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Environmental cost accounting methodologies

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Summary

The importance of so three-pillar sustainability (environmental, economic, social) in decision making and research is rising as can be seen in the accelerated pace of published sustainability studies, societal climate goals and the growing adoption of corporate social responsibility (CSR) as well as environmental management schemes in companies. The depletion of certain resources and possible future legislative changes may raise prices of certain pollution types and may be driving companies towards more sustainable operation through environmental management accounting (EMA) where identification, allocation and management of environmental costs are key elements.

Especially life cycle methodologies are needed to evaluate and verify value chain specific targets and development goals. There is a demand for more comprehensive understanding of the value chains due recent development needs of various production processes and new application possibilities of value verses separated from side flows and bio-waste: each action should add value to the product or to reduce production costs in order to make the development of value chains possible. According to the target of bio-economy, natural resources should be used and recycled effectively in *both* the economic and environmental point of view. The lack of a detailed economic assessment next to the environmental life cycle assessment (LCA) limits its value in the eyes of decision makers who always need to consider economic priorities and not only the social and environmental ones.

In addition to LCA a comparative look at the costs and revenues of products, systems and services (Life Cycle Costing, LCC) for the entire chain creates opportunities to find the most critical points to minimise environmental impacts and production costs and add value. Also, integrating these environmental impacts and costs to environmental-economic methods (Environmental-LCC & Societal-LCC) together is required for sustainable solutions. However, E-LCC and S-LCC methods are still relatively young: the definitions of even basic terms can vary from study to study and there are no international standards for conducting them. Further development of E-LCC methods and their results becoming mainstream could enable environmental effects (positive or negative) impacting product prices in the future, either by taxation or change in consumer demand.

The methodology development work needs identification, allocation and management of company's internal environmental costs but also monetarizing externalities (e.g. environmental impacts). So far, there is no consensus on how to best assign relative weight to different environmental impact categories in monetary terms. Even though many databases and methods exist, there is still a need for new customisable valuation systems and databases that could more reliably provide valuation for different aspects of various ecosystem services or products. Assessments should always clarify what aspect of the assessed site exactly is valuated, and with what assumptions or data.

Keywords: Environmental costs, Life Cycle Costing, Environmental Life Cycle Costing, Societal Life Cycle Costing, Environmental management accounting, Externality valuation, Corporate Social Responsibility

Summary in finnish

Kestävyyden kolmipilarimallin (ympäristö, talous, sosiaalinen) merkitys päätöksenteossa on kasvamassa, minkä voi havaita kiihtyvässä kestävyystutkimusten julkaisutahdissa, yhteiskunnallisissa ilmastotavoitteissa ja yritysvastuun (CSR) sekä ympäristöjohtamisen yleistymisessä. Resurssien ehtyminen ja tulevaisuuden mahdolliset lakimuutokset saattavat nostaa tiettyjen saastetyyppien hintoja sekä ajaa yrityksiä kohti kestävämpää toimintaa ympäristöjohtamismalleja (EMA) hyödyntämällä jossa avaintekijöinä on ympäristökustannusten tunnistaminen, allokointi ja hallinta.

Erityisesti elinkaarimenetelmiä tarvitaan arvioimaan ja tunnistamaan arvoketjujen tietyt tavoitteet ja kehitystarpeet. Kysyntää ketjujen kokonaisvaltaiselle ymmärtämiselle luo tarve arvoketjujen ja korkeamman lisäarvon tuotteiden kehittämiselle sivu- ja biojätevirroista: jokaisen toimenpiteen ketjussa tulisi lisätä tuotteen arvoa tai pienentää tuotantokustannusta jotta arvoketjujen kehitys olisi mahdollista. Biotalouden tavoitteiden mukaisesti luonnonvaroja tulisi käyttää kestävästi ja kierrättää niin talouden kuin ympäristön näkökulmasta tehokkaasti. Yksityiskohtaisten taloudellisten tutkimusten puute elinkaarianalyysien (LCA) rinnalla vähentää sen arvoa päätöksentekijöiden silmissä, sillä taloudelliset prioriteetit pidetään aina mukana päätöksissä, ympäristötekijöistä ja sosiaalisista tekijöistä huolimatta.

Ympäristövaikutusten arvioinnin (LCA) ohella vertailtavat elinkaariset tuotteiden, järjestelmien ja palvelujen kustannukset ja tulot (Life Cycle Costing, LCC) luo mahdollisuuden löytää kriittisimmät pisteet ympäristövaikutusten ja kustannusten vähentämiseksi, sekä lisätä ketjun arvoa. Lisäksi näiden yhdistäminen ympäristö- ja talousvaikutuksia yhdessä käsitteleviin menetelmiin (Environmental -LCC ja Societal-LCC), on tarpeen kestävien ratkaisujen aikaansaamiseksi. Tästä huolimatta, näitä ympäristö ja sosiaalisia vaikutuksia käsittelevät laajennetut elinkaarikustannusmenetelmät ovat edelleen suhteellisen nuoria: jopa peruskäsitteiden määritelmissä on vaihtelua tutkimusten välillä, eikä menetelmien toteuttamiseen ole olemassa kansainvälisiä standardeja. E-LCC-menetelmien jatkokehitys ja niiden tulosten valtavirtaistuminen saattaa tulevaisuudessa mahdollistaa ympäristövaikutusten (positiivisten tai negatiivisten) vaikuttamisen tuotteiden hintaan, joko verotuksen tai kuluttajakysynnän muutosten myötä.

Menetelmäkehitystyö vaatii yrityksen sisäisten ympäristökustannusten tunnistamista, mutta myös ulkoisvaikutusten (esim. ympäristövaikutusten) rahamääräistämistä. Toistaiseksi tutkimuksissa ei ole yhteisymmärrystä siitä, miten ympäristövaikutuskategorioille voisi parhaiten suhteellisesti painottaa näiden rahallisen arvottamisen mahdollistamiseksi. Vaikka arvotusmenetelmiä ja tietokantoja on olemassa useita, on silti olemassa tarve uusille tutkijoiden muokattavissa oleville arvotusjärjestelmille sekä tietokannoille, jotka voisivat nykyisiä luotettavammin arvottaa kohteita ja tuotteita. Tutkimusten tulisi aina selkeästi ilmaista, mitä puolia tutkitussa kohteessa tarkalleen arvotetaan, ja mihin oletuksiin sekä tietoon perustuen.

Avainsanat: ympäristökustannukset, elinkaarikustannukset, elinkaariset ympäristökustannukset, elinkaariset yhteiskunnalliset kustannukset, ympäristölaskenta, ympäristövaikutusten arvottaminen, ulkoisvaikutusten arvottaminen, yritysvastuu

Contents

1. Introduction

According to the target of bio-economy, natural resources should be used and recycled effectively from both the economic and environmental point of view. It is well known that financial constraints affect the companies' decisions on e.g. used energy sources and major technology implementations in modern societies. Therefore, lack of a detailed economic assessment next to the environmental life cycle assessment (LCA) limits the value of LCA in the eyes of decision makers who always need to consider economic priorities and not only the social and environmental ones. Environmental impacts are systematically undervalued in traditional business calculations since it is usually seen that external costs do not influence the formation of the company's result.

Corporate social responsibility (CSR) with environment related activity costs and benefits is becoming more mainstream due to forward-thinking companies that embed sustainability in their operations to create shared value for society in addition to the companies. Through the rising emphasis of CSR the importance of environmental management accounting (EMA) has grown from a mere external reporting method to a supportive tool in total management decision-making processes, and is now seen as a strategic competitive factor. In EMA identification, allocation and management of environmental costs are key elements.

There is growing need for research-based knowledge that links environmental (LCA) and economic (LCC) aspects of products and projects together. Both internal and external environmental costs are needed to be internalized as part of companies' decision makin process. Also, the indirect cost effects caused by industrial activities, energy production, infrastructures and agricultural land-use are becoming more and more important both globally and from the European perspective. The methodology development work needs identification, allocation and management of company's internal environmental costs but also monetarizing external (e.g. environmental impacts) costs. Tools have been and are being developed to make environmental-economic interrelations clearer and enable the internalisation of environmental and social externality costs into product prices. Another goal is to enhance the communication of environmental impacts to non-scientists and help bring environmental considerations into societal decision making and company operations.

This literature review examines existing research on environmental costs and methodologies that links together environmental and economic assessments so that the results of both worlds can be viewed and compared together. It surveys how life cycle costing methods account for internal and external (environmental) costs and how these methods connect to traditional LCA methodology. The review also discusses the possibilities of integrating the environmental impacts into the life cycle costs of projects and products, to produce a more comprehensive cost evaluation methodology. The aim of this report is to explore the state, development and applicability of current environmental-economic costing methods and, in addition, analyze the needs for further research.

First, chapter 2 classifies cost types (i.e. environmental costs) in traditional business accounting sector. Chapter 3 explores environmental costs in corporate social responsibility. Chapter 4 explains more specifically environmental accounting sector and different environmental cost types as well as the monetisation methods that have been developed to estimate the externality costs (i.e. environmental impacts) of corporate and societal operations. Chapter 5 explores environmental accounting methodologies and more specifically the main aspects of environmental life cycle costing are assessed, together with traditional life cycle analysis. In addition, it includes more general sustainability assessment methods incorporating social considerations (cost benefit analysis, societal life cycle costing, social life cycle analysis), with descriptions of their differences and similarities. Chapter 6 includes a review of case studies that portray the results of monetizing environmental impacts and life cycle cost assessment methodologies (i.e. interrelations of environmental and economic aspects of products, projects and sites, from the viewpoints of corporations and/or society). Finally, chapters 7 and 8 end the report with general discussion about the methods and their applications in contemporary studies, as well as needs for future research.

2. Environmental costs in traditional business accounting

Traditional cost types of companies and organisations are divided into **internal** (private) and **external** costs. Another division is between "**direct costs**" and "**indirect costs**". In other words, there can be direct internal costs, indirect internal costs, direct external costs and indirect external costs. So called **environmental costs** can be internal, external or direct and indirect. The line between direct and indirect costs is not always clear, since costs that are direct to some companies or organizations can be indirect to others, depending on the accounting system and how costs are allocated.

2.1. Direct and internal costs

Businesses only deal with costs that they have **internalised**, and influence the formation of the company's result (EPA 1995). All the costs which companies are accountable and responsible for are internal. Internal costs can be categorised into **budget costs** (see more in chapter 5.3.1.) and **transfers** (see chapter 2.4.) and can be measured either in **market prices** or **factor prices**, which are market prices excluding transfers (Nordic Council of Ministers, 2007).

A direct cost is completely and clearly attributed to the production of a specific good or service. Direct costs can be the costs of materials, machinery, facilities and taxes. **Direct internal costs**, also called the conventional or usual costs of a company, include e.g. the costs of raw materials, capital goods, salaries and supplies. Conventional costs are important in environmental accounting (see chapter 4) since savings achieved via e.g. efficient use of materials and reduced waste also lead to environmental benefits (EPA 1995, Russo 2008).

2.2. External costs

External costs, also termed ''externality'' costs or "non-marketed goods/services", are defined either as social or *environmental costs* which are caused to other actors outside the company and/or costs that occur completely outside the economic system because they have no direct monetary value in the market (Martinez-Sanchez et al. 2015). Environmental benefits and damages often fail to receive a market price due to e.g. undetermined property rights. It is usually seen that external costs do not influence the formation of the company's result (EPA 1995) and therefore environmental impacts are systematically undervalued in traditional calculations while the direct economic benefits of projects are emphasized (Hanley et al. 2007). Externalities are caused by the operations of companies and other actors who are not legally responsible for them (Martinez-Sanchez et al. 2015). They represent uncompensated effects on the welfare of individuals or the environment. In order to place external environmental costs "on the same line" with internal costs, economic means can be utilised to describe how citizens valuate and appreciate environmental assets.

In the usual meaning of the word, externalities are environmentally or socially harmful impacts, though they can also sometimes be beneficial. Typical harmful environmental externalities are emissions into air, water and soil that disturb natural environments, damage human health and cause climate problems as well as disamenity impacts. They are generated by most industrial activities, notable examples including waste facilities, transportation, power plants (both fossil and renewable), agriculture, textile production, mining and production of electronic devices. Positive environmental externalities have resulted from e.g. biogas initiatives in developing countries, through improved indoor air quality, less time needed to collect firewood or other sources of heating power as well as the manurial potential of the slurry produced from the digestion process (Srinivasan 2008).

Depending on the context, external costs have also been called "social costs", but in some contexts social costs can also alternatively refer to non-environmental externalities (EPA 1995 & Shapiro 2001). However, in this report **social costs (or societal costs) are defined as the sum of internal and**

external costs, as used by e.g. Porter (2002), Martinez-Sanchez et al. (2015) and Culyer (2014), which means they are only partly external.

Also an environmental externality can have impacts on society and cause **indirect** social externalities. Society can obtain costs as well as gain significant benefits and savings through environmental protection measures carried out by companies. For example, if a company develops a technology that improves its performance, the same technology can potentially be adopted by other actors as well.

2.3. Indirect costs

An indirect cost is any cost not directly identified with a single final cost objective but identified with two or more final cost objectives. In accounting, an indirect cost is an expense (such as for advertising, computing, maintenance, security, supervision) incurred in joint usage and, therefore, difficult to assign to or identify with a specific cost object or cost center (department, function, program). Indirect costs are usually constant for a wide range of output, and are grouped under fixed costs (AACE International 2004). Generally, also costs whose generation processes are unclear are labelled indirect. In contexts outside accounting, indirect costs can also refer to costs that are incurred as any indirect and perhaps unforeseen consequence of company operations, such as the environmental costs of groundwater contamination due to pesticide use (see e.g. Pimentel 2005). The indirect effects caused by industrial activities, energy production, infrastructures and agricultural land-use are becoming more and more important both globally and from the European perspective.

2.4. Transfers

Transfers, or income transfers, are taxes, subsidies, fees and duties which are used to distribute income between different agents in society. More generally, they are monetary flows that lead to income redistribution between stakeholders but do not represent any resource (e.g. land or labour) reallocation or welfare change in society (Møller & Martinsen 2014). An externality can be made internal to companies if it becomes priced by an authority as a transfer via e.g. environmental taxation in the form of air emission taxes (Vigsø 2004) or as environmental taxes for emissions and energy use (Martinez-Sanchez 2015).

Another kinds of transfers**, pecuniary externalities,** are generated when the activities of agents impose costs on (or create benefits for) third parties, by causing increases or decreases in market prices (Holcombe & Sobel 2001). Unlike externalities (see 2.1.2) in general, they happen inside the economic system by definition, but the effects are indirect and do not seem to affect the actor who caused them. For example, increased heat production at a waste incineration plant can force other heat producers to operate below their design capacity, especially if waste incineration has legal priority over other forms of heat production. This in turn could increase the costs of heat production and result in higher heat market prices for consumers. These costs are not usually related to resource reallocation or welfare changes so they are considered transfers if the heat demand on consumers is not altered (as is likely in Northern Europe where heat demand is almost inelastic) (Martinez-Sanzchez et al. 2015).

3. Environmental costs in Corporate Social Responsibility (CSR)

CSR is defined as a "concept whereby companies integrate social and environmental concerns in their business operations and in their interaction with their stakeholders on a voluntary basis" (Commission of the European Communities 2001). Corporate sustainability ''recognizes that corporate growth and profitability are important, and it also requires the corporation to pursue societal goals relating to sustainable development — i.e. environmental protection and economic development'' (Wilson 2003). As a management tool, CSR is becoming more mainstream due to forward-thinking companies that embed sustainability in their operations to create shared value for society in addition to the companies. CSR focuses mainly only on the production phase and uses management information at the corporate phase. This differs from social LCA (S-LCA) which analyzes the whole life cycle and uses information gathered at company, plant and process levels (Ramirez & Petti 2011).

A short-term orientation in corporate sustainability has its origin in the endeavour of firms to turn sustainability into a concrete business issue (Hahn et al. 2015). As a short-term orientation, firms have used corporate sustainability to turn sustainability into concrete business issues (Hahn et al. 2015). It should be noted that many CSR activities are business oriented and therefore take the profit seeking path (Santos 2011). The study of Tilley (2000) about SME's attitudes toward environmental issues found that economical interest predominantly prevails over environmental or social interest.

The business case of CSR follows an alignment logic, which prioritises economic attributes (Hahn et al. 2014). It investigates the costs and benefits of CSR activities (ISO 26000, Sprinkle & Maines 2010, Nurn & Tan 2010, Exter, Cunha & Turner 2011, Sprinkle & Williamson 2010, European Commission 2008). Social and environmental aspects are only considered when they can be aligned with financial performance in line with the business case for sustainability (Carroll & Shabana 2010). This frame is based on the controversial belief that addressing environmental and social issues contributes to profit maximization (Andersson & Bateman 2000 & Byrch et al. 2007).

According to Hahn et al. (2014), the managers with *a business case frame* focus on environmental and social aspects that align with economic objectives. Sustainability issues are interpreted as either positive or negative for business and responses often follow existing routines and solutions. As a result, sustainability issues can be only narrowly observed since mostly quantitative information with business relevance is focused on (Daft & Weick 1984).

Firms seek to balance often divergent economic, social, and environmental goals and therefore corporate sustainability is rife with tensions. According to Van der Byl and Slawinski (2015), one of the total four general approaches how tensions are examined is "Integrative approach to bring balance to the three elements (economic, environmental and social) of sustainability".

Stakeholder requirements make companies implement CSR practices along their supply chains (Wiese & Toporowski 2013). Sustainability is vital for business success as consumers' awareness about global social issues continues to grow as does the importance these customers place on CSR when choosing where to shop (International Trust 2017). According to Alniacik et al. (2011), positive CSR enhances consumers' intentions to buy products from the company. Mutually beneficial cooperation between corporations and non-profit organisations, i.e. cause-related marketing, can be employed for an integrative approach which combines commercial gains from social and environmental activities with societal benefits to related stakeholders (Liu, 2013). Similarly, some organisations have developed hybrid business models that blur the boundary between for-profit and non-profit worlds and try to promote a sustainability mission while simultaneously being oriented towards the market (Haigh & Hoffman 2012).

In general, stakeholders are increasingly interested in making sure that the products they are affiliated with are free from e.g. sweatshop exploitation and employee discrimination. Good corporate reputation has significant economic value and social irresponsibility can tarnish the brand as well as damage customer loyalty (Slaughter & Everatt 1999).

Five dimensions are frequently used in CSR definitions (Dahlsrud 2008): the environmental, social, economic, stakeholder and voluntary dimension. Especially the food industry meets various challenges in implementing CSR where eight areas of responsibility have to be considered: animal welfare, biotechnology, environment, fair trade, health and safety, labour and human rights, procurement and community (Maloni and Brown 2006). Mainly successes regarding CSR in food chains are reported in e.g. CSR reports or best practice recommendations. However, failures occur and for example animal welfare or environmental protection can be neglected (Wiese & Toporowski 2013). Food supply chains have some special challenges for CSR, including hugely varying origins of products (including developing countries) and a large number of companies involved in the production processes (e.g. producers of feedstuffs and suppliers).

An integrative view on corporate sustainability (Berger et al. 2007, Gao and Bansal 2013, Hahn et al. 2010, Kleine and Hauff 2009 & Liu 2012) argues that firms need to pursue the economic, environmental and social dimensions of sustainability at the same time — even if they seem to contradict each other. Managers and decision-makers then need to accept and embrace the tensions between conflicting sustainability aspects, not dismiss them. The integrative view can be seen as an objection to the presently dominant instrumental logic which addresses environmental and social aspects of CSR only through the lens of profit maximisation, both in the conceptual (Dentchev 2004 & Husted & de Jesus Salazar 2006) as well as empirical (Barnett & Salomon 2012, Margolis & Walsh 2003, Orlitzky et al. 2003) sense. Porter & Kramer (2011) also criticise CSR in their widely cited article published in Harvard Business Review, saying that it is harmful for companies to get stuck in a "social responsibility" mind-set, in which societal issues are at the periphery of business strategies, not at the core. They emphasise the meaning of shared value, which involves creating value for society at large, by addressing its needs and challenges. Their main argument is that the purpose of a corporation should be redefined as a creator of shared value, not just profit, which could positively reshape capitalism and legitimise business as a truly responsible shaper of society.

3.1.1. Costs due to environment-related CSR activities

The costs of doing CSR vary depending on the subject. Environment-related CSR activities mainly cause costs in terms of capital and minor recurrent costs. In contrast, recurrent costs of CSR activities aimed at improving social aspects of business operations often exceed capital costs. In addition, CSR implementation may bring considerable costs on suppliers or export–oriented companies (certification and auditing), such as:

- **Opportunity costs** – possible lost revenues from the activities that could not be undertaken due to labour and capital bound to CSR activities.
- **Sunk costs** – all initial investments in new equipment, buildings and infrastructure (invested money and opportunity cost of investment, including the interest rate on the bound investment).
- **Recurrent costs** – labour costs for increased wages and overtime payments, an increase in management time, social insurance, trainings, benefits for workers, monitoring and reporting, equipment update and maintenance (Sprinkle & Maines 2010).

There is a belief held especially by small-to-medium-sized businesses that CSR schemes (including environmental information collection) are too expensive to implement, time-consuming to maintain and non-profitable. Collecting and processing comprehensive and varied information about sustainable issues can certainly be time-consuming and expensive. Some scholars argue that the ability of managers to collect detailed and broader information about sustainability issues will be enhanced with the greater availability of ready resources (Bansal 2005, Bowen 2002 & Sharma 2000). Hahn et al. (2014), on the other hand, question this by claiming that companies are not limited as much by time and resources as they are by their alignment structure and main focus on economic attributes, so that even with more readily available information the managers "will still fail to notice information on sustainability issues that is presented in nonfinancial, qualitative terms and that has an ambiguous relation to financial outcomes".

3.1.2. Benefits gained from environment-related activities

According to Golicic et al. (2010), companies that integrated sustainability practices throughout their supply chains were experiencing clear benefits though, according to Grover (2008), each situation also carries the potential for the supply chain to contribute to higher costs. Small businesses may adopt many easy and affordable changes to their CSR schemes that bring not only social but also financial benefits. Companies that employ CSR may attract more motivated workers, reduce operational costs as well as gain competitive advantages and new contracts (Sino-German Corporate Social Responsibility Project 2012).

However, it is usually difficult to monetize CSR benefits since many of them only get visible in the long run and are indirectly induced. Understanding the causal relationship between direct and indirect benefits can help trace improvements in competitiveness and the financial performance of firms that use CSR. Businesses affect many different people – employees, customers, suppliers and the local community – and it also has a wider impact on the environment. Considerable environmental benefits with simultaneous cost savings can be reached from optimising basic operations such as use of lighting, equipment, water, paper and other resources. Even more can be saved by thinking about waste implications when designing new products and production processes. Companies can also gain revenues from positive image and relevant marketing, since many customers prefer to support-responsible businesses Some companies use this fact by making social responsibility a core of their operations, such as Ben and Jerry's and Starbucks (Ballou et al. 2006).

The environmental impact of businesses can be reduced by employing environmental assessment techniques and using the gained information e.g. for (NI Business Info 2017):

- creating recyclable products,
- sourcing responsibly (e.g. using recycled materials and sustainable timber),
- minimising packaging,
- buying locally to save fuel costs,
- creating an efficient (and fuel-efficient) distribution network and
- working with suppliers and distributors who take steps to minimize their environmental impact.

Reducing environmental impacts through CSR can also create **benefits** as cost savings internally and to external stakeholders, including (Setyadi et al. 2013):

- **Internal direct benefits**: better employee commitment, deeper talent pool, operational effectiveness, reduced emissions trading costs.
- **Internal indirect benefits**: innovation, increased productivity, improved quality.
- **External direct benefits**: positive publicity and reputation, improved stakeholder relationships.
- **External indirect benefits**: capital and market access, customer satisfaction, risk reduction, higher price premiums (i.e. possibility of higher-than-average pricing without negatively affecting demand).

4. Environmental accounting

Through the rising emphasis of corporate social responsibility (CSR) (see ch. 3), the importance of environmental accounting has grown from a mere external reporting method to a supportive tool in total management decision-making processes, and is now seen as a strategic competitive factor (Kolehmainen & Riuttala 2012). Environmental costs and savings (benefits) are formed in the reduction of the environmental impact of the company. Environmental costs and savings are generated to the company or to society during the entire product **life cycle** by different measures relating to air, soil or water protection, waste management, environmental management or prevention of noise and odor. Environmental accounting sectors include National environmental accounting, Environmental financial accounting and Environmental management accounting (EMA) (Figure 1).

National environmental accounting is performed at the governmental level and is concerned with the social and societal costs of operations. National accounts are crucial for national policy development: they track the evolution of the economy as a whole and are the source of many familiar indicators such as GDP, economic growth rates and productivity figures (Hecht 2005). Today, there are international standards for national accounting. The system of national accounts (SNA), adopted by the United Nations Statistical Commission, is an “internationally agreed standard set of recommendations on how to compile measures of economic activity” and “describes a coherent, consistent and integrated set of macroeconomic accounts in the context of a set of internationally agreed concepts, definitions, classifications and accounting rules” (United Nations 2008). To more specifically portray the interrelations between the economy and the environment in a way that is consistent with the national accounts, another statistical standard was developed in 2012: the *System of Environmental-Economic Accounting 2012 – Central Framework* (SEEA Central Framework) and finalised by the Statistical Commission. The SEEA Central Framework aims to aid policy development and produce indicators that relate to e.g. resource use and changes in stocks of natural resources, water and energy productivity, waste and emission intensity, contribution of environmental activities to GDP, environmental taxes as well as environmental assets and their role in the economy (SEEA 2012 Applications and Extensions, 2017). In addition, the *European Environmental Accounts* (consistent with SEEA 2012 CF) were established in Regulation (EU) 691/2011 to provide a legal framework for all EU member states and EFTA countries (Eurostat).

Environmental financial accounting (EFA) is one environmental accounting sector from companies’ perspective which stands for the more “neutral” part of environmental business accounting since its active purpose is not to affect decision-making at the management level. EFA assists in the identification and proper allocation of environmentally related costs and is used to ensure that environmental revenue, costs, assets and liabilities are clearly presented in the company’s financial statements in a standardised way: the environmental procedures then follow from international legislation and accounting standards (Godschalk, 2008).

Environmental management accounting (EMA) is a dynamic and evolving accounting sector which has grown from the globally growing need of corporations to report, evaluate and adjust their operations in response to new environmental requirements laid down by legislation and consumers. EMA and environmental cost types are explored more specifically in the next chapter.

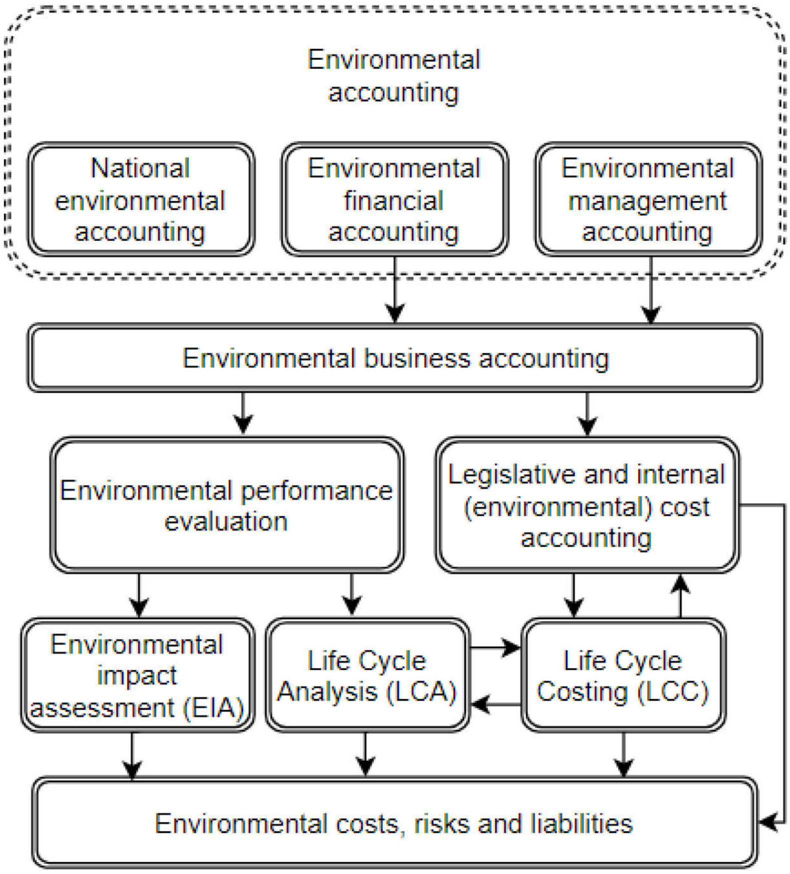

Figure 1. Areas and tools of environmental accounting (simplified from Pohjola 1999 p. 21).

4.1. Environmental management accounting (EMA)

The EMA work includes counting both environmental impacts and environmental costs for an optimal calculation. It consists of identifying, collecting and using both physical and monetary (environmental cost) information with the aim of bringing environmental responsibility to corporate and organisational decision-making as well as minimising wastage of resources. The physical information includes the uses, flows (and destinies) of energy, water, materials and waste (Godschalk 2008).

In EMA literature, there are several different terms, definitions and intepretations of environmental cost accounting methodologies with different system boundaries. System boundaries of accounting also vary depending on if the assessment is done from the perspective of a company or society.

4.1.1. Environmental costs and benefits

Identification, allocation and management of environmental costs, are key elements of environmental accounting. Environmental costs refer to a broad and varyingly defined set of expenses related to the environmental performance and responsibilities of companies and other actors. They can include costs caused by e.g. control, trade and monitoring of emissions, waste treatment, environmental regulations and permits as well as clean-ups from past operations (Shifrin et al. 2015). According to the UN, environmental costs relate to all costs occurred in relation to environmental damage and protection (UN 2001).

Cost savings due environmental benefits are formed with better and effective use of inputs, improvement of nutrient recycling and in some cases by replacing fossil fuels with renewable energy. For example, Ristimäki et al. (2013) found that for a residential area, replacing district heating with geothermal heat pumps can bring significant cost savings over the course of the pump life cycle, even though initial investments are lower for district heating. In addition, production processes can be improved, potential fines and penalty payments avoided as well as the corporate image improved which may have an impact on sales and income. Benefits of using EMA stem from properly identifying and thus avoiding major environmental cost drivers, and may include:

- Reducing of clean-up, compliance, image and liability costs
- Savings via more efficient use of materials, water and energy and avoided wastage
- Reduced environmental taxation
- Profits from emissions trade
- Savings from timely identification and avoidance of to-be-internalised external costs

Reduction of the environmental impact (ie. environmental costs) and the necessary technology and investment will in turn create costs. **Internal environmental costs** are divided into conventional environmental costs, hidden costs, liability costs and promotional image costs (Figure 2).

- Conventional
    - **Direct environmental costs** include e.g. waste management fees, the equipment costs of emission control and environmental taxes.
    - **Indirect (environmental) costs** can be e.g. costs for product design and engineering, permits, environmental training and depreciation of waste treatment equipment.
- **Liability costs or contingent costs** refer to environmental costs that may occur in the future due to legal environmental responsibilities. These costs may still depend on uncertain future events (e.g., costs of remediating future spills).
- **Hidden costs** are unknown to or unobserved by the companies that pay for or cause them (Rogers et al. 2003), and mainly include expenses that are not included in purchase prices of items, such as costs of maintenance, training and environmental damage.
- **Image costs** are expenses incurred for corporate image purposes or for maintaining/enhancing relationships with e.g. regulators, customers, suppliers and the general public (EPA 1995).
- **Shadow prices or accounting prices** are sometimes formed if market prices are not considered to represent the true value of resources used or produced in a project, or market prices do not exist. The definitions vary by source: in Martinez-Sanchez et al. (2015), for example, they represent society's willingness to pay for a good or service, and are used as the measure of value in societal life cycle costing (see chapter 5.3.3.). Curry (1987) defines shadow prices as costs or benefits of producing the same service in another way or from another source (which can be useful e.g. when estimating the true value of monopolised or regulated goods).
- **Opportunity costs** (sometimes also classified as shadow prices) stand for the gained or forgone benefits of choosing one type of activity over another, for example the difference in net earnings from conserving or enhancing forests versus converting them to other land uses (World Bank 2011).

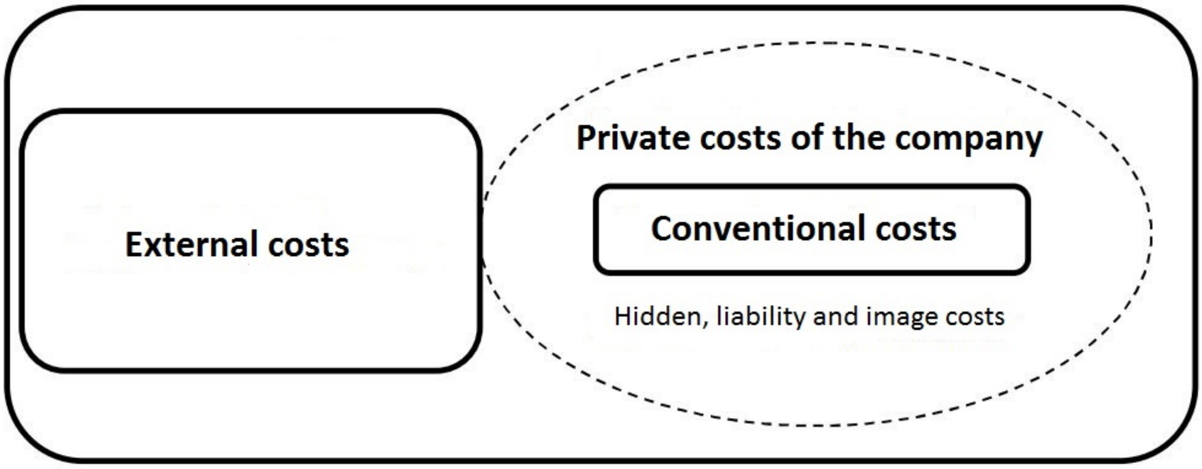

Figure 2. Private and external costs (EPA 1995).

Table 1. Examples of environmental (internal) costs identified by the technology company Pitney Bowes (Rogers et al. 2003).

Environmental costs at Pitney Bowes		
Lobbying regarding environmental legislation	Chemical and hazardous waste storage space	Maintenance time spent on environmental tasks
Contingency plans	Emergency response equipment	Air permit fees
Consultant fees	Energy management/conservation	Facility audits
Engineers' time spent on product design	Office space for environmental staff	Product/packaging end-of-life fees
Inspections	Pre-disposal treatment	Regulated waste disposal
Remediation	Reporting	Wastewater permit fees
Solid waste disposal	Treatment facility depreciation	Supplier environmental costs
Environmental insurance	Environmental protective equipment	Environmental training
Equipment decontamination	Facility engineering	Legal counselling
Marketing	Monitoring	Pollution control
Protective equipment	Public Affairs staff time	Recycling costs
Regulated waste disposal	Take-back costs	Waste and recycling containers

It should be noted that EMA practitioners often only account for **internal environmental costs**. In some studies it is seen that businesses generally only deal with costs that they have internalised, *i.e. external costs are not included in environmental business accountin*g (Jasch 2003), since they do not (directly) influence the formation of the company's result (EPA 1995). In the view of Jasch (2003), it is the role of the government to use necessary political instruments, such as emissions control and eco-taxes, so that external costs will be integrated into business calculations. Burritt (2006) noted that in the competitive business world, considering externalities "becomes a luxury".

Sometimes companies still assess environmental externalities voluntarily within the EMA framework. The Brazilian cosmetics company Natura, for example, bases their choice of suppliers partly on their environmental footprints, including CO_2 emissions, water use and waste generation, among other environmental stressors, and Natura has also conducted life cycle analyses on their products. They evaluate the suppliers using a multidisciplinary team that annually quantifies values to select externalities, answering questions such as “How much does a ton of CO_2 emitted cost in terms of environmental damage or public health cost?” or “What is the social value of one year of education for a given individual?” (World Resources Institute 2013). Monetary valuation of environmental impacts (see chapter 4.2) offers a generalised method to assess risks and opportunities of different operations, products and supply chains.

4.2. Monetising environmental externalities

When assessing environmental pressures for products, projects or systems, multitudes of impact categories may be considered, such as CO_2 emissions, acidification, biodiversity loss, land-use and eutrophication. Collecting the information for any comprehensive environmental analysis is demanding but may not have the intended effect on decision-making if the results are too confusing. Monetisation can help communicate complex environmental information to decision makers, so that the scale and hierarchy of the environmental risks become clearer (Ahlroth 2009). Since externalities are typical market failures, their monetisation and internalisation are also required to achieve optimal resource allocation (Pizzol et al. 2015). However, current markets have only valuated and incorporated into transactions a small subset of all possible ecosystem processes and components. The structural limitations of markets make them unable to provide a comprehensive picture of the ecological values that are relevant to decision processes (MA 2005).

As mentioned in chapter 2.4., one traditional way an externality can be internalised is through becoming priced by an authority as transfers by environmental taxation in the form of e.g. air emission taxes or energy use (Vigsø 2004 & Martinez-Sanchez 2015). However, there is also a broader need for monetary valuation of non-market goods as well as external impacts of market goods and projects. In addition to contamination and cleaning of emissions there should be also information about how people value the quality of environment in situations like the forests for recreation and other uses versus wood production, multifunctional agriculture in which in addition to food production, water protection and biodiversity is produced (what kind of agriculture and water areas & to what extent they are protected from economic exploitation).

In order to place environmental impacts "on the same line" with economic costs, economic means can be utilised to describe how citizens valuate and appreciate environmental assets (Hanley et al. 2007). To monetarise environmental effects such as emissions and resource use, it is possible to use different weighting methods. According to Carlsson-Reich (2005), methods for weighting environmental data to a single monetary unit should always be put through strict scientific scrutiny. The aggregation process should be kept transparent and, when possible, scientific. Valuation results need not be universally applicable, and can also serve as a baseline for discussion for where perceptions of weights differ. Weighting can also be used to decide what should be prioritised in the study, and what can be treated superficially. The Nordic Guidelines on LCA (Lindfors et al. 2005) recommend using many methods for valuation in parallel, to show how they can differ from each other. Differences can arise not only from uncertainty, but also differences in value bases and the chosen details of focus (Carlsson-Reich 2005). **At present, a flawless weighting method does not exist**. There probably will never be a method that is good for all occasions and objects of analysis. Different weighting methods give different results, and it is not always possible to say which is the better method to use for a specific problem. Therefore, it is most important to be aware of the assumptions made and the methods used, and to understand and agree with them if their results are to be used.

The economic value of non-marketed environmental benefits describes how much people are willing to give up on streams of actual economic benefits (income) and consumption opportunities in order to obtain or maintain these environmental benefits. **Willingness to pay (WTP)** generally means the maximum amount of money that a consumer is willing to pay for a commodity. From the environmental point of view, this implies how much consumers are willing to pay so that an environmental protection act is carried out or any environmentally harmful project is abandoned. **Willingness to**

accept (WTA) refers to the amount of monetary compensation the consumers ask for to accept an undesired effect, such as environmental damage or disamenities. If the commodity in question has close substitutes, WTA and WTP are close to each other. However, very often there are no substitutes and values for WTA are greater than for WTP because consumers feel they have environmental ownership rights, which should be at this point be abandoned. (Hanley et al. 2007.)

Studying the willingness to pay (WTP) of individuals for environmental sites and ecosystem services can give some information about how these sites are appreciated, and help develop initiatives that improve the state of the environment (Groot et al. 2012). Cost methods assume that if people are willing to pay a certain amount of money to avoid losing certain ecosystems or their related services, for them the sites must be worth *at least* as much as the measured WTP (Ahlroth 2009). However, the valuation process and its results depend greatly on what aspect of the assessed site is valuated and whose interests towards the site are considered. Some impacts are at least to some accuracy quantifiable in physical units, such as clean air or water, natural fish stocks, or rainforests. On the other hand, e.g. biodiversity and human health are more difficult to measure at all, let alone monetise. In addition, monetary valuation can only measure *marginal* (i.e. small) changes in the availability of non-market goods, and the results are highly site-specific, although **benefit transfer methods** are often used to generalise some of the results from previous studies (Pizzol et al. 2015).

So far, there is no consensus on how to assign relative weight to different environmental impact categories in monetary terms (Nguyen et al. 2016). However, some types of environmental stressors (e.g. CO_2, NOx etc.) have been valuated in general terms with intended universal applicability. Various valuation projects and databases exist that include prices for externalities. The weightings between these databases are different which is why using many methods is recommended. Examples of European databases include ExternE (with the follow-up projects NewExt and NEEDS), Stepwise 2006, EPS2000 and Ecotax.

Monetisation of externalities is used

- commonly (and most traditionally) in cost-benefit analyses (see chapter 5.5.),
- always in societal life cycle costing (chapter 5.3.3.),
- often in social life cycle assessment (chapter 5.4.),
- to some extent in environmental life cycle costing (chapter 5.3.2.),
- infrequently in (environmental) life cycle assessment (chapter 5.1.) and
- very rarely in conventional life cycle costing (chapter 5.3.1.).

Valuation can be divided into biophysical and preference-based methods. Biophysical methods derive value from physical costs, such as energy or material inputs or labor costs, while preference-based methods study the values that rise from the individual preferences and WTF of people.

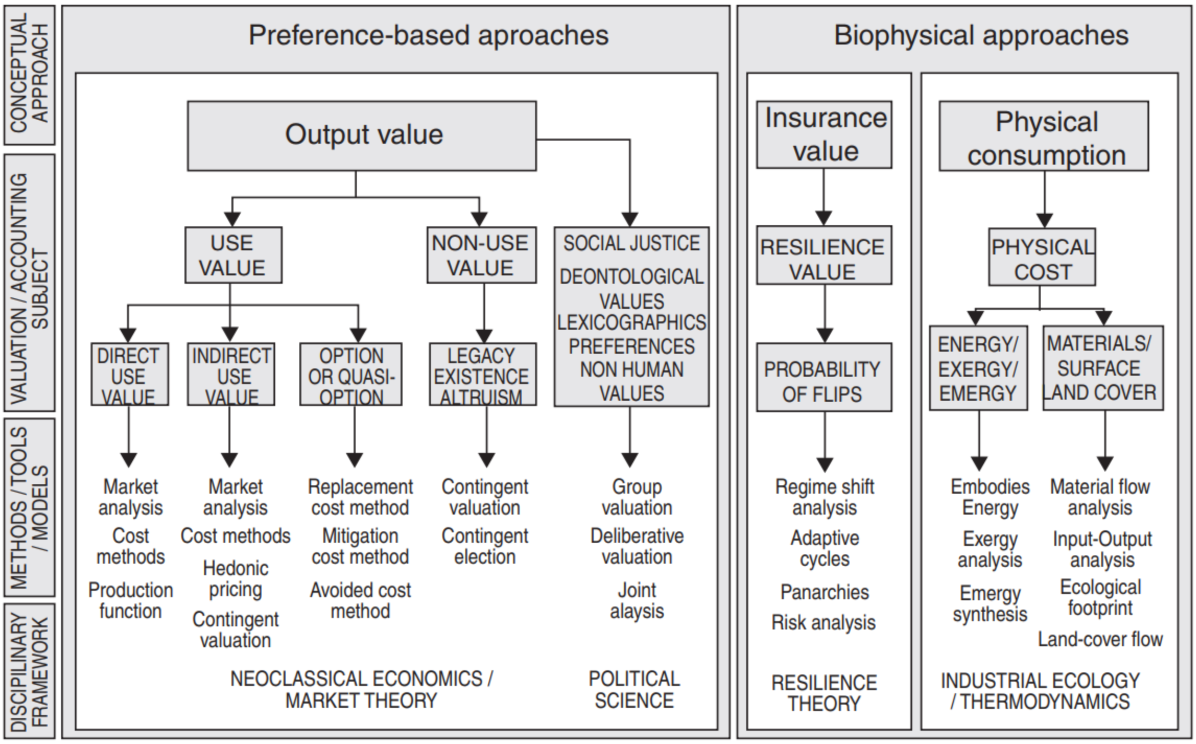

Figure 3. Approaches for the estimation of nature's values (Pascual et al. 2010).

4.2.1. Value types

Values attached to environmental benefits and harms can be classified with a basic distinction to **use and non-use values.** Use values for industries can refer to recreation, fishing, berry picking, bird watching etc. and to industries they can be e.g. extractable resources from forests or other ecosystems. Non-use values are harder to valuate since the usage forms cannot be separated and detected so eadily. **Existence value** is the value people give to species surviving, intact ecosystems, just for existing. The **total economic value** of the system, in the context of valuation, can be seen as the sum of its use and non-use (or existence) values. It should be emphasized that the "total economic value" is summed across categories of values (i.e. use and non-use values) and only measures the value of marginal (small) changes. That is, it cannot be e.g. scaled over complete ecosystems. Values gathered via WTP methods cannot be broken into subgroups of smaller value, either. More explanation about different value types is presented in figure 2. (Pascual et al. 2010.)

Value type	Value sub-type	Meaning
Use values	Direct use value	Results from direct human use of biodiversity (consumptive or non-consumptive).
	Indirect use value	Derived from the regulation services provided by species and ecosystems.
	Option value	Relates to the importance that people give to the future availability of ecosystem services for personal benefit (option value in a strict sense).
Non-use values	Bequest value	Value attached by individuals to the fact that future generations will also have access to the benefits from species and ecosystems (intergenerational equity concerns).
	Altruist value	Value attached by individuals to the fact that other people of the present generation have access to the benefits provided by species and ecosystems (intragenerational equity concerns).
	Existence value	Value related to the satisfaction that individuals derive from the mere knowledge that species and ecosystems continue to exist.

Figure 4. Use and non-use values of ecosystem services (Pascual et al. 2010).

4.2.2. Preference-based approaches: Revealed and observed preference methods

Market prices as well as supply and demand data can provide some help in the valuation process of non-market goods. **Revealed preference methods** seek to valuate non-market commodities by studying how they affect the value or consumption of related marketed items. They aim to measure the WTP indirectly, based on actual consumer choices. In other words, the methods search for paid costs which indirectly represent how much e.g. an environmental commodity is valued. *The advantage of these methods* is that they measure actual behavior and are therefore (locally) reliable, but *they are limited* e.g. by available market data.

Examples of these methods include the **travel cost method** and the **hedonic pricing method**. The travel cost method (which will not be treated in detail here) measures the WTP for travel costs required to access recreational resources, such as national parks. The hedonic pricing method values commodities by estimating how they affect the value of e.g. real estates around them (Ahlroth 2009). In hedonic pricing, detailed data is needed about sales transactions and other characteristics of the sold estates around the valuated commodity, as well as some mathematical tools (e.g. linear regression models). For example, biogas stations generally operate with biowaste and/or animal by-products which can cause odor externalities and decrease the prices of nearby houses. Pechrova & Lohr (2016) studied how the distance to biogas stations affected the value of surrounding real estates by gathering prices of 318 real estates located within a 15-mile radius from eight biogas stations in the Jehomoravsky region of the Czech Republic. They found that, on average, the value of real estate seemed to drop by about 0.4% with every kilometre closer to a biogas station. In addition, a US study by Reichent, Small and Mohanty (1992) found that, in Cliveland, Ohio, placing landfills near expensive housing areas had a much greater lowering effect (5.5%–7.3%) on estate values than placing them near less expensive or predominantly rural areas, where there might be no measurable effect at all.

Environmental valuation, in this context, refers explicitly to gathering WTP or WTA values from agents relevant to the study. In other words, the value of an asset, such as a natural resource, is attributed to it by the economic agents relevant to the study in question. Therefore, the results of valuation vary and depend greatly on human preferences, institutions, culture and other socio-cultural aspects of the study, and are not generally transferable to other contexts (Pearce 1993 & Barbier et al. 2009).

Sometimes the alternative term **observed preference method** is used when WTP is determined directly from a market existing for the product in question, instead of examining surrogate markets (Pizzol et al. 2015). As an example, the **market price method** estimates the actual market value of already priced natural resources extractable from e.g. an ecosystem service. Some of the benefits of cleaning up a polluted lake could be estimated with the market price method by estimating the economic value of fish that could be extracted from the lake if it was clean. The objective is to calculate the total economic surplus gained from the target system. This is done by estimating the market demand for the assessed product, using market data on the WTP of consumers, and adding together the consumer and producer surpluses (for more information, see King & Mazzotta 2000).

4.2.3. Preference-based approaches: Stated preference methods

If both direct and indirect price information on ecosystem services are unavailable, hypothetical markets may have to be created (Pascual et al. 2010). So called **stated preference methods** estimate how people value non-market commodities by, as the name suggests, asking them to state their preferences. The most commonly used method is **contingent valuation** in which individuals are asked how much they would be willing to pay for an increase in environmental quality. *The advantage of these methods* is that they allow measuring the kind of nature values which could not be approached through the market. They are also more comprehensive than revealed preference methods since both non-use and use values are acknowledged. Despite their usefulness, several biases may be involved in contingent valuation as well other stated preference methods: results seem to depend on how the questions are asked in the study (**design bias**), respondents might be insensitive to the scope of the valuated commodity (**scope bias**) and might underestimate their WTP if they believe they will actually have to pay (**strategic bias**), or overestimate it if they want the good to be provided (**free-riding bias**) (Ahlroth 2009, Hanley & Spash 1993). With choice modelling, stated preferences are gathered by asking the partakers to rank different alternatives, e.g. visual landscapes (Rambonilaza 2005), in varying ways, such as contingent ranking, paired comparisons and choice experiments. Alternative goods are given different attributes, including monetary cost, and based on the choices made by the respondents, the other attributes can be derived monetary values as well (Ahlroth 2009).

Since applying stated or revealed preference methods is often time-consuming and expensive, ways to integrate valuation results from previous studies have been developed. Due to the highly site-specific nature of non-market good valuation, utilising valuations from other sites should be approached carefully. **Benefit transfer** stands for the practice of using values from certain sites as proxies for another site: the process usually involves adjusting the values based on the socio-economic differences between the sites and their inhabitants (Ahlroth 2009). As an example, the US Environmental Protection Agency (EPA) has heavily relied on benefit transfer methods to assess benefits gained from marginal improvements in water quality, e.g. from reduced groundwater contamination in private wells (Griffiths et al. 2012).

4.2.4. Abatement cost methods

Valuation methods can generally be classified as either WTP methods or **abatement cost methods** (Oka 2005). Analogous terms for the latter are **mitigation, reduction, control, restoration or replacement cost methods**, as they all use the same potential cost approach (Pizzol et al. 2015). After the environmental damages caused by the assessed product is known, the abatement cost method can be used to calculate the costs of reducing a corresponding amount of pollutants or environmental impacts elsewhere in society, or somehow provide a substitute (ecosystem) service (Oka 2005). These methods assume that these "replacement costs" provide a useful minimum estimate of the value of the assessed site. For example, wetlands can act as sieves that filter excess nutrients and dangerous pollutants from water flowing through them, and abatement costs of replacing these ecosystem services could be the costs of industrial filtering and chemical treatment of the water (Michaud 2001). A contrasting approach for the abatement cost method is the **averting cost method**, which measures preventive or offsetting expenses (Pizzol et al. 2015).

5. Environmental accounting methodologies

In environmental accounting literature, there are several different terms, definitions and intepretations of environmental cost accounting methodologies with different system boundaries (Figure 1). System boundaries of accounting also vary depending on if the assessment is done from the perspective of a company or society. Environmental costs and savings are generated to the company or to society during the entire product **life cycle** by different measures relating to air, soil or water protection, waste management, environmental management or prevention of noise and odor.

Environmental and economic **objectives are sometimes conflicting** and the need to include economic parameters to the life cycle assessment (LCA) tools has been recognized in the literature. Environmental weighting can be seen as a step in the interpretation and communication of LCA results, and therefore it is relevant to refer to ISO 14043 [16]: ''...communication has to be maintained through the life cycle interpretation phase. Therefore, transparency throughout the life cycle interpretation phase is essential. Where preferences, assumptions or value-choices are involved, these need to be clearly stated by the LCA practitioner''.

A life-cycle perspective means accounting for the whole life-cycle of a researched subject, often a product or a product system. The full life cycle of a product consists of all the phases gone through by the product and its constituent materials as well as packaging, starting from resource gathering and ending in some sort of waste management or recycling ("cradle-to-grave"). Since acknowledging the entire life cycle requires great efforts and resources and all information may not be of interest to the researcher, partial life cycle analyses are also done: "cradle-to-gate" analyses, for example, do not treat the phases after production, such as product use and disposal (Bierer et al. 2015).

5.1. Life Cycle Assessment (LCA)

Life-cycle assessments generally analyse a system that receives **inputs** and produces **outputs**. Life Cycle Assessment (LCA) refers to environmental life cycle analyses (E-LCA) and studies the environmental impacts that each life cycle phase inflicts on the environment. From business point of view, a company can gain substantial benefit for its activities by understanding the life cycle environmental impacts of its operations and implementing environmental management accounting methods. This knowledge enables tackling the most significant emission sources and, in optimal "win-win" cases, also generates savings through e.g. more efficient processes and reduced energy use. Environmental impacts are examined using the LCA method throughout the whole product chain so that the essential emission sources can be found and preferably tackled. LCA has the potential to pinpoint critical points along the production chain that enable considering the most effective actions to minimize the environmental impacts.

The environmental LCA has a long history and there are several established standards and methods. The ISO 14040 (2006) and ISO 14044 (2006) standards provide the standardised framework for environmental LCA studies. The ISO 14044 standard describes a life cycle analysis framework consisting of **goal and scope definition (system boundaries), inventory analysis (collection of necessary data), environmental impact assessment and interpretation of results**. Some methodological guidelines have also been published, e.g. the International Reference Life Cycle Data System (ILCD) handbook (JRC 2010).

Life Cycle Impact Assessment (LCIA) is the phase of LCA where the data collected during inventory analysis (quantities of materials, energy use, emissions to water and soil etc.) is linked to the respective environmental impact categories. These categories include potentials for e.g. global warming (CO_2-eqv.), eutrophication, acidification, resource depletion as well as human toxicity. The impact categories portray varying kinds of environmental burdens and are measured with different units, which often makes unambiguous ranking of products with different impact profiles impossible. Developing LCIA methods is an ongoing and often complex process, but applying the methods is usually a simple task of multiplying emissions with predefined characterisation factors, obtained from LCIA databases (Jolliet et al. 2015). Examples of often used LCIA databases include Eco-indicator 99, ReCiPe and CML 2001.

Understanding **functional units** is essential for correctly interpreting the results of (especially environmental) life cycle assessments. The functional unit of an assessment ultimately defines what is being studied: it stands for a reference unit for the quantified performance of the researched product system. That is, functional units are used to scale the collected and/or calculated data to a common metric. This is necessary for comparing data within and between life cycle assessments (ISO 14040/44 2006, Heijungs et al. 2013).

Functional units in LCAs of bioenergy systems can be "(1) input unit related (e.g. unit of input biomass or energy unit), (2) output unit related (e.g. unit of heat produced), (3) units of agricultural land (e.g. hectares of agricultural land needed to produce a certain amount of biomass feedstock) or (4) yearly-basis related" (Cherubini & Strømman 2011). An example of an input unit related functional unit in a bioenergy context is the treatment of 1 Mg (tonne) of biomass feedstock (Lu and Hanandeh 2017). For systems with a singular input and multiple outputs, using an input-based functional unit (such as 1 Mg of feedstock) helps to avoid allocation issues.

One challenge might be how to allocate environmental impacts between different products. Inside the system boundaries some processes produce more than one product (i.e. "the main product") and total chain impacts are caused because of these different products (ie. "side products"). If the production processes cannot be separated for every product, there is a need to allocate total system impacts between different products. Most common allocation methods in LCA are:

- mass allocation (based on masses of products)
- **economic allocation (based on market prices of products)**
- physical allocation (based on physical properties, e.g. energy contents of products).

5.2. Environmental Input-Output models

Input-output analysis, developed in the late 1930s, is one of the most widely applied methods in economics. The analysis makes use of "input-output tables" produced by statistical agencies: these tables include the purchases of each industrial sector from all other sectors, i.e. they are "matrices of inter-industrial flows of goods and services" expressed in monetary units (OECD 2017). The **environmental Input-Output (EIO) model** is one of the first indicators developed for environmental management accounting, and especially for accounting the environmental performance and effects of companies. Environmental input-output balance links the economic IO-table data to physical units by comparing all production inputs (used materials and energy) and outputs (emerging products, waste and emissions) of a given period (Kolehmainen & Riuttala 2012).

Input-output financial data is usually well documented and readily available in organisations, which makes IO-based life cycle assessments well suited for internal purposes. Since readily available accounting or other documented data is utilised, IO-LCA is faster to conduct than traditional LCA and can reduce the workload by an order of magnitude (Junnila, 2008). It can be used for a quick screening of environmental hotspots which can be used in decision-making or e.g. early phases of product design. However, sector-specific economic inputs are used to estimate the environmental impacts, which can make the used data too coarse and aggregated for some applications. In these cases, the EIO data can be combined with "bottom-up" data from traditional process-based LCAs (Kjaer et al. 2015).

There are also many benefits to employing input-output methods in aligning Life Cycle Costing (LCC) and environmental Life Cycle Assessment (LCA) methods. Using an EIO model enables an LCA using the same economic input data as LCC. This is because the input-output table can be relatively easily extended into a hybrid database called the Environmental extended IO (EEIO) table which translates the economic inputs into physical units. The EEIO table can be used to link life cycle costs to environmental indicators: this enables the calculation of life cycle impacts per monetary unit for each sector output. The relatively easy translation of an input-output LCC (IO-LCC) into an input-output LCA (IO-LCA) might further help bring environmental considerations into decision-making (Kjaer et al. 2015).

5.3. Life Cycle Costing (LCC)

Life Cycle Costing (LCC) methods cover the cost impacts on teach life cycle phase. The methods collect all the life cycle costs of the chosen project or product and present their total sum, or several possible total sums that vary according to the possible assumptions and alternatives chosen during the life cycle. LCC analyses have the potential to pinpoint critical points along the production chain that enable considering the most effective actions to minimize cost impacts, often through growing energy efficiency and cost efficiency of production and add value. There is an aim for creating higher added value products from traditional biomass production and fractionation in different parts at various stages of the processing chain. Each action should add value to the product or reduce production costs in order to make the development of value chains possible.

Traditional life cycle costing is an investment calculus tool that can be used to rank different investment alternatives (Gluch & Baumann 2004). The basis of LCC theory was properly developed by Flanagan et al. (1989) and Kirk & Dell'Isola (1995) along with the following steps (summarised by Ristimäki et al. 2013) to undertake an LCC analysis:

1. **"Defining alternative** strategies to be evaluated: specifying their functional and technical requirements
2. **Identifying relevant economic criteria**: discount rate, analysis period, escalation rates, component replacement frequency and maintenance frequency
3. **Obtaining and grouping of significant costs**: in what phases different costs occur and to what cost category
4. **Performing a risk assessment:** a systematic sensitivity approach to reduce the overall uncertaint"

ISO 15686-5:2008 gives guidelines for performing life cycle cost (LCC) analyses but only for buildings, constructed assets and their parts. This has been revised by ISO 15686-5:2017 providing requirements and guidelines for performing LCC analyses of buildings and constructed assets and their parts, whether new or existing. ISO 15686 (2008) defines LCC as "a technique which enables comparative cost assessments to be made over a specified period of time, taking into account all relevant economic factors, both in terms of initial costs and future operational costs". The life cycle costs of a system are obtained as the sum of the costs associated with all activities included in a scenario.

The industry for life cycle costing (LCC) is still relatively young, but it is developing rapidly. Many terms used in the field still do not have well-established definitions, no standard or widely accepted detailed specification for any of the terms used when estimating life cycle costs. Interpretations vary substantially in the literature which makes it difficult to clarify what the terms actually imply. The most common terms used in literature is Life Cycle Costing (LCC) and Life Cycle Cost Assessment (LCCA). In order to avoid confusion, we decided to use the term LCC (Life Cycle Costing) in this report.

Life cycle costing can be applied either from a **"planning"** or **"analysis"** perspective. Planning LCCs focus on how economic performance is affected by changes in the system while analysis LCCs are interested in the system at its current state (Martinez-Sanchez et al. 2015). System boundaries define which parts of the life cycle and which processes belong to the analysed system. It is important that all relevant processes and monetary effecs relevant to the respective decision maker are included in the assessment. For LCC studies, classifications like "cradle-to-gate" or "cradle-to-grave" are unusual (Bierer et al. 2015). *The system boundaries* of the LCC naturally depend on the study in question. The SETAC working group has stressed that the functional unit should be consistent with ISO 14040/44 (2006) provisions especially if LCA and LCC are used to study the same system (either consequently or in parallel).

Basic economic tools in LCC are the **time value of money** (interest rate, discounting, present value) and **annuity calculations** (allocation of investments over time). These tools are used to allocate costs correctly and realistically model the viability of investments. In conventional life cycle costing the received cost data is to be indexed, discounted and presented in a net present value (NPV) context as well as divided into annual costs to make each option comparable with each other from a life cycle perspective.

Allocation of emissions in multi-output systems can be challenging and affect both the environmental and economic results when the emission costs are internalised. For example, there is no standard protocol for apportioning the energy inputs or GHG emissions to the heat and power outputs of CHP systems. However, the Energy Efficiency Council (Energy Efficiency Council 2013) suggests using the so-called proportion or exergy method which calculates emission allocations with the following equation:

$$\mathrm{Emissions_{Heat}} = \mathrm{Emissions_{Total}} \times \frac{\frac{\text{Heat output}}{\text{Efficiency (heat)}}}{\frac{\text{Heat output}}{\text{Efficiency (heat)}} + \frac{\text{Electricity output}}{\text{Efficiency (electricity)}}}$$

Here

$\mathrm{Emissions_{Heat}}$ = the emissions share attributable to heat

$\mathrm{Emissions_{total}}$ = the combined emissions of heat and electricity production.

From a life cycle perspective, LCC studies mainly focus on those life cycle phases that are relevant for the respective decision-maker or company. For example, in conventional LCC studies a classification into cradle-to-gate or cradle-to-grave studies is rather unusual. However, in the study by Luo et al. (2009) the system boundaries were incorporating all processes upstream of the delivered energy product (i.e., extraction of raw resources) and proceeding to consumer use.

LCC acknowledges operational costs through the project's life-time, highlighting investment decisions that bring life cycle cost reductions even if an additional increase in the initial investment is necessary. LCC analyses process and simplify huge amounts of information into a common monetary unit, while providing a valuable life cycle perspective (Gluch & Baumann 2004).

On the other hand, the estimations and valuations of LCC assessments are based on uncertain future events and so contain subjective factors which influence the results (Gluch & Baumann (2004). In addition, there are opportunity costs that must be taken into account when costs and savings have income effects and different expenses meeting alternative consumption needs. (e.g. Martinez-Sanchez et al (2016).

Using the life cycle costing method:

1. enables better evaluation of process planning efficiency for companies, by comparing real costs to life cycle budget costs and showing the distribution of these costs to different parts of the life cycle (Clinton & Graves 1999, Dunk 2004).
2. improves the capacity of companies to make better pricing solutions/decisions (Adamany & Gonsalves 1994).
3. improves the evaluation of production efficiency (Hansen & Mowen 1992).
4. helps to design more environmentally friendly products (Kreuze & Newell 1994, Madu et. Al. 2002).

5. improves the understandability of environmental impacts and their generation throughout the life cycle (Sutton 1992, Weltz et. Al. 1994, Brady et. Al. 1999)
6. helps to focus on post-production phases, including warranties, component costs, services and upkeep. The importance of post-consumer phases has grown in consumer purchase decision-making as has their share of the total life cycle costs (Shields & Young 1991, Murthy & Blischke 2000).

In recent studies, Life Cycle Costing (LCC) can be seen to consist of three different methods: **conventional LCC (C-LCC), environmental LCC (E-LCC) and societal LCC (S-LCC)** (Hunkeler et al. 2008). The three types of LCCs (see more in chapter 5.3) offer an overall framework for systematic economic assessments either in combination with LCAs or, in the case of C-LCC and S-LCC, as stand-alone indicator. Each of the three LCC types supports also different goals. A consistent and comprehensive LCC framework for the economic assessment of systems can be achieved by (modified from Martinez-Sanchezin et al. 2015):

1. developing systematic cost models for all main activities related to system based on transparent technical parameters associated with the involved technologies,
2. implementing the cost model framework on case study examples illustrating the system, and on this basis
3. evaluating applicability as well as identifying critical methodological aspects related to LCC on the targeted system.

5.3.1. Conventional LCC (C-LCC)

Conventional life cycle costing (LCC) methodology is developed only for financial analysis so it is purely economical and, in general, only accounts for the environmental aspects that are manifested directly as internal costs. This kind of LCC is called traditional, conventional LCC (C-LCC), financial LCC (f-LCC) or economic LCC, depending on the source. Traditionally, LCC has been applied to financial assessments (i.e. accounting for marketed goods and services) carried out typically by individual companies focusing on their ‘’own’’ costs and is for the assessment of direct internal costs, private costs and savings only. They may often exclude specific parts or costs of the life cycle: for example, externality costs of environmental impacts are excluded and typically included only in socio-economic assessments (see chapter 2.2) (Nordic Council of Ministers 2007).

In C-LCCs, **functional units** are not always explicitly stated. Conventional LCCs have been traditionally carried out separately from the LCA (though exceptions exist, e.g. Mohamad et al. 2014), employing different assumption, functional units (if any) as well as system boundaries, and their results cannot thus be presented together (Norris 2001, Carlsson Reich 2005, Hunkeler et al. 2008, Swarr et al. 2011).

According to Hunkeler et al. (2008), C-LCC is the assessment of all costs associated with the life cycle of a product that are directly covered by the main producer or user in the product life cycle. **The assessment is focused on real, internal costs, sometimes even without end-of-life or use costs if these are borne by others.** A C-LCC usually is not accompanied by separate LCA results. The perspective is mostly that of 1 market actor, the manufacturer or the user or consumer.

According to Martinez-Sanchez et al. (2015), C-LCC includes the sum of the budget costs and transfers (see cost types in chapter 2.1. and 2.4.) for activities involved in the scenario. A term **budget cost** is meaning the same as direct internal costs in traditional business accounting (see chapter 2). Budget costs can either occur only once in the lifetime of an investment, or be recurring (for example operational and maintenance costs) and are accounted for in factor prices. As stated by Martinez-Sanchez et al. (2015), C -LCC is commonly used in order to:

1. “assess the economic feasibility/viability of treatment solutions (for example Coelho and De Brito 2013 & Franchetti 2009)
2. identify the economically best-performing solution (for example Karagiannidis et al. 2013; Groot et al. 2013)
3. evaluate the economic consequences of implementing a specific waste solution (for example Gomes et al. 2008)”.

According to the study of Martinez-Sanchez et al. (2015) about waste management systems, the C-LCC can be presented as a sum of the budget costs and transfers for all the n activities included in the scenario. Every activity (such as source separation, waste collection, transportation, treatment and disposal) is disaggregated into relevant cost items (such as machinery, salaries, fuel or maintenance costs) which in turn are divided into budget costs and transfers. Martinez-Sanchez et al. (2015) presented the costs as “euros per tonne of waste input” and combined them with the total waste input (in tonnes) of each activity. The total life cycle costs were then presented as the following sum:

$$\text{Conventional LCC} = \sum_{i=1}^{n} [W_i * (UBC_i + UT_i)]$$

Here
W_i = waste input of activity i (in tonnes),
UBC_i = unit budget cost of activity i (in euros per tonne of waste) and
UT_i = unit transfer of activity i (in euros per tonne of waste).

The conventional LCC has also been used as a parallel/complementary analysis tool to an LCA: they both analyze the same problem, but from different aspects (Martinez-Sanchez et al. 2015), and the C-LCC might leave out some costs and stakeholders that are treated in the LCA. According to Carlsson Reich (2005), financial LCC can be used to add another ‘‘effect category’’ to the LCA results, namely the economic dimension: they are then used as complementary tools and no monetary valuation of environmental aspects is done.

5.3.2. Environmental Life Cycle Costing (E-LCC)

Environmental life cycle costing (E-LCC) is an extension to the conventional LCC method.It still mainly focuses on internal costs but is especially designed to be complemented by an environmental LCA. That is, E-LCC should always be accompanied by an LCA and serves as the economic part of the environmental-economic assessment. Despite the term “environmental”, E-LCC still also includes all conventional life cycle costs, but unlike C-LCC, it also acknowledges and accounts for such environmental externalities that are expected to be internalised during the project-relevant time perspective, due to legislative (e.g. environmental taxation) or other causes. However, E-LCC only needs to calculate externalities that will probably be manifested as actual costs (e.g. environmental taxation) to the relevant agents of the study. To make E-LCC compatible with the LCA, both methods need to have the **same system boundaries**, i.e. the system boundaries of C-LCC must be extended to meet those of the LCA, as well as the **same functional unit** (Hunkeler et al. 2008). According to De Menna et al. (2016), the E-LCC may add or leave out one or more stakeholder or actor, and have a different goal and scope than the complementary LCA.

Economic and environmental systems are generally built differently. The economic chain is often cut off by economic borders that do not exist (or are ignored) in logical LCA systems, and vice versa. Therefore it is important to keep both the economic and environmental systems in mind when combining assessments with environmental and economic life cycle perspectives: often the economic framework must be turned into a hypothetical system which diverges from existing economic systems (Carlsson-Reich 2005).

According to Hunkeler et al. (2008), environmental LCC assesses all costs associated with the life cycle of a product or project that are directly covered by one or more of the actors in the product life cycle (supplier, manufacturer, user or consumer, and/or EoL actor). This is another difference to C-LCC which more often focuses on the costs of one actor only. All the phases of an LCA (ISO 14040) below can be (with small variations) adapted to the environmental LCC:

1. Goal and scope definition: this step provides the context for the assessment and defines the functional unit, system boundaries, assumptions, impact categories and allocation method selection.
2. Inventory: All resources extracted from the environment and emissions released into the environment along the whole life cycle of a product are inventoried. In E-LCC, this phase consists of cost information gathering.
3. Impact assessment: Inventory results are translated into impact categories (midpoint or endpoint) with the help of an impact assessment method. This means that all elementary flows within same category (e.g. climate change) are converted to a common unit using characterization factors.
4. Interpretation: In this step, the results of the inventory and impact assessment is checked and evaluated. It should generate a set of conclusions and recommendations.

In E-LCC, phase two translates to cost information gathering and phase three into identification of cost hotspots. The environmental and economic hotspots of the subject in question are then compared and and examined in relation to each other in the interpretation phase (Hunkeler et al. 2008).

Environmental LCC still has no international standard and its precise character varies among studies. For example, the act of adding together the costs of many actors in E-LCC has been a source of some confusion among researchers. Since a cost for one actor is revenue for another, all of the incurred costs can not simply be added together since that would result in double counting: some costs need to cancel each other out. Furthermore, the demand of identical system boundaries and differences in treatment of time between LCA and LCC methods cause challenges for combining data and presenting LCC and LCA results together.

After Hunkeler et al. (2008) and Swarr et al. (2011) laid out the general framework for conducting environmental LCC, Heijungs (2013) noted that their work did not include a clear and precise form for its computational structure, and had only few concrete formulae. Such a structure (which was omitted from this report due to its technicality) was then formulated by Heijungs (2013) and later improved by Moreau & Weidema (2015), based on matrix equations as well as **value added** during different phases of the life cycle of the studied product. Added value is the value of output minus the value of all intermediate inputs, and it represents the contribution of, and payments to, primary factors of production (Deardorff 2014). According to Moreau & Weidema (2015), the total life cycle cost in E-LCC should then be “the sum of the value added for each activity in the product life cycle for each and every actor involved, including externalities which are foreseen to be internalised in the decision-relevant future”.

There are still some misunderstandings and terminological differences in the literature, and the field of environmental life cycle costing seems to still be relatively under-developed. There is also a lack of details in studies which limits transparency and the general applicability of the results. According to Martinez-Sanchez et al. (2015) this is illustrated by many studies which lack details cost calculation principles and clear definitions of used terminology, system boundaries and assumptions. Waste management related exceptions include the study of Groot et al. (2013) which developed a comprehensive cost model to calculate expenses of plastic waste source separation, and Eriksson et al. (2005) which uses transparent and clear definition for assessing the welfare economics of easily degradable waste, plastic and paper.

According to Martinez-Sanchez et al. (2015), E-LCC expands C-LCC by adding future externalities priced by authorities as transfers (see 2.4), such as environmental taxes for emissions and energy use. However, sometimes the costs included in the E-LCC might be very similar to those of C-LCC, if no externalities are considered necessary to include in the calculations and there are only few actors in the system. However, Hunkeler et al. (2008) recommend that, for comparison with the long-term effects of the LCA, E-LCC should be kept as a steady-state system, meaning its results are time-invariant and no discounting of its results is done. More comprehensive calculations of externalities is done in societal life cycle costing (S-LCC) (see ch 5.3.3.).

Martinez-Sanchez et al. (2015) provided a detailed and comprehensive cost model allowing calculation of E-LCC. The main purpose was to show the applicability of the cost model, not to give a deep analysis. To calculate the environmental life cycle costs of a waste management system, anticipated future transfers are added to the conventional LCC formula. As presented by Martinez-Sanchez et al. (2015) for a total of n activities:

$$\text{Environmental LCC} = \sum_{i=1}^{n} [W_i * (UBC_i + UT_i + UAT_i)]$$

Here
W_i = waste input of activity i (in tonnes),
UBC_i = unit budget cost of activity i (in euros per tonne of waste),
UT_i = unit transfer of activity i (in euros per tonne of waste) and
UAT_i = unit anticipated transfer of activity i (in euros per tonne of waste).

5.3.3. Societal Life Cycle Costing (S-LCC)

Societal life cycle costing is the most comprehensive of the life cycle costing methods. It expands the E-LCC method by adding the costs of externalities that *could be relevant in the long term* for both the stakeholders directly *and indirecly* affected by them. This differs from E-LCC which focuses on externalities that will *probably and directly* (monetarily) affect the main stakeholders of the system (Hunkeler et al. 2008). Typically, S-LCC is used to examine the economic efficiency of projects or scenarios on a societal level. Societal LCC **connects environmental and social aspects in monetary terms** and can be described as a ''socio-economic'' or ''welfare-economic'' assessment. That is, it evaluates environmental and social impacts by monetarising (see more in chapter 2.2 & 4.2) the respective effects from a societal perspective (Martinez-Sanchez et al. 2015). Unlike E-LCC, S-LCC does not include transfer payments, such as taxes or subsidies, because they are considered to happen inside the system and cancel each other out. Since S-LCC aims to include all environmental and social effects, it is not accompanied by an LCA or other additional assessments but is considered a stand-alone method (Hunkeler et al. 2008). As a method it is similar to and borrows from social life cycle analysis (S-LCA) and cost-benefit analysis (CBA), which are older methods: the similarities and differencies are explored in chapters 4.4 and 4.5.

Societal LCC can qualitatively consider externalities that are not easily monetarised, such as biodiversity damage as well as effects on social well-being, human rights and public health. Category-specific non-monetary scoring methods can also be used. Since especially the quantification of social effects has high uncertainty, it is recommended by Hunkeler et al. (2008) to present the social impact assessment score and other impact categories separately (disaggregated) and carry out sensitivity analyses (as would also be done with an LCA by itself).

There are no strict rules on how societal LCC costs should be calculated since the method is still new and under development. As an example, the study of Martinez-Sanchez et al. (2015) calculated the societal life cycle costs of a waste management system as the sum of budget costs and externality costs in accounting prices, depicting society's willingness to pay for the considered services (see chapter 4.2). The study did this by converting the factor prices (market prices from which transfers are excluded) to accountingprices by multiplying them with a so-called "net tax factor" (NTF) of 1.17, proposed by the Danish Ministry of Finances. The NTF depicts the average net tax for the whole economy, and is used in socio-economic assessments to calculate a theoretical WTP for goods that were never produced and do not have a real market value (Nordic Council of Ministers 2015). The formula for the total costs for a total of n activities was then:

$$\text{Societal LCC} = \sum_{i=1}^{n} [W_i * (UBC_i * NTF + UEC_i)]$$

where
W_i = waste input of activity i (in tonnes),
UBC_i = unit budget cost of activity i (in euros per tonne of waste),
NTF = Net Tax Factor and
UEC_i = unit external cost of activity i.

5.4. Social Life Cycle Assessment (S-LCA)

Social life cycle assessments can internalise and monetarise same impacts as societal LCCs, but its focus is more on social welfare issues. In addition, some S-LCA studies assess social costs and therefore overlap with the calculations of S-LCCs so that a risk for double accounting exists when doing comparative assessment between S-LCC and S-LCA or integrating the results. According to Nordic Council of Ministers (2007), traditionally social costs (i.e. the sum of private and externality costs) are included in socio-economic assessments and companies' private/internal costs are addressed in financial assessments (see chapter 2).

Social life cycle assessment (S-LCA) can be used to incorporate social and socio-economic dimensions to the analysis while keeping the same methodological steps as LCA. Social LCA collects data on possible social drivers of impacts for each stage of a product life cycle and assesses these impacts based on calculations of relevant indicators of e.g. poverty, cultural heritage, human rights, child labour, worker safety and the health of the community (Sala et. al. 2015).

The environment and the human-industrial sphere, economic exchanges and socio-economic conditions also affect human health and conditions. However, the focus of traditional life cycle assessment is not primarily on the well-being of humans, unless there is damage to people caused by environmental impacts (Hoogmartens et al. 2014).

In social LCAs, social and socio-economic issues are often classified using so-called **stakeholder categories and impact categories**. Stakeholder categories are defined as clusters of stakeholders that are expected to have shared interests due to their similar relationship to the investigated product systems. These categories include e.g. workers, society, consumers and the local community. Social LCA impact categories include human rights, working conditions, health and safety, cultural heritage, governance and socio-economic repercussions. A central challenge in s-LCA is that the social impacts are closely connected to the certain processes and companies and are not easily expressed per unit of process output. However, the indicator values can be weighted and aggregated together using an **activity variable** (e. g. monetary value or worker hours). Activity variables are measures of process activity or scale which can be related to the output in each process and therefore to the functional unit (UNEP/SETAC 2009).

In terms of methodologies, application and harmonisation, s-LCA is not yet as developed as e-LCA and LCC. A core issue with s-LCA is the consistency among standards between studies. In general, s-LCA practitioners need to gather large shares of qualitative data: social LCAs have a high site-specific nature and there are few databases for specific social and socio-economic impacts. Numeric data is useful but its meaning will often have to be interpreted with additional information to estimate social impacts. Wages of a particular enterprise or national minimum wage levels, for example, could need extra information to tell if the wages are livable (UNEP/SETAC 2009).

5.5. Cost Benefit Analyses vs. S-LCC

Cost-benefit analysis (CBA) has its theoretical roots in welfare economics and has been used already in the early 1900s to assess the financial attractiveness of projects (Hoogmartens et al. 2014). Since the late 1960s, it has also been practised in environmental policy planning to evaluate social and societal effects of investments and policies. In the context of CBA, benefits and costs are defined as increases and decreases, respectively, in human wellbeing: a project is qualified on cost-benefit grounds if its social benefits exceed its social costs (OECD 2006). The aim of CBA is to merge all decision-relevant welfare-related factors into one number, the net present value (NPV) of the project (Hoogmartens et al. 2014).

Similarly to the life cycle methods discussed above, cost-benefit analysis methods can as well be crafted to focus on financial, environmental and/or social aspects. **Financial CBA (fCBA)**, which will not be further treated in this report, is for private profitability assessment and is concerned with the discounted cash flows of only one actor. **Environmental CBA (eCBA)** adds to this the challenging task of valuating environmental external costs such as ecosystem damage, pollution and damage to neighbours, as well as integrating them to the traditional fCBA results. **Social CBA (sCBA)** is most clearly a public assessment tool for welfare effects on a societal level, and includes e.g. health costs (including sickness caused by e.g. air pollution), recreational benefits and safety deteriorations (Hoogmartens 2014).

The differences between cost-benefit studies and different life cycle assessments cannot always be clearly defined, and their contents can overlap. As methods, life cycle analyses and costing are more often product-related (excluding IO-LCA and hybrid LCA which treat e.g. product sectors) while CBA focuses on the benefits and costs of projects and policies (Ness et al. 2007 & Rorarius 2007). LCA and LCC studies often compare competing products while CBA is usually used for autonomous project evaluation. There are many similarities between s-LCC and CBA: indeed, s-LCC uses some of the techniques originally developed for CBA (contingent valuation, hedonic pricing etc.) and s-LCC results can also be used as input to CBA (Hoogmartens et al. 2014). Cost-benefit analysis frequently uses information from environmental LCAs as “physical counterparts” of the Environmental Impact Assessment (EIA). Both CBA and LCA start with an EIA, but CBA more often adds the step of monetising the assessed impacts and LCA generally goes further in paying attention to impacts through the whole life cycle (Pearce et al. 2006 & Tukker 2000).

There are also different interpretations of multiple impact categories. For example, S-LCA and S-LCC often consider need for labour a benefit since additional jobs are being created and social welfare generated, while sCBAs consider labor a cost. Job creation also has external effects which are generally positive if the labor markets in question have high unemployment rates, but can also be negative if there are already too many workers in the field (Bartik 2012 & Masur and Posner 2012).

Being an economic tool, the benefits and costs of CBA are expressed in monetary terms, meaning that monetary valuation is needed when there are no ready market prices for the considered asset. The choice of whose benefits and costs are included in the analysis, must be assessed separately. It is also important to consider how the project, if implemented, would affect other policy areas or projects as well as the competitiveness of nations and/or industries (OECD 2006).

6. Results for integrating economic and environmental life cycle dimensions

This literature review explored studies including externality (environmental impact) cost valuation, comparative analysis of C-LCC and LCA as well as more comprehensive E-LCC assessments where C-LCC was expanded to be consistent with the system boundaries of the LCA. Some studies also assessed S-LCC where externalities (environmental and social impacts) are monetarized (i.e. by utilising LCA and S-LCA results) as externality costs to the respective effects from a societal perspective by using e.g. accounting prices.

As mentioned in chapter 4, LCC and environmental LCA have been traditionally carried out separately with different system boundaries and functional units, and therefore the results have not been presentable together (Norris 2001, Carlsson-Reich 2005, Hunkeler et al. 2008, Swarr et al. 2011). A more comprehensive perspective on sustainability is achieved by conducting mutually compatible LCC and LCA together for the entire value chain. More specifically, LCC helps in the ranking of investment decisions when the operational phase costs have significant impacts on the total life cycle costs. LCC assessments highlight investment decisions that can bring life cycle cost reductions for the operational phase, even if an additional increase in the initial investment is necessary (Ristimäki et al. 2013, Gluch & Baumann 2004). Overall, assessing LCA in comparison with LCC shows the environmental impacts and life cycle costs together, enabling clear comparison of the results while revealing potential hotspots for cost and impact reductions.

Economic-environmental assessments create opportunities for finding the most critical points for costs and environmental impacts in the product chain. By showing these critical points, these methodologies provide also an indication of what strategic options and aspects should seriously be considered to most effectively optimise and minimise environmental impacts and costs. Comparing different chain scenarios and options with LCC and LCA helps to find most optimal options from an environmental management perspective to reduce costs and to create more value while at the same time accounting for the environmental impacts of different solutions and impact mitigation potential.

In addition, sorting out together both LCC and LCA for the entire chain and comparing them is helping to see the possible **correlations** between environmental impacts and costs as well as cost saving and impact reductions. This helps to better understand the direct and indirect connections between economic and ecological perspectives in sustainability assessments.

According to Carlsson-Reich (2005), the logical boundaries for an environmental and economic analysis sometimes differ.The definition of an object of analysis can be difficult between the two method approaches. According to Martinez-Sanchez et al. (2015), in practice the system boundaries have not always been equivalent between the economic and environmental parts of assessments, which has made the interpretation of their results difficult. Also, due to differences in framework conditions, published LCC studies naturally reach a variety of conclusions, transparency is limited and results are subsequently not applicable for new studies. Few of them include details of cost calculation principles for the involved technologies, details on assessment focus, definitions of system boundaries and assumptions, or clear, transparent terminology for describing assessment principles.

However, with some cases system boundaries were the same. For example, Mohamad et al. (2014) had the same system boundaries in their study, consisting of a partial (cradle-to-gate) LCA combined with C-LCC. Daylan and Ciliz (2016) study conducted a “cradle-to-wheel” LCA of lignocellulosic second-generation bioethanol combined with a simple environmental LCC.

In addition, also the functional unit also needs to be the same so that the data and the results between two or more systems’ life cycle assessment can be compared (ISO 14040/44 2006 & Heijungs et al. 2013). According to Hunkeler et al. (2008), the environmental results such as energy needed for production, GHG emissions and eutrophication effects as well as life cycle costs (labour, machinery etc.) can both be presented “per functional unit”, which makes comparing the economic and environmental results easier. For example, Mohamad et al. (2014) presented the life cycle costs and environmental impacts per 1-ha olive-growing area while Daylan & Ciliz (2016) presented them per one kilometre travelled with a middle-sized flex-fuel vehicle. However, there is no uniformity concerning chosen functional units (e.g. whether using per kilogram of final produc or per hectare of growing area) and results from studies with different functional units can sometimes be difficult to compare.

The chosen functional unit affects the environmental and economic results of studies and can sometimes lead to misleading conclusions. For example, using a functional unit such as production of 1 MJ of fuel (output-related) is not recommended in studies on transportation biofuels because there is variation in the mechanical efficiency of different fuel types (Cherubini et al. 2009). Singh et al. (2010) and Campbell et al. (2011) have suggested that, instead, a "per vehicle-km" functional unit should be used: this way mechanical efficiency is considered and the results can be compared with conventional fossil fuels. It was seen that that the functional unit is essential when interpreting results. As another example, the study of Mohamad et al. (2014) favoured organic olive agriculture over non-organic production but since the productivity of non-organic olive trees is 1.58 times higher (40.8 kg/tree/a and 25.8 kg/tree/a respectively), the environmental impacts of the increased land area required in organic farming may have been understated in this study, especially since "one hectare of olive growing area" was used as a functional unit instead of an output-related functional unit, such as "1,000 kg of produced olives".

6.1. Externality cost valuation

Many externality types are difficult to measure, and new measuring units (that can be difficult to understand) have been developed to help quantify them. In some valuation methods, such as Stepwise2006, human health and ecosystem quality (or biodiversity loss) are measured in **Quality Adjusted Life Years (QALY) and Biodiversity Adjusted Hectare Years (BAHY)**, respectively (Nguyen et al. 2016). These physical scores are given monetary prices, ideally based on location-specific data. QALY stands for a life-year lived at full well-being: the monetary value of QALY is chosen to be equal to "the potential average annual income" (at full well-being) since it is considered the maximum an average person can pay for one life-year. For example, Weidema (2009) calculated a value of 74,000 EUR2003 (the 2003 value of euro) for one QALY in Denmark, with an uncertainty range of 62,000–84,000 EUR2003. BAHY, in turn, has been developed to measure biodiversity loss and can be valuated in terms of QALY as the fraction of well-being an average person is ready to sacrifice to protect the ecosystem. Weidema (2009) used 1,400 €/BAHY as a temporary proxy value with a very large uncertainty range of 350–3,500 €/BAHY, and stated the need for a future choice modelling study for estimating it more accurately. However, the uncertainty of this value has not been acknowledged in all subsequent research: e.g. Nguyen et al. (2016) used it as a "suggested value" without stating the uncertainty range.

Some studies have assessed the potential of bringing positive and negative externality costs into the prices of products as taxes. Nguyen et al. (2016) used three European monetisation models (EPS 2000, Ecotax and Stepwise2006) to monetise externalities of generating electricity either from renewable (e.g. biomass) and non-renewable (e.g. coal, oil or natural gas) sources in Denmark. They then weighted the possible impacts on prices and the economy if these externalities were internalised either as corrective taxes or modified (i.e. "green") VAT. The study calculated the environmental costs and benefits of using burning straw (a by-product of cereal production) for electricity production instead of fossil fuels from a CHP plant. They found that the three externality valuation methods provided results so differing that the relative rankings depended on the method used. However, the study used the highly uncertain 1,400 €/BAHY value of Weidema (2009) for ecosystem impact valuation, adding to the uncertainty of the end results. Despite this, the study concluded that **internalising the externality costs of electricity would remove the price disadvantage of renewable electricity**, making in most cases it a preferable choice to the fossil-based alternative. This would discourage consumers from buying environmentally unfriendly goods and therefore have negative impacts on the economy in isolation, but the net consequences could be economically positive if the revenue generated from these taxes would be used for e.g. lowering income tax rates or lowering employment insurance premiums. However, renewable electricity did not in all cases receive clearly lower prices than natural gas. For example, according to the Stepwise2006 method, straw should receive a subsidy of 26% in green VAT, which would make the biomass price competitive with coal and oil but not with natural gas.

Patrizio et al. (2017) quantified externality costs of the environmental impacts caused by airborne emissions related to biogas-based energy and their corresponding fossil substitutes. The authors monetised environmental damages associated with various pollutants, including welfare losses from general emission impacts: however, the externality costs of e.g. eutrophication, impacts on water and acidification were not included in the study. Pollutant-specific damage cost factors were estimated with the EcoSenseWeb software, which was developed as part of the ExternE program. All phases of the supply chain, including farming of the biomaterial, were accounted for, and the total (internal and external) costs were analysed with a spatially explicit optimisation model called BeWhere. The BeWhere model constructs least-cost biogas supply chains to optimise plant locations, capacity and conversion technologies. The results of Patrizio et al. (2017) showed that the externality costs of biogas were only slightly lower than those of the fossil alternatives, or even larger in the local scale if the biogas was allocated to local heating. This is largely due to the high impacts of the farming processes which are often left unconsidered in similar assessments since biogas is generally prepared from organic waste material. The results support the idea that the global food waste problem cannot sustainably be solved just by turning the waste into biogas.

According to the hedonic pricing analysis by Chen (2017), river restoration can reverse negative externalities caused by polluted waters to positive externalities. The study assessed housing prices near watercourses which have been restored during the last decade in Guangzhou, China, where the degradation of rivers had previously become a serious threat for sustainable urban development. Extensive sets of apartment transaction data were acquired from real estate agent companies and processed to minimise the effects of locational attributes and other changes in the treated areas (in addition to the river restorations) that might have impacted housing prices. The results of the study showed that apartment values had risen by up to 4.61% after the restoration, reflecting a preference for greening riverscapes among the local residents.

As another example of hedonic pricing, Pechrova & Lohr (2016) studied how the distance to biogas stations affected the value of surrounding real estates by gathering prices of 318 real estates located within a 15-mile radius from eight biogas stations in the Jehomoravsky region of the Czech Republic. They found that, on average, the value of real estate seemed to drop by about 0,4% with every kilometre closer to a biogas station. In addition, a US study by Reichent, Small and Mohanty (1992) found that, in Cliveland, Ohio, placing landfills near expensive housing areas had a much greater lowering effect (5,5%–7,3%) on estate values than placing them near less expensive or predominantly rural areas, where there might be no measurable effect at all.

Dupras et al. (2017) used contingent valuation and choice experiment to assess the WTP of farmers and citizens for improving the environmental situations of agricultural areas. The study focused on valuating “landscape aesthetics”, which can refer to open views, crop diversity, interesting architectural elements, diversity of land use as well as personal attributes, such as emotional attachment to the area and family heritage. The environmental improvements concerned the quality of water and wildlife habitats in 10 agricultural sites located in Quebec, Canada. Participants were asked to state their WTP for environmental improvements and to evaluate different landscape variations, shown as modified images representing possible future scenarios for the local areas with differing levels of intensive agriculture. After dismissing irregular results (e.g. unreasonably high WTPs), the study showed that more than half of the respondents were ready to pay for practices that would improve landscape aesthetics.

6.2. Comparative analysis of LCA and LCC

Some studies have made comparative assessments for finding correlations between costs and environmental impacts through combined LCA and LCC use. Luo et al. (2009) presented a comparative life cycle assessment using LCA and LCC with same system specification on gasoline and ethanol as fuels and with two types of blends of gasoline with bioethanol from sugarcane in Brazil. A steady-state cost model was used in LCC, i.e. no discounting and depreciation was done. Also, only the production costs were taken into account, to provide a first indication of the economic feasibility of the process. Luo et al. (2009) stated that while in the real market the prices of fuels are heavily dependent on taxes and subsidies, technological development can help in lowering both the environmental impact and the prices of the ethanol fuels. The functional unit in this study is defined as power to wheels for 1 km driving of a midsize car. All relevant processes were included within the boundary of the fuel systems. Data was obtained from literature reports, databases and Ecoinvent or was estimated by using methods in reports or assumptions were made in case of data unavailability. The LCA results show that the overall evaluation of fuel options depends on the importance attached to different impacts. It was observed that ethanol fuels are better options than gasoline in terms of e.g. GHG emissions while gasoline is a better fuel where e.g. eutrophication is concerned. In addition, the LCC results show that in all three scenarios driving on ethanol fuels is much cheaper in both base and future case (however, the outcomes depend very much on the assumed price for crude oil).

Some studies claim that using a combined LCC and LCA approach can show which systems are preferable from both the environmental and economic viewpoints. For example, in the study of Resurreccion et al. (2012), algae cultivation methods for bioenergy production were compared by using a combined LCA and LCC approach. With algae there is no food versus fuel competition (Passos et al. 2013) and algae cultivation has been seen as an attractive alternative for energy production due to e.g. low emissions of its production. However, industrial scale production has not been viably achieved due to high prices and technological challenges (Pathak et al. 2015). The analysis considered all phases from plant construction to transport ("cradle-to-wheel") but was still considered only a partial LCA by the researchers since it leaved out some of the internationally recognized LCA impact categories such as photochemical ozone depletion and acidification. The LCA models were complemented with LCC, accounting for startup costs, revenues and expenses associated to the operation, cultivation and processing in each of the four models involving photobioreactors (PB) and open pond (OP) systems in fresh and brackish-to-saline water (BSW). The results showed that open pond systems are preferable both from the environmental and economic viewpoints. The systems were assumed to have a 30-year useful life. Salvage values at the end of the system useful lives were considered minimal and were ignored as were environmental remediation services, such as removal of N and P from wastewater, since the researchers had no basis for estimating their value on the market. The importance of non-energy byproducts (e.g. water treatment and fish meal) was also considered for each case. Results of the study showed that BSW systems support denser algae growth and so generate biomass with greater energy density. Open pond systems with BSW were economically most viable, although none of the systems were yet profitable. Still, sensitivity analyses showed which systems had most potential to increase their profitability index through e.g. better digestion and methane production efficiency. Economically, the market price of biodiesel and discount rates were the most important factors and therefore subsidies or other financial incentives could therefore improve the profitability of algae biodiesel.

Some studies combining LCA and LCC results aimed to give tools and valuable information for decision makers. Ristimäki et al. (2013) conducted both LCC and LCA and cross-examined the results to see if residential development can bring simultaneous environmental and economic benefits (i.e.

sustainable viability) over plain fossil-based district heating via geothermal heat pumps and/or building integrated photovoltaic panels. This was done to add valuable information for decision makers and future residents. LCA and LCC were done separately but together complemented each other. The LCC and LCA were divided into the construction phase and the use phase. The construction phase of the LCC includes investment costs and the use-phase includes estimated costs of energy consumption, operation, maintenance and component replacement schedules. The results showed that economic and ecological aspects clearly support each other from a life cycle perspective and at the same time contradict the investment-cost approach. For example, district heating had the highest GHG emission levels and life cycle costs in all life cycle times, though its initial investment costs were the lowest (partly due to existing infrastructure in the area). The results also showed that by selecting a slightly higher investment, a significant proportion of energy costs and emissions could be avoided. Combining economic and ecological dimensions can complement each other in residential development since lower energy consumption leads to lower running costs.

According to Carlsson-Reich (2005), there might be difficulties in practise that make aligning LCA and LCC tools very difficult. These difficulties, which stem from the differences in dealing with timing of flows and in system boundaries, are presented in more detail in the study. The tools chosen for combining LCA and LCC depend on what data is needed and possible to gather as well as the decision maker's preferences. The relevant parts of the value chain vary depending on the question and the primary beneficiary or decision maker posing it. For example, Kuisma et al. (2013) determined biorefining efficiency according to the choices made in the entire value chain.

According to Martinez-Sanchez et al. (2015), the lack of a balanced economic evaluation restricts the value of traditional environmental LCA in the eyes of decision makers, as it detaches the economic priorities from the environmental point of views. E-LCC and S-LCC methods strengthen the potential of life cycle management in the early design stages of urban development. According to Ristimäki et al. (2013), combining LCC and LCA portrays a life cycle management perspective and supports decision-making on a long-term basis. Enhancing the position of life cycle management can help to identify and implement profound sustainable solutions.

Mohamad et al. (2014) combined a partial (cradle-to-gate) LCA and LCC to compare organic and conventional olive agricultural practises in Italy. The LCC and LCA had the same system boundaries and functional units, but since the economic part did not account for any externalities, the LCC was still labeled conventional. Other studies, e.g. Daylan & Ciliz 2016, have however used the term E-LCC even without externality valuation while Lu & Hanandeh 2017 did not use the term even though they calculated carbon prices that are currently external.

The study of Mohamad et al. (2014) aimed to identify environmental and economic hotspots and compare different scenarios of both organic and conventional practices, for potential optimisation of olive agricultural practises. The LCA used three end-point damage categories as human health, ecosystem quality and resources depletion. The LCC considered revenues (net present value and internal rate of return) as well as most of the costs (the initial investment costs, operational costs, input prices and wages, olive market prices and subsidies). Taxes were omitted since only some of them were mandatory in the region and others concerned the farm as a whole and were difficult to allocate to olive cultivation practices. Results favoured organic olive agriculture both in terms of lower total environmental impacts and profitability. Organic practices contributed less to resource depletion due to lower fossil fuel consumption, especially during weed and pest control activities which are more intensive and machined in conventional practises. Also, the net present value and internal rate of return were also higher in the organic system, reflecting a better investment due to the 25% higher market prices of organic olives and subsidies for organic farming. However, without subsidies organic olives would need to be 36% more expensive than conventional olives (with unaffected sales) to be profitable. The total operating costs were higher in organic than in conventional agriculture: pruning, fertilisation and soil management were significantly more expensive in the organic system, although conventional weed and pest control costs were higher. Also, organic manure fertilisation resulted in higher costs as well as higher environmental impacts on human health and ecosystem quality than synthetic fertilisation. Fertilisation had the highest environmental impact of all the agricultural activities in both organic and non-organic systems.

Daylan & Ciliz (2016) assessed and compared the production costs and life-cycle environmental effects of conventional gasoline (CG) and second generation (made of farming residues) bioethanol E10 and E85 (10% and 85% of bioethanol mixed with gasoline). Consequently, the agricultural production of bioethanol feedstock was not considered in this study. The study found out that climate impact potential (measured in CO_2eqv) was reduced by 4,7% with E10 and as much as 47,1% with E85 when compared to gasoline. Using E85 also resulted in lower acidification and stratospheric ozone depletion potential. In contrast, however, E85 was the highest contributor to aquatic and terrestrial eutrophication potential as well as photochemical oxidant formation (e.g. ground-level ozone formation) which agrees with previous research (Luo et al. 2009). System processes were categorised into three subsystems: 1) feedstock acquisition, 2) bioethanol production (logistics, distillation, dehydration, saccharification, co-fermentation, related wastewater treatment etc.) and 3) combustion of the fuel blends. Since fuel market prices are highly dependent on taxes and subsidies, only the production costs were considered to provide an indication of the economic feasibility of each fuel type. However, the study did not involve a sensitivity analysis, so the potentially remarkable sensitivity of e.g. production costs to oil and agricultural residue prices was neglected (though it was mentioned that increasing oil prices will make bioethanol fuel economically more viable in the future). Life cycle costs of a kilogram of pure bioethanol were lower than those of conventional gasoline by 56% but since the fuel efficiency of bioethanol is lower than that of gasoline, the E10 and CG fuels had equal life cycle costs per functional unit (one driven kilometre by a flex-fuel vehicle). The driving costs of E85 were 23% lower than those of CG. However, the researchers admitted to having used a theoretical figure for ethanol yields per glucose gram, which is significantly higher (up to 46%) than those measured in laboratory studies, meaning that that both the life cycle costs and environmental impacts of bioethanol might have been understated in this study.

Lu & Hanandeh (2017) conducted comparative LCA and LCC for six different bioenergy generation processes: 1) woodchips gasification for power, 2) wood pellets for combined heat and power, 3) wood pellet combustion for domestic water and space heating, 4) pyrolysis for power, 5) pyrolysis with bio-oil upgrading to transportation fuel and 6) bioethanol for transportation fuel. The functional units and system boundaries for the LCA and LCC are the same (‘cradle-to-grave’ system boundaries based on ISO 14040) for making comparison possible. Global warming, acidification, eutrophication, fossil depletion, human toxicity and land use impact categories were considered in the LCA. The study analysed which system had best performance from environmental and economic perspective and the results highlighted that the systems with most intensive processing generally had the highest environmental impacts as well as highest life cycle costs. The study concluded that woodchips gasification had lowest environmental impacts in all categories and the lowest LCC (177,6 AUD/Mg), as well as the highest energy return. The system with second lowest LCC, bioethanol production, had generally the worst environmental performance, being the only option with positive global warming potential. As an emerging technology, however, the environmental as well as economic performance of ethanol production is expected to be enhanced in the future. Pyrolysis for power generation was the most energy-intensive process (and the only option with a negative energy return).

6.3. Integrating LCA results for E-LCC and S-LCC

Recently developed Environmental Life Cycle Costing (E-LCC) methods have improved the comparability and integration of LCC and LCA studies by using the same functional units and system boundaries for the environmental and economic aspects and, unlike conventional LCC, focusing more on environmental externality costs that may be internalised in the future. E-LCC studies have enabled better identification of links between financial costs and environmental impacts.

In Martinez-Sanchez et al. (2015), E-LCC costs incurred by all stakeholders are included in calculations: in this way, not only net costs and savings are seen but also the distribution of costs between stakeholders. Showing which stakeholders incur the highest or lowest costs, the results could be used to evaluate if financial compensation is needed between stakeholders. The study found out that organic waste source segregation and subsequent activities resulted in an extra financial cost per a household and contributed with environmental loads for example to global warming and terrestrial acidification but also provided environmental savings for e.g. noncarcinogenic human toxicity and freshwater eutrophication.

Some studies claimed they are doing comparative assessment between LCA and (traditional) LCC altough the LCC includes internalisation of decision-relevant externalities and therefore the method can be classified as an environmental LCC, though they do not use this term. Lu & El Hanandeh (2017) account for all relevant costs, such as acquisition, operation, maintenance and disposal throughout the entire life stages (Australian/New Zealand Standards Life cycle costing AS/NZS-4536 1999-R2014). In addition they maintained that the life cycle cost related to environmental impacts should also be included in order to internalise the environmental cost. Therefore, the life cycle GHG emissions costs were also included. The environmental costs of GHG emissions, i.e. carbon prices of 29 AUD/t CO_2eqv, (Treasury Australian Government 2016) were internalised in the LCC method. Carbon pricing in Australia took place via taxation during 2012-2014 but the scheme was then repealed, meaning that the carbon price is presently an external cost internalised in the study. However, carbon costs were negligible in all options, mainly because so-called biogenic CO_2 from wood combustion was not considered to count towards the GWP impact. Global warming, acidification, eutrophication, fossil depletion, human toxicity and land use impact categories were considered in the LCA. Monte Carlo simulations were conducted to assess the effects of combined uncertainties in the processes but the rankings within the impact and cost categories were largely unaffected. Ranking of the scenarios (except for woodchips gasification) was considered difficult due to conflicting performance of the alternatives under different impact categories.

According to Lu & Hanandeh (2017), monetary valuation of different impact categories should be conducted to help internalise more of the LCA costs in the LCC. In addition, complete environmental ranking could be simplified if proper weighting for the impact categories were developed. However, as Hoogmartens et al. (2014) have noted, weighting and aggregation are controversial since they require subjective judgement on the priority of different impact categories and might undervalue the categories that are hard to quantify, such as biodiversity and human health. From allocation perspective, in the CHP process the emissions were calculated by utilizing so called exergy method (Energy Efficiency Council 2013, see more in chapter 3.2) to be evenly shared between heat and electricity production ($Emissions_{Heat} = Emissions_{Electricity} = 0.5$).

Martinez-Sanchez et al. (2016) used E-LCC and a rather qualitative S-LCC to assess the costs and social impacts of food waste management in Denmark, and included indirect costs related to the so-called **rebound effect** or, more specifically, **income effect**. They proposed that while buying less food might contribute to lower food waste levels, the money that is saved through lower food expenses might induce other marginal consumption associated with environmental impacts, and so null the positive effect of lowered food waste. The S-LCC did not produce an absolute value for these indirect costs due to their high uncertainty, but focused on overall trends and relative differences. They concluded that environmental effects related to income effects can be reduced if food waste prevention measures also aim at allocating the monetary savings of consumers towards low-impact goods and services.

6.4. Direct and indirect effects through efficiency improvement activities

Efficiency improvement studies may view environmental impact reductions and costs savings in comparison between effiency improvement actions. Some studies view life cycle perspective and even mention life-cycle perspective but ay lack the actual life cycle method utilization. There are also many more aspects effecting efficiency than costs and environmental impacts or the last two might have indirect effects through these aspects/factors. Energy efficiency may have correlations between cost efficiency and eco-efficiency.

According to the literature, LCC improves the evaluation of production efficiency (Hansen & Mowen 1992) and also enables better evaluation of process planning efficiency for companies, by comparing real costs to life cycle budget costs and showing the distribution of these costs to different parts of the life cycle (Clinton & Graves 1999, Dunk 2004).

According to Resurreccion et al. (2012) it is expected that sometimes the economic and environmental results are directly linked, e.g. energy efficiency usually corresponds to cost savings, but it is not possible to know the overall environmental and economic connections of the analysed processes without a dedicated comprehensive financial analysis.

Pöschl et al. (2010) study is providing bases for more detailed assessment of environmental compatibility of energy efficiency pathways on biogas production and utilization of digestate. According to the study, analyses of energy balance in the life-cycle of biogas systems lack bases for coparison due to varying accounting systems and boundaries. None of the analyses reviewed in the study have coupled multiple feedstock scenarious to viable energy conversion pathways to assess impact of plant size to minimise GHG emissions or potential for integrated efficiency enhancement to minimise costs and overall system sustainability. Study evaluated the energy efficiency of different biogas systems, including single and co-digestion of multiple feedstocks, different biogas utilization pathways, and waste-stream management strategies. According to the results there could be significant variation in energy efficiency arising from feedstock resource and process adopted, conversion echnology and digestate management echnique.

Walla (2008) study estimated the costs of biogas and electricity production from maize silage in relation to biogas plant size. Study did not mention or use LCC method but the costs of electricity production from biogas per kWh are calculated from the annual capital costs, substrate costs, labour costs and other costs (maintenance, insurance, administration etc.) for producing the required amount of electricity.The study deals with the optimum size of the plant due to the investment and support available and the graduated tariff for green electricity. Study developed a model to derive cost curves also for the transport costs for maize silage and biogas slurry. The costs of delivering the substrate and removing the biogas slurry are calculated separately. Study also answered to the questions on how electricial efficiency changes due plant size, how plant size effects costs of biogas production and transport of substrate and which plan size is most cost efficienct one. The study does not assess environmental impacts alongside with costs. The study results showed that as plant size increases so does also the electricial efficiency of CHP meaning also less consumption of substrate per KWH of electricity. This is alos slowing down the growth of transport distances and costs. Generating revenues from the excess heat was not considered but it was taken into account in the study that the revenues would increase due selling the excess heat contributing to lower costs. The results demonstrate the role of tariffs as investment grants and price grade mean smaller plants can cover costs through sales but larger ones need lower costs than those in study calculations to turn to profit at the relevant electricity price. The sensitivity alanysis of the study shows that there are opportunities to lower the costs during life cycle: cost reductions were gained e.g. through longer effective life of a biogas plant, a greater availability of substrate with greater yields, increasing the operating hours.

In the study of Kuisma et al. (2013), the aim was to determine agrifood-waste-based biorefining efficiency according to the choices made in the entire value chain. It was discussed how choices for every step from biomass supply (types and quantity covered), collection and conversion (processes, location), to markets (distribution, energy consumption, fields) and demand substituted for (energy, fertilisers) may affect the overall efficiency of biorefining. According to the study, biorefining increases nutrient and energy efficiency in comparison with current use of waste and system boundaries decisively influence the relative efficiency. Nutrient, carbon (C) and energy efficiency are being key to both environmental and economic performanc e. Efficiency is, however, multifaceted and it can mislead decision- making if its dependence on the choices along the entire biorefining chain is not revealed. It was observed that the design, system boundaries, combustion and location close to the heat demand influence the relative efficiency of biorefinery scenarios. Also regional differences in agricultural structure, the extent of the food industry and population density have a major impact on biorefining systems. Vice versa, also the regional conditions may affect the appropriate design and efficiency of biorefinery systems (Kuisma et al. 2013, Kahiluoto et al. 2011, Kokossis & Yang 2010). It was also acknowledged that from cost efficiency perspective biorefining is typically organised on a local or regional scale due to high costs of biomass (digestate) transportation. The keys to sustainable biorefining are high degrees of exploitation of feedstock potential and substitution efficiency, rather than efficiency in conversion.

7. Discussion

This review discusses the integration possibilities and challenges of environmental impact and costing methods to produce a more comprehensive cost evaluation methodology. According to the literature viewed in this report concerning different life cycle costing methodologies (LCC, C-LCC, E-LCC and S-LCC), the challenges of combining and comparing environmental and economic assessments are often related to unconsistent terminology, lacking details in the cost calculation models, poorly defined or differing system boundaries, impractical or differing functional units and limitations of data collection. For example, since there is no uniformity concerning chosen functional units (e.g. whether using per kilo or per hectare), results can sometimes be difficult to compare. The functional units of the LCC and LCA need to be the same so that the data and the results between two or more life cycle assessment methodologies can be compared or integrated. The environmental results and life cycle costs can both be presented "per functional unit", making comparing results easier. However, the chosen functional unit affects the environmental and economic results of studies and can sometimes cause misleading conclusions. Also, different impact categories (e.g. euros and CO_2-eqv) per functional unit (e.g. per kilo of a product) is a challenge to integrate together for a single value indicator.

These different life cycle costing methodologies offer an overall framework for a total economic sustainability assessment by integrating social and environmental aspects in economic assessment work. Conventional LCC does not usually include environmental considerations and is interested only in direct costs and savings, being traditionally used to rank investment decisions. Sometimes, however, C-LCC even on its own includes some environmental effects and it has also been combined with LCA: in this case, the results may not be directly comparable with each other due to e.g. differences in system boundaries and the indifference of the C-LCC towards including all costs and stakeholders. Environmental LCC should always be connected to an LCA while S-LCC is a comprehensive standalone method and not meant to be combined with other assessments. In general, there seems to be some confusion in the literature about the contents and interrelations of socio-economic assessments (S-LCA, S-LCC, CBA): they often use similar methodologies and sometimes their differences can be hard to pinpoint (chapter 5).

Assessing total sustainability is challenging and there is no universally agreed definition for sustainability. The concept of sustainability is defined by the Brundtland commission and it includes three 'pillars': environment, economy and society. It is "development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (Brundtland 1987). Therefore it might be seen that life cycle perspective is a most suitable tool for assessing sustainability goals. Also, economic sustainability is often seen as a prerequisite for the realisation of environmental and social sustainability. Economic sustainability is often very much linked to ecological and social sustainability and the latter is the least developed and has currently no established definition (Partridge 2014).

It was observed from the literature that based on differencies in framework conditions LCC studies naturally reach a variety of conclusions. *The system boundaries* of the LCC naturally depend on the study in question and the preferences of the ones ordering the assessment. For example, LCC does not always consider all life cycle stages from cradle to grave but only a gradle-to-gate perspective. Life cycle perspective should consider all life cycle stages from raw material extraction and acquisition, through energy and material production and manufacturing, to use and end-of-life treatment as well as final disposal.

Assessing costs in any economic analysis involves three important questions (Martinez-Sanchez et al. 2015):

1) which type of cost types should be assessed (for example internal or social costs),
2) for whom should these costs be assessed (which stakeholders are taken into account) and
3) which cost calculation principles should be applied?

Critical aspects in the existing literature regarding cost assessments combining LCA and LCC include:

1. system boundary equivalency: It is important that all relevant processes are included in the assessment
2. accounting for temporally distributed emissions and impacts,
3. the level of environmental impact internalisation and the coverage of shadow prices or indirect costs
4. consistent terminology

When doing comparison between LCC and LCA, there are still many challenges concerning about terminology, system boundaries, functional units and data collection. Developing comprehensive environmental LCC and social LCC methods is still under development. The definition of object of analysis and system boundaries can be difficult as the logical boundaries for an environmental and economic analysis sometimes differ. After all, the LCA is focused on environmental stressors and the LCC methods on costs (though S-LCC can also include qualitative considerations on social welfare issues): not all objects with costs are relevant in the LCA (e.g. employee salaries) and not all environmental externalities can be seriously accounted for even in the S-LCC. The system boundaries of LCAs (especially) are always limited by practical issues, such as working time, availability of information and the ability to quantify or otherwise analyse complex impact chains. Moreover, evaluating (direct, indirect, internal or external) future costs possibly caused by these emissions might require the LCC to consider aspects that lie outside the LCA system and are highly uncertain. In this sense, the ”equivalency” of system boundaries is more of an ideal than a strict precondition for combining LCA and LCC. That said, the two different scientific disciplines need to be as coherent and consistent as possible, with all relevant processes included in the assessment and results presented in the same time frame, so that the economic calculations can be matched with those of the LCA.

While C-LCC might only focus on the costs of one main actor (e.g. a business), E-LCC should assess the costs of most or all actors that are directly a part of the life cycle of the analysed subject. Social LCC, as the most comprehensive method, should assess the costs of even the stakeholders indirectly affected by the impacts of the research subject. Dealing with the stakeholder costs separately and aggregating them together requires a systematic approach since only the added values of each life cycle phase should be summed to get the total life cycle costs (see chapter 4.3.2). There is also limited guidance on how these costs should be related to the LCA results.

One challenge might be how to allocate correctly environmental and cost impacts between different products. Inside the system boundaries some processes produce side products in addition to the main product, and the total impact of the process may not be easily separated into parts. If the production processes cannot be separated for every product, there is a need to allocate total system impacts between different products (e.g. electricity and heat in CHP plants). Another concern is cost allocation over time. In order to be able to allocate costs accordingly, standard economic tools are often used, such as the time value of money (interest rate, discounting, present value), and annuity calculations (allocation of investments over time).

LCA and LCC could be seen as comparative assessments that pinpoint environmental and economic hotspots throughout the life cycle of products and guide the decision-making processes of companies and society towards sustainability. Life cycle costing highlights investment decisions that bring cost reductions even if an additional increase in the initial investment is necessary. In turn, LCA examines environmental impacts throughout the whole product chain and has the potential to highlight critical emission sources along the production chain that enable considering the most effective actions to minimize the environmental impacts. The comparative LCC and LCA framework has great potential to help produce and consume more sustainable products while reducing environmental degradation, using natural resources in a cost-effective manner and contributing to social welfare.

The same functional unit (e.g. "per 1000 kg of treated waste") enables integrating LCA results with LCC for an E-LCC and S-LCC assessment work. However, integrating different impact categories (e.g. global warming & acidification) is difficult, as is aggregating the economic and environmental results to a single monetary figure. In E-LCC, it is recommended to keep the environmental impacts and economic costs separate, and instead show the life cycle costs in relation to each separate environmental impact category.

According to the literature, the monetisation methods of environmental externality costs still need development. Using different weighting methods in order to monetarise environmental effects, such as emissions and resource use, is very challenging. The user should always acknowledge the limitations and assumptions behind the weighting methods to evaluate their applicability to each new site and possibly utilise benefit transfer methods to generalise results from previous studies. Both the benefit transfer methods and the previous studies should also be critically analysed. Some valuation methods are very specific and narrow, e.g. the travel cost method, and the results do not portray the true consumer WTP of environmental sites. Stated preference methods (see chapter 4.2) are very dependent on interviews and questionnaires, and are thus subjects to various biases. All in all, there is a need for customisable valuation systems and databases that could more efficiently provide valuation for different aspects of various sites. In the future, externalities should be internalised and taken into account in environmental management accounting in a more comprehensive way. However, valuation methods will always give a very limited and uncomplete picture of the total use and non-use value of the assessed site. For this reason, it should always be clarified in assessments what aspect of the assessed site exactly is valuated, and with what assumptions or data.

One problem is also that in some studies transfers are sometimes included in Societal LCCs, although this should not be the case. The internalisation of environmental damages in Societal LCC are often carried out but with poor explanations despite the fact that valuation principles may affect the results.

Further development of E-LCC methods and their results becoming mainstream could enable environmental effects (positive or negative) impacting product prices in the future, either by taxation or change in consumer demand. Applying the results of these methods to decision making can enhance understanding of indirect environmental costs or benefits, improve resource efficiency as well as create new business opportunities and jobs. There is still further need for assessments that credibly present so-called shadow prices to portray indirect environmental costs which are caused by industrial and other business operations. It is important to understand how environmental costs affect revenues as well as the underlying reasons for these costs (e.g. resource inefficiency), so that environmental management actions can be taken to allocate resources towards reducing them.

8. Conclusions

This literature review examined existing environmental cost accounting methodologies inside from traditional business accounting to environmental accounting sectors from national and companies' point of view. Review especially focuses on different life cycle costing methods taking into account environmental impacts by comparing them with LCA results or integrating them with cost results by monetraizing them. The review also discussed the methodology development needs and integration possibilities and challenges of environmental impact and costs to produce a more comprehensive cost evaluation methodology.

It was perceived that environmental impacts are systematically undervalued in traditional business calculations since it is usually seen that external costs do not influence the formation of the company's result. Recently, corporations have started to pay more attention to their social and societal impacts with the rise of corporate social responsibility (CSR) taking into account with environment related activity costs and benefits. Also, the importance of environmental accounting has grown through CSR and from a mere external reporting method to a supportive tool in total management decision-making processes. National environmental accounting is performed at the governmental level, is concerned with the social and societal costs of operations and is crucial for national policy development. The environmental management accounting (EMA) includes counting both company's environmental impacts and environmental costs for an optimal calculation.

The environmental management accounting (EMA) is seen at this moment as a strategic competitive factor where identification, allocation and management of environmental costs are key elements. Environmental costs can be both internal costs (conventional environmental costs, hidden costs, liability costs and promotional image costs) and exernal costs (environmental impacts internalised as transfers by environmental or subsides). In EMA environmental costs are usually internal but both internal and external environmental costs are needed to be internalized as part of companies' decision makin process. One traditional way an externality can be internalised is through becoming priced by an authority as transfers by environmental taxation in the form of e.g. air emission taxes or energy use. However, there is also a broader need for monetary valuation of non-market goods as well as external impacts of market goods and projects. Monetisation of environmental impacts can help communicate complex environmental impact information to decision makers, so that the scale and hierarchy of the environmental risks become clearer. According to the literature, the monetisation methods of environmental externality costs still need development and using different weighting methods is very challenging. There is a need for customisable valuation systems and databases that could more efficiently provide valuation for different aspects of various sites.

Environmental costs and savings (benefits) are generated to the company or to society during the entire product life cycle by different measures relating to air, soil or water protection, waste management and environmental management. Usually, environmental management costs due impact reduction are having negative cost effects but environmental impact reduction may also positive indirect financial effects (e.g. image of the company). The indirect cost effects caused by e.g. industrial activities, energy production and agricultural land-use are becoming more important both globally. However, when talking about monetizing externalities, valuation methods will always give a very limited and uncomplete picture of the total use and non-use value of the assessed site and it should always be clarified in assessments what aspect of the assessed site exactly is valuated, and with what assumptions or data.

In environmental accounting literature, there are several different terms, definitions and intepretations of methodologies with different system boundaries. The most well-known life cycle methodology is life cycle assessment (LCA) that is accounting for environmental impacts. On the other hand, life cycle costing (LCC) methodology work is developing rapidly but it is still relatively young and for many terms in the field there are still no well established definitions. Traditionally, life-cycle costing (LCC) or sometimes called financial LCC and conventional -LCC (C-LCC) has been applied only for investment decisions, when examining only private costs and savings and has taken into account only the direct costs and for example the costs of the impacts on the environment are excluded. Environmental Life Cycle Costing (E-LCC) is an extension to the conventional LCC method and is designed to be complemented by an environmental LCA. It accounts for environmental externalities that are expected to be internalised during the project-relevant time perspective, due to legislative (e.g. environmental taxation) or other causes. Societal life cycle costing (S-LCC) expands the E-LCC method by adding the costs of externalities that *could be relevant in the long term* for both the stakeholders directly *and indirecly* affected by them. It **connects environmental and social aspects in monetary terms** and can be described as a ''socio-economic'' or ''welfare-economic'' assessment.

Lack of a detailed economic assessment next to the environmental life cycle assessment (LCA) limits the value of LCA in the eyes of decision makers who always need to consider economic priorities and not only the social and environmental ones. A comparative look at product's, system's or service's environmental impacts (LCA) and costs and revenues (LCC), as well as to integrate these with environmental-economic methods (E-LCC and S-LCC) together is required for sustainable solutions. The commensurate the information makes it more easily to assess a variety of food and bioeconomy chains' overall economy and evaluate its development.

Exploring environmental life-cycle impacts (LCA) and costs (LCC) for the entire chain creates opportunities to find the most critical points to minimize environmental impacts and production costs and add value. LCA provides a systematic frame of reference for calculating the environmental impacts associated with administrative matters, but it is well known that financial constraints affect on decisions on the major technology implementations in modern societies. The studies examined in this review detected that economic and environmental life cycle outcomes often have same trends. It was also demonstrated that while some life cycle phases were not critical for the economic assessment itself, a significant influence on environmental impacts could be observed and vice versa. This illustrated that unbalanced decisions for system cut-off (examining LCC and LCA outcomes separately) cannot be advised. There was also lack of common terminology, same system boundaries and transferability of the studies. One problem was that in some studies transfers are sometimes included in Societal LCCs, although this should not be the case. The internalisation of environmental damages in Societal LCC is often carried out but with poor explanations despite the fact that valuation principles may affect the results.

Further development of E-LCC and S-LCC methods and their results becoming mainstream could enable environmental effects (positive or negative) impacting product prices in the future, either by taxation or change in consumer demand. Applying the results of these methods to decision making can enhance understanding of indirect environmental costs or benefits, improve resource efficiency as well as create new business opportunities and jobs. There is still further need for assessments that credibly present so-called shadow prices to portray indirect environmental costs which are caused by industrial and other business operations. It is important to understand how environmental costs affect revenues as well as the underlying reasons for these costs (e.g. resource inefficiency), so that environmental management actions can be taken to allocate resources towards reducing them.

There is growing need for research-based knowledge that links environmental (LCA) and economic (LCC) aspects of products and projects together. Both internal and external environmental costs are needed to be internalized as part of companies' decision makin process. Also, the indirect cost effects caused by industrial activities, energy production, infrastructures and agricultural land-use are becoming more and more important both globally and from the European perspective.

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