate matter on human health. Moreover, the ReCiPe method uses a fate factor calculated based on the atmospheric fate model EUTREND (van Zelm et al., 2008), which is only representative for Europe, and not India. Therefore, there is a need to geographically differentiate the fate factors of substances in different countries or continents.

5. Conclusion

In this chapter, the sustainability of the current supply of cooking energy, fertilizers and rice straw management in rural Chhattisgarh (India) was compared with a prospective scenario in which cooking fuel is replaced by biogas produced from the co-digestion of rice straw and cow dung used today as a fertilizer and cooking fuel. The digestate is used as a fertilizer and soil conditioner. From a methodological point of view, coupling substance flow analysis and life cycle assessment provides additional results than using LCA only. It allows calculating SFA indicators such as the self-sufficiency ratios, which inform on the level of dependence on imported resources of a region, which are expected to be reduced when implementing circular systems. Moreover, the perturbation analysis shows that the parameters that most affect the results of the study are related to the characteristics of the soil conditioners and the emissions of carbon during their storage/processing, which are defined when conducting the SFA. Therefore, there is a need to accurately characterize the composition of soil conditioners at substance level in such analyses. This chapter also shows that LCA alone does not reflect the full impacts from local emissions and a risk assessment approach using specific fate factors at the foreground system level should be followed. This study shows the high potential of anaerobic digestion to increase the environmental sustainability of Chhattisgarh, especially to reduce the impact on human health and increase resource efficiency. Moreover, it shows that while the potential of the technology to reduce the dependency of the communities towards synthetic nitrogen and phosphorus is limited, there is an interesting potential regarding the decrease of farmers' dependency towards synthetic potassium, especially if all the cow dung produced by farmers would be valorized through anaerobic digestion. The barriers to its practical implementation are important to consider to evaluate the full success potential of the technology. These barriers can be related to the social and political context, which is key to consider when evaluating how this scenario could contribute to tackle the challenges India is facing today.

Chapter 5: Improving the evaluation of the resource footprint of household sewage sludge valorisation products in the context of a circular economy: a discussion on allocation approaches

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Draft of

Sfez, S., De Meester, S., & Dewulf, J. (2017). *Improving the evaluation of the resource footprint of household sewage sludge valorisation products in the context of a circular economy: a discussion on allocation approaches.* To be submitted in Resources, Conservation and Recycling.

1. Introduction

This chapter aims to feed the discussion on the fourth recommendation presented in Chapter 2, i.e., the need to revise the way resources consumed by circular systems are accounted for today. This methodological discussion is illustrated by the case of municipal sewage sludge valorisation in the Netherlands.

Until recently, household wastewater treatment was mainly considered as a step to reduce the emission of harmful substances to the environment and recover water as a resource for human activities. However, wastewater produced by households contains large amounts of substances that could have a secondary use in the economy. This is for example the case for nutrients and organic matter which could be valorized as fertilizers and biogas, amongst others. Resource recovery from wastewater streams is more and more seen as one option to help tackling challenges such as the resource efficiency of regions and countries and the low revenues from wastewater treatment today (IWA, 2016; Spinosa et al., 2011). Using sewage sludge as a fertilizer has been considered for many years but is often limited by a content in heavy metals that exceeds the maximum allowed in regulation (Linderholm et al., 2012). To overcome this challenge, technologies to extract the useful compounds of sewage sludge and produce “heavy metal free” fertilizers such as struvite are being developed. The wastewater sector is also developing several other innovative technologies, e.g., to recover metals and ammonia or to produce bio-plastics, biodiesel and biogas from sewage sludge (Puyol et al., 2017). Therefore, the wastewater treatment sector is increasingly positioning itself as a key player in the shift towards a circular economy (IWA, 2016). However, this requires a paradigm shift related to the main goal assigned to wastewater treatment today, i.e., to produce clean water. There should be a shift from the “water cleaning” to the “resource recovery” approach that considers giving a second life to resources consumed by households and discarded in wastewater as a major goal of the wastewater treatment chain (i.e., wastewater treatment and sludge management). This paradigm shift has consequences on the way the sustainability of products obtained from wastewater treatment chains is assessed and some methodological approaches commonly used today to conduct the Life Cycle Assessment of such systems become questionable when it comes to compare products from sewage sludge valorisation with virgin material-based products. If wastewater streams are considered as a resource and not as a waste, it implies that part of the upstream environmental burdens should be allocated to the downstream products to allow a fair comparison with the equivalent products obtained from raw materials. This means that the “zero burden” assumption (Ekvall et al., 2007) usually followed when evaluating the impact of wastewater treatment systems in LCA studies is not valid anymore. Note that a similar paradigm shift can be observed in the sector of solid waste management in which there is a growing discussion on the necessity to allocate part of the impact from the upstream processes (i.e., the processes that produce the products which will turn into waste) to the recycled products (Chen et al., 2010; Oldfield & Holden, 2014). The recent ecoinvent model “allocation at the point of substitution” also follows this approach and allocates the environmental burden of primary production to solid waste streams by considering them as co-products (Weidema et al., 2013). However, this approach is not yet applied to wastewater streams. It has been recently discussed by Pradel et al. (2016), who reviewed the modelling approach followed by 44 LCA studies assessing the environmental sustainability of sewage sludge management. This study shows that the sludge is always considered as a “burden free” flow. The authors stress the fact that such an approach can be followed when comparing different sewage sludge management options but becomes questionable when comparing the environmental sustainability of products obtained from the valorisation of sewage sludge with products originating from virgin raw materials. In these cases, Pradel et al. (2016) argue that part of the environmental burden of the wastewater treatment plant should be allocated to the sewage sludge. However, the products from sludge valorisation do not only rely on the treatment of the wastewater to be produced. They also rely on the production of the products ending in the wastewater streams (i.e., consumer goods). Therefore, the rationale of Pradel et al. (2016) could be extended to the allocation of part of the environmental burden from consumer goods’ production to the products from sludge valorisation. This implies considering the wastewater treatment chain and its upstream processes as a cascade system in which natural resources are first used to produce the consumer goods and then partly used to produce new products from sludge valorisation. The sector of material recycling is already dealing with such a situation and developed several approaches to allocate the impact of virgin raw material processing to the different products of a cascading chain. These approaches also allocate part of the impact of recycling to the products of the chain. They are regularly discussed in literature, especially in the context of the Product Environmental Footprint (PEF) initiated by the European Commission. In this context, Allacker et al. (2017) present different “end-of-life formulas” commonly used in literature. Examples are the 50:50 approach (the material being recycled and the recycled material each bear 50% of the environmental burden of the recycling process) and the “adapted 50:50” approach (the material being recycled and the recycled material each bear 50% of the environmental burden of the virgin raw material processing and recycling process) (Allacker et al., 2017). The recovery of resources from consumer goods discarded by households in the sewage system is similar to the recycling of materials. The used products enter a “recycling” process, which starts with the wastewater treatment plant producing clean water and sewage sludge and ends with the sludge treatment processes to obtain final products. Therefore, the “end-of-life formulas” applied to recycled materials could also be applied to the products used by households and used to produce products from sewage sludge valorisation.

The goal of this chapter is to test different approaches that discard the zero burden assumption usually followed in LCA studies by applying approaches inspired by the so-called “end-of-life” formulas to assess the resource footprint of products from sewage sludge valorisation. The consequence of such an approach on the resource footprint of the consumer goods ending in the sewerage system is also investigated. This methodological approach is tested on a case of the valorisation of sewage sludge from the wastewater treatment plant of the city of Eindhoven (The Netherlands). The products recovered from sewage sludge valorisation are compared with equivalent benchmark products. Moreover, the consequence of producing biogas and struvite from sludge on the difference of resource footprint between the recovered products and the benchmark products is tested by defining an alternative scenario which includes anaerobic digestion and struvite precipitation as supplementary steps.

2. Materials and method

2.1 Description of the case study

The two scenarios used as case studies are presented in Fig. 1 and 2.

2.1.1. Baseline scenario

This chapter takes the wastewater treatment chain of the city of Eindhoven in the Netherlands as a case to test the different methodological approaches. First, consumer goods ending up in the sewage system are produced and consumed by households. They are the food and water that are partly uptaken by the human body and partly turned into feces and urine, and the non-food products such as detergent, cleaning water and soap that are flowing into the sewer after use. The consumption of food products results in the production of food waste and kitchen waste (e.g., vegetable peel). These waste streams are assumed to be incinerated. This assumption is most probably not valid as in Europe today, at least 17% of municipal waste is composted (Eurostat, 2017). It is however considered here, as the focus of the chapter is on the wastewater treatment chain, and not on solid waste management. This aspect is further discussed in the discussion section. The sewage ends up in the wastewater treatment plant of Eindhoven, which has a capacity of 680000 person equivalent (PE), where 1 PE is defined as 150 g COD per day. The wastewater first flows through coarse grids and is then pumped through finer grids before flowing through sand beds. The wastewater is then directed to the primary sedimentation tanks in which the primary sludge is separated. The influent is directed to the activated sludge tanks where nitrogen, phosphorus and additional organic compounds are removed. After the biological treatment, the water is directed to secondary sedimentation tanks. Finally, the effluent flows into the nearby river, the Dommel. Secondary sludge is sent to gravitational sludge thickeners before being mixed with primary sludge and pumped to another facility located in Mierlo via a 7 km pipeline. In Mierlo, the sludge is mixed with the sludge of four other wastewater treatment plants and dewatered in centrifuges. The dry matter of the output sludge is 24.8%, against 2.2% for the influent sludge. The centrate is pumped back to the wastewater treatment plant of Eindhoven and mixed with influent wastewater. The dewatered sludge is then treated. The products obtained from the treatment of sludge are called “recovered products” and the processes from the wastewater treatment plant to the production of the recovered products are called the “resource recovery processes”, including the disposal of waste from the incineration plant (surplus ashes and adsorbents). After dewatering, the sludge is transported by truck to an incineration plant located in Moerdijk (N.V.

Slibverwerking Noord-Brabant (SNB)), 100 km from Mierlo. There, the dewatered sludge is incinerated. Part of the CO_2 produced during incineration is used by a neighboring plant to produce calcium carbonate ($CaCO_3$). All the energy produced during incineration is self-consumed. In 2013, 36359 tons of incineration ashes were produced, 58% of which were used as roadfilling material, 21% to produce a landfill capping material and 3% to produce phosphoric acid for fertilizer production. No detailed information is available on the current process used to produce phosphoric acid from ashes, but the Ecophos process was assumed to be used (Jossa & Remy, 2015). This process produces two other products: calcium chloride ($CaCl_2$) and an iron chloride solution ($FeCl_3$). The remaining fraction of ashes was transported to a salt mine in Germany for a long-term storage (18%) and the waste adsorbents were landfilled after immobilization.

2.1.2. Alternative scenario

The alternative scenario tests the implementation of an additional valorisation step along the sludge management chain. It is based on upcoming improvements that the organization in charge of the management of the Dommel basin (Waterschap De Dommel) is currently implementing and which consists in directing the output sludge of the different wastewater treatment plants they manage to an anaerobic digester before incineration. The sludge is transported by truck from Mierlo to Tilburg (50 km), pre-treated in a thermal hydrolysis process (THP) and digested. The biogas produced (1.4×10^6 Nm^3 $year^{-1}$) is then pumped via pipelines to a company that purifies and compresses it to produce green gas used in city buses. The digestate is dewatered and transported to the incineration plant. The same valorisation pathways for ashes are considered. The centrate from digestate dewatering is used in a struvite precipitation process to produce struvite. The output water is pumped to the nearest wastewater treatment plant before being released into water bodies.

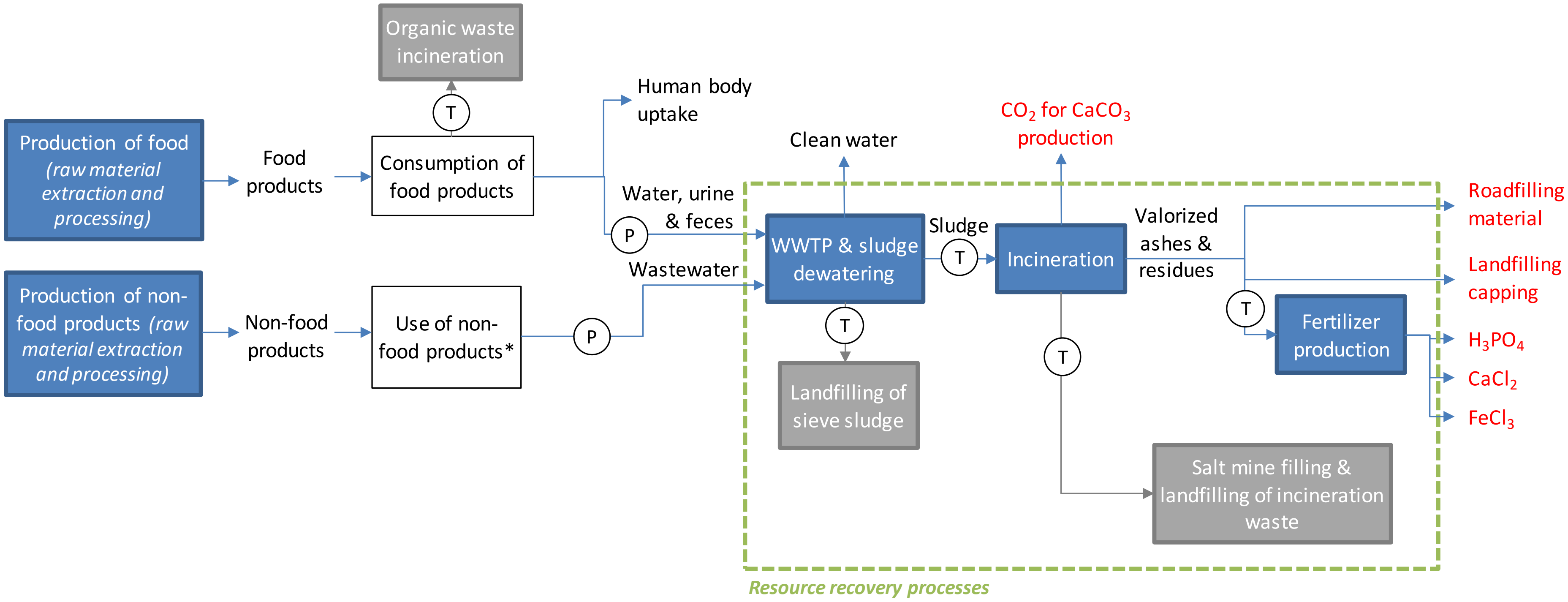


Figure 1: *Baseline scenario (the blue and grey process boxes are included in the system boundaries; the grey boxes represent the disposal processes; the white process boxes are excluded from the system boundaries; the products in red are the products obtained from the processing of sewage sludge; WWTP: Wastewater treatment plant; T: Transport by truck; P: Transport by pipeline; * non-food products ending in the sewerage system).*

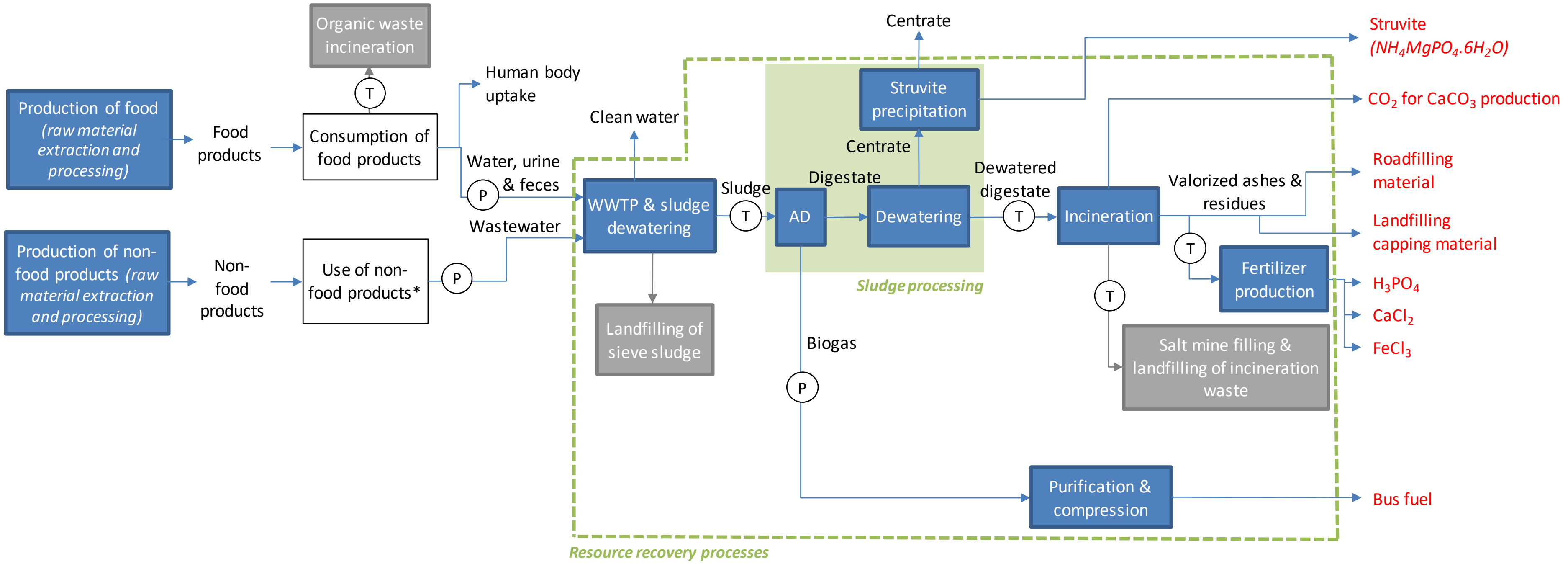


Figure 2: *Alternative scenario (the blue and grey process boxes are included in the system boundaries; the grey boxes represent the disposal processes; the white process boxes are excluded from the system boundaries; the products in red are the products obtained from the processing of sewage sludge; WWTP: Wastewater treatment plant AD: Anaerobic Digestion; T: Transport by truck; P: Transport by pipeline; * non-food products ending in the sewerage system).*

2.1.3. Benchmark scenarios

The baseline and alternative scenarios are each compared with benchmark scenarios producing equivalent products based on benchmark technologies. In the baseline scenario, six products are produced from the treatment of sewage sludge: roadfilling material, landfill capping material, carbon dioxide used for the production of calcium carbonate, phosphoric acid, iron chloride solution and calcium chloride. Ashes used as roadfilling material are assumed to replace gravel (Birgisdóttir et al., 2007). Ashes used as landfill capping material are assumed to replace bentonite clay, a material commonly used to cover landfills (Guyonnet et al., 2009). Carbon dioxde is assumed to replace carbon dioxide produced from the treatment of different industrial gases as described in the ecoinvent database. The phosphoric acid, iron chloride solution and calcium chloride obtained from the Ecophos process are assumed to replace their equivalent product produced from virgin raw materials and to contain no impurities that could decrease their value compared to virgin raw materials-based products. In the alternative scenario, two other products are obtained, i.e., green gas used to feed city buses and replace diesel fuel (1 Nm^3 of biogas is estimated to replace 0.7 kg of diesel fuel (see 2.2.2.3)) and struvite, which replaces synthetic nitrogen and phosphorus fertilizers.

2.2 Life cycle-based resource footprint

2.2.1. Goal and scope

2.2.1.1. Functional unit and system boundaries

The functional unit of the studied scenarios is defined as a basket of products. For the comparison of the baseline scenario with its benchmark scenario, the basket of product presented in Table 1 is chosen. It is based on the products recovered from sewage sludge produced by the households of the city of Eindhoven during one year. Clean water produced by the wastewater treatment plant is not included in the basket of products because it is released in the Dommel river and thus not used in a downstream industrial process.

Table 1: *Basket of products chosen to compare the resource footprint of the current and baseline scenarios with their benchmark scenarios.*

Products	Current scenario	Alternative scenario
Roadfilling material	$2.1x10^6$ kg year^{-1}	$2.1x10^6$ kg year^{-1}
Landfill capping material	$7.3x10^5$ kg year^{-1}	$7.3x10^5$ kg year^{-1}
Phosphoric acid (H_3PO_4)	$1.2x10^5$ kg year^{-1}	$6.6x10^4$ kg year^{-1}
Calcium chloride ($CaCl_2$)	$3.0x10^5$ kg year^{-1}	$1.7x10^5$ kg year^{-1}
Iron chloride solution 40% ($FeCl_3$)	$1.5x10^4$ kg year^{-1}	$8.5x10^3$ kg year^{-1}
Carbon dioxide for $CaCO_3$ production	$2.5x10^6$ kg year^{-1}	$2.5x10^6$ kg year^{-1}
Kilometres driven by city buses		$2.7x10^6$ km year^{-1}
Phosphorus fertilizer, as P_2O_5		$1.2x10^6$ kg year^{-1}
Nitrogen fertilizer, as N		$2.4x10^5$ kg year^{-1}

Note that the amount of roadfilling material, landfill capping material, phosphoric acid and calcium chloride are the same in both basket of products because it was assumed that anaerobic digestion of the sludge does not affect the amount and composition of the ashes contained in the sludge. Less CO_2 is produced during the incineration of the sludge after the implementation of anaerobic digestion as the carbon content of the sludge is reduced due to the production of biogas. However, as the amount of CO_2 delivered to produce $CaCO_3$ would remain constant over time to allow a continuous supply to the $CaCO_3$ producer, it is assumed to only have an effect on the amount of CO_2 released in the atmosphere.

To evaluate the impact of the tested methodological approach on the resource footprint of the consumer goods, another functional unit is defined: the basket of consumer goods, which represents the amount of food and non-food products consumed by households and which end up in the sewerage system.

The processes included in the system boundaries are presented in Fig. 1 and Fig. 2. The packaging of consumer goods is not considered in the analysis. The impact from food consumption itself (e.g., energy for cooking) is considered negligible as Notarnicola et al. (2017) showed that is represents less than 5% of the resource footprint of food consumption. For non-food products (e.g., cleaning products), only the impacts from the ingredients and their transport to the processing plant were accounted for. This is due to a lack of data on the processing itself but also because the contribution of the processing step is negligible compared to the production and the transport of the ingredients (Golsteijn et al., 2015).

2.2.1.2. Allocation between co-products

As it can be seen in Fig. 1 and 2, several processes along the chain produce more than one product. Therefore, before performing the analysis, the system should be partitioned to allow evaluating the resource footprint of the products of interest only (i.e., the basket of products defined above). The three processes that produce several products not included in the basket of products are:

- The consumption of food products: it produces two products, i.e., the proper function of the human body through nutritional uptake of a fraction of ingested food, and the feces and urine (brown water). Note that this process also produces organic waste (e.g., kitchen and food waste) that are not considered as useful products, as they are assumed to be incinerated with no further valorisation. As mentioned above, this assumption is probably not valid but considered here to simplify the studied system, as the aim of this chapter is to test the application of a new approach;
- The wastewater treatment plant: it produces two products, i.e., the clean water and the sewage sludge.

- The sludge processing (in green in Figure 2): it produces four products, i.e., the biogas, the dewatered digestate, the struvite and the centrate.

For each of these processes, allocation factors need to be defined. As mentioned in Chapter 1, these allocation factors can be defined based on different properties of the products (e.g., mass, energy content, exergy value, monetary value etc). Here, an exergy-based allocation is chosen for each of these processes to allow consistency between processes, but also with the exergy-based method chosen to account for resources (see 2.2.3).

Allocation between nutritional uptake and feces/urine - To fulfill the body's vital functions, our organism assimilates a fraction of the food we consume and discards the remaining fraction through feces and urine. Mady and Oliveira Junior (2013) showed that the difference between exergy and energy metabolisms in the human body is lower than 5%. Therefore, the ratio of the energy contained in feces and urine over the energy intake is used as a proxy to estimate the allocation factor. Based on the daily energy requirement per age group provided by the British Nutrition Foundation (BNF, 2017) and the structure of the Dutch population per age group (CBS, 2017), the average energy requirement in the Netherlands can be estimated at 2114 kcal capita^{-1} day^{-1}. The daily energy content of feces is calculated as the energy content of one kilogram of feces (2.7×10^{7} J kg^{-1} dry feces; water content of 72.6%; Van den Neucker et al. (2002)) multiplied by the amount of feces produced per day (on average 175 g day^{-1}; Encyclopedia Britannica (2017)). The energy content of urine is estimated to 3.7×10^{5} J capita^{-1} day^{-1} based on Jumpertz et al. (2011). The energy content of feces and urine excreted daily is thus estimated to be 1.7×10^{6} J capita^{-1} day^{-1}. Therefore, 19% of the intake energy ends up in the feces and urine and this value is taken as allocation factor.

Allocation between clean water and sewage sludge - To calculate this allocation factor, the exergy value of the sewage sludge and the clean water need to be calculated. They are both calculated based on a mass balance and the COD value and water content of the input wastewater and the clean water, as reported in the environmental performance report of Waterschap De Dommel (Blom, 2013). 32% of the exergy of the wastewater ends up in the sewage sludge. This value is chosen as an allocation factor.

Allocation between the centrate and the struvite, dewatered digestate and biogas – 55.6% of the exergy of the input sludge ends in the biogas, 42.8% in the dewatered digestate and 0.9% in the struvite. Therefore, 99.3% of the exergy of the input sludge ends up in the struvite, dewatered digestate and biogas.

The allocation factors are represented in Fig. 3 for the baseline and alternative scenarios.

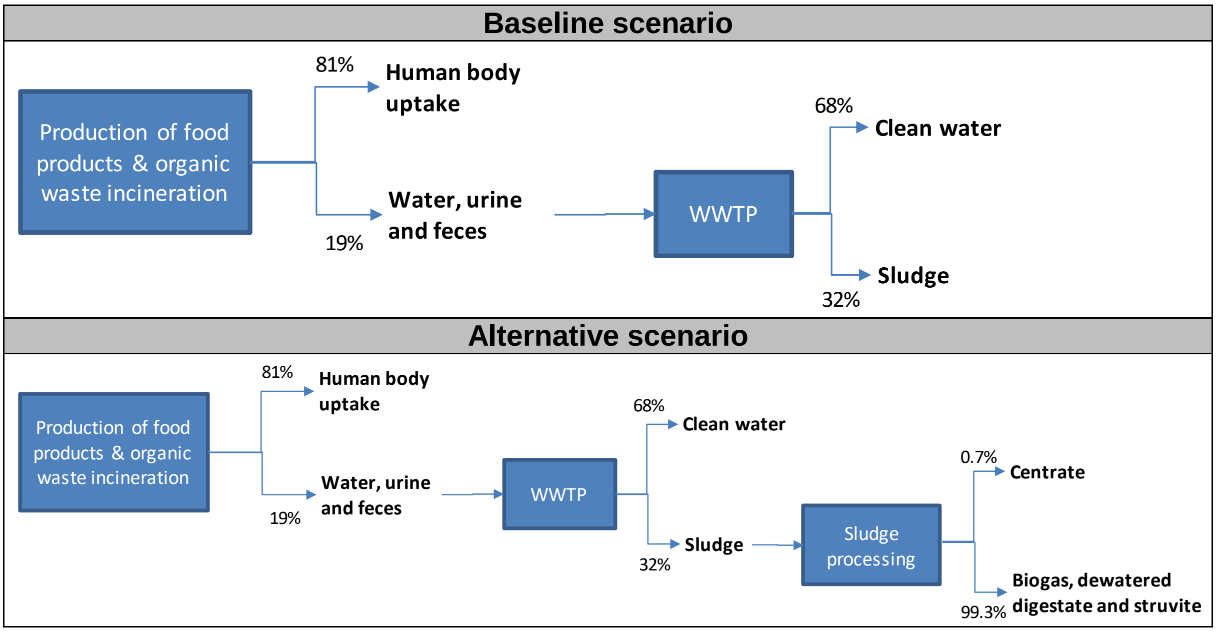


Figure 3: Allocation factors calculated for each of the processes producing more than one product

Applying the allocation factors as defined above results in partitioning the process chain in sub-chains that each delivers one single product or basket of products. These sub-chains are represented in Fig. 4 for the baseline scenario.

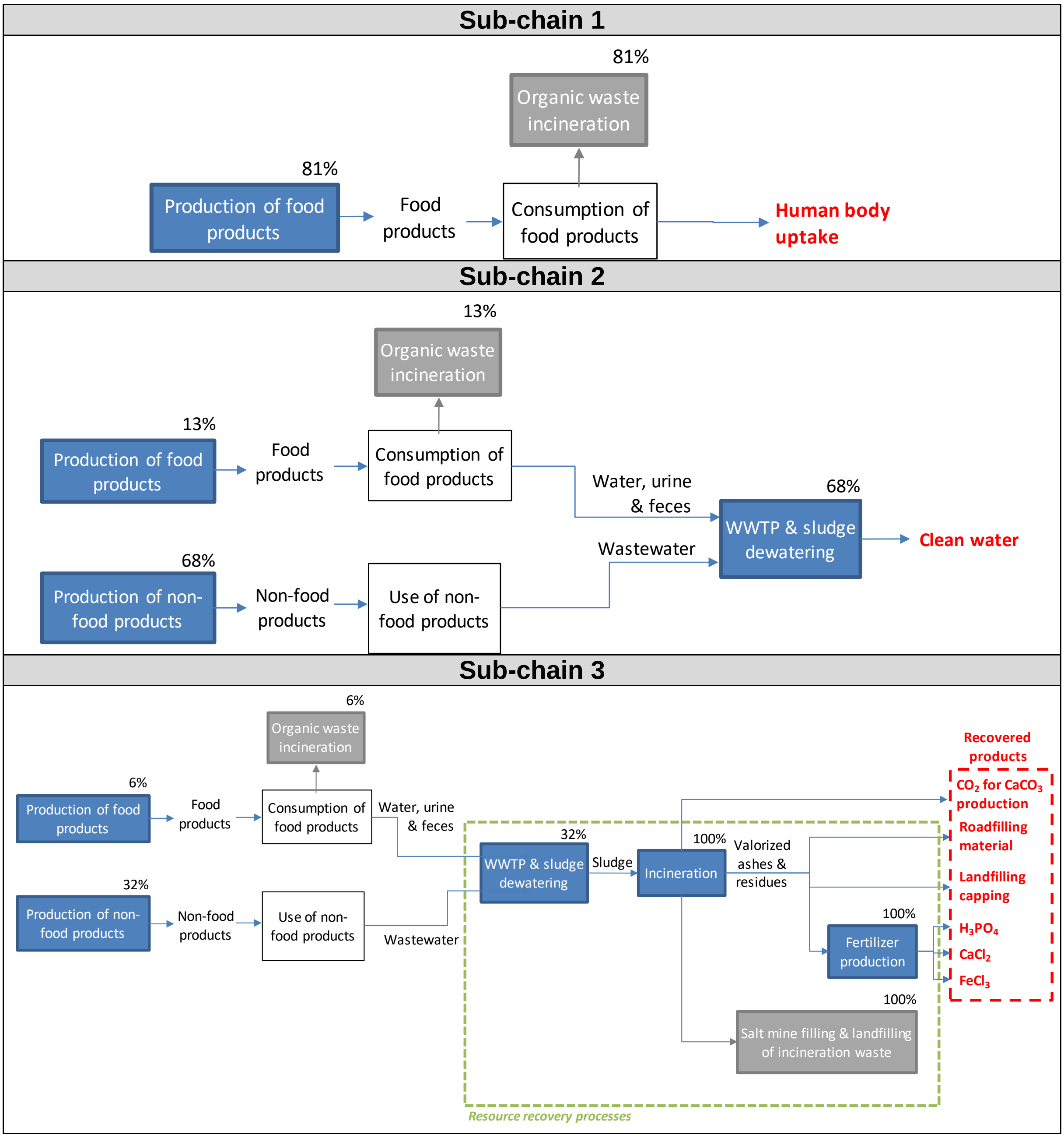


Figure 4: *Partitioning of the studied system (baseline scenario) based on the defined allocation factors. The percentages next to the process boxes represent the fraction of the resource footprint of the process allocated to the product(s) of the sub-chain. For example, the percentage of the resource consumption of the production of food products allocated to the recovered products (6%) is calculated by multiplying the factor allocating the impact of food production to water, urine and feces (19%), with the factor allocating the impact of the wastewater treatment and dewatering plant to the sludge (32%) (represented in Fig. 3).*

2.2.1.3. Application of the "end-of-life" formulas

Once the partitioning of the system has been done, sub-chain 3 is obtained, in which resources are consecutively used to produce consumer goods (food and non-food products) and the basket of recovered products. Then, a similar approach as followed in the sector of material recycling and which allocates the resource footprint of the processes along the chain to the different products of the chain (in this case the consumer goods and the basket of products) can be applied. Allacker et al. (2017) present 11 end-of-life formulas that can be applied to products used consecutively in a cascade system. Some simply differ by the fact that they account for avoided virgin production by the recycled product. In our case, recovered products (i.e., the recycled products) are compared with benchmark products (i.e., from the processing of virgin material). Therefore, these methods are discarded from the analysis. Moreover, Allacker et al. (2017) discuss four methods based on the 100:100 principle, meaning that 100% of the impact of recycling is allocated to the recycled products and 100% is allocated to the product producing the recycled material, which results in a double counting of the impact when considering the overall system. To keep a consistent system, this end-of-life formula was not considered in the analysis either. The five remaining approaches are presented in Table 3, which presents a description of the allocation of the burden of the different processes along the chain between the first intended product (i.e., producing the secondary material at its end-of-life) and the downstream products produced from secondary material.

The 0:100, 50:50, "50:50 adapted" and "degressive linearly" approaches imply to know if the recovered products are disposed or recycled after use. If recycled, the burden from this recycling step should be fully or partly allocated to the recovered products. In the case of the example taken in this chapter, it implies for example to know if the roadfilling material is disposed when the lifetime of the road ends, or if it is recycled or reused for another application.

Table 3: *Description of the five approaches tested in the analysis.*

Name	**Description**	**Rationale** ***(when considering a cascade with 2 products)***	**Part of the burden from consumer goods production is allocated to recovered products**	**Part of the burden from resource recovery processes is allocated to consumer goods**
0:100	Full allocation of the recycling impact to the intended product and no burden allocated to downstream products using secondary materials	The recycling process is considered the responsibility of the product that generates the material to be recycled.	No	Yes
100:0	Full allocation of the recycling impact to the product using secondary material, with no burden from recycling operations allocated to the intended product	The recycling process is considered the responsibility of the final recycled material.	No	No
50:50	50% allocation of the recycling impact to the intended product and 50% to the product using the secondary material	The responsibility of the recycling process is equally shared between the two products.	No	Yes
50:50 adapted	Distributes the impacts due to recycling in a 50:50 manner over the different products in the overall product cascade system but also the virgin material and disposal impact	The responsibility of the recycling process is equally shared between the two products. The consumption of virgin material is necessary for the production of the intended product, but also assumed to be necessary for the production of the 2nd product. Similarly for disposal.	Yes	Yes
Degressive linearly	Uses the 50:50 approach for the allocation of the recycling impact. Allocates the impact of the virgin material in a linearly degressive way to all products in the product cascade system, allocating the highest share of impact to the first product. Same approach with disposal, but allocating the highest share of impact to the last product.	The responsibility of the recycling process is equally shared between the two products. Both products are responsible for the extraction of virgin material, but the first material has a larger responsibility than the second material. Similarly, both products are responsible for disposal but the last material has a larger responsibility.	Yes	Yes

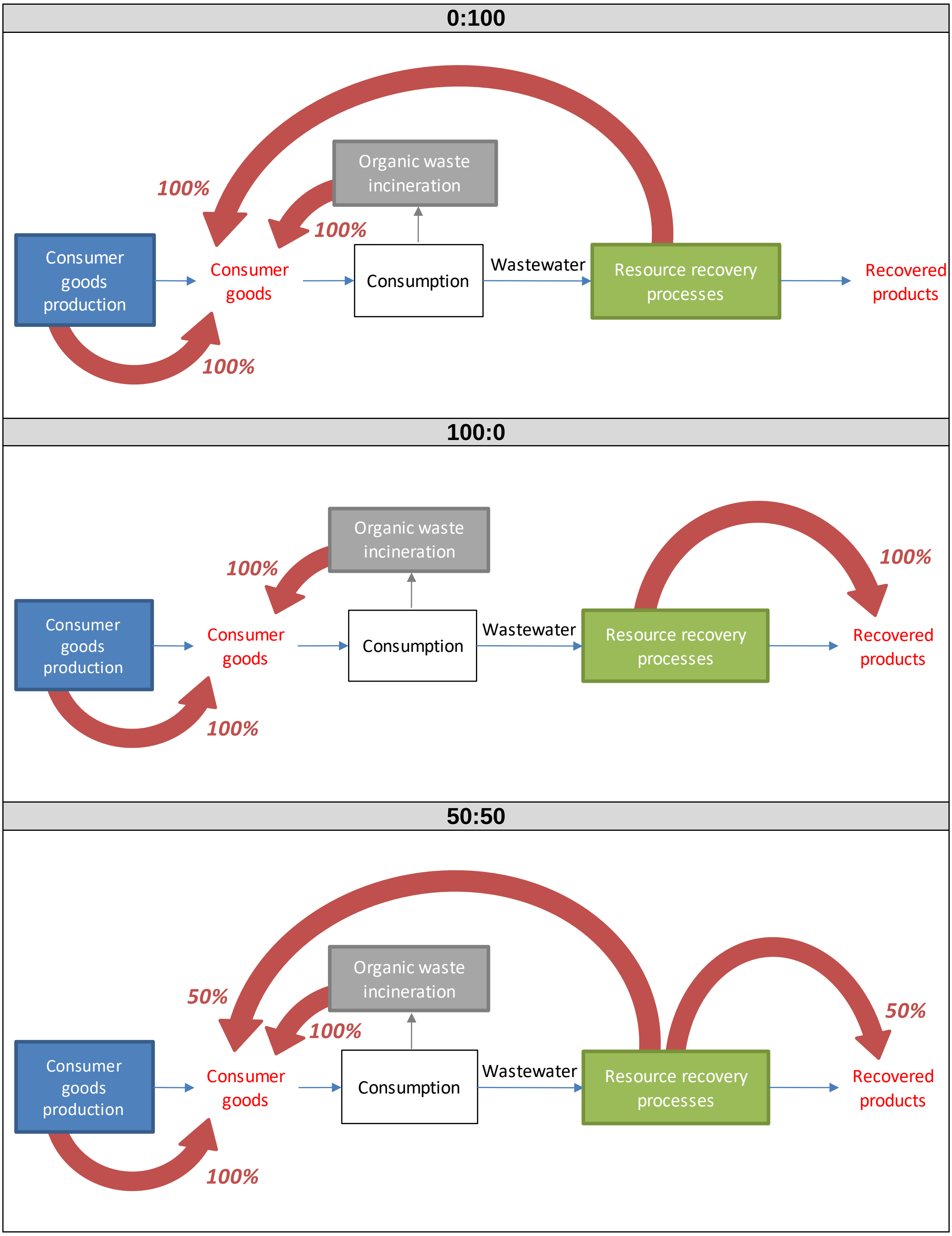
However in this study, the recovered products are compared with benchmark products for which the disposal or recycling steps are the same than the recovered products. Therefore, the impact of the downstream steps that should be allocated to the recovered products are not included in the analysis. This does not have any consequence for the 0:100, 50:50 and “50:50

adapted” approaches but does have a consequence for the “degressive linearly” approach for which the percentage of impact allocated along the chain depends on the number of times a product is recycled before final disposal. In the case study presented here, this information cannot be known because of a lack of tracking of materials during their whole lifetime. Therefore, the approach “degressive linearly” was slightly modified compared to the one described in Allacker et al. (2017). Instead of being shared between all the products of the chain until final disposal, the responsibility of the extraction and processing of virgin material is shared between the virgin material-based product (in this example, the consumer goods) and the first product from recycling of this material (in this example, recovered products), but in a degressive manner. This allows applying the principle of degressive allocation without having to know how the recycled products are then used for. Allacker et al. (2017) propose to use the following factor to allocate the impact of virgin material to the different products of the chain:

$$f = \frac{2 \times n - 1}{n^2} \tag{1}$$

Where *n* is the number of products along the chain. In the case study presented in this Chapter, two baskets of products are obtained. Therefore, 75% of the burden of virgin material extraction and processing is allocated to the virgin material-based product, and 25% is allocated to the product obtained from the first recycling process. The responsibility of the recycling processes is equally shared between both products (50% for both). The approaches applied to the case study are presented in Fig. 5 for the sub-chain 3, which produces the recovered products.

To calculate the resource footprint of the consumer goods, the approach presented in Fig. 5 should also be applied to the sub-chains 1 and 2 in order to quantify the resource use from the downstream processes that will be allocated to the consumer goods. The resource footprint of the consumer goods in the sub-chains 1, 2 and 3 are then summed up to obtain the total resource footprint of the consumer goods.


0:100
Organic waste incineration
100%
100%
Consumer goods production
Consumer goods
Consumption
Wastewater
Resource recovery processes
Recovered products
100%
100:0
Organic waste incineration
100%
100%
Consumer goods production
Consumer goods
Consumption
Wastewater
Resource recovery processes
Recovered products
100%
50:50
Organic waste incineration
50%
100%
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Resource recovery processes
Recovered products
100%

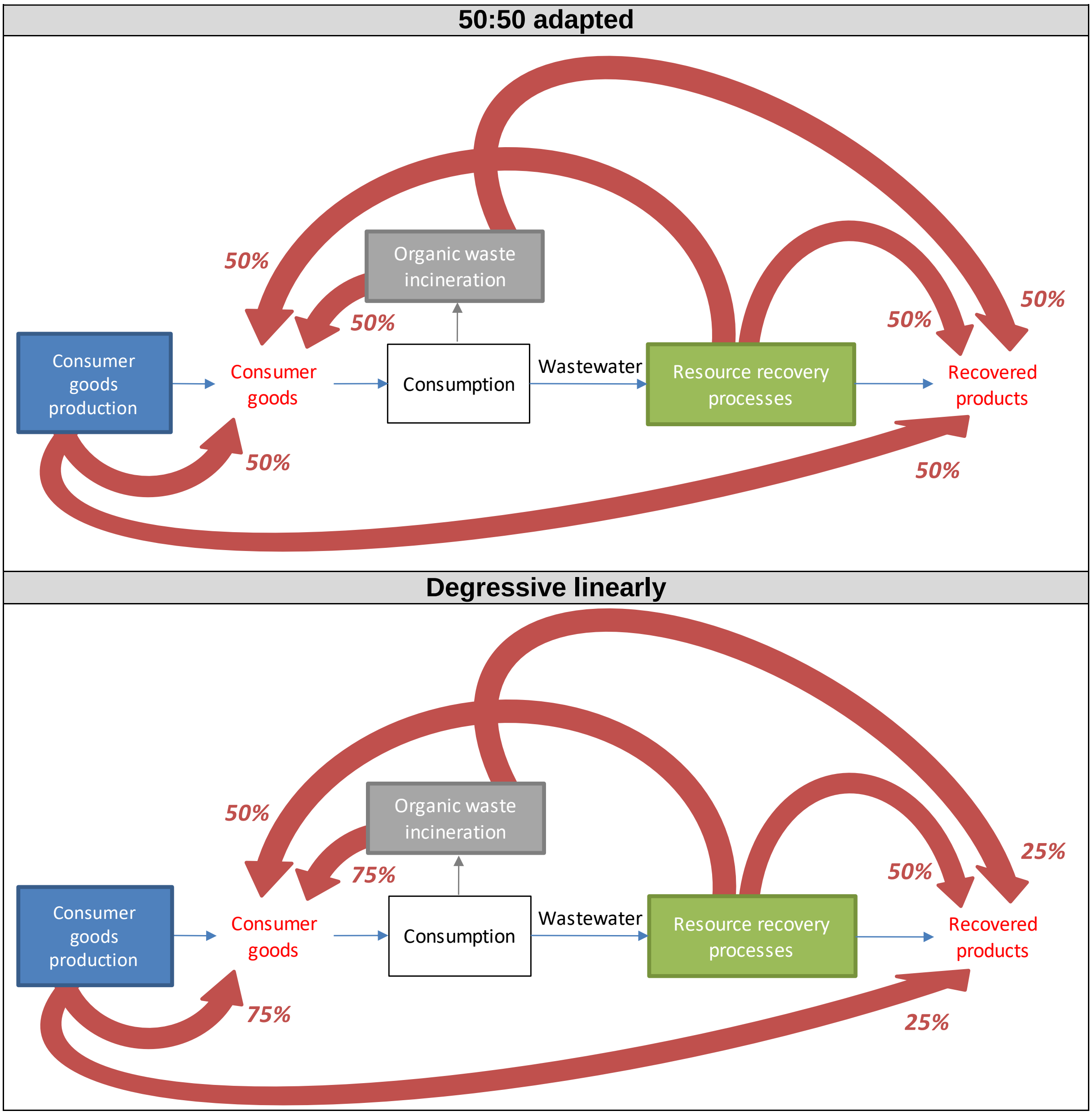


Figure 5: *Visualization of the allocation procedure for each allocation approach. One red arrow represents the allocation of the environmental burden of one process to one specific product (in red: consumer goods or recovered products). Percentages represent the share of the environmental burdens allocated to the products.*

2.2.2. Data inventory

2.2.2.1. Consumer goods production

To estimate the resource footprint of the raw materials extraction and processing necessary to produce the consumer goods, the consumption patterns of food and non-food products released in the wastewater stream had to be estimated.

The food consumption habits of the population of Eindhoven was estimated based on the Dutch National Consumption Survey (RIVM, 2011), which provides the daily food consumption of one person per category of products (e.g., vegetables, legumes, meat and meat products etc) and per type of product in each category (e.g., tomato products, onions, cabbage etc, in the category “vegetables”). The survey gathers consumption patterns for more than 1700 food products. Therefore, for each category, only the products representing 60% (or more) of the consumption habits for this category were considered in the analysis. In total, 47 products were selected and assumed to represent the complete diet of the Dutch population. The resource footprint of each product was calculated using the life cycle databases ecoinvent version 3.3 (Frischknecht & Rebitzer, 2005), the Agri-footprint database (version 3.0; (Blonk Consultants, 2017) and the LCA Food database (2.-0 LCA Consultants, 2003). The amount of food waste was estimated 10% of the consumed food (LNV, 2010) and the amount of kitchen waste (peels, shells etc.) was estimated based on literature data (e.g., Mahmood et al. (1998) for potato peel) and on the author’s estimation.

The non-food consumption patterns were estimated based on several sources such as the RIVM factsheet for cleaning products and cosmetics (RIVM, 2002, 2006) and the results from the “PAN-European consumer survey on sustainability and washing habits” (AISE, 2014). The composition of the body and house care products was based on the RIVM reports and Golsteijn et al. (2015). The background processes were modelled based on the ecoinvent database (v3.3). The transport of ingredients with renewable origin were assumed to be transported by boat (8000 km) and the ingredients of non-renewable origin to be transport by truck (2000 km) (Golsteijn et al., 2015).

2.2.2.2. Resource recovery processes

Data on the materials, water and energy consumption of the wastewater treatment plant in Eindhoven and the dewatering plant in Mierlo were taken from the environmental performance report from Waterschap De Dommel (Blom, 2013). The wastewater treatment plant treats both household and industry water. The inventory from the plant was allocated to the household stream based on the COD of each stream (74% to the household wastewater).

Data for digestate dewatering and the struvite precipitation process was taken from literature (see Table 4).

Table 4: *Literature data used for the modelling of the digestate dewatering and struvite precipitation processes*

	Data	Value	Reference
Digestate dewatering	Electricity consumption – belt filter press	1.5 to 2 kWh m^{-3} digestate	Drosg et al. (2015)
	Total solids in solid fraction	20 to 30%	Drosg et al. (2015)
Struvite precipitation	Amount of P transferred in the liquid phase	45%[1]	Drosg et al. (2015)
	Amount of MgO added for struvite precipitation	1.2 mol mol^{-1} P	Ishii and Boyer (2015)
	Electricity consumption – struvite precipitation	0.2 kWh m^{-3} liquor	Jossa and Remy (2015)
	Heat consumption – struvite drying	0.9 kWh kg^{-1} P_{out}	Jossa and Remy (2015)
	Amount of P mobilized in struvite	80% of input P	Lowest recovery rate reported for struvite production in Desmidt et al. (2015)
	Substitution rate struvite/synthetic fertilizers	1	Amann et al. (2018); Ishii and Boyer (2015)

[1]Drosg et al. (2015) reports a range of 35 to 45% of P transferred in the liquid phase during digestate dewatering. As sludge is biologically treated in activated sludge tanks in the WWTP, a high amount of unbonded P is expected in the treated sludge and thus in the digestate. Therefore, the highest value of the range proposed by Drosg et al. (2015) is used.

Data on materials, water and energy consumption of the incineration plant, as well as on the destination of bottom ashes for disposal or valorisation were extracted from the environmental annual report of N.V. Slibverwerking Noord-Brabant (Sijstermans & van der Stee, 2013). The consumed chemicals were not included in the assessment. The incineration plant also incinerates sludge from other wastewater treatment plants. The resource consumption of the plant was allocated to the sludge from the Eindhoven plant based on the dry solids content of the input sludge. Therefore, 16% of the resource use from the incineration plant was allocated to the sludge from Eindhoven.

The ashes valorized as landfill capping and roadfilling material are used as such, without any other processing step. The ashes valorized as a fertilizer need further processing and enter the Ecophos process. Data on materials and energy consumption as well as the yields of the three by-products obtained from this process are based on Jossa and Remy (2015).

2.2.2.3. Background processes

The background processes (e.g., the production of electricity from the grid and the production of the benchmark products) are modelled based on the ecoinvent database version 3.1 (Frischknecht & Rebitzer, 2005). To be consistent with the co-products allocation approach followed in the foreground system, the ecoinvent modelling approach "allocation at the point of substitution" is used.

Regarding the equivalence of the recovered products with the benchmark products, ashes used as roadfilling material are assumed to replace gravel and ashes used as landfill capping material to replace bentonite clay, both with a 1:1 ratio (Birgisdóttir et al., 2007). The same 1:1 ratio is used to estimate the equivalence between the recovered phosphoric acid, iron chloride solution and calcium chloride and the virgin material-based products, as no impurities are assumed to be present in the obtained products. To estimate the amount of diesel fuel replaced by biogas in city buses, the following data is used. One city bus drives 1.9 km per Nm^3 of green gas (Ahmadi Moghaddam et al., 2015) so the use of biogas would allow driving 2.7×10^6 km per year. One city bus drives around 2.4 km per litre of diesel (Ally & Pryor, 2007; Nylund et al., 2007) so 1 Nm^3 of biogas is estimated to replace 0.7 kg of diesel fuel. Based on Amann et al. (2018) and Ishii and Boyer (2015), it was assumed that 1 kg of phosphorus contained in the struvite would replace 1 kg of phosphorus in synthetic fertilizer. The same approach is followed for nitrogen.

2.2.3. Impact assessment

The resource accounting method that considers the widest types of resources as presented in Chapter 2, i.e., the CEENE method, is chosen to conduct the impact assessment (Dewulf et al., 2007).

3. Results

3.1 Resource footprint of the recovered products

Fig. 6 shows the resource footprint of the recovered products following the zero burden assumption and the five allocation approaches for the baseline and alternative scenarios. Two approaches result in a lower resource footprint than with the zero burden assumption, i.e., the 0:100 approach, which does not allocate any impact from the resource recovery processes to the recovered products, and the 50:50 approach, which allocates 50% of the impact from the resource recovery processes to the recovered products. For the baseline scenario, the resource footprint with the zero burden assumption is 27, 79 and 63% lower than with the 100:0, “50:50 adapted” and “degressive linearly” approaches, respectively. This difference slightly decreases when implementing the alternative scenario: it becomes 22, 73 and 53% lower than with the 100:0, “50:50 adapted” and “degressive linearly” approaches, respectively. With the 0:100, 100:0 and 50:50 approaches, no impact from consumer goods production is allocated to the recovered products. For the baseline scenario, the process mainly contributing to the resource footprint when following the 100:0 and 50:50 approaches is incineration, which represents 49% of the footprint. The second contributor is the wastewater treatment plant (27%), followed by the Ecophos process (27%).

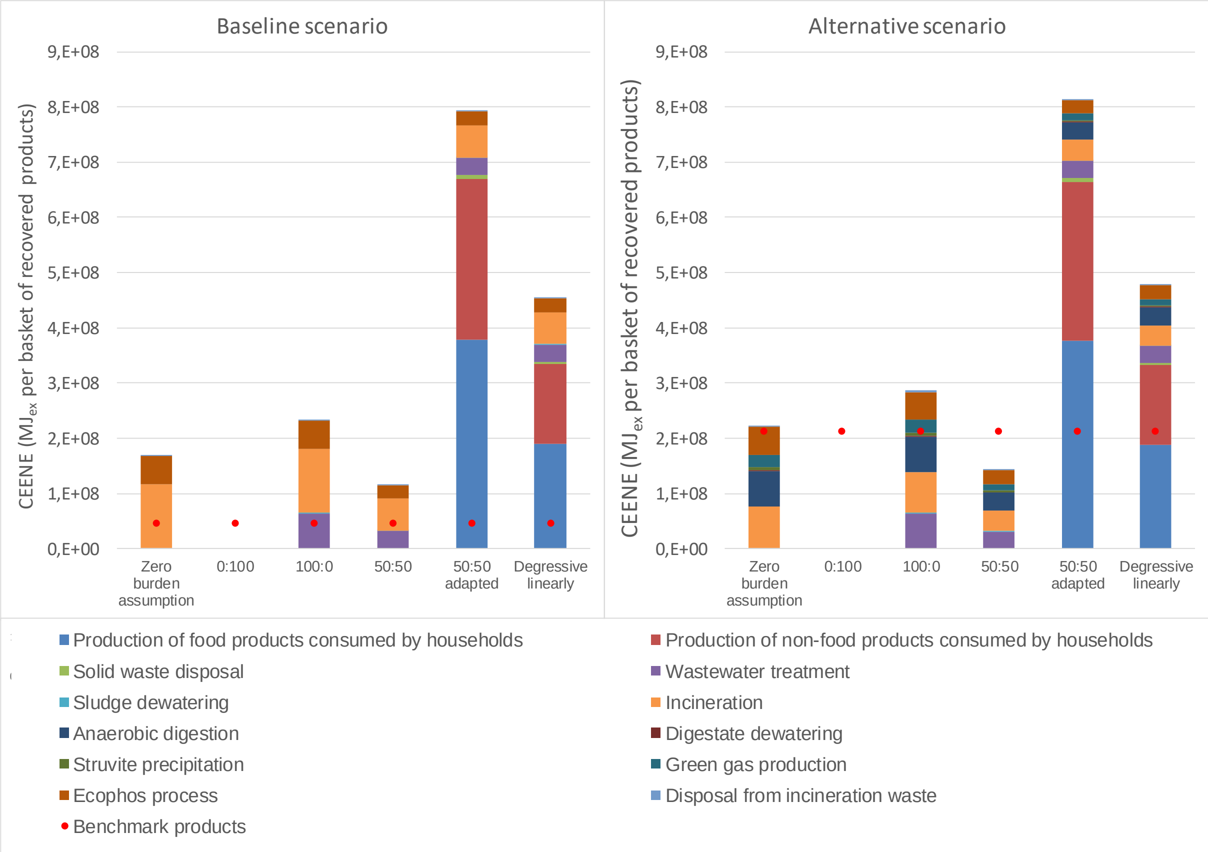


Figure 6: *Comparison of the resource footprint of the recovered products (bars) and the benchmark products (red dots) for the baseline and alternative scenarios, following the zero burden assumption and the five allocation approaches.*

In the alternative scenario, the contribution pattern changes, as the second contributor becomes anaerobic digestion (23% of the footprint with the 100:0 and 50:50 approaches), wastewater treatment (22%), and the Ecophos process (18%). Including a digestion step between sludge dewatering and incineration reduces the amount of sludge sent to incineration and thus decreases the contribution of incineration. For example with the 100:0 approach, the impact from incineration decreases from 1.2×10^{8} to 7.4×10^{7} MJ_{ex} per basket of recovered products.

With the “50:50 adapted” and “degressive linearly” approaches, part of the impact from the production of consumer goods is allocated to the recovered products. The production of consumer goods becomes the first contributor to the footprint, with 84 and 74% of the impact for the baseline scenario for the “50:50 adapted” and “degressive linearly” approaches, respectively. The share of the impact from food products is slightly higher than the share from non-food products (e.g., 46 and 37% of the footprint for the baseline scenario following the “50:50 adapted” approach).

The resource footprint of the benchmark products with the 0:100 approach is higher than the recovered products for both scenarios. This is due to the fact that no impact is allocated to the recovered products. The resource footprint of the benchmark products with the 50:50 approach is lower than the recovered products for the baseline scenario, but becomes higher for the alternative scenario. This is due to the large resource consumption avoided by replacing synthetic fertilizers by struvite. In the study, a 1:1 substitution ratio of synthetic nitrogen and phosphorus by nitrogen and phosphorus contained in struvite was used. Note that when applying a 1:2 ratio, the resource footprint of the benchmark products with the 50:50 approach remains lower than the benchmark products. For all the other approaches, the resource footprint of the recovered products is higher than those of the benchmark products. For example, the resource footprint of the recovered products with the zero burden assumption is 73% higher than the benchmark products (1.7×10^8 MJ_{ex} and 4.6×10^7 MJ_{ex} for the recovered and benchmark products, respectively). This is line with the results from Linderholm et al. (2012) who compared the resource footprint of mineral P fertilizer and P fertilizer obtained from the valorisation of the bottom ashes from wastewater sludge incineration. The authors found that the resource footprint of mineral P is around 85% lower than the resource footprint of the P fertilizer obtained from bottom ashes. In the case presented in this chapter, this difference highly decreases when implementing the alternative scenario (e.g., the resource footprint of the recovered products with the zero burden assumption becomes only 5% higher than the benchmark products). This is due to the large resource footprint of synthetic fertilizers replaced by struvite (56% of the avoided resource footprint) and bus diesel replaced by biogas (25% of the avoided resource footprint). Moreover, the valorisation of the sludge as biogas reduces the amount of sludge that needs to be incinerated, and therefore reduces the amount of resources consumed for incineration.

This case shows that today, for four of the allocation approaches out of the six applied, using products from the valorisation of the ashes of wastewater sludge incineration consumes more resources than using products from raw materials. However, it also shows that including valorisation steps among the resource recovery processes reduces the resource footprint of the recovered products. Other improvement options are still possible. For example, nitrogen is completely lost during incineration, and the inclusion of nitrogen recovery steps such as air stripping of ammonia and membrane-based processes could reduce the resource footprint of the recovered products.

As expected, allocating part of the resource footprint of consumer goods strengthens the conclusions of the comparison and the potential of recovered products to compete with the benchmark products becomes rather limited. However, in the context of a circular economy, considering waste streams as resources is a requirement for a successful implementation of the concept. This also implies that impact assessment approaches account for this change of paradigm and thus discard the zero burden assumption. This is not favourable for the products obtained from resource recovery processes and which resource footprint becomes even larger than the virgin material based products. This is especially because the resource footprint of consumer goods is more than 30 times higher than the resource footprint of the resource recovery processes. It implies that measures to improve the resource footprint of recovered products should also include measures to reduce the contribution of consumer goods.

3.2 Resource footprint of the consumer goods

Fig. 7 shows the resource footprint of the consumer goods with the zero burden assumption and the five allocation approaches.

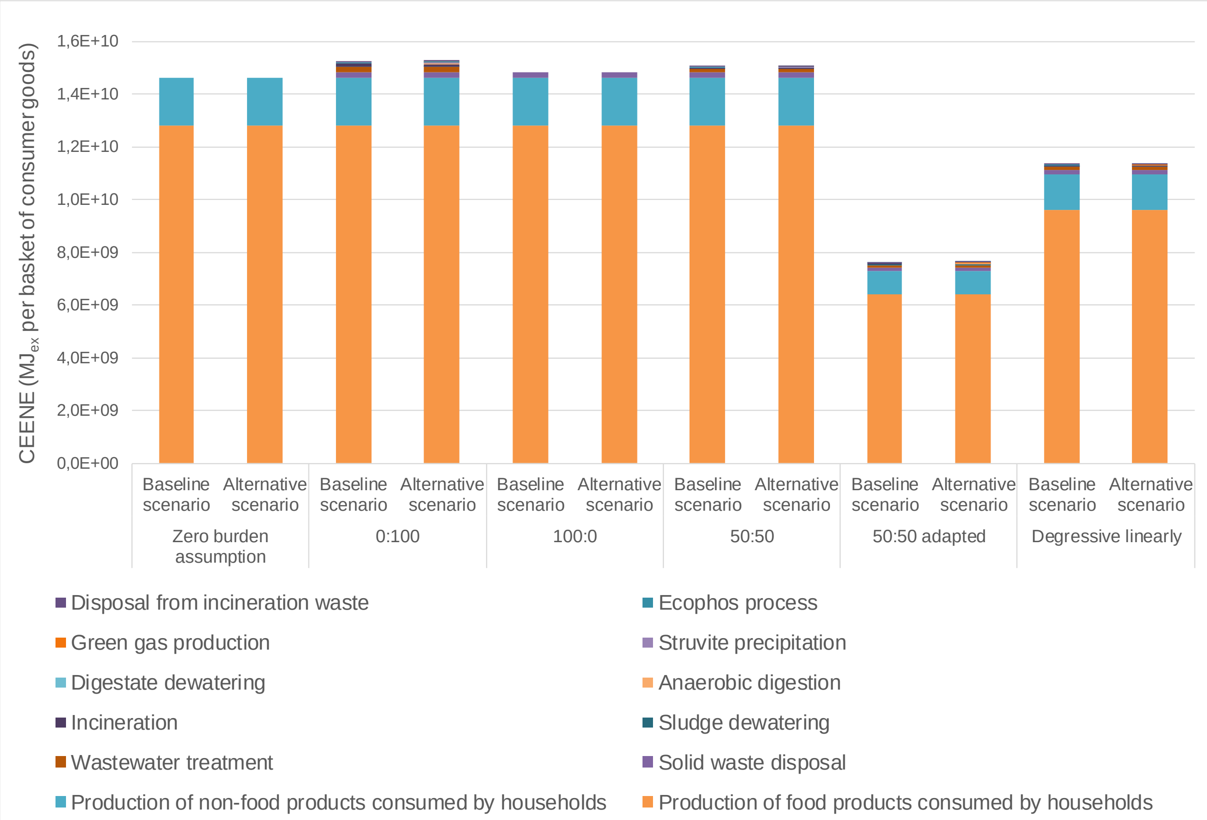


Figure 7: *Resource footprint of the consumer goods with the zero burden assumption and following the five allocation approaches.*

First, the order of magnitude of the resource footprint of the consumer goods is more than ten times higher than the one of the recovered products. This is due to the large resource footprint of the raw material extraction and processing for the production of consumer goods, which represents more than 96% of the resource footprint of the consumer goods. The first contributor of the resource footprint is the production of the food products, which represents 84 to 88% of the footprint. It is followed by non-food products, which represent 12% of the footprint for all approaches. With the zero burden assumption and the 100:0 approaches, no impact from the resource recovery processes is allocated to the consumer goods. However, while no impact from solid waste disposal is allocated to the consumer goods with the zero burden assumption, it is the case with the 100:0 approach. The 0:100 and 50:50 approaches result in a slightly higher footprint as part of the impact from the resource recovery processes is allocated to the consumer goods. However, they only represent less than 3% of the footprint. The 0:100, 100:0 and 50:50 approaches result in a resource footprint which is only 4, 2 and 3% higher than when considering the zero burden assumption for both scenarios. The "50:50 adapted" and "degressive linearly" approaches result in footprints 48 and 23% lower than with the zero burden assumption for the baseline scenario and 49 and 24% lower for the alternative scenario. Therefore, while allocating part of the impact of the resource recovery processes to the consumer goods barely changes the resource footprint of these, allocating part of the impact of the consumer goods production to the recovered products highly contributes to decrease the footprint of the consumer goods.

4. Discussion

Choosing one approach over another can appear arbitrary. However, the compliance of the approaches with the concepts of industrial ecology can still be discussed for the case study presented here. Industrial ecology is based on the concept of waste-as-a-resource. It considers products, which are intended to be produced, and secondary resources, which are unintended but can contribute to obtain new products. These new products depend on the intended products to be produced. On the other hand, the unintended secondary resources should be safely managed as a consequence of the production of the intended products. The concept of industrial ecology highlights a "hierarchy of intent" (intended products and secondary resources), as well as a dependence of all products from the system to one another. First, some allocation approaches do not allocate any impact of virgin raw materials extraction and processing to the recovered products for products used consecutively (the zero burden, 0:100, 100:0 and 50:50 approaches). This does not reflect the dependence of these products to the intended products as the recovered products could not be produced without the extraction and processing steps. On the other hand, the 100:0 approach fully allocates the impact of these processes to the recovered products while these processes are a consequence of the production and consumption of consumer goods. Therefore, based on the concept of the producer's responsibility, which is often used as a tool to promote the implementation of the industrial ecology principles, part of the burden from recovery processes should be allocated to the consumer goods. The "50:50 adapted" approach allocates equally the impact from the raw materials extraction and processing to the consumer goods and the recovered products, while the original goal of these processes is to produce consumer goods. This approach thus considers the dependence of products but does not consider the "hierarchy of intent". Compared to the other approaches, the "degressive linearly" approach appears to consider both the dependence of the products to one another and the "hierarchy of intent" and thus to translate best the concepts of industrial ecology in the LCI modelling.

In this study, the "degressive linearly" approach considers an allocation of the environmental burdens based on a 75:25 ratio based on Allacker et al. (2017). Other approaches could be investigated to define the values used for the allocation of the impact along the chain. One possibility could be to consider that the ratio of the gate fee at the entrance of the recovery processes (here the wastewater treatment plant) over the cost to run the recovery processes represents the share of the impact from these processes that can be allocated to the waste treatment function, and thus allocated to the consumer goods. The remaining fraction can be fully allocated to the recovered products. A similar approach can be applied to allocate the impact of consumer goods production between the consumer goods and the recovered products.

The results presented in this chapter are obtained using the resource-based accounting method CEENE. However, other conclusions might be drawn when using other resource-based methods that consider issues related to resource availability or scarcity such as the ADP (van Oers et al., 2002) and the Ecological scarcity (Frischknecht & Büsser Knöpfel, 2013) methods. Using such methods could potentially change the difference of resource footprint between the recovered and benchmark products. Similarly, other results might be obtained when conducting an emission-based impact assessment in which the emissions of the different processes along the chain would be allocated to the different products following the same allocation approaches. For example, if human toxicity is analysed and the allocation approach "degressive linearly" is applied, the environmental impact from releasing heavy metals or other chemicals in the Dommel river after the treatment of the wastewater should be allocated to the recovered products and the consumer goods. A similar approach should be followed for other emission-based impact categories such as Climate change.

Another point of attention when applying the proposed approach is the consistency of the modelling approaches followed in the foreground and background systems. Indeed, several allocation approaches were tested in the foreground system but the allocation approach used to model the background system is "fixed", as it is based on a database. The ecoinvent modelling approach "allocation at the point of substitution" was used to model the background system. Similarly to the approach followed to allocate the burden of the processes to the different co-products in the foreground system (e.g., clean water and sludge), the approach "allocation at the point of substitution" should in principle consider all waste streams as co-products of the process they are produced from. However, some discrepancies and unclarity can be found with this approach. While the approach is applied to municipal solid waste, it is not clear in what extend it is also applied to other waste streams such as sewage sludge. When looking at the modelling of the production of biogas from sewage sludge ("treatment of sewage sludge by anaerobic digestion", ecoinvent v3.1) with the "allocation at the point of substitution" approach, it can be seen that the process does not consider sewage sludge or its precursors as an input and thus applies the "zero burden" assumption. This aspect should be kept in mind during results interpretation. Similarly, the end-of-life formulas applied in the foreground system are not applied in the background system modelled with the ecoinvent database. The application of the end-of-life formulas in the background system would make the study more consistent and probably change the results of the analysis. However, the implementation of such an approach in LCI databases would require a deep rethinking of how products and processes are linked to each other in those databases.

In the two studied scenarios, solid waste from food consumption is assumed to be incinerated without valorisation of the produced energy or heat. This assumption was made to simplify the scenarios, as the focus was on the wastewater treatment chain and not on solid waste management. This assumption is most probably unrealistic, as in Europe today, at least 17% of municipal waste is composted (Eurostat, 2017). If solid waste valorisation steps are considered, a similar approach applying the end-of-life formulas as for the wastewater treatment chain should be applied to the solid waste treatment processes as well. It highlights the complexity of the practical implementation of such an approach, especially for the calculation of the resource footprint of the consumer goods. Moreover, the approach presented in this chapter can only be applied when comparing sewage sludge valorisation products with benchmark products, or to account for the credits of avoided production. As discussed in section 2.2.1.3, a study that would not compare the recovered products with benchmark products and would not account for the credits from avoided production would require knowing the fate of these products, i.e., if they are further recycled after use or disposed. However, as highlighted in Allacker et al. (2017), the feasibility to access such information is very low as producers most of the time lose track of their products after use.

5. Conclusion

The goal of this chapter was to evaluate the consequence of discarding the zero burden assumption on the resource footprint of the products obtained from the valorisation of household wastewater sludge, as well as on the resource footprint of the consumer goods that end up in the sewage system. First, the process chain had to be partitioned based on allocation factors. In this study, exergy-based allocation factors were chosen. Other physical properties used as a basis to define these allocation factors could be tested to evaluate their impact on the results (e.g., COD-based for the allocation between the sludge and the clean water; mass-based for allocation between the CO_2 and the ashes and residues from incineration). Secondly, five approaches presented in Allacker et al. (2017) were tested. The results show that discarding the zero burden assumption and applying the different allocation approaches has only a large impact on the resource footprint of the consumer goods when following the “50:50 adapted” and “degressive linearly” approaches. However, it has large consequences on the resource footprint of the recovered products. Except with the 0:100 and the 50:50 approaches, discarding the zero burden assumption results in a resource footprint 22 to 79% higher than with the zero burden assumption. While a shift of paradigm from considering wastewater as a waste to considering it as a resource is necessary and should be considered in environmental sustainability assessment methods, the interest of discarding the zero burden assumption in this case becomes questionable for stakeholders producing these recovered products. A discussion on the “fairness” of each of these approaches resulted in selecting the “degressive linearly” approach as the one sharing the impacts over the process chain the most consistently according to the principles of industrial ecology. However, it is a data intensive approach as data on consumer goods consumption need to be gathered. The selection of an approach could depend on the incentives that policy makers want to give to each of the actors along the chain. A similar idea is followed in the BPX30-323-0, the French repository for good practices for the communication of the environmental impacts of products. It proposes to choose different allocation factors to pull the market of recycled products depending if the market for secondary materials is in equilibrium or not. If there is a high demand for secondary materials, all the impacts of recycling are allocated to the recycler, thus encouraging the producers of secondary materials to put their materials on the market. If there is no disequilibrium, the impacts of recycling are equally shared between the producer of secondary material and the recycler. The 0:100 and 50:50 approaches are the most favourable for the producers of recovered products compared to the zero burden assumption followed so far in LCA studies. The "50:50 adapted" and "degressive linearly" approaches are the least favourable but might be interesting approaches for policy makers as it provides an overview of the contribution of consumption to the resource footprint of recovered products. The results of this analysis encourage policy makers to take action towards less resource intensive consumption patterns. A future interesting analysis could be to evaluate the impact of those consumption patterns on the resource footprint of the recovered products.

Chapter 6: Conclusion and future perspectives

1. General conclusions

1.1 Advantages and limitations of the implemented recommendations

The concept of circular economy being integrated in industry introduces a risk of using this concept as a marketing argument rather than as a real way to allow our own development without compromising the ability of future generations to meet their own needs, as stated in the definition of sustainable development. To do so, the environmental sustainability of newly developed products and services has to be measured using objective and scientifically sound methods. Existing methods have been presented in Chapter 1. The diversity of the approaches which can be followed in assessment studies reflects the complexity of the field of environmental sustainability and the numerous challenges it aims to tackle. One single method answering all questions on sustainability does not exist. The diversity of life cycle impact assessment methods for some impact categories shows that the field is constantly evolving and debated to catch the most complete picture possible on the sustainability of products and services. This is especially the case for the assessment of the impact of resource use for which many uncertainties still remain. Examples of major uncertainties are the amount of resources still available in the Earth crust, which technologies will be able to extract resources which are not reachable today and what will be the demand of future generations for natural resources. It results in a wide range of impact assessment methods, as presented in Table 1 of Chapter 2. These uncertainties leave room for lobbying, for example on the debate on which types of reserves should be considered in sustainability assessment studies, or if resource depletion is at all an environmental problem or belongs more to the field of economics. It is very likely that the field of sustainability assessment will keep evolving for many years as new challenges to tackle appear every year. That is why sustainability assessment methods should not be expected to provide the one and only truth on the sustainability of products and services, but a set of insights that should help the decision making process.

This diversity of methods and approaches leaves the freedom to choose the approach to follow. This can introduce a risk of choosing the approach providing the most favourable results, but also a challenge when it comes to compare these products/services. Policy makers need to be able to compare the resource efficiency of research and innovation projects to define new policies or orientate new research and innovation funding programs. Similarly, assessing the resource efficiency of products at the research and innovation stage can help industry orientate its business strategy. Some recommendations to support decision makers in their choice of assessment method are provided in Chapter 2 and applied to case studies in Chapters 3, 4 and 5. Implementing these recommendations brings additional insights to decision makers on the environmental sustainability of the studied technologies. However, they also require "efforts" that need to be invested to apply them. The benefits brought by the application of each of these recommendations and the "efforts" that were necessary to implement them are summarized in Table 1 and presented in the next sections.

1.1.1. Upscaling technologies so far only developed at lab or pilot level to allow a fair comparison with benchmark products

This recommendation was implemented in Chapter 3. The analysis of the environmental sustainability of a MaB-flocs raceway pond system showed that upscaling is a key step to provide useful insights on the potential resource and emission savings of new technologies to policy makers and industry. It provides the order of magnitude of the additional or avoided potential environmental burdens that could occur if the technology is implemented.

Table 1: *Overview of the insights provided and efforts to be invested when implementing the four recommendations in Chapters 3 to 5 of this thesis.*

Chapter	Implemented recommendation	Insights provided by implementing the recommendation	"Efforts" to be invested and limitations of the implementation
Chapter 3	Upscaling technologies so far only developed at lab or pilot level to allow a fair comparison with benchmark products	• Identifies the potential magnitude of reduction of the environmental burden of a technology developed at pilot scale; • Identifies the processes contributing to a higher footprint compared to the benchmark system after upscaling.	• Involvement of an expert of the process to be upscaled; • Requires literature screening and assumptions to be made; • Might require gaining knowledge on upscaling tools such as learning curves.
Chapter 4	Conducting a consistent assessment complemented with a material or substance mass balance	• Allows the evaluation of a consistent system where inputs are related to the outputs.	• Need for a thorough literature study to estimate the substances' transfer coefficients for each process: ➢ Time consuming; ➢ Requires a good understanding of the functioning of the processes; • Lack of data on transfer coefficients for organic waste management in tropical countries.
Chapter 4	Coupling gate-to-gate and life cycle analyses	• Provides additional insights on resource efficiency, especially related to resource self-sufficiency; • Results in using specific local characterization factors for emissions; • Highlights the difference of order of magnitude of the impacts from emissions on the local and the global populations.	• Requires to conduct a material/substance flow analysis; • Requires to have data on the concentration of local emissions, or to apply a dispersion model based on air flows in the surrounding environment (open air or indoor environment), ➢ Time consuming; ➢ Requires to gain knowledge on dispersion modelling;

Chapter	Implemented recommendation	Insights provided by implementing the recommendation	"Efforts" to be invested and limitations of the implementation
			• Can introduce a bias due to the asymmetry between the handling of the emissions from the foreground and background systems.
Chapter 5	Reviewing the way resources consumed by circular systems are accounted for today	• Contributes to apply the principles of industrial ecology in LCA ➢ Allows a consistent consideration of the concepts in life cycle assessment studies.	• Requires the definition of allocation factors between the intended products and unintended secondary resources: ➢ Often arbitrary. • Results in conclusions that are not favourable to the waste valorisation; processes: ➢ Could discourage their implementation. • The more valorisation products the process chain delivers, the more partitioning of the chain is necessary to know the impact of each single product; • Implies to calculate the environmental burden of consumer goods ending in the sewage system: ➢ Requires large amounts of data on consumption habits.

The upscaling of the MaB-flocs raceway pond reduced the resource footprint of the pond by a factor 3, especially because of the economy of scale that highly benefits the energy consumption for stirring and the resource use for infrastructure. However, the results show that upscaling is not sufficient to make the system competitive with the current situation (baseline scenario), and other improvements are still necessary. Therefore, the method allows identifying the processes that could still contribute to a higher environmental footprint compared to the benchmark system after upscaling. The upscaling of the pilot system was made together with the developer of the technology, who provided estimations on areas, energy consumption and yields of an up-scaled set-up. Improvement options were also defined with the project developer, but information for modelling had to be completed by a literature review and additional expert consultation. This was especially the case for the estimation of the energy consumption of the paddle wheels, which depends on the physical characteristics of the pond. Other tools such as learning curves could be used but would require the person in charge of the life cycle assessment study to gain knowledge on this field, as today this is not an expertise that the LCA community has acquired.

The approach followed to analyse the system in Chapter 3 could also be applied to other technologies. First, the pilot plant is thoroughly described and an upscaled plant is modelled based on the data available at the pilot plant to obtain an upscaled system as close as possible to what has already been implemented, thus ensuring that the modelled upscaled system could function. For example in Chapter 3, rather than directly implementing in the upscaled system the paddle wheels with which the functioning of the MaB-flocs pond has not been tested, the same propeller pumps as in the pilot plant are considered in the linearly upscaled plant. This can be done together with the technology developer. The analysis of the linearly upscaled technology allows identifying the processes contributing the most to the environmental impact and the upscaled system can be refined by testing some improvement options while ensuring that a reality check is conducted for parameters having a large impact on the results.

1.1.2. Conducting a consistent assessment based on a material or substance mass balance

This recommendation was implemented in Chapter 4 by conducting a substance mass balance on carbon and three nutrients, and on which the LCA calculations were based. Most LCA studies do not conduct a full and complete substance balance at the level of the foreground system. However, this is particularly important in the case of LCA studies analyzing waste treatment and valorisation options as the properties of products and emissions from waste are defined based on their chemical composition. By not closing the substance balance, the LCA community takes the risk to model unrealistic systems in which inputs and outputs are not linked via the understanding of the substance pathway along the process chain. In this case, the conclusions are weaker. In this work, the SFA conducted on carbon and nutrients allows studying a consistent system, in which the mass balance is respected all along the chain and the emissions are linked with the inputs, which strengthens the conclusions of the study. Moreover, it allowed calculating gate-to-gate resource use indicators that are useful as well. The drawback of implementing this recommendation is its need for a thorough literature study to estimate the transfer coefficients of the substances for each process such as composting and anaerobic digestion. It required an intensive search for data in literature as well as expert consultation. This is partly because the tropical conditions under which the system is implemented were not often represented in the scientific literature. It resulted in using data from processes implemented under temperate climate, which is not always representative for tropical conditions. Moreover, even under temperate climate, there is a large variation of the fate of substances within the studied processes due to different conditions of process implementation (e.g., open versus covered composting and type of aeration of the compost). Therefore, implementing this recommendation requires the person in charge of the LCA study to gain knowledge on the chemical processes occurring within the studied processes, especially in the case of biomass processing.

1.1.3. The importance to couple gate-to-gate and life cycle analyses

This recommendation was implemented in Chapter 4 by calculating gate-to-gate indicators of resource use. Differentiating resource use at both gate-to-gate and life cycle levels resulted in making the same differentiation to evaluate the impact of local emissions on the health of the surrounding population, which also brought additional insights to the analysis.

The calculation of the resource-based gate-to-gate indicators allowed gaining information on the self-sufficiency of the rural population on nutrients for fertilization purposes in Chhattisgarh. The implementation of processes with a higher life cycle-based resource efficiency can result in an increase of the self-sufficiency of the region where they are implemented. However, the self-sufficiency of a region or a community can only be evaluated based on a gate-to-gate approach, i.e., by focusing on the foreground system. For example in Chapter 4, the implementation of the prospective scenario results in a lower life cycle-based resource footprint of the studied products, but also in a higher amount of resources imported by the rural community to fulfil its needs. Moreover, the calculation of the self-sufficiency indicators allows pointing out the resources that are the most impacted by the implementation of the technology and those which are not. In Chapter 4, the flows of potassium appear to be much more impacted by the implementation of the prospective scenario than the flows of nitrogen and phosphorus. When calculated at substance level, the drawback of such indicators is that they require conducting a substance flow analysis of the system. As discussed in section 1.1.2, it is time and knowledge intensive.

The impact of emissions on human health was also characterized at gate-to-gate level, i.e., at the level of the foreground system. It resulted in replacing the generic characterization of the impact of emissions on human health, which would be used in traditional LCA, by specific characterization factors for emissions affecting the local population. The results highlight the difference of order of magnitude of the impacts from emissions on the local and the global population. This difference cannot be seen when using generic characterization factors as usually done in LCA. This is shown in Fig. 1, which presents the impact of the basket of products on human health when using specific and generic characterization factors.

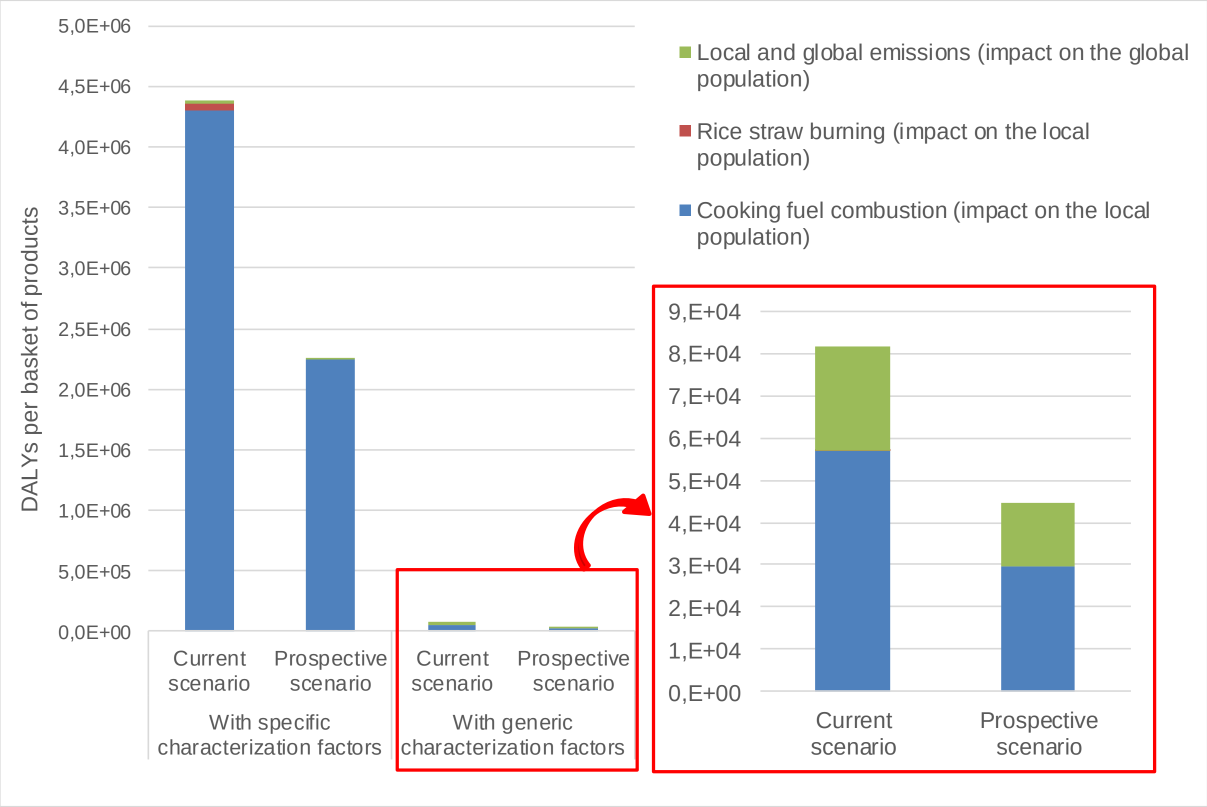


Figure 1: *Comparison of the impacts on human health calculated with specific (as done in Chapter 4) and generic (based on the ReCiPe method) characterization factors.*

The impact of emissions on the local population drops by 99% when using the generic characterization factors provided by the ReCiPe method. Moreover, even though the impact on the local population is still the main contributor to the total impact on human health, it represents 70% of the impact while it represents 99% of the impact when using specific characterization factors. This approach provides a more accurate vision on the contribution of local impacts to the total results and allows policy makers to have a clearer view on the prioritization of measures to be implemented to reduce this impact. However, it is a time intensive process. It requires gathering data on the concentration of local emissions, or, when not available, to apply dispersion models based on air flow in the surrounding environment (open air or indoor environment).

1.1.4. The need to review the way resources consumed by circular systems are accounted for today

Today, there is a paradigm shift from considering waste as disposable towards considering waste as a resource that can be used in further processes. This implies considering waste differently in sustainability assessment studies. The common way of dealing with waste streams used as inputs in processes today is to consider that it does not bear any environmental burden related to its production. However, if they are considered as resources, a burden should be allocated to them and the zero burden assumption should be discarded. This recommendation was discussed in Chapter 5, in which the impact of discarding the zero burden assumption on the resource footprint of sewage sludge valorisation products was tested. Discarding the zero burden assumption can contribute to apply the concept of industrial ecology in LCA. However, simply discarding the zero burden assumption and considering waste as a product in a similar way as the intended product is still not fully in line with the concept of industrial ecology and the dependence of the intended products toward the waste valorisation processes should still be considered. This can be done by following similar approaches than the ones already applied in the sector of material recycling in LCA, i.e., by applying the so-called “end-of-life” formulas. The approach “degressive linearly” appears to be the one that reflects best the concept of industrial ecology by considering the dependence of the processes to one another along the chain, as well as the “hierarchy of intend”. Applying such an approach is time intensive. First, it requires defining allocation factors between the intended products and unintended secondary resources. This choice is often arbitrary and can result in different conclusions with regard to the sustainability of the studied products (De Meester, 2013). Moreover, the more valorisation products are obtained along a waste treatment chain, the more partitioning is necessary to assess the environmental burden of each single product and the more confusing the studied system becomes. Discarding the zero burden assumption also requires assessing the environmental burdens of the intended products that result in the production of the valorized waste. In the case of household waste, it implies calculating the environmental burdens of the consumer goods producing the studied waste stream, which is based on large amounts of data on households' consumption habits. Finally, this approach results in conclusions that are not favourable to the waste valorisation processes as a large environmental burden is allocated to them. This can discourage the implementation of circular systems. Applying the approach described in Chapter 5 to other waste and wastewater valorisation chains could provide additional insights on its added value for industry and policy makers.

1.2 The potential of the studied technologies to increase the resource efficiency at macro-scale

The innovative resource recovery processes analyzed in this thesis show a high potential to increase resource efficiency. They can contribute to save primary resources as well as lower the impact of emissions. In Chapter 3, the upscaled MaB-flocs technology shows a potential to divide the resource footprint of the treatment of aquaculture wastewater by 9 when MaB-flocs are valorized as shrimp feed. In Chapter 4, the implementation of anaerobic digestion to digest cow dung and rice straw, which are combusted in India today, shows a potential to decrease the dependency of farmers to potassium from synthetic fertilizers by 13% and to increase the resource efficiency of the system by 60%. In Chapter 5, the implementation of anaerobic digestion and struvite precipitation as intermediary steps between wastewater treatment and sludge incineration decreases the difference of resource consumption between the sludge valorisation products and the benchmark products by 34% (applying the "degressive linearly" approach).

However, the results also show that the benefits obtained from these innovative resource recovery processes cannot always be observed and can depend on several conditions of implementation. In Chapter 3, the upscaled plant only becomes more competitive with the baseline scenario when the three discussed improvement measures are all implemented (i.e., improvement of the stirring efficiency of the pond, changing the electricity mix and improving the MaB-floc productivity). Similarly, in Chapter 5, the resource footprint of the recovered products is 61 to 94% higher than the benchmark products, and this difference is only reduced when anaerobic digestion and struvite precipitation are implemented as intermediary steps. One challenge of the newly developed resource recovery processes presented in this thesis is that they often result in high energy consumption that contribute to make them non-competitive compared to the current alternatives from an environmental point of view. For example in Chapter 3, the modelled up-scaled plant shows that the electricity consumption needed to stir the pond contributes 88% to the total resource footprint as well as the carbon footprint of the system. Similarly, in Chapter 5, adding an anaerobic digestion step to produce biogas from the wastewater sludge does contribute to lower the impact from incineration but this decrease is partly compensated by the energy required for the pre-treatment of the sludge in the THP system and the electricity needed to clean and compress the biogas. Therefore, the energy efficiency of newly developed resource recovery processes should be a major focus for improving the environmental sustainability of these systems. Another outcome of this work is that on top of consuming resources as utilities, technologies such as incineration (Chapter 5) and the combustion of organic waste to provide cooking energy (Chapter 4) contribute to loose material resources contained in the waste. This is especially the case for nitrogen, which is 100% lost during the combustion of biomass, but also for carbon, phosphorus and potassium. This stresses the need to replace the only production of energy via combustion of organic waste by more specialized resource recovery processes able to recover specific resources with high value. Not only should the large quantity of treated waste (allowed by incineration) be considered, but also the quality of the recovered products. Therefore, additional steps aiming to recover specialty products from waste and wastewater should be included in the treatment chain before their incineration. It can be implemented through the concept of biorefinery.

The IEA Bioenergy Task 42 ‘‘Biorefineries” defines biorefining as “the sustainable processing of biomass into a spectrum of marketable products and energy” (IEA, 2007). It was developed at the same period as the production of the first generation biofuels, during which the similarities between biofuel production and the refining of fossil fuels that results in several products were pointed out (Cherubini, 2010). The biorefining of organic waste started to develop with the production of the next generations of biofuels. From the strict production of biofuels, it extended to the production of high value products such as chemicals and enzymes (Yang et al., 2015). The potential of wastewater to feed biorefineries was discussed later in time by the scientific community. Similar to raw biomass, many different products can be obtained from wastewater. Puyol et al. (2017) showed that carbon-based products such as biopolymers and methane can be recovered, as well as metal-complexes that can be used as fertilizers or as a source for metal production. The development of the concept in the wastewater sector is slow but starts to impregnate the design of the next generation of wastewater treatment plants. For example in 2017, the Billund biorefinery plant was inaugurated in Denmark and is presented as a demonstration of the wastewater treatment plant of the future (Fig. 2).

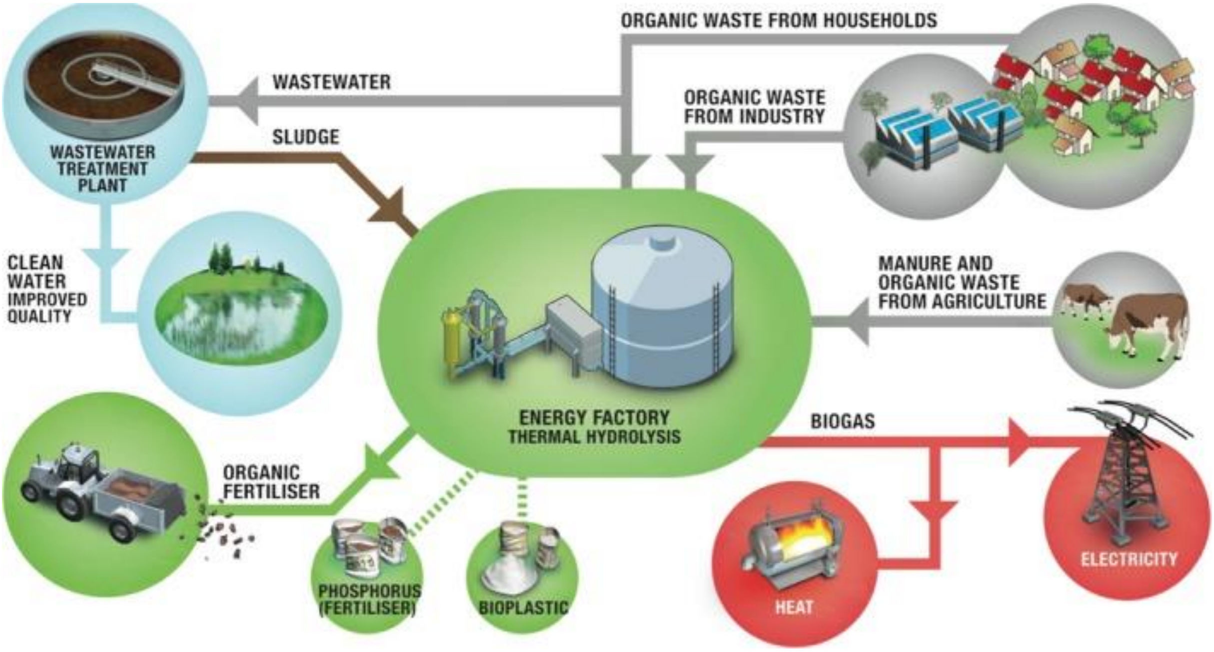


Figure 2: *The biorefinery concept as applied in the wastewater treatment plant of Billund, Denmark (retrieved from Nielsen (2017)).*

The plant treats both sewage sludge and solid organic waste produced by the local population and livestock. It processes it into energy via anaerobic digestion and organic fertilizer, and intends to produce two other by-products, i.e., phosphorus and polyhydroxyalkanoates for bioplastics production. The implementation of such a concept in new facilities will require assessing their environmental sustainability to insure that they indeed contribute to a more sustainable society, with a special attention to the energy needs of the processes, as highlighted above.

2. Perspectives

The outcomes of this thesis leave some challenges for further research. First, additional elements could complete the information provided to policy makers to allow them identifying the technologies that are the most promising in terms of resource savings. This is especially the case for information on the potential of these technologies to save resources at macro-scale (e.g., if implemented at EU level). Moreover, the way consumed resources are accounted for today needs to be re-thought to consider the multiple use of resources. Today, resources consumed are considered as the amount of resource extracted, but taking into account the dissipative use of resources could be of added value when resources are used several times. Finally, this thesis showed that the proposed recommendations could improve the outcome of the resource efficiency and environmental impact assessment studies. However, it also showed that it requires integrating external expertise to the LCA practice. These elements are developed in the following sections.

2.1 Fine-tuning the framework to assess the resource efficiency of new technologies

Chapter 2 presents first elements for the definition of a framework that could be used to assess the resource efficiency of new technologies. It provides preliminary guidance to project developers but could be further developed to more specifically orientate them towards the preferred approach, e.g., under the form of a decision tree. In the context of research funding programs, it is especially relevant for general calls in which it is difficult to define specific requirements on resource efficiency evaluation. Note that the framework can be refined but it is unlikely that it will result in a decision tree ending with the exact method to be used. For more specific calls, e.g., aiming at developing specific technologies, it would be interesting to test the effect of adding requirements on resource efficiency evaluation in the calls on the outcome of the projects and see if the harmonization of resource efficiency evaluation in the different projects provides valuable insights that can be used to orientate policy. To do so, a collaboration with stakeholders in charge of writing research and innovation calls should be built.

The recommendations presented in Chapter 2 can be applied to other sectors but some of them could be refined when they are of particular importance for specific sectors. For example in the sector of electrical and electronic equipment, more specific recommendations on how the criticality of some materials used in components should be included in the analysis should be drawn. Moreover, recommendations on how to account for the recoverability of these materials should also be provided. In the energy sector, the way the energy flows are accounted for in gate-to-gate analyses should be harmonized as many approaches are followed today, as presented in Chapter 1 (e.g., based on primary energy, feedstock energy or energy embedded in energy carriers).

2.2 Improving the evaluation of the potential of innovative technologies to increase the resource efficiency at macro-scale

One of the major goals of the research and innovation funding programs in the EU is to foster the development of new processes and technologies that can help the EU reaching its resource efficiency targets. However today, there is no consensus on how to evaluate resource efficiency and each project follows its own approach. The consequence is that projects cannot be compared and public authorities lack information to conduct a proper evaluation of project outcomes that would help outlining a strategic agenda and orienting the focus of future calls towards the most resource efficient fields of research. First elements to improve the outcomes of innovation projects regarding resource efficiency evaluation were proposed in Chapter 2.

One recommendation for future work is to link the resource efficiency indicator at micro-scale to a resource efficiency indicator at macro-scale to see how each innovative technology could contribute to the overall policy goal. Today the lead resource efficiency indicator at the EU level is the GDP/DMC ratio. In the future, DMC is expected to be replaced by the Raw Material Consumption (RMC), expressed in Raw Material Equivalents (RME), which is able to account for raw materials consumed along the whole supply chain of the products and services consumed by the EU, and not only those which cross the border of the EU (Eurostat, 2016a). The mass weight of extracted material is therefore generally higher than the mass weight of goods crossing the EU border, as shown in Fig. 3.

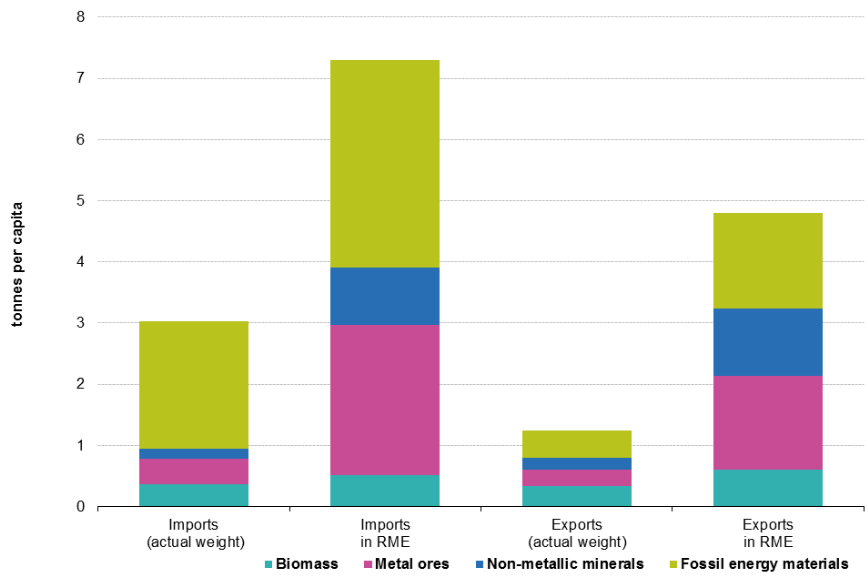


Figure 3: *Comparison of the actual weight of traded goods with trade in raw material equivalents (RME) for the EU-28 in 2014 (in tonnes per capita; retrieved from Eurostat (2016b))*

Raw material equivalents are calculated for 182 product groups (NACE classification) based on 51 raw material categories. This means that the raw material consumption of the 182 product groups is calculated as the sum of the amount of each of the 51 raw materials considered and necessary to produce these products. For example, the product group "Electrical equipment" (NACE code 27) consumes 0.921 tons of the 51 raw materials per 1000 euros of product (e.g., 0.01 ton of nickel, 0.086 tons of gold – gross ore, 0.046 tons limestone and gypsum) (Eurostat, 2016c). Therefore, the raw material consumption of this product group is 0.921 tons RME. They are calculated every year based on an annual high resolution monetary input-output table, complemented by information for some product groups on regionalised information such as metal recycling ratios in exporting countries or the energy mix of electricity generation, to account for the difference of production technologies in the member states and in the non-EU exporting countries. Therefore, different RME coefficients are obtained for imported and exported goods.

In case innovative technologies would be implemented in the EU, they are expected to have an impact on the lead resource efficiency indicator of the EU. One option to evaluate the potential of innovative processes and technologies is to estimate, at the stage of technology development, how these technologies would impact the RMC of the EU if implemented at full scale. The result of this estimation would be a variation of RMC in percentage, which could allow a comparison between technologies in different sectors. One possible approach to do so is:

1. Making an inventory of the input and output flows from the innovative process;
2. Upscaling the technology as if implemented at industrial scale;
3. Converting the innovative process input flows into RME;
4. Converting the input flows of the process avoided by the new technology into RME;
5. Estimating the potential at the EU level;
6. Calculating the variation of RMC of the EU after implementation.

The variation of RMC of the EU after implementation for two innovative products A and B can be calculated as:

$$\Delta RMC_A = \frac{RMC_{EU} - RMC_{avoided\ product\ A} + RMC_A}{RMC_{EU}} \quad (1)$$

$$\Delta RMC_B = \frac{RMC_{EU} - RMC_{avoided\ product\ B} + RMC_B}{RMC_{EU}} \quad (2)$$

Where RMC_{EU} is the RMC of the EU, $RMC_{avoided\ product\ i}$ is the amount of RME consumed to produce the product avoided by the production of the innovative product *i* at EU scale and RMC_i is the amount of RME consumed to produce product *i* at EU scale.

The implementation of such an approach would first require to check if the same conclusion can be obtained regarding the most resource efficient technology when conducting a conventional LCA approach and when applying the proposed approach. A first rough comparison can be made here by comparing the resource footprint of ecoinvent processes when using a conventional life cycle impact assessment method with the RME of the NACE product group they belong to. As land and water are not accounted for as resources consumed by the different product groups to calculate the RMC, an LCIA method that does not consider these two resources is chosen, i.e., the Cumulative Energy Demand (CED). One remark is that in LCA software packages, no mass-based LCIA accounting method is available, even though it would be a preferred choice when making the comparison with the RME-based approach. Then, ecoinvent processes need to be selected for comparison. The selection of the processes is based on Huijbregts et al. (2010). The authors selected 498 products divided in 8 product groups (metals, glass, paper and cardboard, organic and inorganic chemicals, agricultural products, construction materials and plastics) and evaluated the correlation of their environmental burdens when applying six different environmental life cycle impact assessment methodologies. To compare the resource footprint obtained when applying the CED method and the RME-based approach, two products of each product group defined by Huijbregts et al. (2010) are randomly selected. The product categories from the NACE classification to which each of these products belong are identified. For example, the product "cement mortar" from the ecoinvent database belongs to the NACE category "*23.5. Cement, lime and plaster"*. The amount of RME consumed to produce one unit of these products is then identified using the RME coefficient for imported goods (i.e., for goods consumed) (Eurostat, 2016b) and compared with the results obtained with the CED method. When the unit of product is expressed in monetary terms, it is converted in mass units based on the unit price for intra trade and extra trade import of these goods (Eurostat, 2016b). The results are presented in Fig. 4.

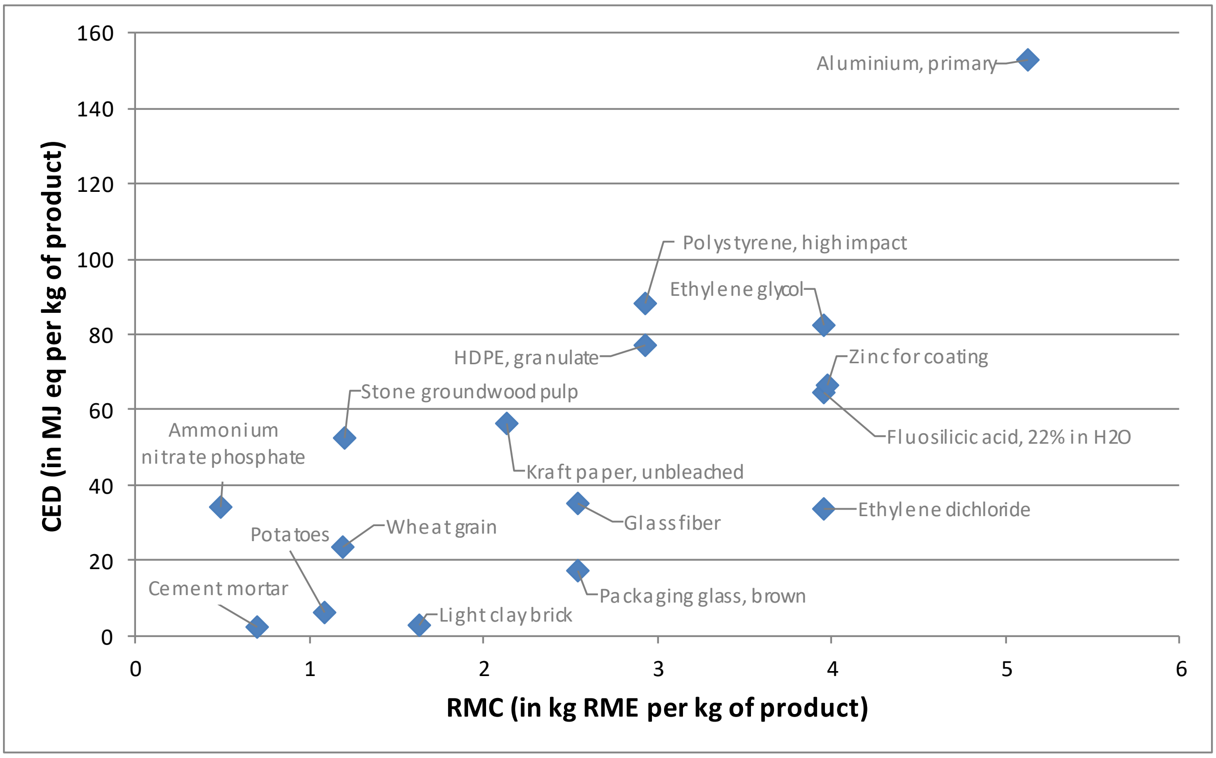


Figure 4: *Comparison of the resource footprint of 16 products from the ecoinvent database using the CED method and the proposed RME-based approach.*

The number of products used for this comparison is not enough to calculate a correlation coefficient between the results obtained with the CED method and with the RME-based approach, but a trend suggesting a positive correlation can be observed. This rough comparison should be refined to make sure that the background processes are consistent in both approaches. For example, the background activities considered to calculate the RME coefficient of the NACE category “Natural water; water treatment and supply services” could be compared to the background processes used to model the product “tap water” in ecoinvent. Today, the available documentation on the RME model does not provide a clear information on these background activities. Moreover, the raw materials considered to calculate the RME coefficients are different from the natural resources considered in LCA databases. For example, biomass such as cereals and fruits are considered as raw materials in the calculation of the RME coefficients, while they are considered as products in LCA databases. This discrepancy between both approaches can limit the application and relevance of the proposed RME-based approach and should be further investigated.

2.3 Considering resource dissipation in LCA

Different approaches can be followed to compare the resource consumption of the products obtained from innovative processes and products designed to save resources, e.g., waste and wastewater recovery processes and products designed to facilitate recycling. The conventional approach in LCA considers that the resources consumed by a product equals the sum of the amount of raw materials extracted from the natural environment during its life cycle, which can be expressed in different units (MJ, MJ_{ex}, kg etc.). However, this approach assigns all resources consumed to the first product of the chain. This is valid in the case of the single use of resources followed by disposal, but becomes debatable in the case of a multiple use of resources, as reuse and recycling allow the conservation of resources in the anthropogenic system. Another approach has been proposed by Frischknecht (2014), based on what is already done in the case of water consumption accounting. The scientific community working on water footprint makes the distinction between the amount of water withdrawn from the natural environment and the water lost during the process. Frischknecht (2014) proposes to apply the same principle to other resources, as resources are extracted from the environment but part is lost and part is still available for further use in the economy. This is typically the case for products produced from waste and wastewater streams. The resource consumed is then defined as the amount of resources used dissipatively: the amount of resources consumed is estimated as the amount of resources lost during the production of the product. This approach is illustrated in Fig. 5, with the example of the extraction and dissipation of phosphorus along a process chain.

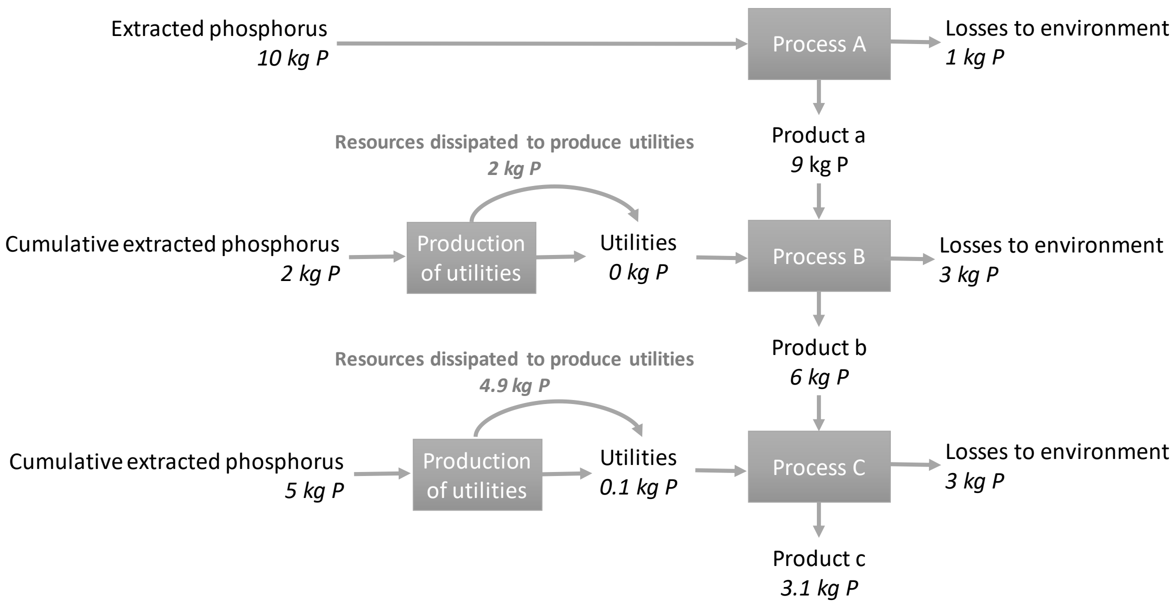


Figure 5: *Example of the amount of the phosphorus extracted and dissipated along a process chain.*

The phosphorus consumption of product c expressed as extracted resources is the sum of all the phosphorus extracted from the natural environment to produce the utilities used by the different processes of the chain. It is 17 kg per unit of product c. The amount of phosphorus used dissipatively to produce product c is calculated as the difference between the phosphorus content of product c and the amount of phosphorus extracted to produce product c. Therefore, it is 13.9 kg per unit of product.

In a hotspot analysis, the way contributions of processes A, B and C to the resource footprint of product c are estimated are different depending on how the resource use is estimated as either extracted or dissipated. Indeed, when assessing the amount of resources extracted, the contribution of process B is calculated as the natural resources *extracted* to produce the utilities used by the process. In the example here, it is 2 kg of phosphorus. The contribution of process B to the amount of natural resources *used dissipatively* to produce product c is calculated as the amount of resources dissipated to produce the utilities used in the process (2 kg of phosphorus) to which the amount of resources dissipated by the process itself should be added (3 kg phosphorus). The total is thus 5 kg phosphorus. Therefore, while the total amount of resources used dissipatively will always be lower than the amount of resources extracted, the individual contribution of processes taken separately can be higher when evaluated as resources used dissipatively than as resources extracted in a hotspot analysis. This is illustrated in Fig. 6 in the case of the example presented in Fig. 5.

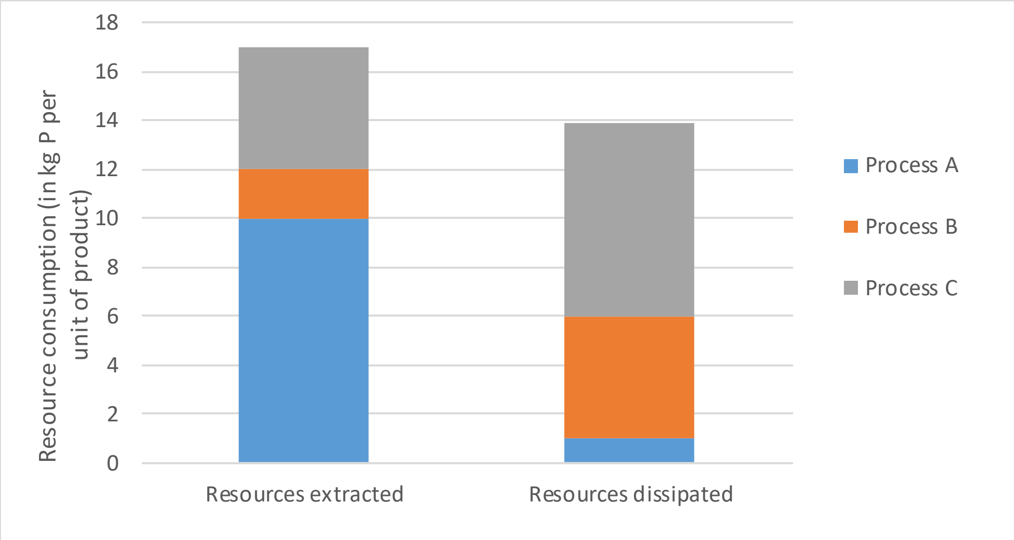


Figure 6: *Resource footprint of product c estimated as the amount of resources extracted and the amount of resources dissipated.*

Today the interest of industry and the scientific community for considering the dissipative use of resources in the anthropogenic system in LCA is growing, but the concept of resource dissipation as introduced by Frischknecht (2014) has not been tested yet. From a practical point of view, implementing this approach would require characterizing all the products in LCI databases in terms of their content in the resource of interest. For example, this could be done for critical raw materials such as phosphorus, cobalt or magnesium. This is a tedious task but when done, the approach can be implemented in software tools, at the condition that software developers check the consistency of the mass balance of each process. This approach allows identifying the processes that contribute the most to the loss of resources. As shown in the hotspot analysis of Fig. 6, it would result in different conclusions on which process contributes the most to the resource footprint of a product than when considering extracted resources. This could result in different measures to be taken, for example with regard to the choice of suppliers. For policy makers, as products would be characterized in terms of their content in specific resources, such an approach could allow “tracking” these resources in the economy and facilitate the management of resources at macro-scale. One important point of attention when developing this method is to consider the availability of resources in the products. Some resources are still present in some products but their recoverability might differ from one product to another. This aspect should be taken into account. The method should be tested on several products to evaluate its added value compared to considering the amount of resources extracted and evaluate the potential trade-offs between the benefits and the efforts to be put to implement it.

2.4 Complementing LCA with external expertise

The recommendations implemented in Chapters 3, 4 and 5 aim to improve the practice of LCA by providing additional insights necessary to a more informed decision-making process. As discussed in section 1.1, they require the LCA community to invest efforts to implement them. Upscaling a technology to evaluate its full potential to contribute to a decrease of resource use or emissions requires the involvement of experts in the field of the process under study or in upscaling tools such as learning curves. Similarly, conducting an LCA based on a closed mass balance requires to gain knowledge on the substance flows within processes and thus to have a deep understanding of industrial processes. The characterization of the impact of local emissions at the local level also requires developing knowledge on the fate of emissions in the environment surrounding the source of emissions. These examples show that while they improve the outcomes of LCA studies (see Table 1), these recommendations require integrating external expertise to the practice of LCA (Fig. 7).

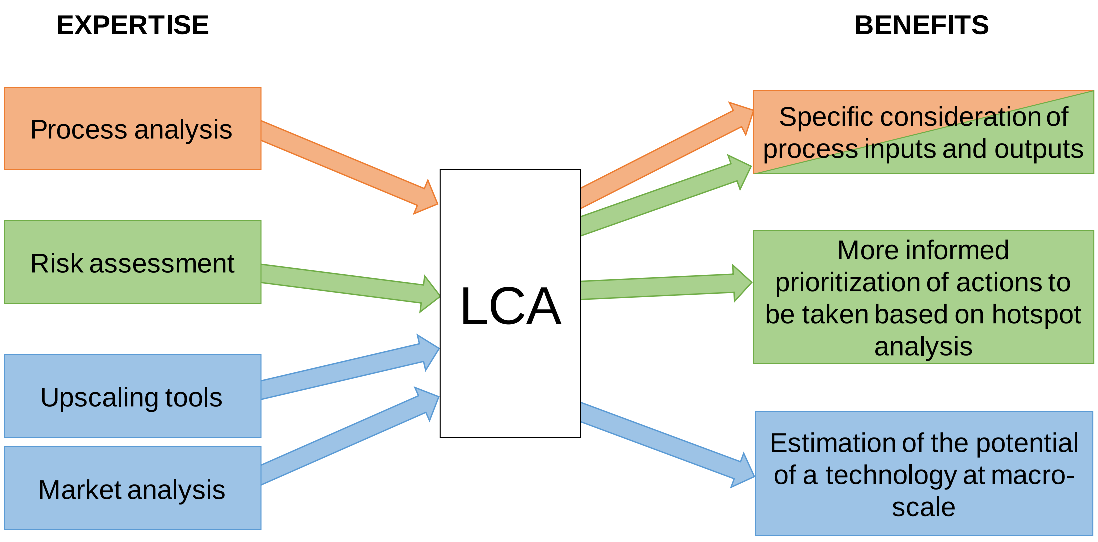


Figure 7: *Schematic representation of the fields of additional expertise to be integrated in LCA and their benefits.*

One major improvement that external expertise could bring in the practice of LCA is specificity, especially regarding substance and material flows within a studied process and the characterization of the impact from emissions.

The LCA community mostly follows a generic approach regarding substance and material flows within a studied process, without linking resource use or emissions with the specific characteristics of the process. This is especially the case in the field of waste-LCA. However in this field, mass balance checks are necessary to ensure a consistent consideration of emissions based on local waste composition (Laurent et al., 2014). It requires integrating more systematically a process-based analysis in LCA. One way to study more consistently a process chain is to use the information on feedstock or waste composition available in LCI databases and look how this parameter affects emissions or resource use in literature. Therefore, it also requires that feedstock and waste compositions are consistently reported in LCI databases. For example, this is already the case for some products in the ecoinvent database (e.g., biowaste and sewage sludge to incineration). In the case of organic waste treatment, a review on the fate of substances within the processes under different conditions (e.g., tropical versus temperate) could provide an overview of the substance transfer coefficients of these processes and facilitate their substance flow analysis.

A generic approach is also followed when characterizing the impact of emissions on the environment surrounding the source of emissions. As shown in section 1.1, using specific characterization factors to assess the impact on the local environment in an LCA study provides a more accurate vision on the priority of the measures to be taken to lower this impact. It is equivalent to coupling a life cycle-based with a risk assessment approach. Today, different experts usually conduct the two methods apart and there is not a standardized way of combining both methods. Harder et al. (2015) reviewed 30 studies blending elements from LCA and risk assessment and showed that authors follow three main approach types when doing so. This highlights a lack of consensus in this field, which should first be improved by a more consistent use of terminology. Moreover, Harder et al. (2015) stress the risk of introducing a bias related to the asymmetry between the handling of the emissions emitted in the foreground and in the background systems. For example in the case presented in Chapter 4, the emissions accounted for as having an effect on the global population (e.g., emissions from fossil fuel refinery to produce diesel for transportation) are also emitted at local level and should also be characterized using specific characterization factors to evaluate their effect on the local population. Therefore, the scientific community should conduct more work to improve the integration of risk assessment in LCA studies.

In the context of the sustainability assessment of innovative products and technologies, expertise on upscaling and market analysis should be used to complete the information provided to policy makers by an LCA study. This is essential to estimate the full potential of a new product or process to contribute to increase sustainability at macro-scale and compare products and processes between each other to orientate policy. Today, project developers need support when calculating the market share and upscaling the studied system. In the context of EU funding programs, working groups could be formed to recommend the methods to follow. They can follow the same approach as when forming the technical working groups in charge of writing the Product Environmental Footprint Category Rules (PEFCRs) in the framework of the Product Environmental Footprint pilots, or the Reference Document on Best

Available Techniques (BREFs) developed under the IPPC and the Industrial Emissions Directives. In parallel of the calls for projects, working groups focusing on the market share calculation could be formed per type of products (e.g., feed and energy) and provide guidelines, for example to choose the period under which the market is described and the potential penetration percentage of the new product/technology. Working groups focusing on upscaling could be formed per type of process (e.g., extrusion and stirring processes) and provide guidelines on the effect of upscaling on resource use and emissions. A guideline providing databases on the materials used per type of application in industry (e.g., for heating water circulation) or known upscaling effects on energy consumption would facilitate the upscaling of the studied process and thus encourage to add this analysis to the results of projects' evaluation. These initiatives could contribute to make the process of upscaling and market analysis more accessible to the LCA community, which has low expertise in these fields today.

In conclusion, the evaluation of the environmental sustainability of innovative products and technologies could be improved by integrating external expertise in LCA. Today, this process is time and resource intensive and requires that companies, research project consortia or public authorities allocate appropriate resources until best practices are well established. Finally, the true value of LCA is when it is used during the design phase and in collaboration with all.

Appendix A: Supplementary material for Chapter 3

Appendix A1: System boundaries of the three studied scenarios

Appendix A2: Devices, infrastructures and energy used for pilot and up-scaled MaB-floc-based wastewater treatment plants

Appendix A3: Description of devices and infrastructures used for the three studied scenarios

Appendix A4: Calculation of land occupation of the pilot and up-scaled wastewater treatment plants

Appendix A5: Ecoinvent processes, flows and values used for the comparison of the MaB-floc-based wastewater treatment plants at both scales

Appendix A6: Ecoinvent processes, flows and values used for the comparison of the three scenarios at industrial scale

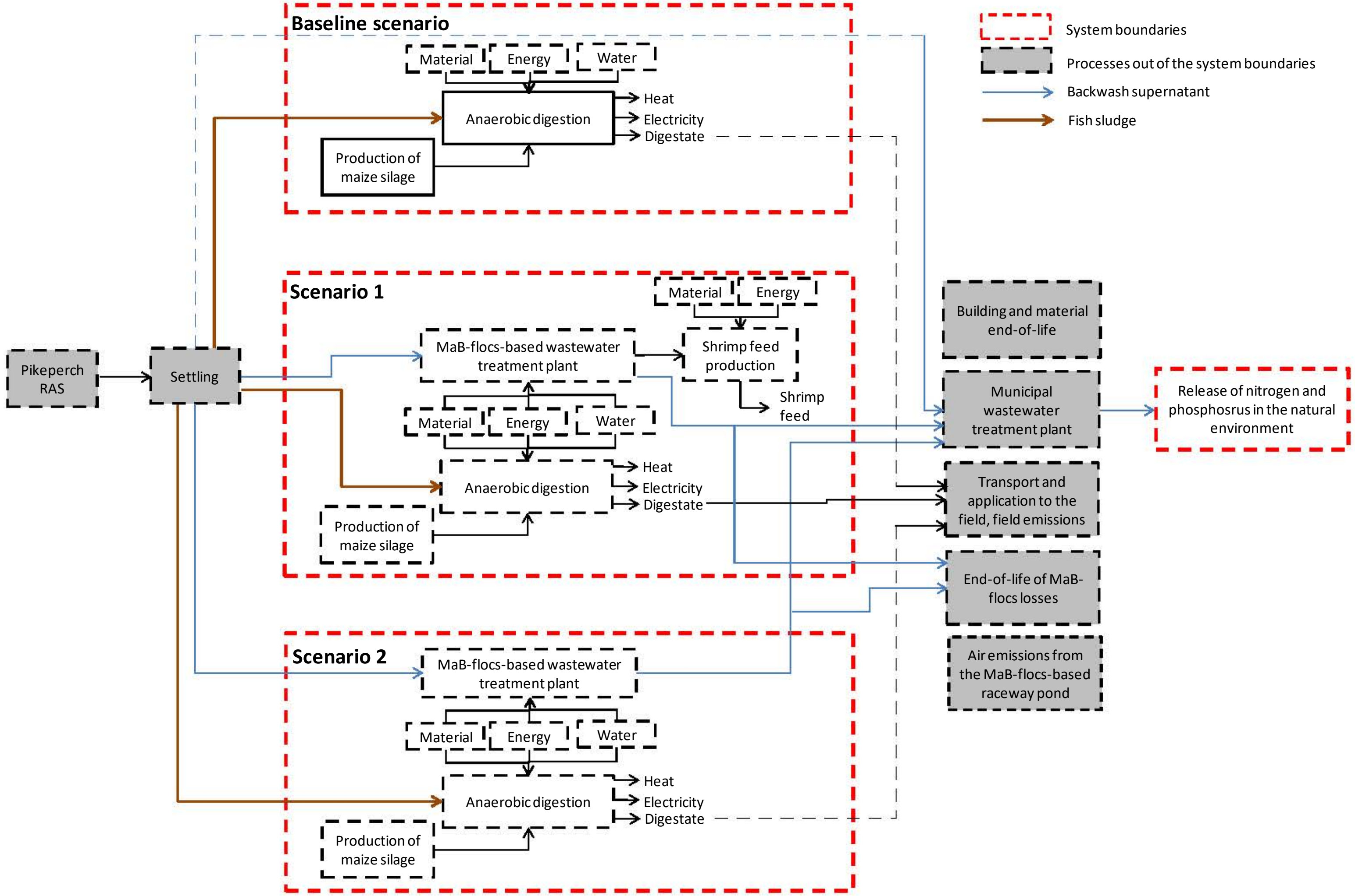
Appendix A7: Calculation of MaB-floc harvesting data for the pilot and up-scaled plants

Appendix A8: Estimation of fish sludge production

Appendix A9: Estimation of biochemical methane potentials and energy production from anaerobic digestion

Appendix A10: Description of avoided processes

Appendix A1: System boundaries of the three studied scenarios



Appendix A2: Devices, infrastructures and energy used for pilot and up-scaled MaB-floc-based wastewater treatment plants

(1) MaB-floc-based wastewater treatment plant – pilot scale

Pond – The pond is made of steel plates (width: 0.55 cm), with polyurethane foam insulation (width: 5 cm). Given the pond width (2.5 m), length (11.7 m) and height (0.5 m) as well as the steel and polyurethane densities (7850 kg m^{-3} and 62 kg m^{-3}), the required amount of steel and polyurethane foam are respectively 1631 kg and 26.4 kg for the pond. Lifetimes of 50 years for steel and 30 years for polyurethane foam (PU Europe, 2013) were chosen.

Pumps – The characteristics of each pump are detailed in Table A2-1.

Table A2-1: *Characteristics of pumps used at pilot scale (Van Den Hende et al., 2014a)*

Pump[a]	**Type**[a]	**Weight**[b] **(kg)**	**Power**[a] **(kW)**	**Efficiency**[c] **(%)**	**Working time**[a] **(h day^{-1})**
Propeller pump (x2)	MXD 230/90/075; Dreno	20	0.75	80	14.1
Influent pump	Emerged pump; EUS, EVAK Taichung, Taiwan	14.9	0.75	80	0.3
Effluent pump	Submerged pump; Industrial pump system bvba, Belgium	14.9	0.75	80	0.3
Harvesting pump	EP Midex, Liverani, Italy	14.3	0.75	80	0.001[d]

[a] Van Den Hende et al. (2014a)
[b] Van Den Hende et al. (2014a) and supplier information
[c] Taelman et al. (2013)
[d] 0.35 min perharvest

The type and amount of material of the pumps were extrapolated from the process ‘pump 40W, at plant’ (ecoinvent 2.2, see Table A2-2), based on the weight of each pump.

Heating of the pond

- Heating tubes: the heating tubes below the pond are made of copper. The total length of the tubes is 600 m and their diameter is 1.2 cm. A copper tube wall thickness of 0.10 cm was assumed (Copper Development Association Inc., 2011). Using a density of 8960 kg m^{-3}, a copper weight of 206 kg below the pond was estimated. A copper lifetime of 50 years was chosen.

- Boiler: data for the boiler was taken from the process ‘gas boiler’ in ecoinvent 2.2.
- Energy consumption: the consumption of heat was estimated based on the volume of gas consumed over the period of operation of the plant (Table A2-3).

Table A2-2: *Material, weight and proportion of each component of the product ‘pump 40W, at plant’ (ecoinvent 2.2)*

Material	Weight	Unit	Proportion (%)
Aluminium, production mix, wrought alloy, at plant	0.02	kg	1
Cast iron, at plant	1.2	kg	49
Chromium steel 18/8, at plant	0.92	kg	38
Copper, at regional storage	0.25	kg	10
Polyvinylchloride, at regional storage	0.03	kg	1
Synthetic rubber, at plant	0.007	kg	0.3
TOTAL	2.4	kg	

Table A2-3: *Gas consumption recorded during the pilot plant operation*

	From 21/01/2013 to 04/03/2013	From 04/03/2013 to 30/09/2013	Unit
Gas consumption[a]	850	931	L
Average daily consumption	19.8	5.4	L day^{-1}

The heat consumption was then allocated to each period defined in Van Den Hende et al. (2014a) based on their duration (periods 1 to 8), and calculated the average of the daily consumption for the periods considered in this study (periods 4 to 8). The average of the daily natural gas consumption is 5.4 l day^{-1}. Using a calorific value of 35.2 MJ m^{-3} for natural gas (UK Biomass Energy Center, 2014), a consumption of 0.19 MJ day^{-1} of natural gas was calculated.

Tubing – We estimated the polyethylene tubing length, thickness and diameter of 2 m, 2 cm and 10 cm. With a density of 950 kg m^{-3} (Frank et al., 2009), a weight of 3.8 kg of tubing was calculated. The lifetime of polyethylene is 50 years (Frank et al., 2009).

Settling tank – The settling tank is a 1000 m^3 cubitainer (HDPE + galvanized steel structure). Based on suppliers website, a weight of 20 kg of HDPE and 20 kg of steel were estimated.

(2) MaB-floc-based wastewater treatment plant – up-scaled

Pond – The pond is dug in the ground and covered with a 286 m^2 HDPE foil. The thickness of the wall is 2 mm (producers: RKW SE Philippsthal, Germany, and Numa Industrial, Spain).

Pumps – The characteristics of each pump are detailed in Table A2-4.

Table A2-4: *Characteristics of pumps used in the theoretical up-scaled plant (data for 1 pond)*

Pump	Type	Weight (kg)	Power (kW)	Efficiency[a]	Working time (h day^{-1})
Propeller pump (x6)	Same as pilot scale	20	0.75	80%	14.5 (per pump)
Influent pump (x1)	CO(M) 500/22, Lowara	20	2.2	80%	0.5
Effluent pump (x1)	CO(M) 500/22, Lowara	20	2.2	80%	0.5
Harvesting pumps (x3)	CO(M) 500/22, Lowara	20	2.2	80%	0.3 (in total)

[a] *Taelman et al. (2013)*

We considered that 1 influent pump was used for 2 ponds. One hour is necessary to pump 49 m^3 of wastewater in 2 ponds (two times 24.5 m^3). The pump with the reference CO(M) 500/22 from Lowara allows such a flow, with a power of 2.2 kW. The same pump is chosen for the effluent pumps, working 0.5 h day^{-1} per pond, and the harvesting pump, working 19 min day^{-1} per pond (7.4 m^3 harvested from the pond and then pumped back both in the pond and to the filter press).

Flue gas blower – For the up-scaled scenario, a flue gas blower was included. The calculation of the flow and working time of the blower is extrapolated from data of the pilot scale (for which there was just a flue gas bottle and no blower) and detailed in Table A2-5.

Table A2-5: *Calculation of the required flow of the flue gas blower for the up-scaled plant*

	Pilot scale[a]			**Up-scaled**			
	CO_2 concentration in flue gas	Duration of injection (h day^{-1})	Quantity of flue gas injected (L day^{-1})	CO_2 concentration in flue gas	Working time[b] (h day^{-1})	Quantity of flue gas injected (Nm^3 day^{-1})	Required flow of the blower (Nm^3 h^{-1})
Period 4	12%	2.5	711	5%	6.0	14.9[d]	0.061
Period 7	No flue gas	-	-	5%	7.8[c]	18.3[e]	0.058
Period 8	5%	9.5	2481	5%	9.5	21.7[f]	0.056
					Average (weighted by periods duration)		**0.057**

[a] *Van Den Hende et al. (2014a)*
[b] *Data from pilot scale multiplied by 12/5 (CO_2 concentration rate at pilot scale and up-scaled)*
[c] *Average between period 4 and 8, as the need in CO_2 injection is increasing from winter to summer (pH decreases with light increase)*
[d] *Data from pilot scale adapted to the up-scaled size of the pond and multiplied by 12/5*
[e] *Average between period 4 and 8*
[f] *Data from pilot scale adapted to the up-scaled size of the pond*

The Bosa blower SER-8, with a power of 50 W, allows the required flow. The weight of this blower was not available. However, the weight of the smallest Bosa blower of the CB series was used (CB-820-4T, 12 kg).

Heating of the pond

- Heating tubes: the heating tubes of the up-scaled plant are assumed to be made of steel. The length of the steel tubes was calculated so that the amount of heat per square meter delivered to the pond is the same as at pilot scale, and that the temperatures of the water entering and exiting the copper tube are the same as the water entering and exiting the steel tube. Therefore, the logarithmic mean temperature difference between outside and inside the heating tube are the same in both cases. The calculation was made using the heat equation based on Fourier's law:

$$Q = k_{copper} \times \frac{A1}{d} \times \Delta T_{LM} = k_{steel} \times \frac{A2}{d} \times \Delta T_{LM}$$

Where:

Q = heat transfer through the tube (W)

k = conductivity of the material; k_{steel}=50.2 W m^{-1} K^{-1}; k_{copper}=401 W m^{-1} K^{-1}

A = heat transfer area (m^2)

d = diameter of the heating tube (m)

ΔT_{LM} = logarithmic temperature difference between the outside and inside of the heating tube (K)

The factors d and ΔT_{LM} have the same value in both cases.

Based on this equation, 4602 m of tubing is necessary below each pond. The diameter and thickness of the tubes are considered the same as the copper tube of the pilot plant. Using a density of 7850 kg m^{-3}, calculated a steel weight of 1384 kg below each pond. A lifetime of 50 years for steel was chosen.

- Boiler: Data for the boiler was taken from the process ‘gas boiler’ in ecoinvent 2.2.
- Energy consumption: assumed that the energy consumption of the pond was proportional to its volume. calculate a average natural consumption of 47 L day^{-1} (or 1.7 MJ day^{-1} of natural gas).

Tubing –The same material and amount of tubing than for the pilot scale were assumed.

Settling tank – Each settling tank has a volume of 8 m^3. It is dug in the ground and coated with a 18 m^2 HDPE foil (same type as for the pond).

Belt filter press – The energy consumption of the filter press was assumed to be 0.55 kWh kg^{-1} DM MaB-flocs (0.4-0.7 kWh kg^{-1} DM algae; Van Den Hende et al. (2016)). For the filter press, a weight of of 415 kg was used (steel, based on the model SF:1000, Salsnes Filter).

APPENDIX A3: Description of devices and infrastructures used for the three studied scenarios

(1) Baseline scenario

The materials used for the digester of the baseline scenario are based on the process 'anaerobic digestion plant, agriculture' of ecoinvent version 2.2. Their amount as well as land occupation are assumed to be proportional to the digester capacity. For the baseline scenario, 3.5 m^3 of feedstock and water is supplied to the digester per day. With a solid retention time of 30 days, the capacity of the digester is assumed to be 106 m^3.

Table A3-1: *Description of infrastructure and land occupation of the digester of the baseline scenario*

	Anaerobic digestion plant, agriculture[a]	**Digester of the baseline scenario**	**Unit**
Capacity	300	112.6	m^3
Occupation, industrial area, built up	2220	833.4	$m^2.a$
		0.1	m^2 a day^{-1}
Chromium steel 18/8, at plant	780	292.8	kg
Polystyrene, high impact, HIPS, at plant	342	128.4	kg
Synthetic rubber, at plant	180	67.6	kg
Concrete, normal, at plant	78.5	29.5	m^3
Polyethylene, HDPE, granulate, at plant	25.5	9.6	kg
Glued laminated timber, outdoor use, at plant	5.5	2.1	m^3
Copper, at regional storage	75	28.2	kg
Polyvinylchloride, at regional storage	49.5	18.6	kg
Reinforcing steel, at plant	6580	2470.2	kg

[a] *ecoinvent 2.2*

The energy requirements for anaerobic digestion are based on the process 'biogas, from agricultural co-digestion, not covered, at storage' (ecoinvent version 2.2). The consumption of electricity and heat are assumed to be proportional to the volume of biogas produced.

Table A3-2: *Energy consumption of the digester in the baseline scenario*

	Biogas, from agricultural co-digestion, not covered, at storage[a]	**Baseline scenario**	**Unit (per day)**
Biogas production	1	408	Nm^3
Electricity consumption	0.14	58.9	kWh
Heat consumption	4.9	2020.7	MJ

[a] *ecoinvent v2.2*

(2) Scenario 1 – valorisation of MaB-flocs as shrimp feed

Digester - Data for anaerobic digestion in scenario 1 (infrastructure and energy consumption) is the same as for the baseline scenario.

Drying of MaB-flocs – The use of a drum dryer with a consumption of 3556 kJ_{gas} kg^{-1} of water removed was assumed (Taelman et al., 2013). Data on the amount of MaB-flocs (TSS) harvested and associated energy required to dewater the MaB-flocs (water content of 18%; Borowitzka and Moheimani (2013)) are presented in Table A2-3. For the estimation of the marine and freshwater eutrophication potentials, no increase of MaB-floc productivity was considered, therefore the values for drying MaB-flocs are the same for case 1 and 2.

Table A3-3: *Energy required for the drying of MaB-flocs*

Case	**Harvested MaB-flocs (kg TSS day^{-1})**	**Volume of water removed (kg)**	**Energy required for dewatering (MJ_{heat}day^{-1})**
$Up_{L,shrimp\ feed}$	154	700	2537
$Up_{SEM,shrimp\ feed}$	199	909	3108

Based on sizing data from R. Simon (Dryers) Ltd. (length: 2.5 m; diameter: 0.9 m) and a hypothetical wall thickness of the dryer of 5 cm, the weight of the drum dryer was estimated to 2774 kg (steel).

Milling of MaB-flocs – On average, 154 kg TSS have to be milled per day. Two industrial mills were found to be able to mill this amount of dried MaB-flocs within a reasonable time (1h). Their characteristics are presented in Table A3-4.

Table A3-4: *Characteristics of the two industrial mills*

Supplier	**Type**	**Power (kW)**	**Capacity**
Wintech Pharmachem equipment PVT. LTD	Multi mill	2.24	50 to 200 kg $hour^{-1}$
Jas Enterprise	Jas 1310BL	7.50	175 to 200 kg $hour^{-1}$

An average power of 4.87 kW and a working time of 1h per day were chosen. Therefore, the electricity consumption of milling is 4.9 kWh day^{-1}. The same energy consumption is used for both estimations.

For the infrastructure of the mill, the characteristics of the Multi mill were chosen (steel, 250 kg).

(3) Scenario 2 – valorisation of MaB-flocs as feedstock for anaerobic digestion

To estimate data for anaerobic digestion (infrastructure and energy consumption), the same methodology as for the baseline scenario was used. Data on infrastructure and land occupation is presented in Table A3-5. Data on energy consumptions is presented in Table A3-6.

Table A3-5: *Description of infrastructure and land occupation of the digester of scenario 2*

	Anaerobic digestion plant, agriculture[a]	**Up$_{L,AD}$**	**Up$_{SEM,AD}$**	**Unit**
Capacity	300	172.1	189.9	m3
Occupation, industrial area, built up	2220	1273.2	1405.3	m2.a
		0.17	0.19	m2.a/day
Chromium steel 18/8, at plant	780	447.3	493.7	kg
Polystyrene, high impact, HIPS, at plant	342	196.1	216.5	kg
Synthetic rubber, at plant	180	103.2	113.9	kg
Concrete, normal, at plant	78.5	45.0	49.7	m3
Polyethylene, HDPE, granulate, at plant	25.5	14.6	16.1	kg

	Anaerobic digestion plant, agriculture[a]	$Up_{L,AD}$	$Up_{SEM,AD}$	Unit
Glued laminated timber, outdoor use, at plant	5.5	3.2	3.5	m3
Copper, at regional storage	75	43.0	47.5	kg
Polyvinylchloride, at regional storage	49.5	28.4	31.3	kg
Reinforcing steel, at plant	6580	3773.7	4165.2	kg

[a] *ecoivent v2.2*

Table A3-6: *Energy consumption of the digester in scenario 2*

	Biogas, from agricultural co-digestion, not covered, at storage[a]	$Up_{L,AD}$	$Up_{SEM,AD}$	Unit (per day)
Biogas production	1	603	661	Nm^3
Electricity consumption	0.14	87.1	95.6	kWh
Heat consumption	4.9	2988.0	3278.2	MJ

[a] *ecoinvent v2.2*

APPENDIX A4: Calculation of land occupation of the pilot and up-scaled wastewater treatment plants

At both scales, the land occupation of the wastewater treatment plants was estimated as the rectangular surface occupied by the pond and the settling tank, adding 1 meter on each site (Fig. A4-1). This surface was assumed to give enough space for the other devices such as the tubing and the pumps.

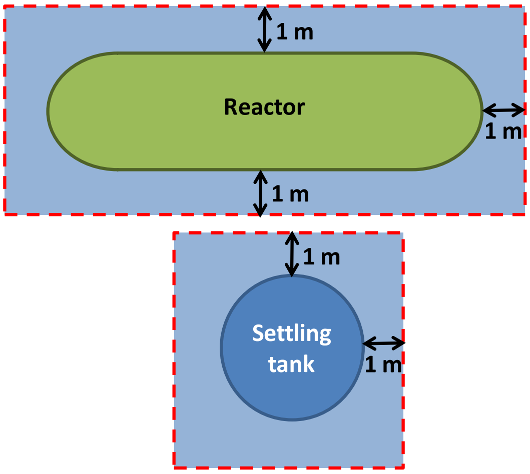


Figure A4-1: *Surface taken into account to estimate direct land occupation of the wastewater treatment plants (red dotted line)*

The surface occupied by the pilot plant is 71 m^2. The surface occupied by the up-scaled plant is 382 m^2 $pond^{-1}$.

APPENDIX A5: Ecoinvent processes, flows and values used for the comparison of the MaB-floc-based wastewater treatment plants at both scales

All values are expressed per kg TSS of MaB-flocs.

(1) MaB-floc-based wastewater treatment pond – pilot scale

INPUTS			ecoinvent processes and flows (v2.2)	Value	Unit
Infrastructure	Pond	Steel	steel, low-alloyed, at plant	2.3E-01	kg
		Polyurethane	polyurethane, rigid foam, at plant	6.2E-03	kg
	Pumps	Propeller pump 1	modified process pump 40W, at plant	1.2E-02	item
		Propeller pump 2	modified process pump 40W, at plant	1.2E-02	item
		Influent pump	modified process pump 40W, at plant	8.7E-03	item
		Effluent pump	modified process pump 40W, at plant	8.7E-03	item
		Harvesting pump	modified process pump 40W, at plant	8.4E-03	item
	Others	Heating tubes	copper, at regional storage	2.9E-02	kg
		Boiler	gas boiler	5.4E-04	item
		Tubing	polyethylene, HDPE, granulate, at plant	5.4E-04	kg
	Settling tank	HDPE	polyethylene, HDPE, granulate, at plant	1.4E-02	kg
		Galvanized steel	steel, low-alloyed, at plant	1.4E-02	kg
Energy		Stirring pumps	electricity, low voltage, at grid (BE)	6.6E+01	kWh
		Influent pump	electricity, low voltage, at grid (BE)	7.4E-01	kWh
		Effluent pump	electricity, low voltage, at grid (BE)	6.6E-01	kWh
		Harvesting pump	electricity, low voltage, at grid (BE)	3.8E-01	kWh
		Heating of the pond	natural gas, burned in boiler condensing modulating <100kW	4.9E-01	MJ
Land occupation		Direct occupation of pond and settling tank	Occupation, industrial area	5.0E-01	m²
OUTPUTS			**ecoinvent processes and flows (v2.2)**	**Value**	**Unit**
Nutrients emission in the sewage system		Total Nitrogen	Nitrogen, total; water/unspecified	9.7E-03	kg
		Total Phosphorus	Phosphorus, total; water/unspecified	2.7E-04	kg

(2) MaB-floc-based wastewater treatment pond – up-scaled (L)

INPUTS			ecoinvent processes and flows (v2.2)	Value	Unit
Infrastructure	Pond	Plastic foil	polyethylene, HDPE, granulate, at plant	8.3E-06	kg
	Pumps	Propeller pumps (6)	modified process pump 40W, at plant	4.3E-02	item
		Influent pump	modified process pump 40W, at plant	6.0E-04	item
		Effluent pump	modified process pump 40W, at plant	1.2E-03	item
		Harvesting pump	modified process pump 40W, at plant	3.6E-03	item
		Flue gas blower	steel, low-alloyed, at plant	5.5E-04	item
	Others	Heating tubes	steel, low-alloyed, at plant	2.0E-02	kg
		Boiler	gas boiler	3.6E-05	item

	Tubing	polyethylene, HDPE, granulate, at plant	5.5E-05	kg
Settling tank	Plastic foil	polyethylene, HDPE, granulate, at plant	5.3E-07	kg
Energy	Stirring pumps	electricity, low voltage, at grid (BE)	2.1E+01	kWh
	Influent pump	electricity, low voltage, at grid (BE)	3.5E-01	kWh
	Effluent pump	electricity, low voltage, at grid (BE)	3.5E-01	kWh
	Harvesting pump	electricity, low voltage, at grid (BE)	2.2E-01	kWh
	Flue gas blower	electricity, low voltage, at grid (BE)	1.4E-01	kWh
	Heating of the pond	natural gas, burned in boiler condensing modulating <100kW	4.4E-01	MJ
Land occupation	Direct occupation of pond and settling tank	Occupation, industrial area	2.8E-01	m^2
OUTPUTS		**ecoinvent processes and flows (v2.2)**	**Value**	**Unit**
Nutrients emission in the sewage system	Total Nitrogen	Nitrogen, total; water/unspecified	8.7E-03	kg
	Total Phosphorus	Phosphorus, total; water/unspecified	2.1E-04	kg

(3) MaB-floc-based wastewater treatment pond – up-scaled (S)

The processes and values are the same than for the up-scaled plant (a), except the stirring pumps which consume now 4.8 kWh kg^{-1} TSS harvested in the MaB-floc liquor.

(4) MaB-floc-based wastewater treatment pond – up-scaled (E)

The processes and values are the same than for the up-scaled plant (a), except that the processes ‘electricity, low voltage, at grid (BE)’ are replaced by ‘electricity, at wind power plant (RER)’.

(5) MaB-floc-based wastewater treatment pond – up-scaled (M)

The amount of TSS harvested in the MaB-floc liquor is 199 kg. Therefore, the processes are the same than for the up-scaled plant (L) but the values are multiplied by 154/199 = 0.77, except for the electricity consumption of the harvesting pump. Indeed, the only parameter modified when MaB-floc productivity increases is the volume of water pumped from the reactor to the settling tank (and the volume of supernatant pumped back into the reactor). As the amount of harvested MaB-flocs increases as well, the energy consumed for harvesting is similar than with the actual productivity (0.22 kWh kg^{-1} TSS).

APPENDIX A6: Ecoinvent processes, flows and values used for the comparison of the three scenarios at industrial scale

All values are expressed per m^3 of water treated.

(1) Baseline scenario

INPUTS		ecoinvent processes and flows (v2.2)	Value	Unit
Anaerobic digestion	Silage maize	silage maize IP, at farm	1.7E+00	kg
	Water	tap water, at user	3.0E-01	m^3
	Iron sulphate	iron sulphate, at plant	3.0E-03	kg
	Heat	heat, natural gas, at boiler condensing modulating <100kW	0.0E+00	MJ
	Electricity	electricity, low voltage, at grid (BE)	5.9E-02	kWh
	Direct occupation of the digester	Occupation, industrial area	1.1E-04	m^2
	Infrastructure of the digester	See appendix A2		

OUTPUTS		ecoinvent processes and flows (v2.2)	Value	Unit
Nutrients emission in the sewage system	Total Nitrogen	Nitrogen, total; water/unspecified	1.9E-03	kg
	Total Phosphorus	Phosphorus, total; water/unspecified	1.3E-04	kg

AVOIDED PROCESSES		ecoinvent processes and flows (v2.2)	Value	Unit
Energy production from AD	Electricity	electricity mix BE	7.1E-01	kWh
	Heat	heat, natural gas, at boiler condensing modulating <100kW	2.4E+00	MJ
Soil conditioner	Compost	compost, at plant	4.9E-01	kg

(2) Scenario 1

INPUTS			ecoinvent processes and flows (v2.2)	System		Unit
				$Up_{L,shrimp\ feed}$	$Up_{SEM,shrimp\ feed}$	
Infrastructure	Pond	Plastic foil	polyethylene, HDPE, granulate, at plant	1.3E-06	1.3E-06	kg
	Pumps	Propeller pumps (6)	modified process pump 40W, at plant	6.6E-03	6.6E-03	item
		Influent pump	modified process pump 40W, at plant	9.2E-05	9.2E-05	item
		Effluent pump	modified process pump 40W, at plant	1.8E-04	1.8E-04	item
		Harvesting pump	modified process pump 40W, at plant	5.5E-04	5.5E-04	item
		Flue gas blower	steel, low-alloyed, at plant	8.5E-05	8.5E-05	item
	Others	Heating tubes	steel, low-alloyed, at plant	3.1E-03	3.1E-03	kg

	Boiler	gas boiler	5.6E-06	5.6E-06	item
	Tubing	polyethylene, HDPE, granulate, at plant	8.5E-06	8.5E-06	kg
Settling tank	Plastic foil	polyethylene, HDPE, granulate, at plant	8.1E-08	8.1E-08	kg
Energy	Stirring pumps	electricity, low voltage, at grid (BE)[a]	3.2E+00	0.7E00	kWh
	Influent pump	electricity, low voltage, at grid (BE) [a]	5.4E-02	5.4E-02	kWh
	Effluent pump	electricity, low voltage, at grid (BE) [a]	5.4E-02	5.4E-02	kWh
	Harvesting pump	electricity, low voltage, at grid (BE) [a]	3.3E-02	4.3E-02	kWh
	Flue gas blower	electricity, low voltage, at grid (BE) [a]	2.2E-02	2.2E-02	kWh
	Heating of the pond	natural gas, burned in boiler condensing modulating <100kW	0	0	MJ
	Filter press	electricity. low voltage. at grid (BE) [a]	8.4E-02	8.4E-02	kWh
Land occupation	Direct occupation of pond and settling tank	Occupation, industrial area	4.3E-02	4.3E-02	m^2
Anaerobic digestion	Silage maize	silage maize IP, at farm	1.7E+00	1.7E+00	kg
	Water	tap water, at user	5.8E-01	5.8E-01	kg
	Iron sulphate	iron sulphate, at plant	2.1E-03	2.1E-03	kg
	Heat	heat, natural gas, at boiler condensing modulating <100kW	0.0E+00	0.0E+00	MJ
	Electricity	electricity, low voltage, at grid (BE) [a]	5.9E-02	5.9E-02	kWh
	Direct occupation of the digester	Occupation, industrial area	1.1E-04	1.1E-04	m^2
	Infrastructure of the digester	See appendix A2			
Shrimp feed production	Drum dryer				
	Infrastructure	steel, low-alloyed, at plant	3.8E-04	3.8E-04	kg
	Natural gas	natural gas, burned in boiler condensing modulating <100kW	2.3E-01	9.1 E-01	MJ
	Mill				
	Infrastructure	steel, low-alloyed, at plant	1.4E-04	1.4E-04	kg
	Electricity	electricity, low voltage, at grid (BE) [a]	4.9E-03	4.9E-03	kWh
OUTPUTS		**ecoinvent processes and flows (v2.2)**	**System**		
			Up$_{L,shrimp\ feed}$	**Up$_{SEM,shrimp\ feed}$**	**Unit**
Nutrients emission in the sewage system	Total Nitrogen	Nitrogen, total; water/unspecified	1.3E-03	1.3E-03	kg
	Total Phosphorus	Phosphorus, total; water/unspecified	3.2E-05	3.2E-05	kg

AVOIDED PROCESSES		ecoinvent processes and flows (v2.2)	System $Up_{L,shrimp\ feed}$	$Up_{SEM,shrimp\ feed}$	Unit
Energy production from AD	Electricity	electricity mix BE	7.1E-01	7.1E-01	kWh
	Heat	heat, natural gas, at boiler condensing modulating <100kW	0.0E+00	0.0E+00	MJ
Soil conditioner	Compost	compost, at plant	4.9E-01	4.9E-01	kg
Wheat-based shrimp feed	Wheat production	wheat grains conventional, Barrois, at farm	6.8E-02	8.8E-02	kg
		wheat grains conventional, Castilla-y-Leon, at farm	6.8E-02	8.8E-02	kg
		wheat grains conventional, Saxony-Anhalt, at farm	6.8E-02	8.8E-02	kg
	Flour production (milling)	electricity, low voltage, at grid (BE)	1.2E-01	1.5 E-01	kWh

[a] *Replaced by 'electricity, at wind power plant (RER) when improvement E is implemented*

(3) Scenario 2

INPUTS			ecoinvent processes and flows (v2.2)	System $Up_{L,AD}$	$Up_{SEM,AD}$	Unit
Infrastructure	Pond	Plastic foil	polyethylene, HDPE, granulate, at plant	1.3E-06	1.3E-06	kg
	Pumps	Propeller pumps (6)	modified process pump 40W, at plant	6.6E-03	6.6E-03	item
		Influent pump	modified process pump 40W, at plant	9.2E-05	9.2E-05	item
		Effluent pump	modified process pump 40W, at plant	1.8E-04	1.8E-04	item
		Harvesting pump	modified process pump 40W, at plant	5.5E-04	5.5E-04	item
		Flue gas blower	steel, low-alloyed, at plant	8.5E-05	8.5E-05	item
	Others	Heating tubes	steel, low-alloyed, at plant	3.1E-03	3.1E-03	kg
		Boiler	gas boiler	5.6E-06	5.6E-06	item
		Tubing	polyethylene, HDPE, granulate, at plant	8.5E-06	8.5E-06	kg
	Settling tank	Plastic foil	polyethylene, HDPE, granulate, at plant	8.1E-08	8.1E-08	kg
Energy		Stirring pumps	electricity, low voltage, at grid (BE)[a]	3.2E+00	0.7E-01	kWh
		Influent pump	electricity, low voltage, at grid (BE)[a]	5.4E-02	5.4E-02	kWh
		Effluent pump	electricity, low voltage, at grid (BE)[a]	5.4E-02	5.4E-02	kWh
		Harvesting pump	electricity, low voltage, at grid (BE)[a]	3.3E-02	4.3E-02	kWh

	Flue gas blower	electricity, low voltage, at grid (BE) [a]	2.2E-02	2.2E-02	kWh
	Heating of the pond	natural gas, burned in boiler condensing modulating <100kW	0	0	MJ
	Filter press	electricity. low voltage. at grid (BE) [a]	8.4E-02	8.4E-02	kWh
Land occupation	Direct occupation of pond and settling tank	Occupation, industrial area	4.3E-02	4.3E-02	m^2
Anaerobic digestion	Silage maize	silage maize IP, at farm	2.6E+00	2.8 E+00	kg
	Water	tap water, at user	2.4E-01	6.6 E-01	kg
	Iron sulphate	iron sulphate, at plant	3.1E-03	3.1E-03	kg
	Heat	heat, natural gas, at boiler condensing modulating <100kW	0.0E+00	0.0E+00	MJ
	Electricity	electricity, low voltage, at grid (BE) [a]	8.7E-02	9.5E-02	kWh
	Direct land occupation of the digester	Occupation, industrial area	1.6E-04	1.9 E-04	m^2
	Infrastructure of the digester	See appendix A2			
OUTPUTS		**ecoinvent processes and flows (v2.2)**	**System**		
			Up*L,AD*	**Up*SEM,AD***	**Unit**
Nutrients emission in the sewage system	Total Nitrogen	Nitrogen, total; water/unspecified	1.3E-03	1.3E-03	kg
	Total Phosphorus	Phosphorus, total; water/unspecified	3.2E-05	3.2E-05	kg
AVOIDED PROCESSES					
Energy production from AD	Electricity	electricity mix BE	1.1E+00	1.2 E+00	kWh
	Heat	heat, natural gas, at boiler condensing modulating <100kW	3.5E+00	3.8 E+00	MJ
Soil conditioner from AD	Compost	compost, at plant	7.7E-01	8.8E-01	kg

[a] *Replaced by ‘electricity, at wind power plant (RER) when improvement E is implemented*

APPENDIX A7: Calculation of MaB-floc harvesting data for the pilot and up-scaled plants

(1) Pilot plant

The amount of MaB-flocs (TSS) in the pumped liquor is presented in Table A7-1 (based on Van Den Hende et al. (2014a)).

Table A7-1: *Calculation of the amount of TSS harvested in the MaB-floc liquor at pilot scale*

Period	**Period duration** *(days)*	**Total TSS pumped out of the reactor** *(g period^{-1})*	**TSS losses during settling**	**TSS in the MaB-floc liquor after settling** *(g period^{-1})*
4	9	2055	4.7 %	1958
5	11	4079	13.9 %	3512
6	28	2574	15.6 %	2172
7	42	26778	8.8 %	24422
8	52	23606	3.3 %	22827

The total amount of MaB-flocs harvested in the liquor is the average of the amount of TSS in the liquor for each period, weighted by their duration. It is 387 g TSS day^{-1}.

(2) Up-scaled plant

For the up-scaled plant, the amount of TSS in the MaB-floc liquor and the amount of TSS after dewatering need to be calculated.

The amount of biomass that has to be removed from the pond after one day corresponds to the sum of the daily biomass production in the pond and the losses occurring during settling (as these losses are pumped back into the pond): at the beginning of the day, the concentration of MaB-flocs is 0.5 g TSS L^{-1}, and should be the same at the beginning of the next day after harvesting. Thus, the amount of biomass withdrawn from the pond corresponds to the e_x factor in Table A7-3, and the amount of TSS in the MaB-floc liquor corresponds to the a_x factor. An average (weighted by each period duration) of 155 kg TSS harvested in the MaB-floc liquor per day was calculated. Because a HRT of 4 days was chosen, only data of periods with the same HRT is used (periods 4, 7 and 8).

To calculate the amount of dewatered MaB-flocs harvested in the integrated scenarios, the losses occurring during dewatering were withdrawn from the amount of MaB-flocs in the liquor. At pilot scale, MaB-flocs are dewatered in two steps: first they are filtered by gravity in a filter bag and then the bag is pressed in a hydropress (Van Den Hende et al. (2014a); not included in this study). The total losses during filtering and pressing are presented in Table A7-4. The losses are assumed the same when using a filter press (up-scaled scenario).

When the improvements are implemented, the calculation is the same with a_x factors increased by 30%.

Table A7-2: *Data used to calculate the MaB-floc production for the up-scaled scenario*

	Value	**Unit**	**Abbreviation**
Pond volume	97.85	m^3	*v*
Concentration of MaB-flocs maintained in the pond	0.5	g TSS L^{-1}	*c*

Table A7-3: *Calculation of the volume of water pumped out of the pond to harvest the right amount of MaB-flocs (data used for $Up_{L,shrimp\ feed}$ and $Up_{L,AD}$)*

Data from pilot scale[a]				**Data calculated for up-scaled plant**				
	Period duration	**Biomass production**	**Losses during settling**	**Biomass production**	**Biomass quantity after 1 day**	**Biomass concentration**	**Quantity of biomass removed after 1 day**	**Volume of water pumped out of the reactor**
	days	*mg TSS $L_{reactor}^{-1}$ day^{-1}*		*g TSS $pond^{-1}$ day^{-1}*	*g TSS $pond^{-1}$*	*g TSS m^{-3}*	*g TSS day^{-1}*	*m^3 day^{-1}*
				$p_x.v$	$v.c.1000 + a_x$	b_x/v	$a_x + (l_x/100).a_x$	e_x/d_x
Period 4	9	34.12 (p_1)	4.7% (l_1)	3339 (a_1)	52266 (b_1)	534 (d_1)	3496 (e_1)	6.5
Period 7	42	46.54 (p_2)	8.8% (l_2)	4554 (a_2)	53481 (b_2)	547 (d_2)	4955 (e_2)	9.1
Period 8	52	33.10 (p_3)	3.3% (l_3)	3239 (a_3)	52166 (b_3)	533 (d_3)	3346 (e_3)	6.3
							Average volume of water pumped *(weighted by periods duration)*	**7.4 m^3 day^{-1}**

[a] *Van Den Hende et al. (2014a)*

Table A7-4: *Calculation of the total dewatered biomass from the up-scaled plant (data used for $Up_{L,shrimp\ feed}$ and $Up_{L,AD}$)*

Data from pilot scale[a]			**Data calculated for up-scaled plant**
	Period duration	Total losses during dewatering *(filtering and pressing)*	Total dewatered biomass
	days		*g TSS day^{-1} $pond^{-1}$*
			a_x-$(tl_x/100)$*a_x
Period 4	9	1.2% (tl_1)	3301
Period 7	42	1.4% (tl_2)	4491
Period 8	52	0.4% (tl_3)	3226
		Average harvested biomass *(weighted by periods duration)*	**3749 g TSS day^{-1} $pond^{-1}$**

[a] *Van Den Hende et al. (2014a)*

APPENDIX A8: Estimation of fish sludge production

In order to estimate the quantity of fish sludge produced from the fish farm, data was collected at the pike perch farm of the Aquaculture Practice Centre of Inagro (Belgium), from where the aquaculture wastewater was coming from. The pikeperch center has an average stocking density of 35 kg fish m^{-3}. Two methods were used to calculate the amount of sludge produced.

- Method 1 : It is based on a value given by Gebauer (2004), assessing that 15-20% of the feed used in fish farms is recovered as sludge dry matter. Thus, as around 3500 kg of feed is used per year at Inagro, 1.7 kg of fish excretion are produced per day.
- Method 2: This method is based on a formula given by Lekang (2013):

$$Sludge\ DM\ (g\ kg_{fish}^{-1} day^{-1}) = 0.2 \times 10^{0.5 \times FCR}$$

where FCR = Feed Conversion Rate

The average FCR for the studied system was estimated to 1 (unpublished results from Inagro) so the quantity of sludge produced is estimated to 0.63 g kg_{fish}^{-1} day^{-1}. Knowing the stocking density (35 kg m^{-3}) and the volume of the fish tanks (84 m^3), the amount of sludge produced is estimated to 1.9 kg DM day^{-1}. The amount of fish sludge produced is calculated as the average of the results obtained from methods 1 and 2. It is estimated to 1.8 kg DM day^{-1}. To estimate the amount of sludge for the up-scaled scenarios of our study, assumed a fish farm releasing around 1000 m^3 of water per day (41 ponds treating each 24.5 m^3 of water per day). assume that the production of fish sludge is proportional to the amount of water released in the sewage system. This value was therefore first estimated for the fish farm of Inagro. Among the 174 m^3 of water circulating in the fish farm systems (84 m^3 in the fish tanks and 90 m^3 in the drum, piping, sump and moving bed biofilter), 5 to 10% is renewed per day. An average value of 7.5% is taken. Therefore, around 13 m^3 of water is pumped out of the fish farm and released in the sewage system per day (after particles settling). Based on these values, the amount of fish sludge produced by the theoretical up-scaled fish farm is estimated to 136 kg DM day^{-1}.

APPENDIX A9: Estimation of biochemical methane potentials and energy production from anaerobic digestion

(1) Biochemical methane potential

Biochemical methane potential of fish sludge

Data used to estimated the biochemical methane potential of fish sludge is expressed in liters of biogas per gram of COD (Mirzoyan et al., 2010).

First, the amount of wet sludge produced by the fish farm per day was calculated. The amount of dry sludge is given in appendix A8. Data used to make the calculation is presented in Table A9-1.

Table A9-1: *Dry content of fish sludge*

	Value	Source
Dry matter content of fish sludge (%)	5.3	Mirzoyan et al. (2010)[a]
	8.2	Gebauer (2004)
	10.2	Gebauer (2004)
	7.9	Average

[a] Average of data given in the paper

The quantity of wet fish sludge produced is therefore estimated to 1725 kg. Then, the COD content of fish sludge was estimated. Data found in literature is presented in Table A9-2.

Table A9-2: *COD content of fish sludge*

	Value	Source	Type of farm
COD content (g L^{-1})	60.3	Gebauer (2004)	saline fish farm
	74.1	Gebauer (2004)	saline fish farm
	75	Kugelman and van Gorder (1991)	striped bass farm
	95	Kugelman and van Gorder (1991)	striped bass farm
	78	Westerman et al. (1993)	trout farm
	113	Westerman et al. (1993)	trout farm
	110	Gebauer and Eikebrokk (2006)	salmon farm
	193	Gebauer and Eikebrokk (2006)	salmon farm
	99.8	Average	

A density of 1 kg L^{-1} was considered for fish sludge (likening fish sludge to pig manure based on Seydoux et al. (2008)). Therefore, the amount of COD brought to the digester is estimated to 172 169 kg day^{-1}.

The biogas and biochemical methane potentials of fish sludge found in Mirzoyan et al. (2010) are presented in Table A9-3.

Table A9-3: *Methane production from the digestion of fish sludge*

BMP ($Nl.g^{-1}$ COD)	Methane content in biogas (%)	Source
0.125-0.164	36-71	Gebauer and Eikebrokk (2006)
0.198-0.250	>80	Gebauer and Eikebrokk (2006)
0.114-0.184	49-58	Gebauer (2004)
0.14-0.151	59-61	Gebauer and Eikebrokk (2006)
0.02	30-60	Mirzoyan et al. (2008)
0.15	**56**	**Average**

Based on this data, estimated a production of 0.0263 Nm^3_{biogas} kg^{-1} of wet fish sludge and 0.0148 Nm^3_{CH4} kg^{-1} of wet fish sludge.

Biochemical methane potential of MaB-flocs

The calculation of the biochemical methane potential of MaB-flocs is based on measures made on harvested MaB-flocs during the operation of the pilot plant. An average value of 0.169 Nm^3_{CH4} kg^{-1} VS was calculated for periods 4 to 8 (average weighted by each period duration). A dry content of MaB-flocs of 18% after dewatering (through the belt filter press) was assumed and, based on the composition of dewatered and dried MaB-flocs (Van Den Hende et al., 2016), a VSS content of 7.4% in dewatered MaB-flocs was calculated. Therefore, 0.0125 Nm^3_{CH4} kg^{-1} of dewatered MaB-flocs is produced. The measured methane content of biogas from MaB-flocs is on average 67.6%v (Van Den Hende et al., 2014b). The biogas potential of MaB-flocs is therefore 0.0186 Nm^3_{biogas} kg^{-1} of dewatered MaB-flocs.

Biochemical methane potential of silage maize

Based on the UK Official Information Portal on Anaerobic Digestion, a biogas potential of 210 Nm^3 ton^{-1} of silage maize was assumed, with a methane content of 55% in biogas from silage maize (Hutnan et al., 2010) and therefore a methane production of 0.114 Nm^3 kg^{-1} of silage maize. A wet content of 64% in silage maize (communication from OWS, Belgium) was considered.

(2) Electricity and heat production from anaerobic digestion

The volumes, wet content, biogas and BMP of each feedstock used in for the baseline scenario, scenario 1 and scenario 2 are presented in Table A9-4 and Table A9-5.

Table A9-4: *Feedstock's characteristics and biogas production for Baseline scenario and Scenario 1*

	Feedstock quantities		**Wet content**	**Biogas potential (Nm^3 kg^{-1} fresh weight)**	**BMP (Nm^3 kg^{-1})**
Fish sludge	1.7	tons day^{-1}	92%	0.026	0.015
Silage maize	1.7	tons day^{-1}	64%	0.210	0.114
TOTAL	**3.5**	**tons day^{-1}**	**78%**	**0.118**	**0.065**
Biogas production	**408**	**Nm^3 biogas day^{-1}**			
Methane production	**223**	**Nm^3 CH_4 day^{-1}**			

Table A9-5: *Feedstock' characteristics and biogas production for Scenario 2*

	Feedstock quantities			**Wetcontent (%)**	**Biogas potential (Nm^3 kg^{-1})**	**CH_4 production (Nm^3 kg^{-1})**
MaB-flocs	0.9	1.1	tons day^{-1}	82	0.019	0.013
Fish sludge	1.7	1.7	tons day^{-1}	92	0.026	0.015
Silage maize	2.6	2.8	tons day^{-1}	64	0.210	0.114
TOTAL	**5.2**	**5.7**	**tons day^{-1}**	**76**	**0.117**	**0.064**
Biogas production	**603**	**661**	**Nm^3 $_{biogas}$ day^{-1}**			
Methane production	**332**	**364**	**Nm^3 $_{CH4}$ day^{-1}**			

Water was added to reach a wet content of 85% in the input feedstock (Braun et al., 2009).

Based on ecoinvent v2.2, electricity and heat production efficiencies of 32% and 55% in CHP were chosen. Using a calorific value of 35.8 MJ m^{-3} of methane (Nzila et al., 2010), the electricity and heat production from anaerobic digestion for each scenario was calculated (Table A9-6).

Table A9-6: *Electricity and heat production from anaerobic digestion for the 3 studied scenarios*

	Baseline scenario and scenario 1		
Electricity	709.7		kWh day^{-1}
Heat	4391.6		MJ day^{-1}
	Scenario 2		
	$Up_{L,AD}$	$Up_{SEM,AD}$	
Electricity	1054.8	1158.3	kWh day^{-1}
Heat	6526.7	7167.2	MJ day^{-1}

APPENDIX A10: Description of avoided processes

(1) Energy production avoided by anaerobic digestion

Electricity – All the electricity produced from biogas is delivered to the grid. It avoids electricity production (Belgian production mix). Therefore, based on Table I6, the production of 710 kWh (baseline scenario and scenario 1) and 1055 kWh to 1158 kWh (scenario 2) of electricity is avoided per day.

Heat – The produced heat is used on site. If heat remains after fulfilling the on-site needs, it is used to heat the fish tanks. This remaining heat avoids to produce heat from natural gas. Heat production and use for each scenario is detailed in Table A10-1.

Table A10-1: *Heat production, use and total avoided heat production for each scenario*

	Heat production from anaerobic digestion	Heat requirements		Total avoided heat production
		Purpose	Value	
Baseline scenario	4392 MJ day^{-1}	Heating of the digester	2021 MJ day^{-1}	2371 MJ day^{-1}
		Remaining heat	2371 MJ day^{-1}	
Scenario 1	4392 MJ day^{-1}	Heating of the digester	2021 MJ day^{-1}	0 MJ day^{-1}
		Heating of the pond	69 MJ day^{-1}	
		Drying	2537 MJ day^{-1}	
Scenario 2	6395 MJ day^{-1}	Heating of the digester	2928 MJ day^{-1}	3470 MJ day^{-1}
		Heating of the pond	69 MJ day^{-1}	

Remark: 0.98 MJ of gas are necessary to produce 1 MJ of heat (ecoinvent v2.2)

(2) Avoided production of wheat-based shrimp feed

Based on Van Den Hende et al. (2016), the assumption that MaB-flocs replace wheat in a typical shrimp diet is made: 154 kg to 199 kg TSS of MaB-flocs are produced per day in $Up_{L,shrimp\ feed}$ and $Up_{SEM,shrimp\ feed}$ respectively. Processes associated with the production of wheat-based shrimp feed are wheat production and flour production (milling). The yield of wheat flour production is 75% (Chambre d'agriculture d'Île-de-France, 2014) and on average, 0.21 kWh kg^{-1} of flour is necessary for milling wheat (Steerneman, 2013). Therefore, the production of 192 kg and 249 kg of wheat is avoided per day for $Up_{L,shrimp\ feed}$ and $Up_{SEM,shrimp\ feed}$ respectively.

(3) Avoided compost production

The calculation of the avoided amount of compost is based on a methodology proposed by Hermann et al. (2011) which calculates volume equivalences between soil conditioners based on their content of carbon contributing to humus formation, called humus factor. First, the amount of organic carbon in the digestate has to be calculated and multiplied by the humus factor of digestate (35%). The result gives the amount of organic carbon in the digestate contributing to humus formation. Then, knowing the humus factor of compost, the amount of compost replaced by the digestate can be calculated.

This section presents the calculations made for $Up_{L,shrimp\ feed}$ and $Up_{L,AD}$. The principle of the calculation is the same for $Up_{SEM,shrimp\ feed}$ and $Up_{SEM,AD}$, using the correct values for the amounts of co-digested MaB-flocs and silage maize.

Estimation of carbon in the digestate

The amount of carbon in the digestate is calculated as the difference between the amount of carbon in the feedstock and in the biogas.

a) Amount of carbon in feedstock

In fish sludge – The carbon content of fish sludge is calculated based on data presented in Table A10-2.

Table A10-2: *Calculation of C content in fish sludge*

C content (g kg^{-1} DM sludge)	**Source**	**Comment**
318	Mirzoyan et al. (2008)	Brackish aquaculture
429		
398		
250	Willett and Jakobsen (1986)	Trout farm
364	Stewart et al. (2006)	Trout farm
Average	**352 g C kg^{-1} DM sludge**	

In silage maize – The carbon content of silage maize based on its composition was calculated,

Table A10-3: *Composition of silage maize (communicated by the company OWS)*

Macronutrient	Mass fraction (%)
Water	64.1
Fat	1.4
Ash	1.3
Proteins	2.9
Carbohydrates	27.8

The amount of carbon in proteins and carbohydrates was based on Rouwenhorst et al. (1991), giving an equivalence of 0.53 g C g^{-1} protein and 0.44 g C g^{-1} carbohydrate.

The carbon content in the fat of silage maize was estimated based on its composition in fatty acids (Khan et al. (2012); Table A10-4).

Table A10-4: *Composition and carbon content of fat in silage maize*

Fatty acids	Comment on data from *Khan et al., 2012*	Assumed proportion *(simplified)* *(%)*	Molecular weight (g mol^{-1})	C content in fat from silage maize (g g^{-1})
C18:2n-6	0.52g.g^{-1} of total FA	50	278.4	0.39
C18:1cis-	2nd main fatty acid	25	282.5	0.19
C16:0	3rd main fatty acid	25	256.4	0.19
			Total	**0.77**

Table A10-5: *Total carbon content of silage maize*

	Baseline scenario + scenario 1	Scenario 2	Unit
Amount of silage maize	1.7	2.6	tons day^{-1}
Fat	24.3	36.4	kg day^{-1}
Proteins	50.7	75.8	kg day^{-1}
Carbohydrates	478.9	715.9	kg day^{-1}
Carbon content			
in fat	18.6	27.9	kg day^{-1}
in proteins	26.9	40.2	kg day^{-1}
in carbohydrates	210.7	315.0	kg day^{-1}
C content in silage	**256.2**	**383.1**	**kg day^{-1}**

In MaB-flocs – The protein, lipid and carbohydrate content was analysed (Table A10-6).

Table A10-6: *Composition of MaB-flocs*

	Content	Unit	Source
Dry weight	994.5	$g\ kg^{-1}$ MaB-flocs	Van Den Hende et al. (2016)
Crude proteins	209.9		
Crude lipids	33.9		
Ash	616.7		
Carbohydrates	134.0	$g\ kg^{-1}$ MaB-flocs	Dry weight – (protein content + lipid content + ash content)[a]

[a] *Rouwenhorst et al. (1991)*

As done previously, the carbon content of proteins and carbohydrates was calculated based on Rouwenhorst et al. (1991), giving an equivalence of 0.53 $g\ C\ g^{-1}$ protein and 0.44 $g\ C\ g^{-1}$ carbohydrate. The carbon content in the fat of MaB-flocs was estimated based on its composition in fatty acids (Van Den Hende et al., 2016). The result presented in Table A10-7 (23.86 $g\ C\ kg^{-1}$ MaB-flocs) is the amount of carbon present in 95% of the lipids. This estimation is then extrapolated to reach 100%.

Table A10-7: *Carbon content from lipids in MaB-flocs (from Van Den Hende et al. (2016))*

Lipid	Amount in MaB-flocs (g kg^{-1} MaB-flocs)	Molecular weight (g mol^{-1})	C content from lipids in MaB-flocs (g kg^{-1} MaB-flocs)
C12:0	0.37	200.32	0.27
C13:0	0.17	214.34	0.12
C14:0	1.36	228.37	1.00
C14:1	0.65	226.36	0.48
C15:0	2.48	242.4	1.84
C16:0	13.23	256.42	9.91
C17:0	0.71	270.45	0.54
C17:1	0.37	268	0.28
C18:0	1.56	284.48	1.18
C18:1c	4.73	282.46	3.62
C18:2(n-6) (LA)	2.41	278.43	1.87
C18:3(n-3) (ALA)	4.05	278.43	3.14
		TOTAL	**23.86**

Based on Table A10-8 and on a MaB-floc production of 154 kg TSS day^{-1}, a carbon supply from MaB-flocs to the digester of 28.2 kg day^{-1} was estimated.

Table A10-8: *Carbon content in MaB-flocs*

Carbon content in...	Value	Unit
Crude proteins	111.2	g C kg^{-1} MaB-flocs
Crude lipids	25.2	g C kg^{-1} MaB-flocs
Carbohydrates	58.96	g C kg^{-1} MaB-flocs
TOTAL	**195.8**	**g C kg^{-1} MaB-flocs**

Total – In total, the amount of organic carbon supplied by feedstock in the baseline scenario and scenario 1 is 304.1 kg C day^{-1} and 451.4 kg C day^{-1} in scenario 2.

a) Amount of carbon in biogas

The production of biogas and methane for each scenario is presented in Table A10-4 and Table A10-5. The amount of CO_2 can be calculated as the difference between the volumes of biogas and methane. Using a volume of 0.022 m^3 mol^{-1} of gas at standard temperature and pressure, an amount of 218.5 kg C day^{-1} in biogas was calculated for the baseline scenario and scenario 1 and of 323.1 kg C day^{-1} for scenario 2.

b) Carbon content in the digestate

The amount of carbon in the digestate is calculated as the difference between the carbon content in feedstock and the carbon content of the biogas. It is estimated to 85.6 kg C day^{-1} for the baseline scenario and scenario 1 and 137.9 kg C day^{-1} for scenario 2.

Estimation of compost equivalent

a) Estimation of the amount of carbon in digestate contributing to humus formation

The humus factor of digestate is 35% (Hermann et al., 2011). It means that in the digestate, 35% of the organic carbon contributes to humus formation in the soil. Thus, the amount of carbon contributing to humus formation for the three scenarios can be calculated.

Table A10-9: *Organic carbon contributing to humus formation in digestate*

Baseline scenario and scenario 1	Scenario 2	
30.0	48.3	kg C day^{-1}

b) Estimation of avoided compost production

In fresh compost, the amount of organic carbon contributing to humus formation is 61.2 g kg^{-1} of compost (Hermann et al., 2011). Thus, the amount of avoided compost was calculated by dividing the amount of organic carbon contributing in humus formation in the digestate by this value: a production of compost of 490 kg day^{-1} is avoided for the baseline scenario and scenario 1 and of 789 kg day^{-1} for scenario 2.

Appendix B: Supplementary material for Chapter 4

Appendix B1: Estimation of cow dung and rice straw potentials

Appendix B2: Calculation of the biogas potential

Appendix B3: Calculation of the amounts of replaced fuels

Appendix B4: Calculation of the nitrogen leaching factors

Appendix B5: Calculation of CH_4 field emissions due to rice straw remaining on the field

Appendix B6: Calculation of nitrogen emissions from the field

Appendix B7: Estimation of transport distances

Appendix B8: Substance balances for the current scenario

Appendix B9: Substance balances for the prospective scenario

Appendix B10: Life cycle inventory

Appendix B11: Local health impact assessment

Appendix B12: Exergy calculation

Appendix B13: Sankey diagrams for phosphorus

Appendix B1: Estimation of cow dung and rice straw potentials

a) Cow dung potential

The cow dung potential for anaerobic digestion in the prospective scenario is calculated as the sum of the amount of cow dung used as a fertilizer, as a cooking fuel and as feedstock for household digesters today in rural Chhattisgarh.

Cow dung used as fertilizer

In Chhattisgarh, 1921 kt of farmyard manure is applied per year (Agriculture Census Division, 2016). After storage in the pit, 64% of the manure remains available for application in the field (Reddy et al., 2010). Therefore, it can be estimated that **3002 kt of cow dung** is stored to be used as a fertilizer (fresh weight).

Cow dung used as a cooking fuel

Cow dung is used as a cooking fuel in the form of cow dung cakes, which are a mix of crop residues and fresh cow dung. The authors considered that crop residues represent 10% of the weight of cow dung cakes (authors' estimation). Based on Table 1 of Chapter 4, an amount of 504.9 kt of cow dung cakes is estimated to be consumed per year in rural Chhattisgarh. Based on the estimation of the dry content of fresh cow dung (Table B1-1) the amount of fresh cow dung used as a cooking fuel is estimated based on its dry content: 2234 kt of cow dung (fresh weight).

Table B1-1: *Estimation of the dry content of cow dung*

Dry content (%)	Source
25.0	IAEA (2008)
19.6	Ndayegamiye and Côté (1989)
21.0	Chukwuma and Orakwe (2014)
13.4	Liao et al. (2007)
14.5	Amon et al. (2007)
Average	***18.7%***

Cow dung used as feedstock for anaerobic digestion

Based on Table 1 of Chapter 4, 2944 tons of biogas is estimated to be consumed by rural households in Chhattisgarh. Pathak et al. (2009) estimated that 4400 kg of cow dung (dry weights) produces 2200 m^3 of biogas in a family size digester in India. Therefore, it can be estimated that **30 kt of cow dung** (fresh weight) is used during one year to produce biogas in rural Chhattisgarh.

b) Rice straw potential

Table B1-2: *Calculation of the rice straw potential for anaerobic digestion*

Data	Value	Unit	Source
Rice productivity *(a)*	1570	kg ha^{-1}	Pandey et al. (2016)
Area harvested *(b)*	$3.61x10^6$	ha $year^{-1}$	Pandey et al. (2016)
Production of rice *(c)*	$5.67x10^6$	t $year^{-1}$	*(a x b)/1000*
Residue to crop ratio for rice straw *(d)*	1.5	-	MNRE (2010)
Availability rate of rice straw *(e)*	10%	-	MNRE (2010)
Total amount of rice straw available for collection *(f)*	848883	t $year^{-1}$	*c x d x e*
Losses during rice straw collection *(g)*	18%		Mangaraj and Kulkarni (2011)
Total amount of rice straw collected	696084	t $year^{-1}$	*f – g x f*

Appendix B2: Calculation of the biogas potential

The calculation of the theoretical biogas potential of the mix of crop residues and cow dung is conducted according to Ranalli et al. 2007. This methodology is based on cellulose, hemicellulose, lipids and protein contents of each feedstock. Each of these constituents has an oxygen demand, i.e., a specific mass of O_2 needed to degrade it under aerobic conditions, as shown in the following equation:

$$\text{Organic matter} + O_2 \rightarrow CO_2 + H_2O$$

For example, we can calculate the oxygen demand of a carbohydrate with the structural formula $C_6H_{12}O_6$:

$$C_6H_{12}O_6 + 6O2 \rightarrow 6CO_2 + 6H_2O$$

The weight of 6 moles of O_2 in this reaction is 192 g. The weight of one mole of $C_6H_{12}O_6$ is 180g. The oxygen demand of 1g of $C_6H_{12}O_6$ is then 192/180=1.07 g.

Similarly, the oxygen demand of CH_4 can be calculated, which is 4 g g^{-1} of CH_4. Based on CH_4 volume occupation at given conditions, the volume of CH_4 per degraded oxygen demand and thus the volume of CH_4 theoretically produced by the oxygen demand degradation of each feedstock constituents can be estimated. The theoretical constituents oxygen demand equivalent, associated biogas yield and composition are presented in Table B2-1.

Table B2-1: *Oxygen demand, biogas yield and biogas composition per constituent of biomass*

Polymer	Structural formula	COD equivalent (g)	Biogas yield (L g^{-1})	% CH4	% CO2
1g Carbohydrates	$C_6H_{12}O_6$	1.07	0.75	0.5	0.5
1g Lipids	RCOOH	2.91	1.25	0.68	0.32
1g Proteins	$(C_4H_{1,6}O_{1,2})x$	1.5	0.7	0.71	0.29

For cow dung and each crop residue, the dry content, cellulose, hemicellulose, lipids and protein contents were found in literature (Table B2-2).

Table B2-2: *Rice straw and cow dung composition*

	% DM	Cellulose content	Hemicellulose content	Lipids content	Protein content	References
Rice straw	91.0%	31.0%	30.0%	0.0%	0.0%	*IRRI (2016), Sarnklong et al. (2010), Di Blasi et al. (1999)*
Cow dung	18.7%	23.6%	13.7%	3.2%	18.2%	*IAEA (2008), Ndayegamiye and Côté (1989), Chukwuma and Orakwe (2014), Liao et al. (2007), Amon et al. (2007), Chen et al. (2003)*

Appendix B3: Calculation of the amounts of replaced fuels

Considering a cooking stove efficiency of 55% (Singh & Gundimeda, 2014), the biogas produced in the prospective scenario is able to supply 6.68×10^{9} MJ of cooking energy per year. The energy supplied replaces energy supplied by other sources in the current scenario. First, as cow dung is used as feedstock for anaerobic digestion in the prospective scenario, it replaces the energy supplied by cow dung cakes (6.31×10^{8} MJ). Secondly, because the full potential of available cow dung is considered in the prospective scenario and thus also the cow dung which is used as feedstock for household digesters in the current scenario, the remaining energy supplied by biogas in the prospective scenario is assumed to substitute the energy supplied by biogas in the current scenario 2.87×10^{7} MJ. After substitution of cow dung cakes and biogas from household digesters, 6.02×10^{9} MJ remains. It is assumed to replace part of the energy supplied by firewood in the current scenario. Based on the calorific values and thermal efficiency of the cook stoves, the amount of replaced fuels can be estimated (Table B3-1).

Table B3-1: *Calculation of the amount of fuels replaced by biogas in the prospective scenario*

Cooking fuel	**Total heat requirements for cooking (MJ year^{-1})**	**Thermal efficiency of cook stoves^{b} (%)**	**Calorific value (MJ kg^{-1})^{c}**	**Quantity of fuel used in the current scenario (tons year^{-1})**	**Quantity of fuels replaced by biogas in the prospective scenario (tons year^{-1})**	**Quantity of fuel used in the prospective scenario (tons year^{-1})**
Firewood	1.32×10^{10}	18.0	13.9	5258519	2399356	2859162
Crop residues	1.29×10^{8}	11.0	12.8	91347	0	91347
Cow dung cake	6.31×10^{8}	10.5	11.9	504865	504865	0
Coal, lignite, charcoal	4.30×10^{7}	15.5	31.4	8837	0	8837
Kerosene	1.43×10^{7}	47.0	42.9	712	0	712
LPG	2.29×10^{7}	57.0	45.2	890	0	890
Biogas	2.87×10^{7}	55.0^{c}	17.7^{d}	2944	2944	0

[a] Census of India (2011); [b] Venkataraman et al. (2010); [c] Singh and Gundimeda (2014); [d] USEPA (2000)

Appendix B4: Calculation of the nitrogen leaching factors

The leaching factors are taken from IPCC (2006), which considers that 30% of nitrogen leaches from humid and irrigated lands and 0% of nitrogen leaches from dry lands. These factors are assigned to the areas considered as "humid and irrigated" and "dry" in Chhattisgarh. "Humid and irrigated lands" are lands subject to rainfall, irrigation and flooding. The percentage of areas classified as "humid and irrigated" is defined for the two main cropping seasons in Chhattisgarh: Kharif, during which most of the rice is cultivated and during which the rainy season occurs (June to October), and Rabi.

a) Leaching factor for manure, compost and synthetic fertilizers

Manure, compost and synthetic fertilizers are applied during both Kharif and Rabi seasons. It is assumed that 50% of fertilizers are applied during each cropping season.

During the Kharif season, crops are flooded and/or irrigated and/or un-irrigated. However, all crops are rain-fed as the rainy season occurs during the Kharif season. Therefore, the whole area is considered as a "humid and irrigated" land and the leaching factor for the Kharif season is 30%.

During the Rabi season, rice is not considered to be cultivated and there is no rainy period. Therefore, crops are irrigated or un-irrigated, but not flooded or rainfed. Agriculture Census Division (2016) reports that 27% of the cultivated area in Chhattisgarh is irrigated. This percentage is applied for the Rabi season. Therefore, the leaching factor during the Rabi season is 8.1%.

The leaching factor for the yearly amount of fertilizers applied during one year is thus 19.1%.

b) Leaching factor for rice straw left on the field

The leaching of nitrogen contained in rice straw only occurs during the Rabi season, i.e., after the Kharif season during which most rice is cultivated. Therefore, the leaching factor for rice straw is the same as for manure, compost and synthetic fertilizers during the Rabi season: 8.1%.

Appendix B5: Calculation of CH_4 field emissions due to rice straw remaining on the field

The area on which methane is emitted due to rice straw left on the field is estimated as the same as the percentage of rice straw remaining on the field (Table B5-1). Emissions from rice straw left on the field depend on the crop which follows rice cultivation. Frolking et al. (2006) evaluated the area of rice cropping systems in Madhya Pradesh (Table B5-3), before the state of Chhattisgarh was created and separated from Madhya Pradesh. The authors assumed that the shares of the different cropping systems areas are the same as measured by Frolking et al. (2006). The emissions of methane are based on Liu et al. (2016) and Zhang et al. (2015), who measured the difference of methane emissions with and without rice straw left on the field for a rice-fallow system and a rice-wheat system, respectively (

TableB5-4 and Table B5-5) In the prospective scenario, the same approach is followed to estimate CH_4 emissions from rice straw remaining in the field after collection.

Table B5-1: *Calculation of the area on which CH_4 from rice straw left on the field is emitted (current scenario)*

Surplus rice straw (available for other uses) *(a)*	10%	of total rice straw	*MNRE (2010)*
Surplus rice straw remaining on the field *(b)*	38%	of surplus rice straw	*Gadde et al. (2009)*
Rice straw remaining on the field *(c)*	4%	of total rice straw	*a x b*

Table B5-2: *Net area cultivated with rice in Chhattisgarh*

Net area cultivated with rice in Chhattisgarh (ha) *(e)*	3694151	Agriculture Census Division (2016)

Table B5-3: *Calculation of the CH_4 emissions from rice straw left on the field*

	Total area (based on Frolking et al. (2006)) *(d)*	% of total area with rice straw remaining on the field *(c x d = f)*	Area (ha) *(g = f x e)*	**CH_4-C emissions due to straw (kt year^{-1})**	**CH_4 emissions (kt year^{-1})**
upland	20.6%	0.8%	28548	**8.12** *(g x h)/10^6*	**1.1E+07**
rice-fallow	36.8%	1.4%	51013	**14.52** *(g x h)/10^6*	**1.9E+07**
rice-wheat	15.6%	0.6%	21589	**0.17** *(g x i)/10^6*	**2.3E+05**
rice-pulse	20.7%	0.8%	28743	**0.23** *(g x i)/10^6*	**3.0E+05**
rest	6.3%	0.2%	8689	**0.07** *(g x i)/10^6*	**9.2E+04**
TOTAL				**23.11**	**3.1E+07**

Table B5-4: *CH_4 emissions in a rice-fallow system in China - comparison between emissions from conventional NPK treatment and NPK treatment with rice straw mulching (Liu et al., 2016)*

	CH_4-C emissions	
NPK	109.51	kg ha^{-1} year^{-1}
NPK + rice straw mulching	394.06	kg ha^{-1} year^{-1}
Difference *(h)*	284.55	kg ha^{-1} year^{-1}

Table B5-5: *CH_4 emissions in a rice-wheat system in China (Zhang et al., 2015)*

	Wheat season - removal of rice straw		Wheat season - returning of rice straw		Unit
	CH4-C (2012-2013)	CH4-C (2013-2014)	CH4-C (2012-2013)	CH4-C (2013-2014)	
Conventional tillage	5.26	4.39	11.98	13.56	kg ha^{-1}
Average	4.83		12.77		kg ha^{-1}
Difference *(i)*	7.94				kg ha^{-1}

Appendix B6: Calculation of nitrogen emissions from the field

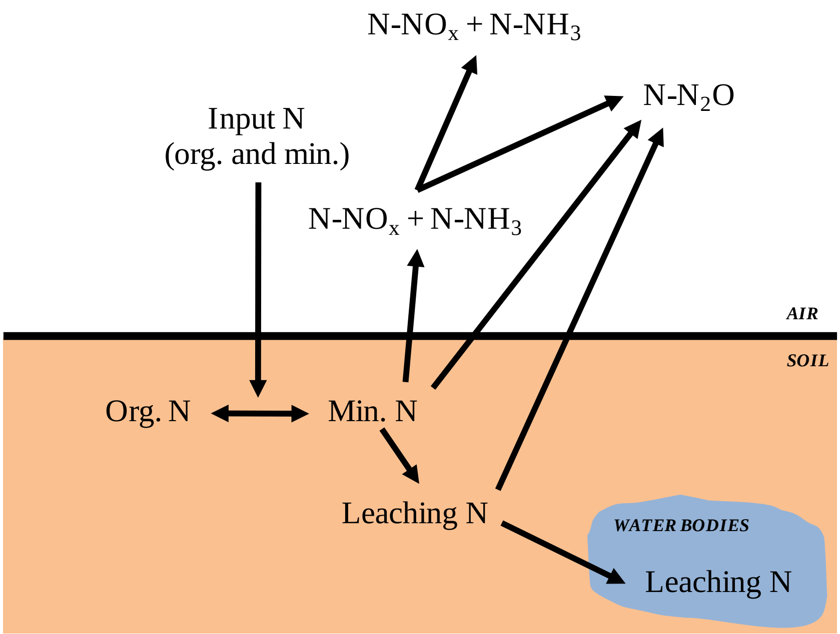


Figure B6-1: *Nitrogen pathways from field application to the compartments air, soil and water bodies (based on IPCC (2006)) (Min.: mineralized; Org.: organic)*

Table B6-2: *Emission factor for direct and indirect N_2O emissions from field application of manure*

	Emissions from manure, compost and crop residues	**Emissions from synthetic fertilizers**	**Unit**
Emissions of NO_x-N and NH_3-N via volatilization	0.2	0.1	kg N emitted.kg N input^{-1}
Indirect emissions of N_2O-N	0.01	0.01	kg N emitted.kg (NO_x-N + NH_3-N) emitted^{-1}
Direct N_2O-N emissions	0.02	0.02	kg N emitted.kg N input^{-1}
Direct N_2O-N emissions from flooded rice	0.03	0.03	kg N emitted.kg N input^{-1}
Indirect N_2O-N emissions from leachage/run-off	0.0075	0.0075	kg N emitted.kg N leaking/run-off^{-1}
Leaching/run-off factor for dry lands	0%	0%	-
Leaching/run-off factor for humid and irrigated lands	30%	30%	-

Table B6-3: *Total N field emissions per type of material applied/left on the field in the current and prospective scenarios*

	Current scenario				Prospective scenario		
	Manure	Rice straw left on the field	Synthetic fertilizers	Burnt rice straw	Compost	Synthetic fertilizers	Rice straw left on the field
NOx-N + NH_3-N	8.45E-01	3.71E-01	3.31E+01	6.79E-02	1.83E+00	3.31E+01	1.78E-01
NOx-N + NH_3-N after conversion to N_2O-N	8.37E-01	3.68E-01	3.27E+01	6.72E-02	1.81E+00	3.28E+01	1.76E-01
Indirect N_2O-N emissions	8.45E-03	3.71E-03	3.31E-01	6.79E-04	1.83E-02	3.31E-01	1.78E-03
Direct N_2O-N emissions	2.93E-02	1.86E-02	2.29E+00	3.40E-03	3.60E-02	2.30E+00	8.90E-3
Direct N_2O-N emissions from flooded rice	3.89E-03	0.00E+00	3.04E-01	0.00E+00	4.79E-03	3.05E-01	0.00E+00
Indirect N_2O-N emissions from leachage	6.04E-03	1.13E-03	4.72E-01	2.06E-04	1.30E-02	4.73E-01	5.41E-04
Total N emissions	8.84E-01	3.91E-01	3.61E+01	7.15E-02	1.88E+00	3.62E+01	1.87E-01

Appendix B7: Estimation of transport distances

a) From shop to house

One shop is considered to be in the center of a square in which the clients of the shop are located. One shop selling cooking fuels for 4 villages is considered here. Based on the number of villages, the state of Chhattisgarh is divided into squares of 28 km^2. Then, the average distance of any point from the square center is estimated. Random coordinates of 5000 points within a 1x1 km square are generated and the distance between each point and the center of the square is calculated. An average distance of 0.38 km was found. Secondly. a tortuosity factor of 3 was applied (Wright & Brown, 2007). The obtained value is multiplied by the square root of 28 km^2. The average distance of households to buy fossil fuels is thus 6 km.

b) From field to digester

One digester is considered to be in the center of a square in which the farmers' fields are located. Based on the number of villages in the state, Chhattisgarh is divided into squares of 6.9 km^2. Following the same methodology as above, the average distance of one field to one digester is found to be 3 km.

Appendix B8: Substance balances for the current scenario

a) Transport fuels

Input

	C	1.12	*kt year^{-1}*	*Based on fuel consumption of processes in the inventory (see section B10) and considering C and N content of diesel and fuel oil (Phyllis2, 2012)*
	N	1.3E-04	*kt year^{-1}*	

Output

Air emissions			*Considering that C is only emitted in the form of CO_2 and CO and N in the form of NOx. Emissions based on the ecoinvent database (Frischknecht & Rebitzer, 2005)*
CO_2-C	1.12	*kt year^{-1}*	
CO-C	3.87E-03	*kt year^{-1}*	
NOx-N	1.3E-04	*kt year^{-1}*	

b) Surplus rice straw management

	Rice straw composition (% dry weight)	Reference
Carbon	39.5	*Jusoh et al. (2013); Oh and Park (2002)*
Nitrogen	0.64	*Jusoh et al. (2013)*
Phosphorus	0.21	*Jusoh et al. (2013)*
Potassium	1.20	*Jusoh et al. (2013)*

Table B8-1: *Carbon, nitrogen, phosphorus and potassium content of rice straw*

Rice straw burnt on the field:

Input

Total amount of rice straw burnt		482.26	kt DW year^{-1}	
Rice straw burnt		429.21	kt DW year^{-1}	*Burn efficiency ratio: 89% (Kanabkaew & Kim Oanh, 2010)*
	C	*169.5*	*kt year^{-1}*	*Based on Table B8-1* Table
	N	*2.75*	*kt year^{-1}*	
	P	*0.90*	*kt year^{-1}*	
	K	*4.81*	*kt year^{-1}*	
Unburnt rice straw left on the field		53.05	kt DW year^{-1}	*Idem*
	C	*20.95*	*kt year^{-1}*	*Idem*
	N	*0.34*	*kt year^{-1}*	
	P	*0.11*	*kt year^{-1}*	

	K	*0.59*	*kt year^{-1}*	

Output from rice straw burning

From burnt rice straw	Air emissions			
	C	*163.5*	*kt year^{-1}*	*Based on emission factors presented in Table 4 of Chapter 4*
	N	*2.75*	*kt year^{-1}*	*Based on Dobermann and Fairhurst (2002): Losses of ▪ 100% N ▪ 25% P ▪ 35% K*
	P	*0.23*	*kt year^{-1}*	
	K	*1.68*	*kt year^{-1}*	
	Ashes			
	C	*6.06*	*kt year^{-1}*	*Difference between Inputs and Outputs*
	N	*0.00*	*kt year^{-1}*	
	P	*0.68*	*kt year^{-1}*	
	K	*3.12*	*kt year^{-1}*	
From unburnt rice straw left on the field	Air emissions			
	CH_4-C	*4.22*	*kt year^{-1}*	*Same methodology followed as described in section B5*
	Other C emissions from respiration	*12.33*	*kt year^{-1}*	*Input C - CH_4-C emissions - Org. C*
	N emissions	*0.07*	*kt year^{-1}*	*See* Table B6-3
	Soil emissions			
	N	*0.01*	*kt year^{-1}*	*Mineralized N/(1-leaching factor for crop residues) – Mineralized N – N emitted from leachate (see sections B4 and B6)*
	P	*0.01*	*kt year^{-1}*	*Input P x leaching factor (5%; Hokazono and Hayashi (2012))*
	K	*0.02*	*kt year^{-1}*	*Input K x leaching factor (3%;Phong et al. (2011))*
	Remaining in the soil			
	Organic C	*4.40*	*kt year^{-1}*	*Inputs C x Humus factor of straw (21%; Hermann et al. (2011))*
	Mineralized N	*0.17*	*kt year^{-1}*	*Input N x mineralization factor (50%; Gabrielle and Gagnaire (2008))*
	Organic N	*0.08*	*kt year^{-1}*	*Input N – Emissions – mineralized N*
	P	*0.11*	*kt year^{-1}*	*Input P – P in leachate*

K	*0.58*	*kt year^{-1}*	*Input K – K in leachate*

Output from ashes remaining in the field			
Soil emissions			
P	*0.03*	*kt year^{-1}*	*P in ashes x leaching factor (5%; Hokazono and Hayashi (2012))*
K	*0.09*	*kt year^{-1}*	*K in ashes x leaching factor (3%;Phong et al. (2011))*
Remaining in the soil			
C in ashes	*6.06*	*kt year^{-1}*	*Substances in ashes - emissions*
P	*0.64*	*kt year^{-1}*	
K	*3.03*	*kt year^{-1}*	

Rice straw remaining in the field:

Input			
Rice straw left on the field	290.22	kt DW year^{-1}	*Based on Table B6-3*
C	*114.64*	*kt year^{-1}*	
N	*1.86*	*kt year^{-1}*	
P	*0.61*	*kt year^{-1}*	
K	*3.25*	*kt year^{-1}*	

Output			
Air emissions			
CH_4-C	*23.11*	*kt year^{-1}*	*See section B5*
Other C emissions from respiration	*67.46*	*kt year^{-1}*	*Input C – emissions – Org. C*
N emissions	*0.39*	*kt year^{-1}*	*See Table B6-3*
Soil emissions			
N	*0.08*	*kt year^{-1}*	*Mineralized N/(1-leaching factor for crop residues) – Mineralized N – N emitted from leachate (see sections B4 and B6)*
P	*0.03*	*kt year^{-1}*	*P in ashes x leaching factor (5%; Hokazono and Hayashi (2012))*
K	*0.10*	*kt year^{-1}*	*K in ashes x leaching factor (3%;Phong et al. (2011))*
Remaining in the soil			
Organic C	*24.07*	*kt year^{-1}*	*Inputs C x Humus factor of straw (21%; Hermann et al. (2011))*
Mineralized N	*0.93*	*kt year^{-1}*	*Input N x mineralization factor (50%; Gabrielle and Gagnaire (2008))*

	Organic N	*0.46*	*kt year^{-1}*	*Input N – Emissions – mineralized N*
	P	*0.58*	*kt year^{-1}*	*Input P – P in leachate*
	K	*3.15*	*kt year^{-1}*	*Input K – K in leachate*

c) Cow dung management

	Cow dung composition (% dry weight)	Reference
Carbon	43.60	*Chukwuma and Orakwe (2014); Vijayaraghavan et al. (2014)*
Nitrogen	1.17	*Reddy et al. (2010)*
Phosphorus	0.23	*Reddy et al. (2010)*
Potassium	0.98	*Reddy et al. (2010)*

Table B8-2: *Carbon, nitrogen, phosphorus and potassium content of cow dung*

Carbon	46.0%	wt
Nitrogen	0.5%	wt
Phosphorus	558.0	mg kg^{-1}
Potassium	8668.0	mg kg^{-1}

Table B8-3: *Carbon, nitrogen, phosphorus and potassium content of crop residues (Phyllis2, 2012).*

Cow dung used as fuel:

Preparation and drying of cow dung cakes

Input

	Fresh cow dung	2233.81	kt WW year^{-1}	*See section 0 (chapter)*
	C	*182.30*	*kt year^{-1}*	*Based on Table B8-2*
	N	*4.89*	*kt year^{-1}*	
	P	*0.96*	*kt year^{-1}*	
	K	*4.10*	*kt year^{-1}*	
	Crop residues (10%)	50.49	kt WW year^{-1}	*10% of the cow dung cakes (weight basis)*
	C	*23.22*	*kt year^{-1}*	*Based on Table B8-3*
	N	*0.26*	*kt year^{-1}*	
	P	*0.03*	*kt year^{-1}*	
	K	*0.44*	*kt year^{-1}*	

Output			
Air emissions			
CH_4-C	*0.53*	*kt $year^{-1}$*	*Based on Maeda et al. (2013); 2g CH_4 kg^{-1} of sun dried feces*
N_2O-N	*0.10*	*kt $year^{-1}$*	*Based on Maeda et al. (2013); 20 g kg^{-1} N of sun dried feces*
NH_3-N	*0.57*	*kt $year^{-1}$*	*Based on Laubach et al. (2013); N losses from deposited cow dung: 12%*
Sun dried cow dung cakes	504.87	kt $year^{-1}$	
C	*204.99*	*kt $year^{-1}$*	*Inputs - Emissions*
N	*4.48*	*kt $year^{-1}$*	
P	*0.99*	*kt $year^{-1}$*	
K	*4.53*	*kt $year^{-1}$*	

Combustion of the cow dung cakes

Output			
Air emissions			
C	*164.15*	*kt $year^{-1}$*	*Based on the emission factors presented in Table 3 of Chapter 4*
N	*4.48*	*kt $year^{-1}$*	
P	*0.17*	*kt $year^{-1}$*	*Input P – P in ashes*
K	*0.009*	*kt $year^{-1}$*	*Based on Sen et al. (2014): PM from cow dung combustion contains 18.23 mg K^+ kg cow $dung^{-1}$*
Ashes			
C	*40.83*	*kt $year^{-1}$*	*Inputs - Emissions*
N	*0.00*	*kt $year^{-1}$*	
P	*0.82*	*kt $year^{-1}$*	*Based on Wang et al. (2015): 80-85% of P of biofuels burnt in boilers is retained in bottom ashes*
K	*4.53*	*kt $year^{-1}$*	*Input K - Emissions*

Dumping of the ashes

Output			
Leaching			
P	*0.04*	*kt $year^{-1}$*	*P in ashes x leaching factor (5%; Hokazono and Hayashi (2012))*
K	*0.14*	*kt $year^{-1}$*	*K in ashes x leaching factor (3%;Phong et al. (2011))*
Remaining in the soil			
C	*40.83*	*kt $year^{-1}$*	*Inputs - Emissions*
N	*0.00*	*kt $year^{-1}$*	

P	*0.78*	*kt year⁻¹*	
K	*4.39*	*kt year⁻¹*	

Cow dung used as a fertilizer:

Storage in the cow dung pit

Input

Cow dung	3001.9	kt WW $year^{-1}$	*See section 0 (chapter)*
C	*245.0*	*kt $year^{-1}$*	*Based on Table B8-2*
N	*6.6*	*kt $year^{-1}$*	
P	*1.29*	*kt $year^{-1}$*	
K	*5.51*	*kt $year^{-1}$*	

Output

Air emissions			
CH_4-C	*5.4*	*kt $year^{-1}$*	*Based on Gupta et al. (2007); 6.6 mg CH_4 kg $dung^{-1}$ day^{-1}*
Other C-emissions	*87.0*	*kt $year^{-1}$*	*Based on average from Sommer (2001) and Vu et al. (2015); 36.6% of initial C*
N_2O-N	*0.10*	*kt $year^{-1}$*	*Based on Pardo et al. (2015): emission of 1.5% and 12.5% of N as N_2O-N and NH_3-N, respectively, from stored unturned organic waste*
NH_3-N	*0.82*	*kt $year^{-1}$*	
Other N emissions	*0.11*	*kt $year^{-1}$*	*Total N emissions (Pardo et al., 2015) – soil emissions*
Soil emissions			
N	*1.31*	*kt $year^{-1}$*	*Based on Reddy et al. (2010); losses from cow dung pit in India via leaching:* ▪ *20% N* ▪ *30% P* ▪ *50% K*
P	*0.39*	*kt $year^{-1}$*	
K	*2.75*	*kt $year^{-1}$*	
Cow dung			
C	*152.61*	*kt $year^{-1}$*	*Inputs - Emissions*
N	*4.23*	*kt $year^{-1}$*	
P	*0.90*	*kt $year^{-1}$*	
K	*2.75*	*kt $year^{-1}$*	

Cow dung application

Output			
Air emissions			
C from respiration	*99.19*	*kt year^{-1}*	*Input C – Org. C*
N emissions	0.88	*kt year^{-1}*	*See Table B6-3*
Soil emissions			
N	*0.30*	*kt year^{-1}*	*Mineralized N/(1-leaching factor for manure) – Mineralized N – N emitted from leachate (see sections B4 and B6)*
P	*0.05*	*kt year^{-1}*	*P in manure x leaching factor (5%; Hokazono and Hayashi (2012))*
K	*0.08*	*kt year^{-1}*	*K in manure x leaching factor (3%;Phong et al. (2011))*
Remaining in the soil			
Organic carbon	*53.4*	*kt year^{-1}*	*Inputs C x Humus factor of manure (35%; Hermann et al. (2011))*
Mineralized N	*1.29*	*kt year^{-1}*	*NH_4^+-N + Mineralized org. N; based on Chowdhury et al. (2014) (19.2% of N in the form of NH_4^+ in composted manure) and Martínez-Blanco et al. (2013) (14% of N mineralized during the first year of application)*
Organic nitrogen	*1.75*	*kt year^{-1}*	*Input N – Mineralized N - Emissions*
P	*0.86*	*kt year^{-1}*	*Input P - Emissions*
K	*2.67*	*kt year^{-1}*	*Input K - Emissions*

Cow dung used as feedstock in household digesters:

Same methodology as in section B9.

d) Cooking fuels use (except cow dung cakes)

Cooking fuels combustion:

Input			
Firewood	*4.84E+03*	kt DW year^{-1}	*See Table 1 of Chapter 4, considering a moisture content of 7.98%*
C	*2842.27*	*kt year^{-1}*	*C emitted/85%; based on Bhattacharya et al. (2002): 85% of C in firewood is emitted into the air*
N	*26.99*	*kt year^{-1}*	*Composition based on Phyllis2 (2012)*
P	*3.62*	*kt year^{-1}*	
K	*17.66*	*kt year^{-1}*	
Crop residues	*9.13E+01*	kt WW year^{-1}	*See Table 1 of Chapter 4*

C	42.02	kt $year^{-1}$	*Composition based on Phyllis2 (2012)*
N	0.47	kt $year^{-1}$	
P	0.05	kt $year^{-1}$	
K	0.79	kt $year^{-1}$	
Other fuels			*See Table 1 of Chapter 4*
C	9.80	kt $year^{-1}$	*Equal to C emissions*
N	0.0	kt $year^{-1}$	*Authors assumption: 0% of nutrients in fossil fuels and biogas*
P	0.0	kt $year^{-1}$	
K	0.0	kt $year^{-1}$	

Output			
Air emissions			
C	2464.28	kt $year^{-1}$	*Based on emissions factors presented in Table 3 of Chapter 4*
N	27.46	kt $year^{-1}$	
P	0.64	kt $year^{-1}$	*Input P – P in ashes*
K	0.0	kt $year^{-1}$	*Based on Sen et al. (2014): PM from cow dung combustion contains 18.23 mg K^{+} kg cow $dung^{-1}$*
Ashes			
C	429.82	kt $year^{-1}$	*Input C - Emissions*
N	0.0	kt $year^{-1}$	*Input N - Emissions*
P	3.03	kt $year^{-1}$	*Based on Wang et al. (2015): 80-85% of P of biofuels burnt in boilers is retained in bottom ashes*
K	18.44	kt $year^{-1}$	*Input K - Emissions*

Dumping of the ashes:

Output			
Leaching			
P	0.15	kt $year^{-1}$	*P in ashes x leaching factor (5%; Hokazono and Hayashi (2012))*
K	0.55	kt $year^{-1}$	*K in ashes x leaching factor (3%;Phong et al. (2011))*
Remaining in the soil			
C	429.82	kt $year^{-1}$	*Inputs - Emissions*
N	0.00	kt $year^{-1}$	
P	2.87	kt $year^{-1}$	
K	17.89	kt $year^{-1}$	

e) Application of the synthetic fertilizers

Input

	N	*330.57*	*kt year^{-1}*	*Input fertilizer in Chhattisgarh; based on Agriculture Census Division (2016)*
	P	*162.38*	*kt year^{-1}*	
	K	*56.07*	*kt year^{-1}*	

Output

Air emissions			
N emissions	*36.13*	*kt year^{-1}*	*See Table B6-3*
Soil emissions			
N	*62.50*	*kt year^{-1}*	*Input N x leaching factor for synthetic fertilizers - N emitted from leachate (see sections B4 and B6)*
P	*8.12*	*kt year^{-1}*	*P inputs x leaching factor (5%; Hokazono and Hayashi (2012))*
K	*1.68*	*kt year^{-1}*	*K inputs x leaching factor (3%;Phong et al. (2011))*
Remaining in the soil			
N	*231.95*	*kt year^{-1}*	*Inputs - Emissions*
P	*154.26*	*kt year^{-1}*	
K	*54.39*	*kt year^{-1}*	

Appendix B9: Substance balances for the prospective scenario

a) Transport fuels

Input

C	*0.91*	*kt year^{-1}*	*Based on fuel consumption of processes in the inventory (see section B10) and considering C and N content of diesel and fuel oil (Phyllis2, 2012)*
N	*9.95E-05*	*kt year^{-1}*	

Output

Air emissions			*Considering that C is only emitted in the form of CO_2 and CO and N in the form of NOx. Emissions based on the ecoinvent database (Frischknecht & Rebitzer, 2005)*
CO_2-C	*0.90*	*kt year^{-1}*	
CO-C	*3.19E-03*	*kt year^{-1}*	
NOx-N	*9.95E-05*	*kt year^{-1}*	

b) Rice straw left in the field

Input

Rice straw left on the field	152.80	kt DW year^{-1}	*Considering that 18% of available rice straw is left in the field after collection (Mangaraj & Kulkarni, 2011)*
C	*54.92*	*kt year^{-1}*	
N	*0.89*	*kt year^{-1}*	
P	*0.29*	*kt year^{-1}*	
K	*1.56*	*kt year^{-1}*	

Output

Air emissions			
CH_4-C	*14.76*	*kt year^{-1}*	*See section B5*
Other C emissions from respiration	*28.63*	*kt year^{-1}*	*Input C – emissions – Org. C*
N emissions	*0.19*	*kt year^{-1}*	*See Table B6-3*
Soil emissions			
N	*0.04*	*kt year^{-1}*	*Mineralized N/(1-leaching factor for crop residues) – Mineralized N – N emitted from leachate (see sections B4 and B6)*
P	*0.01*	*kt year^{-1}*	*P in ashes x leaching factor (5%; Hokazono and Hayashi (2012))*
K	*0.05*	*kt year^{-1}*	*K in ashes x leaching factor (3%;Phong et al. (2011))*
Remaining in the soil			
Organic C	*11.53*	*kt year^{-1}*	*Inputs C x Humus factor of straw (21%; Hermann et al. (2011))*

Mineralized N	*0.45*	*kt year^{-1}*	*Input N x mineralization factor (50%; Gabrielle and Gagnaire (2008))*
Organic N	*0.22*	*kt year^{-1}*	*Input N – Emissions – mineralized N*
P	*0.28*	*kt year^{-1}*	*Input P – P in leachate*
K	*1.51*	*kt year^{-1}*	*Input K – K in leachate*

c) Co-digestion of rice straw and cow dung

Input

Cow dung			
C	*429.76*	*kt year^{-1}*	*Sum of C, N, P and K from cow dung used as fertilizer, cooking fuel and feedstock for household digesters today (see section B8)*
N	*11.53*	*kt year^{-1}*	
P	*2.27*	*kt year^{-1}*	
K	*9.66*	*kt year^{-1}*	
Rice straw			
C	*250.21*	*kt year^{-1}*	*Sum of C, N, P and K in surplus rice straw (see section B8)*
N	*4.05*	*kt year^{-1}*	
P	*1.33*	*kt year^{-1}*	
K	*7.09*	*kt year^{-1}*	

Output

Biogas to distribution	6.21E+08	m^3 year^{-1}	*Biogas produced – fugitive emissions. See section B2 for the estimation of the biogas potential.*
CO_2-C	*136.91*	*kt year^{-1}*	
CH_4-C	*167.63*	*kt year^{-1}*	
Fugitive emissions			
CO_2-C	*9.60*	*kt year^{-1}*	*Based on Bruun et al. (2014): between 3.1% and 10% of fugitive emissions from inlet and outlet pipes → estimation of 6.6% in this case*
CH_4-C	*11.75*	*kt year^{-1}*	
Digestate			
C	*354.1*	*kt year^{-1}*	*Inputs – Substances in biogas – Substances in fugitive emissions*
N	*15.6*	*kt year^{-1}*	
P	*3.60*	*kt year^{-1}*	
K	*16.80*	*kt year^{-1}*	

d) Biogas distribution

Output

Fugitive emissions			
CO_2-C	*0.96*	*kt year^{-1}*	*Based on Evangelisti et al. (2015): 0.7% of biogas losses during distribution in pipelines*
CH_4-C	*1.17*	*kt year^{-1}*	

	Biogas			
	CO_2-C	135.95	kt year^{-1}	Inputs – Fugitive emissions
	CH_4-C	166.46	kt year^{-1}	

e) Management of the digestate

Drying of the digestate:

Air emissions			
CH_4-C	3.06	kt year^{-1}	Based on Amon et al. (2006)
CO_2-C	18.53	kt year^{-1}	Based on Amon et al. (2006)
N_2O-N	0.02	kt year^{-1}	Based on Rehl and Müller (2011)
NH_3-N	6.18	kt year^{-1}	Based on Amon et al. (2006)

Composting of the digestate:

Output			
Air emissions			
CH_4-C	8.14	kt year^{-1}	*Difference between CH4-C emissions from storage manure in the pit and composting of the digestate = Difference between CH4-C between stored and turned composting of organic waste = 11% more initial C emitted as CH_4-C when the compost is turned (Pardo et al., 2015)*
Other C-emissions	148.31	kt year^{-1}	*Difference between CO_2-C emissions from storage manure in the pit and composting of the digestate = Difference between CO_2-C between stored and turned composting of organic waste = 26% more initial C emitted as CO_2-C when the compost is turned (Pardo et al., 2015)*
N_2O-N	0.11	kt year^{-1}	*Difference between N_2O-N emissions from storage manure in the pit and composting of the digestate = Difference between N_2O-N between stored and turned composting of organic waste = 20% less initial C emitted as N_2O-N when the compost is turned (Pardo et al., 2015)*
NH_3-N	1.97	kt year^{-1}	*Difference between NH_3-N emissions from storage manure in the pit and composting of the digestate = Difference between NH_3-N between stored and turned composting of organic waste = 68% more initial C emitted as NH_3-N when the compost is turned (Pardo et al., 2015)*

Other N emissions	*1.86*	*kt year^{-1}*	*(Total N losses x Input N) - N_2O-N - NH_3-N – N leached* *Total N losses when the compost of organic waste is turned: 45% Pardo et al. (2015)*
Soil emissions			
N	*0.24*	*kt year^{-1}*	*Leeaching from covered composting of sild manure (Sommer, 2001):* ▪ *2.6% N* ▪ *1.7% P* ▪ *8.2% K*
P	*0.06*	*kt year^{-1}*	
K	*1.37*	*kt year^{-1}*	
Compost			
C	*176.05*	*kt year^{-1}*	*Input substances - Emissions*
N	*5.20*	*kt year^{-1}*	
P	*3.36*	*kt year^{-1}*	
K	*15.38*	*kt year^{-1}*	

Field application of the compost:

Output

Air emissions			
C from respiration	*86.26*	*kt year^{-1}*	*Input C – Org. C*
N emissions	*1.09*	*kt year^{-1}*	*See Table B6-3*
Soil emissions			
N	*0.36*	*kt year^{-1}*	*Mineralized N/(1-leaching factor for manure) – Mineralized N – N emitted from leachate* *(see sections B4 and B6)*
P	*0.18*	*kt year^{-1}*	*P inputs x leaching factor (5%; Hokazono and Hayashi (2012))*
K	*0.46*	*kt year^{-1}*	*K inputs x leaching factor (3%;Phong et al. (2011))*
Remaining in the soil			
Organic carbon	*89.78*	*kt year^{-1}*	*Inputs C x Humus factor of compost (51%; Hermann et al. (2011))*
Mineralized N	*1.56*	*kt year^{-1}*	*NH_4^+-N + Mineralized org. N; based on Chowdhury et al. (2014) (18.6% of N in the form of NH_4^+ in composted digestate) and Martínez-Blanco et al. (2013) (14% of N mineralized during the first year of application)*
Organic nitrogen	*2.19*	*kt year^{-1}*	*Input N - Emissions – Mineralized N*
P	*3.36*	*kt year^{-1}*	*Inputs - Emissions*
K	*14.92*	*kt year^{-1}*	

f) Application of the synthetic fertilizers

Input			
N	*328.90*	*kt year^{-1}*	*(Total mineralized N in current scenario - Mineralized N from compost in prospective scenario)/(1- %N lost from NOx and NH_3 emissions - % direct N-N_2O losses from dry land * % dry land - % direct N-N_2O losses from wet land * % wet land - N leaching factor) See sections B4 and B6*
P	*160.86*	*kt year^{-1}*	*(Total P made available for crops in current scenario - P made available from compost in prospective scenario)/(1- P leaching factor)*
K	*48.89*	*kt year^{-1}*	*(Total K made available for crops in current scenario - K made available from compost in prospective scenario)/(1- K leaching factor)*
Output			
Air emissions			
N emissions	*36.19*	*kt year^{-1}*	*See Table B6-3*
Soil emissions			
N	*62.61*	*kt year^{-1}*	*Input N * Leaching factor - Indirect N emissions from leachage*
P	*8.04*	*kt year^{-1}*	*P inputs x leaching factor (5%; Hokazono and Hayashi (2012))*
K	*1.47*	*kt year^{-1}*	*K inputs x leaching factor (3%;Phong et al. (2011))*
Available for plants			
N	*232.34*	*kt year^{-1}*	*Inputs - Emissions*
P	*152.82*	*kt year^{-1}*	
K	*47.42*	*kt year^{-1}*	

g) Cooking fuels use

Cooking fuels combustion:

Input			
Firewood	2.86E+03	kt DW year^{-1}	*See section B3, considering a moisture content of 7.98%*
C	*1545.40*	*kt year^{-1}*	
N	*14.7*	*kt year^{-1}*	
P	*1.97*	*kt year^{-1}*	
K	*9.60*	*kt year^{-1}*	
Crop residues	9.13E+01	kt WW year^{-1}	*See section B3*
C	*42.02*	*kt year^{-1}*	

	N	0.47	kt $year^{-1}$	
	P	0.05	kt $year^{-1}$	
	K	0.79	kt $year^{-1}$	
	Biogas			
	C	302.41	kt $year^{-1}$	See section B3
	Other fuels			
	C	8.50	kt $year^{-1}$	
	N	0	kt $year^{-1}$	
	P	0	kt $year^{-1}$	
	K	0	kt $year^{-1}$	
Output				
	Air emissions			
	C	1359.76	kt $year^{-1}$	Same methodology as for the current scenario
	N	15.14	kt $year^{-1}$	
	P	0.35	kt $year^{-1}$	
	K	0.01	kt $year^{-1}$	
	Ashes			
	C	236.16	kt $year^{-1}$	Inputs - Emissions
	N	0	kt $year^{-1}$	
	P	1.66	kt $year^{-1}$	
	K	10.39	kt $year^{-1}$	

Dumping of ashes:

Output				
	Leaching			
	P	0.08	kt $year^{-1}$	P inputs x leaching factor (5%; Hokazono and Hayashi (2012))
	K	0.31	kt $year^{-1}$	K inputs x leaching factor (3%;Phong et al. (2011))
	Remaining in the soil			
	C	236.16	kt $year^{-1}$	Inputs - Emissions
	N	0.00	kt $year^{-1}$	
	P	1.58	kt $year^{-1}$	
	K	10.08	kt $year^{-1}$	

Appendix B10: Life cycle inventory

Table B10-1: *Life cycle inventories of the current and prospective scenarios.*

			Amount	
	Cooking fuels production and use (both scenarios)		**Current scenario**	**Prospective scenario**
Cooking fuels production	Wood	Wood from Tropical and subtropical dry broadleaf forest	4.84E+09	2.86E+09
	Coal, lignite, charcoal	market for hard coal briquettes, alloc rec, U	2.77E+08	2.77E+08
	Kerosene	market for kerosene, alloc rec, U	7.12E+05	7.12E+05
	LPG	market for liquefied petroleum gas, alloc rec, U	8.90E+05	8.90E+05
	Electricity	market for electricity, low voltage, alloc rec, U	0.00E+00	0.00E+00
	Biogas	no burden	1.16E+05	0.00E+00
	Cow dung cakes (drying)	CH_4	7.07E+05	0.00E+00
		N_2O	2.89E+05	0.00E+00
		NH_3	1.29E+06	0.00E+00
Transport of coal to household	By motocycle, scooter, moped	transport, passenger, motor scooter, alloc rec, U	4.52E+06	4.52E+06
	By car	transport, passenger car, EURO 3, alloc rec, U	3.25E+04	3.25E+04
Transport of kerosene to household	By motocycle, scooter, moped	transport, passenger, motor scooter, alloc rec, U	6.51E+04	6.51E+04
	By car	transport, passenger car, EURO 3, alloc rec, U	4.03E+04	4.03E+04
Transport of LPG to household	By motocycle, scooter, moped	transport, passenger, motor scooter, alloc rec, U	1.15E+05	1.15E+05
	By car	transport, passenger car, EURO 3, alloc rec, U	6.66E+03	6.66E+03
	Wood	CO	2.25E+08	1.22E+08

		CH_4 fossil	0.00E+00	0.00E+00
		CH_4 biogenic	5.87E+07	3.19E+07
		NMVOC	5.00E+07	2.72E+07
		NOx	1.07E+06	5.81E+05
		N_2O	4.73E+05	2.57E+05
		PM	5.59E+07	3.04E+07
		CO_2 fossil	0.00E+00	0.00E+00
		CO_2 biogenic	8.16E+09	4.44E+09
		SO_2	4.48E+06	2.43E+06
Direct emissions from cooking fuels	Crop residues	CO	5.99E+06	5.99E+06
		CH_4 fossil	0.00E+00	0.00E+00
		CH_4 biogenic	6.94E+05	6.94E+05
		NMVOC	7.76E+05	7.76E+05
		NOx	4.34E+05	4.34E+05
		N_2O	4.57E+03	4.57E+03
		PM	1.91E+06	1.91E+06
		CO_2 fossil	0.00E+00	0.00E+00
		CO_2 biogenic	1.19E+08	1.19E+08
		SO_2	5.11E+04	5.11E+04
	Cow dung	CO	2.01E+07	0.00E+00
		CH_4 fossil	0.00E+00	0.00E+00
		CH_4 biogenic	2.27E+06	0.00E+00
		NMVOC	1.22E+07	0.00E+00
		NOx	4.19E+05	0.00E+00
		N_2O	1.51E+05	0.00E+00

		PM	5.38E+06	0.00E+00
		CO_2 fossil	0.00E+00	0.00E+00
		CO_2 biogenic	5.28E+08	0.00E+00
		SO_2	1.77E+06	0.00E+00
	Coal	CO	2.43E+06	2.43E+06
		CH_4 fossil	6.98E+04	6.98E+04
		CH_4 biogenic	0.00E+00	0.00E+00
		NMVOC	9.28E+04	9.28E+04
		NOx	2.92E+04	2.92E+04
		N_2O	2.12E+03	2.12E+03
		PM	5.43E+05	5.43E+05
		CO_2 fossil	2.13E+07	2.13E+07
		CO_2 biogenic	0.00E+00	0.00E+00
		SO_2	4.55E+03	4.55E+03
	Kerosene	CO	4.41E+04	4.41E+04
		CH_4 fossil	7.61E+02	7.61E+02
		CH_4 biogenic	0.00E+00	0.00E+00
		NMVOC	1.35E+04	1.35E+04
		NOx	1.93E+03	1.93E+03
		N_2O	7.11E+01	7.11E+01
		PM	5.23E+02	5.23E+02
		CO_2 fossil	2.09E+06	2.09E+06
		CO_2 biogenic	0.00E+00	0.00E+00
		SO_2	2.13E+01	2.13E+01
	LPG	CO	1.33E+04	1.33E+04

		CH_4 fossil	4.45E+01	4.45E+01
		CH_4 biogenic	0.00E+00	0.00E+00
		NMVOC	1.67E+04	1.67E+04
		NOx	2.70E+03	2.70E+03
		N_2O	1.33E+02	1.33E+02
		PM	9.77E+02	9.77E+02
		CO_2 fossil	2.74E+06	2.74E+06
		CO_2 biogenic	0.00E+00	0.00E+00
		SO_2	0.00E+00	0.00E+00
	Biogas	CO	5.74E+03	1.47E+06
		CH_4 fossil	0.00E+00	0.00E+00
		CH_4 biogenic	2.95E+03	7.56E+05
		NMVOC	1.67E+03	4.27E+05
		NOx	2.61E+03	6.66E+05
		N_2O	2.79E+02	7.14E+04
		PM	1.55E+03	3.95E+05
		CO_2 fossil	0.00E+00	0.00E+00
		CO_2 biogenic	4.25E+06	1.09E+09
		SO_2	1.56E+02	4.00E+04
Rice straw management			**Current scenario**	**Prospective scenario**
Direct emissions from rice straw burning		CO	4.21E+07	0.00E+00
		CH_4 fossil	0.00E+00	0.00E+00
		CH_4 biogenic	1.03E+07	0.00E+00
		NMVOC	3.64E+06	0.00E+00

	NOx	1.12E+06	0.00E+00
	N_2O	4.05E+04	0.00E+00
	PM	5.87E+06	0.00E+00
	CO_2 fossil	0.00E+00	0.00E+00
	CO_2 biogenic	5.81E+08	0.00E+00
	NH_3	8.08E+04	0.00E+00
	SO_2	1.93E+05	0.00E+00
Direct emissions from rice straw left on the field	CO	0.00E+00	0.00E+00
	CH_4 fossil	0.00E+00	0.00E+00
	CH_4 biogenic	3.08E+07	1.48E+07
	NMVOC	0.00E+00	0.00E+00
	NOx	7.97E+03	3.82E+03
	N_2O	3.68E+04	1.76E+04
	PM	0.00E+00	0.00E+00
	CO_2 fossil	0.00E+00	0.00E+00
	CO_2 biogenic	0.00E+00	0.00E+00
	NH_3	4.42E+05	2.12E+05
	SO_2	0.00E+00	0.00E+00
Use of cow dung as a fertilizer (current scenario)		**Current scenario**	**Prospective scenario**
Land occupation of the pit	Land occupation	3.75E+04	0.00E+00
Direct emissions from manure pit (and digestate from household digesters drying)	CO_2 biogenic	3.19E+08	0.00E+00
	CH_4	7.26E+06	0.00E+00
	N_2O	1.24E+06	0.00E+00
	NH_3	1.26E+06	0.00E+00

Transport and application of manure (and digestate from household digesters) to the field	Solid manure loading and spreading, by hydraulic loader and spreader GLO, market for, alloc rec, U	1.90E+09	0.00E+00
Direct emissions from manure (and digestate from household digesters) application in the field	NOx	1.83E+04	0.00E+00
	NH_3	1.02E+06	0.00E+00
	N_2O	5.27E+04	0.00E+00
Indirect emissions from manure (and digestate from household digesters) application in the field	N_2O	2.30E+04	0.00E+00
Co-digestion of cow dung and rice straw (prospective scenario)		**Current scenario**	**Prospective scenario**
Mechanical collection of rice straw in the field – mechanized cutting followed by mowing of straw and balling	mowing. by motor mower. alloc rec. U	0.00E+00	4.51E+04
	fodder loading. by self-loading trailer. alloc rec. U [(1)]	0.00E+00	2.64E+06
Mechanical collection of rice straw in the field – combine harvesting (i.e., only balling)	fodder loading. by self-loading trailer. alloc rec. U [(1)]	0.00E+00	2.64E+06
Transport of rice straw to digester	transport, freight, lorry 16-32 metric ton, EURO3, alloc rec, U [(2)]	0.00E+00	1.90E+06
Pre-treatment of rice straws	market for electricity, low voltage, alloc rec, U	0.00E+00	2.37E+04
Electricity for mixing of the digester	market for electricity, low voltage, alloc rec, U	0.00E+00	1.05E+08
Land occupation of the digester	Land occupation	0.00E+00	3.62E+05
Water for the digester	Tap water, market for, Alloc Rec, U	0.00E+00	4.89E+09
Injection of biogas in pipelines	market for electricity, low voltage, alloc rec, U	0.00E+00	8.28E+07
Direct emissions from CH_4 fugitive emissions	CH_4	0.00E+00	1.80E+07
Drying of the digestate (prospective scenario)		**Current scenario**	**Prospective scenario**
Land occupation of the slurry drying beds	Land occupation	0.00E+00	3.09E+06

Direct emissions from digestate drying	CO_2 biogenic	0.00E+00	6.79E+06
	CH_4	0.00E+00	4.08E+06
	N_2O	0.00E+00	2.59E+04
	NH_3	0.00E+00	7.51E+06
Composting and application of the digestate (prospective scenario)		**Current scenario**	**Prospective scenario**
Direct emissions from digestate composting	CO_2 biogenic	0.00E+00	5.44E+08
	CH_4	0.00E+00	1.08E+07
	N_2O	0.00E+00	6.84E+05
	NH_3	0.00E+00	2.18E+06
Land occupation composting	land use	0.00E+00	7.89E+04
Transport and application of the compost to field	Solid manure loading and spreading, by hydraulic loader and spreader (Global Footprint Network), market for, alloc rec, U	0.00E+00	5.38E+08
Direct emissions from composted digestate application in the field	NOx	0.00E+00	2.23E+04
	NH_3	0.00E+00	1.24E+06
	N_2O	0.00E+00	6.41E+04
Indirect emissions from composted digestate application in the field	N_2O	0.00E+00	2.80E+04
Production and application of synthetic fertilizers (both scenarios)		**Current scenario**	**Prospective scenario**
Synthetic fertilizers	market for nitrogen fertiliser, as N, alloc rec, U	3.31E+08	3.31E+08
	market for phosphate fertiliser, as P2O5, alloc rec, U	3.72E+08	1.61E+08
	market for potassium fertiliser, as K2O, alloc rec, U	6.76E+07	4.89E+07

	application of plant protection product, by field sprayer (GLO)	8.07E+04	8.07E+04
Direct emissions from synthetic fertilizers application in the field	NOx	7.10E+05	7.11E+05
	NH_3	3.93E+07	3.94E+07
	N_2O	4.07E+06	4.08E+06
Indirect emissions from synthetic fertilizers application in the field	N_2O	1.26E+06	1.26E+06

(1) The diesel consumption of this process was changed based on Silalertruksa and Gheewala (2013); (2) The diesel consumption of this process was changed based on Soam et al. (2017)

Appendix B11: Local health impact assessment

To calculate the health impact from local PM_{10} emissions on the population of Chhattisgarh, the amount of inhaled PM_{10} per person should be estimated and multiplied by the effect and damage factors from the ReCiPe method (Goedkoop et al., 2013). This amount is estimated by calculating the percentage of emitted PM_{10} which is inhaled by the local population. PM_{10} is mainly emitted by two sources: the combustion of cooking fuels and the burning of rice straw.

a) Inhalation of PM_{10} from the combustion of cooking fuels

Ansari et al. (2010) measured the concentrations of $PM_{2.5}$ and PM_{10} in rural homes during cooking and non-cooking periods in India.

Table B11-1: *Mean concentrations of $PM_{2.5}$ and PM_{10} from the combustion of plant material and cow dung (based on Ansari et al. (2010)). Note that Ansari et al. (2010) did not measure the emissions from the combustion of cow dung alone, but together with plant material. The PM emissions from cow dung are estimated as the difference between the emissions from combustion of plant material and cow dung together and the emissions from the combustion of plant material alone.*

	$PM_{2.5}$	PM_{10}	
Plant material − cooking period	1.19	3.95 *(a)*	mg m^{-3}
Mean Plant material − non cooking period	0.23	0.67 *(b)*	mg m^{-3}
Mean cow dung − cooking period	1.19	4.23 *(c)*	mg m^{-3}
Mean cow dung − non cooking period	0.26	0.67 *(d)*	mg m^{-3}

Table B11-2: *Data used to calculate the amount of inhaled PM_{10} during the cooking periods*

Volume inhaled *(e)*	0.018	m^3 min^{-1}	Respiratory volume per person: 13 m^3 day^{-1} (van Zelm et al., 2008). Assumption of the authors: 2 persons are targeted by the emissions
Duration of the cooking period *(f)*	210	min day^{-1}	3 to 4 hours of cooking per day (Ansari et al., 2010)
Volume inhaled during the cooking period *(g)*	3.79	m^3 day^{-1} inhaled during the cooking periods	*e x f*
Amount of PM_{10} inhaled from the combustion of plant material *(h)*	12.4	mg PM inhaled day^{-1}	*g x (a – b)*
Amount of PM_{10} inhaled from the combustion of cow dung *(i)*	13.5	mg PM inhaled day^{-1}	*g x (c – d)*

The percentage of PM_{10} emitted which is inhaled is then calculated based on the PM emission factors from plant material (considered as wood) and cow dung provided in literature.

Table B11-3: *Estimation of the average rate of inhaled PM_{10}*

	Firewood	Cow dung cake	
Emission factor for $PM_{2.5}$	3.2 *(j)*	3.0 *(k)*	g kg^{-1} fuel (Bhattacharya et al. (2002) and Venkataraman et al. (2010)
$PM_{2.5}$/PM_{10} ratio	30.1% *(l)*	28.1% *(m)*	Based on Table B11-1
Emission factor for PM_{10}	10.6 *(n)*	10.7 *(o)*	g kg^{-1} fuel (*j/l* and *k/m*)
	18.9 *(p)*	38.2 *(q)*	g $meal^{-1}$ $household^{-1}$ Based on Table 1 of Chapter 4, considering 2 meals per day
Percentage of inhaled PM_{10}	0.033% *(r)*	0.035% *(s)*	*r = (h/1000)/(2 x p)* *s = (i/1000)/(2 x q)*
Average	0.0034%		Average between *r* and *s*

Table B11-4: *Estimation of the amount of PM_{10} inhaled due to cooking fuels combustion (kg $year^{-1}$). $PM_{2.5}$ emissions are taken from Table 3 of Chapter 4, PM_{10} emissions are calculated based on the $PM_{2.5}/PM_{10}$ ratio presented in Table B11-3 and the amount of PM_{10} inhaled is calculated by multiplying the PM_{10} emitted by the average rate of inhaled PM_{10} presented in Table B11-5*

		Wood	**Crop residues**	**Cow dung**	**Coal**	**Kerosene**	**LPG**	**Biogas**	**TOTAL**
Current scenario	$PM_{2.5}$ emitted	5.59E+07	1.91E+06	5.38E+06	5.43E+05	5.23E+02	9.77E+02	1.55E+03	6.37E+07
	PM_{10} emitted	1.92E+08	6.56E+06	1.85E+07	1.86E+06	1.80E+03	3.35E+03	5.31E+03	2.19E+08
	PM_{10} inhaled	6.53E+04	2.23E+03	6.30E+03	6.35E+02	6.12E-01	1.14E+00	1.81E+00	**7.45E+04**
Prospective scenario	PM2.5	3.04E+07	1.91E+06	0.00E+00	5.43E+05	5.23E+02	9.77E+02	3.95E+05	3.32E+07
	PM10	1.04E+08	6.56E+06	0.00E+00	1.86E+06	1.80E+03	3.35E+03	1.36E+06	1.14E+08
	PM10 inhaled	3.55E+04	2.23E+03	0.00E+00	6.35E+02	6.12E-01	1.14E+00	4.62E+02	3.88E+04

b) Inhalation of PM_{10} from the burning of rice straw in the field

Table B11-5: *Calculation of PM_{10} inhaled due to rice straw burning*

Concentration of PM10 during burning of rice straw due to rice straw in Chhattisgarh *(a)*	149	µg m^{-3}	Nirmalkar and Deb (2016)
Inhalation volume *(b)*	13	m^3 day^{-1} $person^{-1}$	van Zelm et al. (2008)
Duration of burning season *(c)*	21	days (own assumption: 3 weeks)	Authors' assumption based on Nirmalkar and Deb (2016)
Number of households in rural Chhattisgarh *(e)*	4312213	households	Census of India (2011)
Average members per household *(f)*	4.58		Census of India (2011)
PM_{10} inhaled due to rice straw burning	4.07E-05	kg PM_{10} $person^{-1}$	$g = (a \times b \times c)/10^9$
	8.03E+02	kg PM_{10} $year^{-1}$	$g \times e \times f$

c) Effect and damage factors

Table B11-6: *Effect x Damage factors applied in this study (Goedkoop et al., 2013)*

Chronic mortality	57.59	yr kg^{-1}
Acute mortality	0.21	yr kg^{-1}
Acute respiratory morbidity	0.02	yr kg^{-1}
Acute cardiovascular morbidity	0.02	yr kg^{-1}
TOTAL	**57.84**	**yr kg^{-1}**

Appendix B12: Exergy calculation

a) Wood

The chemical exergy of wood was estimated based on Alvarenga et al. (2015) and considered as a mix of leaves, wood, grass and shrub from tropical and subtropical dry broadleaf forest, grasslands, savannas, and shrublands. The exergy content of firewood is estimated to be 20.05 MJ kg DM^{-1}.

b) Crop residues, cow dung and fossil fuels

The chemical exergy ex_{ch} of crop residues (including rice straw), cow dung and fossil fuels was calculated using equation 1.

$$ex_{ch} = \beta \times LHV \tag{1}$$

Where LHV is the Lower Heating Value of the material and β is the exergy-to-energy ratio of the material. β depends on its elementary composition and is calculated following equation 2, 3 or 4 (Szargut, 2005).

- For solid CHON compounds:

$$\beta\left(\text{for } \frac{O}{C} < 0.5\right) = 1.0347 + 0.014 \times \frac{H}{C} + 0.0968 \times \frac{O}{C} + 0.0493 \times \frac{N}{C} \tag{2}$$

$$\beta\left(\text{for } \frac{O}{C} < 2\right) = \frac{1.044 + 0.016 \times \frac{H}{C} - 0.3493 \times \frac{O}{C} \times \left(1 + 0.053\frac{H}{C}\right) + 0.0493\frac{N}{C}}{1 - 0.4124 \times \frac{O}{C}} \tag{3}$$

- For liquid CHOS compounds:

$$\beta = 1.047 + 0.0154 \times \frac{H}{C} + 0.0562 \times \frac{O}{C} + 0.5904 \times \frac{S}{C} \times \left(1 - 0.175 \times \frac{H}{C}\right) \tag{4}$$

The composition and LHV of each material was collected from the Phyllis database (Phyllis2, 2012).

c) Digestate and compost

During digestion and composting, the temperature of the feedstock being processed increases and therefore, their exergy is calculated as the sum of their chemical and physical exergy. The chemical exergy of the digestate and the compost is calculated based on equation 1. The physical exergy of the digestate and the compost is calculated based on equation 5.

$$ex_{ph} = \left|c_p \times \left[(T-T_0)-T_0 \times \ln\frac{T}{T_0}\right]\right| + |v \times (P-P_0)| \qquad (5)$$

Where c_p is the heat capacity of the considered substrate, T and P are the temperature and the pressure of the substrate, T_0 and P_0 are the temperature and pressure of the reference environment and v is the specific volume of the substrate.

d) Biogas

The exergy content of the biogas is calculated as the sum of the chemical, physical and mixing exergy of the biogas.

The chemical exergy of CO_2 and CH_4 is retrieved from Szargut (2005).

The physical exergy of each of the gas in the biogas is calculated based on equation 6.

$$ex_{ph} = \left|c_p \times \left[(T-T_0)-T_0 \times \ln\frac{T}{T_0}\right]\right| + \left|R \times T_0 \times \ln\frac{P}{P_0}\right| \qquad (6)$$

Where c_p is the heat capacity of the considered gas, T and P are the temperature and the pressure of the gas, T_0 and $P_0$are the temperature and pressure of the reference environment and R is the gas constant.

The mixing exergy of the biogas is calculated based on equation 7.

$$ex_{mix} = R \times T_0 \times \sum_i x_i \times \ln x_i \qquad (7)$$

Where R is the gas constant, $T_0$is the temperature of the reference environment and x_i is the molar fraction of the gas in the biogas.

e) Synthetic fertilizers

The exergy content of the synthetic fertilizers was calculated based on the Gibbs free energy of formation (Szargut, 2005).

$$ex_{ch} = \Delta G_f^0 + \sum_k v_k \times ex_{ch,\,k} \qquad (8)$$

With ΔG_f^0 the standard Gibbs free energy of formation of the compound and v_k and $ex_{ch,k}$ respectively the amount of moles and specific chemical exergy of products and reactants k.

f) Output products

The system produces three outputs products: cooking energy, organic carbon and nutrients (N, P and K).

The exergy of the cooking energy is calculated using equation 9.

$$\beta = 1 - \frac{T_0}{T} \qquad (9)$$

Where β is the exergy-to-energy ratio of the heat flow, T_0 is the temperature of the reference environment and T is the temperature of the cooking pot. T is assumed to be 100 °C.

The exergy of the organic carbon is calculated as the exergy of the amount of humus containing the organic carbon. It is calculated based on equation 5 and its composition is considered the same as compost.

The exergy of the nutrients N, P and K is calculated based on the Gibbs free energy of formation of NO_3^-, PO_4^{3-} and K^+ (equation 8).

Appendix B13: Sankey diagrams for phosphorus

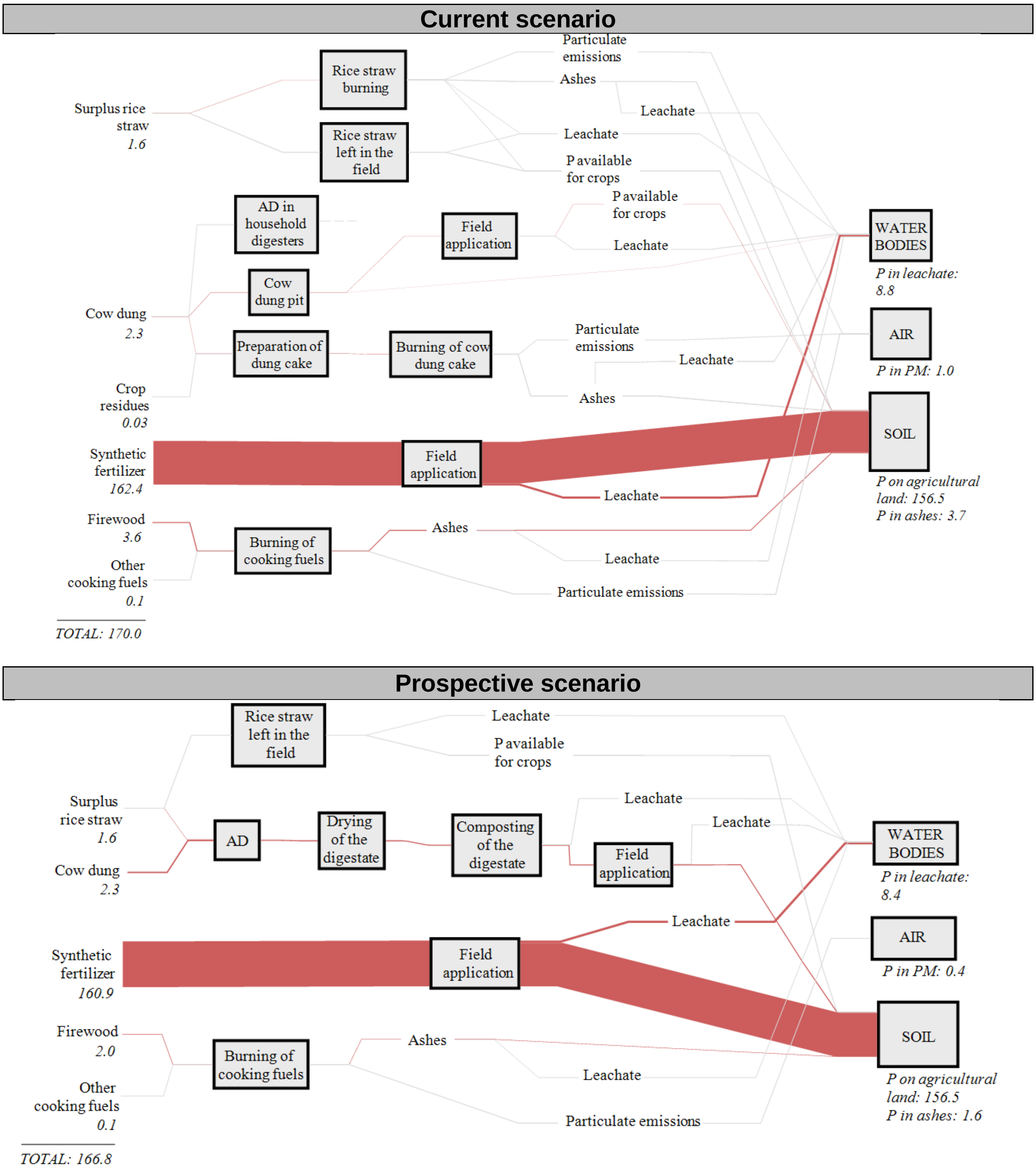


Figure B13-1: *Phosphorus flow diagram. The values correspond to the potassium flows during 1 year (in kt). Flows with a value lower than 1 kt are represented in grey. The flows of phosphorus in the sub-system providing biogas to households in the current scenario are not represented.*

Appendix C: Supplementary material for Chapter 5

Appendix C1: Consumption of food and non-food products

Appendix C2: Data inventory of the resource recovery processes

Appendix C3: Calculation of the exergy-based allocation factors

Appendix C1: Consumption of food and non-food products

Table C1-1: *Amount and type of non-food products consumed by the population of Eindhoven during one year, and released in the sewage system (based on AISE (2014); Golsteijn et al. (2015); RIVM (2002, 2006)).*

Consumed quantities (kg year^{-1})	
Tap water	9.6E+09
Powder laundry detergent	5.1E+06
Liquid laundry detergent	1.2E+06
Hand dishwashing product	1.3E+05
Dishwashing tablet	1.8E+05
Acid toilets cleaners	1.7E+05
Bleach toilet cleaner	2.6E+05

Consumed quantities (kg year^{-1})	
Bathroom trigger spray	1.8E+05
All purpose cleaner	7.4E+05
Shampoo	1.2E+06
Hand soap	4.1E+05
Shower soap	5.1E+05
Toilet paper	5.1E+06

Table C1-2: *Amount and type of products consumed by the population of Eindhoven during one year, per category of food products as defined in RIVM (2011).*

Consumed quantities (kg year^{-1})	
Potatoes and other tubers	
Potatoes	7.5E+06
Vegetables	
Tomato products	1.9E+06
Onions	1.1E+06
Cabbage	8.6E+05
Cauliflower	7.1E+05
Beans	1.3E+06
Carrots	1.2E+06
Cucumber	9.5E+05
Lettuce	1.1E+06
Spinach	7.1E+05
Legumes	
Beans	2.3E+05
Fruits, nuts and olives	
Apple with skin	2.6E+06
Apple without skin	1.5E+06
Banana	2.3E+06
Orange	1.2E+06

Consumed quantities (kg year^{-1})	
Fish and shellfish	
Whitefish	4.7E+05
Salmon	2.3E+05
Herring	1.8E+05
Shrimps	1.3E+05
Tuna	8.1E+04
Eggs and egg products	
Eggs	9.5E+05
Fat	
Margarine	1.9E+06
Butter	1.8E+05
Sugar and confectionery	
Sugar	9.8E+05
Ice cream	1.2E+06
Chocolate, bars and candies	1.8E+06
Cakes	3.9E+06
Non-alcoholic beverages	
Tap water	8.1E+07
Soda	2.3E+07

Manderins	9.6E+05
Strawberries	6.1E+05
Dairy products	
Semi-skimmed milk	1.5E+07
Low fat yoghurt	5.4E+06
Normal Yoghurt	6.2E+06
Cheese	4.1E+06
Cereals and cereal products	
Bread	1.7E+07
Meat and meat products	
Chicken	1.5E+06
Beef	2.2E+06
Pork	5.0E+06

Orange juice	4.7E+06
Mineral water	9.7E+06
Alcoholic beverages	
Beer	1.5E+07
Condiments and sauces	
Mayonnaise	4.8E+05
Peanut sauce	4.3E+05
Tomato sauce	6.6E+05
Salad sauce	9.2E+05
Soups, bouillon	
Soups	4.8E+06

Appendix C2: Data inventory of the resource recovery processes

a) Wastewater treatment plant and dewatering plant

Table C2-1: *Data inventory of the wastewater treatment plant (based on Blom (2013), allocated to the household stream based on COD value).*

Inputs	Amount	Unit
Wastewater	9.71E+09	kg year^{-1}
COD	*1.70E+07*	*kg year^{-1}*
Electricity[1]	1.11E+07	kWh year^{-1}
$AlCl_3$ solution	1.07E+06	kg year^{-1}
$Al_2(SO_4)_3$ solution	3.73E+06	kg year^{-1}
Tap water	4.50E+06	kg year^{-1}
Natural gas	2.40E+04	kg year^{-1}
Sand	3.40E+05	kg year^{-1}
Output products	**Amount**	**Unit**
Clean water	9.59E+09	kg year^{-1}
COD	1.36E+06	kg year^{-1}
Sludge	1.20E+08	kg year^{-1}
COD[2]	1.57E+07	kg year^{-1}

[1]For sewage pumps and wastewater treatment; [2]Calculated based on mass balance

In addition to the output products, 2.35E+05 kg year^{-1} of sieve sludge is landfilled (authors' assumption on the end-of-life scenario, based on *Blom (2013)*). The impact from fat sludge disposal (2.46E+04 kg year^{-1}) is considered negligible.

After use, sand is cleaned for further use (Blom, 2013). Therefore, the sand consumed by the Eindhoven plant is also assumed to have been cleaned. Therefore, only the cleaning step is considered in the inventory of sand production. The consumption of electricity and water for sand cleaning are taken from Hou et al. (2014) (4.4 kWh m^{-3} of washable sediments; 0.1 m^3 m^{-3} of washable sediments).

Table C2-2: *Data inventory of the dewatering plant (based on Blom (2013), allocated to the sludge stream from Eindhoven based on the water content of input sludge streams).*

Inputs	**Amount**	**Unit**
Sludge	1.20E+08	kg $year^{-1}$
Electricity	3.21E+05	kWh $year^{-1}$
Tap water	8.65E+04	kg $year^{-1}$
Natural gas	3.82E+03	kg $year^{-1}$
Output products	**Amount**	**Unit**
Sludge cake	4.03E+07	kg $year^{-1}$
Water	3.03E+07	kg $year^{-1}$
Dry solids	1.00E+07	kg $year^{-1}$

The water extracted during the dewatering step is pumped back to the Eindhoven plant. This water flow is therefore considered as a closed loop system.

b) Incineration plant

Table C2-2: *Data inventory of the incineration plant (based on Sijstermans and van der Stee (2013), allocated to the sludge cake from Eindhoven based on the dry solid content of input sludge streams).*

Inputs	**Baseline scenario**	**Alternative scenario**[1]	**Unit**
Sludge cake	4.03E+07	3.67E+07	kg $year^{-1}$
Natural gas	2.86E+04	1.84E+04	kg $year^{-1}$
Purchased electricity	2.45E+06	1.58E+06	kWh $year^{-1}$
Tap water	1.38E+05	8.89E+04	kg $year^{-1}$
Outputs	**Baseline scenario**	**Alternative scenario**	**Unit**
Valorized ashes			
To road filling	2.07E+06	2.07E+06	kg $year^{-1}$
To production of landfill capping material	7.34E+05	7.34E+05	kg $year^{-1}$
To production of fertilizer	9.38E+04	9.38E+04	kg $year^{-1}$
CO_2 for calcium carbonate production	2.47E+06	2.47E+06	kg $year^{-1}$
Unvalorized ashes to landfill and salt mine	6.41E+05	6.41E+05	kg $year^{-1}$
Residues to landfill	1.78E+05	1.78E+05	kg $year^{-1}$

[1]*Inputs re-calculated based on the assumption that input consumption for incineration is proportional to dry solid content of the sludge cake*

Note that the amount of ashes produced after incineration is the same for both scenarios. This is because the authors assumed that anaerobic digestion does not modify the ash content of the sludge. Moreover, the decrease of carbon content in the sludge because of biogas production is considered to have an effect of the amount of carbon released in the air after incineration, but not on the amount of CO_2 delivered to the calcium carbonate company.

c) Ecophos process

Table C2-3: *Data inventory of the Ecophos process (for the prospective scenario, based on Jossa and Remy (2015)).*

Inputs	**Amount**	**Unit**
Ashes	9.38E+04	kg $year^{-1}$
Electricity	2.81E+03	kWh $year^{-1}$
Steam	2.81E+05	kg $year^{-1}$
HCl (37%)	8.44E+04	kg $year^{-1}$
Outputs	**Amount**	**Unit**
H_3PO_4	6.58E+04	kg $year^{-1}$
$CaCl_2$ solution (100%)	1.69E+05	kg $year^{-1}$
$FeCl_3$ solution (40%)	8.54E+03	kg $year^{-1}$

d) Green gas production

Table C2-4: *Data inventory of the THP and anaerobic digestion processes (based on Blom (2013) and personal communication from Waterschap De Dommel, unless specified).*

Inputs	**Amount**	**Unit**
Sludge cake	4.03E+07	kg $year^{-1}$
Steam (THP process)	7.68E+06	kg $year^{-1}$
Electricity[1]	1.69E+05	kWh $year^{-1}$
Heat[1]	2.69E+06	MJ $year^{-1}$
Water	2.63E+07	kg $year^{-1}$
Outputs	**Amount**	**Unit**
Digestate	6.31E+07	kg $year^{-1}$
Biogas	4.23E+06	Nm^3 $year^{-1}$

[1]*Based on ecoinvent 3.3*

Table C2-5: *Data inventory of biogas transport via pressure line, cleaning and compression.*

Inputs	Amount	Unit
Biogas	4.23E+06	Nm^3 $year^{-1}$
Electricity (pressure line)[1]	5.64E+05	kWh $year^{-1}$
Electricity (cleaning)[2]	6.19E+05	kWh $year^{-1}$
Electricity (compression)[2]	2.48E+05	kWh $year^{-1}$
Outputs	**Amount**	**Unit**
Green gas	4.23E+06	Nm^3 $year^{-1}$

[1]Based on Evangelisti et al. (2015); [2]Based on Ahmadi Moghaddam et al. (2015)

Table C2-6: *Data inventory of digestate dewatering and struvite production.*

Inputs	Amount	Unit
Digestate	6.31E+07	kg $year^{-1}$
Electricity (dewatering)	9.91E+04	kWh $year^{-1}$
MgO (struvite precipitation)	5.64E+05	kWh $year^{-1}$
Electricity (struvite precipitation)	7.43E+03	kWh $year^{-1}$
Heat (drying of the struvite)	1.73E+06	MJ $year^{-1}$
Outputs	**Amount**	**Unit**
Struvite	4.23E+06	kg $year^{-1}$

Appendix C3: Calculation of the exergy-based allocation factors

The calculation of the chemical exergy ex_{ch} of the clean water and sludge is based on equation 1.

$$ex_{ch} = 13.6\,\frac{kJ_{ex}}{g} \times COD \qquad (1)$$

Where COD is the Chemical Oygen Demand.

The exergy values of the other products are presented in Table D1-1.

Table D1-1: *Exergy values of waste, CO_2, ashes and residues used to calculate the allocation factor*

Product	Exergy value (MJ kg^{-1})	Source
Water	0.05	Szargut (2005)
CO_2	0.45	Szargut (2005)
Ashes and residues	2.11	Alvarenga et al. (2013)

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Curriculum vitae

PERSONAL DATA

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First name	Sophie
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EDUCATION

2007-2011	Master of Agricultural Engineering, AgroParisTech, Paris. Specialization : Ecological Engineering and ecosystems management
2005-2007	Undergraduate courses to prepare nation-wide competitive exams in science (prep-school BCPST), Lycée Saint-Louis, Paris
2003-2005	Scientific high school diploma (with honours and European mention), Lycée Marceau, Chartres

PROFESSIONAL ACTIVITIES

2013 - …	Doctoral researcher in Applied Biological Sciences. Environmental Organic Chemistry and Technology Research Group (EnVOC), Faculty of Bioscience Engineering, Ghent University.

Projects:

- INTERREG IVB NWE EnAlgae (Energetic Algae)
- H2020 MEASURE (Metrics for Sustainability Assessment in European Process Industries)
- H2020 REPAIR (Resource Management in Peri-urban Areas: Going Beyond Urban Metabolism)
- FISCH-ICON (IWT) project Omega-extract

2011-2013	Project engineer at RDC Environment, Brussels
Mar. 2011 – Sept. 2011	Intern consultant, BIO Intelligence Service, Paris
Feb. 2010 – Jul. 2010	Intern engineer, NGO Friend In Need, Kameswaram, India
Jun. 2009 – Dec. 2009	Intern student, Department of Biological Sciences, University of Alberta, Edmonton, Canada

TEACHING AND TUTORING EXPERIENCE

2013-2017	Tutor of 3 master thesis students
2014-2017	Teaching exercises Clean Technology
2014-2017	Teaching practical exercises Olfactometry

WORKSHOPS AND COURSES

2016	Spatialization in LCA – Interest, limits and feasibility for eco-design
2016	Social LCA, LCC, Waste modelling and PROSUITE

SCIENTIFIC PUBLICATIONS

- Sfez, S., De Meester, S., & Dewulf, J. Improving the evaluation of the resource footprint of household sewage sludge valorisation products in the context of a circular economy: a discussion on allocation approaches. To be submitted in RESOURCES, CONSERVATION AND RECYCLING.
- Sfez, S., Dewulf, J., De Soete, W., Schaubroeck, T., Mathieux, F., Kralisch, D., & De Meester, S. (2017). Toward a framework for resource efficiency evaluation in industry : recommendations for research and innovation projects. RESOURCES-BASEL, 6(1).
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- Sfez, S., De Meester, S., Dewulf, J., Jeswani, H., & Azapagic, A. (2016). Solid waste management sector: overview of assessment tools and methods. Background document supplementing the “Roadmap for Sustainability Assessment in European Process Industries” (MEASURE deliverable).
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- Sfez, S., De Meester, S., & Dewulf, J. (2014). Environmental footprint of organic carrot and beetroot juices based on organisational data (study in partnership with Colruyt).

CONTRIBUTION TO INTERNATIONAL CONFERENCES, WORKSHOPS AND SYMPOSIA

- Sfez, S., De Meester, S., & Dewulf, J. (2017). Co-digestion of agricultural waste and cow manure to supply cooking fuel and fertilizers in rural India: life cycle assessment and substance flow analysis. Presentation at the 16th International Waste Management and Landfill Symposium, S. Margherita di Pula, Italy.
- Sfez, S., Dewulf, J., De Soete, W., Schaubroeck, T., Mathieux, F., Kralisch, D., & De Meester, S. (2016). Evaluation of resource efficiency in research and innovation projects : challenges and recommendations of the MEASURE project for industry. Presentation at the 22nd SETAC Europe LCA Case Study symposium, Montpellier, France.

- Sfez, S., Van Den Hende, S., Taelman, S. E., De Meester, S., & Dewulf, J. (2015). Environmental sustainability assessment of a microalgae raceway pond treating wastewater from a recirculating aquaculture system: from upscaling to system integration. Presented at the 4th International congress on Sustainability Science and Engineering (ICOSSE 2015), American Institute of Chemical Engineers (AIChE), Balatonfüred, Hungary.
- Sfez, S., Van Den Hende, S., Taelman, S. E., De Meester, S., & Dewulf, J. (2015). Forecasting the environmental sustainability of a microalgae raceway pond treating aquaculture wastewater: from pilot plant to system integration at industrial scale. Presented at the Algae Around the World Symposium, Ghent, Belgium.
- Sfez, S., Van Den Hende, S., Taelman, S. E., De Meester, S., & Dewulf, J. (2015). Forecasting the environmental sustainability of a microalgae raceway pond treating aquaculture wastewater: from pilot plant to system integration at industrial scale. Poster displayed at the Algae Around the World Symposium, Cambridge, United Kingdom.
- Sfez, S., Van Den Hende, S., Taelman, S. E., De Meester, S., & Dewulf, J. (2015). Forecasting the environmental sustainability of newly developed algae-based technologies: case of the upscaling and integration of a MaB-floc raceway pond treating aquaculture wastewater in Belgium. Presented at the 3rd European Workshop on LCA for Algal Biofuels and Biomaterials, Brussels, Belgium.