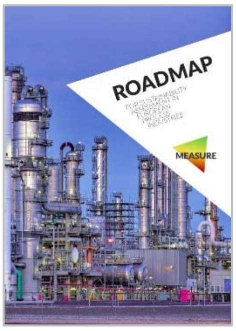
Background document

supplementing the

“Roadmap for

Sustainability Assessment in

European Process Industries”



Sector C:

Solid waste management sector – Overview of assessment tools and methods

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1 Introduction

1.1 Challenges of environmental sustainability assessment in the solid waste management sector

In 2012, 2515 million tons of waste was generated from all economic activities and households in Europe (EU-28), among which 64% was generated by mining, quarrying and construction activities, 11% by manufacturing activities and 9% by households (EC 2015). If not properly managed or disposed, these waste streams can have a high impact on the environment, economy and the society as a whole. One of these environmental issues is related to marine litter, which has a direct impact on wildlife via ingestion but also allows the development of new microorganisms and invertebrates which impact on marine ecosystems is still unknown (Reisser et al. 2014). Moreover, illegal waste trading from developed countries to developing countries is a growing concern as it shifts and/or creates new environmental, economic and social issues in the receiving countries. On the other hand, these waste streams are also a source of resources that can be recovered to fulfill resource needs of the European Union (EU). Therefore, the goal of waste management can be summarized as the “protection of men and the environment, and the conservation of resources such as materials, energy and space” (Brunner, Rechberger 2015).

1.2 Regulatory issues: European legislation and environmental sustainability assessment in the SWM sector

Solid waste management (SWM) projects are often large investment projects, for which specific compulsory sustainability assessments need to be performed at the early stages of project development. The EU regulation 1303/2013 defining the common rules for major EU projects and programs eligible for EU funds (in which waste management projects can often be classified) requires that applicants conduct a cost-benefit analysis (CBA) of their project, in which a risk assessment should be performed. Similarly, two compulsory studies taking into account environmental aspects are required by the Environmental Impact Assessment (EIA) Directive[1] and the Strategic Impact Assessment (SIA) Directive[2] for future projects and plans/programs. EIA is mandatory for certain types of public or private projects, among which the construction and operation of installations for the disposal of non-hazardous waste with a capacity superior to 100 tons per day. The Directive recommends project developers to evaluate the impact of

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1 Directive 2014/52/EU on the assessment of the effects of certain public and private projects on the environment

2 Directive 2001/42/EC on the assessment of the effects of certain plans and programs on the environment

the project on climate change, resources (soil, land, and water) and biodiversity and has the advantage of encouraging the project stakeholders to conduct a first inventory of the future energy demand and use, the nature and quantity of material consumed and the type and amount of waste produced. SIA is very similar to EIA but applies to public plans and programs which have a potential high environmental impact. Both assessments have the advantage to consider criteria that are not often covered in other environmental evaluation studies (e.g. impact on biodiversity). However, they do not provide a framework for impact assessment, leaving to local authorities the evaluation of the final report on a case by case basis. Life Cycle Assessment (LCA) is a tool proposed in the Guidance on Integrating Climate Change and Biodiversity into Environmental Impact Assessment (EC 2013), but its use is not required.

Other legislative frameworks such as the different European waste directives have been developed to help stakeholders choose on a voluntary basis the most sustainable option for the treatment of the waste of their concern. In these Directives, the assessment of the environmental sustainability and the calculation of sustainability indicators are presented as key steps to enhance this choice and are highly encouraged.

The Waste Framework Directive[3] defines the waste hierarchy, which defines the priority order of measures and treatments that should be applied by member states, i.e. waste prevention, preparing for reuse, recycling, recovery and disposal. The Waste Framework Directive also stipulates that some waste streams can depart from the waste hierarchy if it is justified by “life-cycle thinking on the overall impacts of the generation and management of such waste”. Moreover, the Directive stresses the fact that life cycle thinking (LCT) should be more used to link environmental impacts and economic valuation of waste. The Directive also provides the R1 formula to calculate the energy efficiency of incineration and co-incineration plants and which is used to classify these plants as recovery or as disposal facilities.

LCT is also encouraged in some waste-specific Directives. The Directive on waste electrical and electronic equipment (EEE)[4] stresses the fact that the environmental performance of all the operators involved in the life cycle of EEE should be improved, and that the whole life cycle of the product should be taken into account when optimizing reuse and recovery through product design. Similarly, the Directive on Packaging and packaging waste5 requires Member States to conduct LCA studies to justify the hierarchy applied among reuse, recycling and recovery (this has also been briefly discussed in the background document “Sector report: metals and automotive”).

[3] Directive 2008/98/EC on waste (Waste Framework Directive)
[4] Directive 2012/19/EU on waste electrical and electronic equipment (WEEE)
[5] Directive 94/62/EC on Packaging and packaging waste

1.3 Environmental evaluation and impact assessment in the SWM sector: typical applications and guidelines

Environmental impact assessment and evaluation studies are widely conducted in the SWM sector in which they are conducted from technology (e.g. incineration or gasification) to system level (e.g. waste management strategy at regional or national level). Most studies assessing the sustainability of SWM systems are conducted by public organizations such as universities and public authorities, the latter mainly coordinating studies conducted by consultancies. Moreover, these studies often focus on municipal solid waste (Allesch, Brunner 2014) while household waste represent less than one tenth of the waste generated in the EU-28 and construction, quarrying and mining waste represent almost two third of the waste generated but these streams are scarcely the focus of the studies. At system level, some studies are conducted by professional federations for specific waste streams. Few studies at technology level are publicly available in literature. The ones that are found in literature are conducted by universities or in the framework of public-private partnerships.

LCA is the main assessment tool used in the SWM sector (Allesch, Brunner 2014). The other ones are: material and substance flow analysis (MFA/SFA), energy analysis (EA), exergy analysis (ExA), emergy analysis (EmA), risk assessment (RA) and the ecological footprint (EF). Two methods used to integrate environmental impact assessment and evaluation results with other sustainability indicators, i.e. economic and social indicators have been identified: cost-benefit analysis (CBA) and multi-criteria decision analysis (MCDA). Several guidelines and documents providing rules exist to support environmental impact assessment and evaluation studies in the SWM sector (Table 1). They mainly focus on the application of LCA and LCT-based assessment methods.

Three guides to help applying LCA and LCT to SWM systems and technologies were commissioned by the JRC in 2011: one guidance document on how LCT and LCA can be used to identify the best solution among alternatives for SWM in general (JRC 2011a) and two other ones which focus on specific waste streams, i.e. construction and demolition waste and bio-waste (JRC 2011b and JRC 2011c, respectively). All guidelines have been developed for experts in the field of SWM and for LCA practitioners and are accessible for non-LCA experts.

The Norden guideline (Norden 2007) provides guidance on the definition of the goal and scope of CBA studies, suggesting the involvement of several stakeholders concerned by the project in this step as well as in the identification of the studied scenarios. It gives insights and recommendations on the choice of the system and geographical boundaries as well as the time horizon. Economic effects to consider are listed, as well as environmental effects. The guideline proposes to couple CBA with LCA by taking into account the environmental impacts associated with activities outside of the studied system, i.e. the background system and the avoided processes.

Table 1: Existing guidelines to support environmental impact assessment and evaluation studies in the SWM sector (non-exhaustive).

Name of the publication	Method	Leading organisation	Year	Initiating country/ region
Supporting Environmentally Sound Decisions for Waste Management – A technical guide to Life Cycle Thinking (LCT) and Life Cycle Assessment (LCA) for waste experts and LCA practitioners	LCA and LCT-based methods	JRC	2011	EU
Supporting Environmentally Sound Decisions for Bio-Waste Management – A practical guide to Life Cycle Thinking (LCT) and Life Cycle Assessment (LCA)	LCA and LCT-based methods	JRC	2011	EU
Supporting Environmentally Sound Decisions for Construction and Demolition (C&D) Waste Management – A practical guide to Life Cycle Thinking (LCT) and Life Cycle Assessment (LCA)	LCA and LCT-based methods	JRC	2011	EU
Nordic guideline for cost-benefit analysis in waste management	CBA	Norden	2007	Nordic countries[1]
Guide to Cost-Benefit Analysis of Investment Projects	CBA	EC	2014	EU
A Practical Guide to Environmental Risk Assessment for Waste Management Facilities	RA	EPA	2000	UK
Practical Handbook for Material Flow analysis	MFA/SFA	Brunner & Rechberger	2003	Austria

[1] *Denmark, Finland, Iceland, Norway, Sweden, the Faroe Islands, Greenland and Åland*

The “Guide to Cost-Benefit Analysis of Investment Projects” (EC 2014) has been written to help applicants to EU funds for major investment projects comply with the aforementioned EU regulation 1303/2013 (see I.2.a.) and is not specific to SWM projects. However, it includes a section on the application of CBA to the SWM sector. Concerning SWM projects, the guide provides a list of the main engineering features that should be reported. Moreover, it gives insights on how to forecast the evolution of SWM services demand in the time frame of the project as well as how to choose the studied alternative scenarios. It provides a list of typical investments necessary to build such facilities, typical revenue sources and gives insights on how to predict revenues in the future. It also gives guidance on how to monetize non market goods and the avoided environmental burdens (mainly avoided emissions to the air and resources saved by the production of energy from waste). Finally, the document provides a list of parameters to consider when conducting sensitivity analysis.

The “Practical Guide to Environmental Risk Assessment for Waste Management Facilities” was written by the UK EPA in the framework of the national Licensing Regulations for waste management facilities. Indeed, in England and Wales, a risk assessment should be conducted to support demands for permits and licenses for waste treatment facilities (Drew et al. 2009). Therefore, this document is adapted to the UK legislative context. It gives insights on how to conceptually model the studied facility, how to identify the environmental risks and to prioritize them. The guideline does not detail the way risks are actually characterized.

The “Practical handbook for material flow analysis” (Brunner, Rechberger 2003) has not been written specifically for the SWM sector but it still highly relevant. It aims at defining a structured framework to conduct MFA applied to environmental, resource and waste management. Therefore, it includes a specific chapter on the application of MFA/SFA in the SWM sector and case studies are used as illustrations. In addition to the description of the methodology itself (definition of the time and space frames, identification and quantification of flows etc.), this book provides insights on data uncertainty and sensitivity analysis in MFA/SFA studies. It provides information on the existing software tools that can be used to support such studies and on which assessment methods can be used to assess the impacts associated with the results of the MFA/SFA results.

Note that in the framework of the International EPD® System (www.environdec.com), specific rules are set to conduct environmental impact assessment studies on services or products. These rules are gathered in so-called Product Category Rules (PCR) documents, which are not guidelines as such. Two PCR documents have been written for application in the SWM sector: one on solid waste disposal services and one on plastic waste and scrap recovery (recycling) services (The International EPD® System 2008, 2013). These two EPDs provide rules to develop Type III environmental declarations: choice of the functional unit and of specific or generic data, the processes to be included in the system boundaries, the Life Cycle Inventory (LCI) modeling framework and the indicators to be calculated. Note that their implementation in EPD documents could only be found for two services: the EPD of “Collection of hazardous, potentially infective sanitary waste and disposal through incineration” (Mengozzi S.A. 2013) and the EPD of “Polyamide scrap recovery service” (RadiciGroup 2015).

1.4 Goal of this background document

The aim of this background document is to discuss how environmental impact assessment is conducted in the SWM sector, how sustainability indicators are calculated by stakeholders and to identify what gaps need to be filled to help improve the quality of the studies and make the different assessment tools and methodologies easier to apply. It summarizes the state of the art of environmental assessment and evaluation methods used in the SWM sector and provides detailed background information for section 3.2 of the MEASURE roadmap.

This document is based on several previous studies reviewing methods and tools used to assess the sustainability of SWM systems and technologies (Allesch, Brunner 2014; Ekvall et al. 2007; Karmperis et al. 2013; Pires et al. 2011a). Moreover, several review studies can be found on the application of LCA to SWM systems and technologies (Morrissey, Browne 2004; Lazarevic et al. 2010), the latest ones being from Laurent et al. (2014a, b), who reviewed 222 LCA studies and summarized the main gaps related to the practical application of LCA in the SWM sector, and from Astrup et al. (2015), who conducted a similar analysis with a focus on thermal waste-to-energy (WtE) technologies. Moreover, the information provided by these studies was completed by information from several other scientific publications such as case studies or position papers, public reports and guidelines.

This study follows the scheme proposed in Figure 1. The first step of an assessment study is the modelling of the studied system or technology. The second step corresponds to the assessment of the impacts of the defined model. The third step, which is optional, consists in integrating the results from studies focusing on environmental aspects in sustainability assessment studies, i.e. taking into account the three pillars of sustainable development. Therefore, the third part of this study focuses on the integration of environmental issues in sustainability assessment studies on SWM systems and technologies. Finally, after environmental impact assessment and integrated sustainability assessment, the interpretation of the results should be made.

This report focuses on the environmental aspects of sustainability. Economic and social assessment aspects are further discussed in the **background document** “Current state in LCSA” of the MEASURE roadmap. Even if the SWM sector has also its own specificities concerning social aspects (e.g. employment of social workers in recycling facilities or informal recycling), the issues associated with the evaluation and assessment of the social impacts in the SWM sector are similar to the other MEASURE sectors and methods are still at the early stage of development.

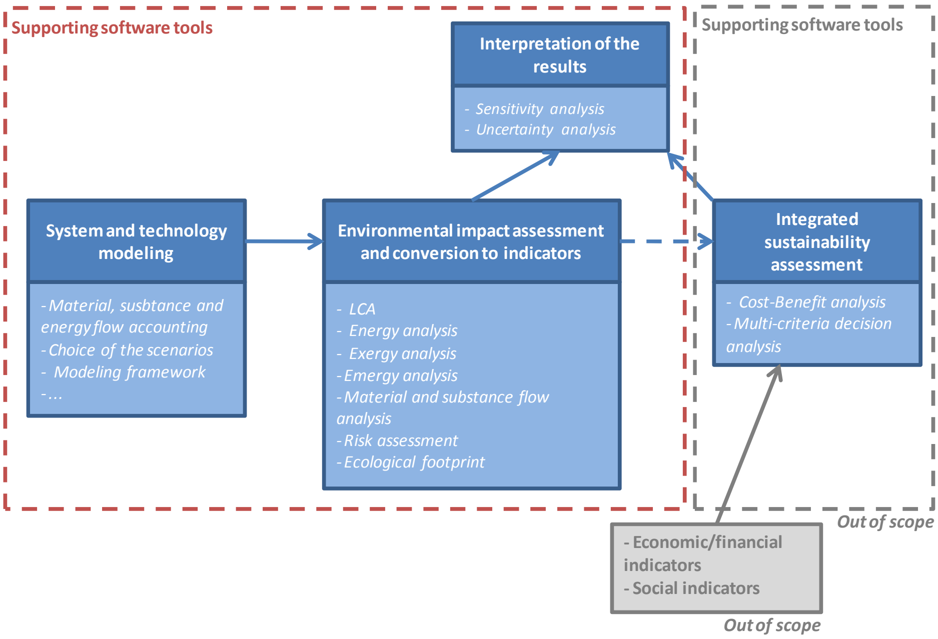

Figure 1: Framework of the waste sector study within the MEASURE project.

2 Challenges and common issues of assessment practices

2.1 Specificities of the SWM sector

Compared to the other industrial sectors, the SWM sector has specificities which require specific modeling and impact assessment/evaluation methods and indicators. These specificities are summarized below:

– Other industrial sectors aim at delivering goods whereas the SWM sector first aims at delivering a service.

– Unlike the other sectors, SWM systems are multi-inputs and multi-outputs systems, i.e. input materials are made of several waste fractions and several final products can be delivered from waste valorization (e.g. energy, precious metals).

– Stakeholders involved in the decision making and the implementation of SWM systems are different from other sectors. Indeed, waste management strategies within a specific area are often driven by public authorities and not private companies. The latter have mainly leeway through process improvement and development, e.g. by improving the recovery process of trace elements in incineration plants bottom ashes or testing new treatment technologies for specific waste streams.

– Specific issues such as odor and visual disamenities are associated with SWM systems. It does not only concern waste treatment facilities, but also pre-treatment waste collection and storage.

– The environmental sustainability of SWM systems is highly dependent on local conditions such as waste composition or climatic conditions. The most sustainable waste management strategy in one region is not necessarily the most sustainable strategy in a different one. Therefore, even if the waste management hierarchy defined by the Waste Framework Directive will generally lead to the most environmentally sustainable choice, SWM systems are often influenced by local conditions which can modify this priority order and it is advised to perform environmental impact assessment or evaluation studies to choose the best option to implement.

2.2 System and technology modelling

Prior to any analysis or impact assessment, the studied technology or system needs to be modelled, i.e. its complexity should be reduced to reach a comprehensive and analysable level. One of the steps to do so is to inventory all material, substance and energy flows entering, leaving or accumulated in the studied technology or system.

2.2.1 Material, substance and energy flow accounting

As in other sectors, environmental impact assessment studies in the SWM sector should be based on an inventory of all substances (elements and small molecules, e.g. CO_2, Pb, Zn) and materials (i.e. made of a large number of combined substances, e.g. wood and plastic) and consumed, emitted and stocked in the system. Similar work should be done on energy flows. The SWM sector is characterized by several key material flows, substance flows and parameters which characterization greatly influences the results of the studies. This paragraph focuses on these flows and parameters, which are unevenly taken into account in studies. Several of these key flows have been reported to be improperly characterized or accounted for when conducting sustainability analysis (Laurent et al. 2014b). This can be due to the type of analysis performed but also to malpractice, to poor availability of data on these flows or to a difficulty to characterize the impact of these flows because of limited literature data on characterization factors, which implies that they are excluded from the analysis.

2.2.1.1 Material and substance flows accounting

Specific waste characterization

Input waste composition highly depends on local conditions and is complex to characterize as this flow is generally composed of several waste fractions. The proper site or region specific characterization of the input waste stream is not common practice. This is mainly due to a lack of data available, as it is only available if sampling campaigns in the region of the study are conducted, or if an analysis of the production and consumption of goods and substances is performed to estimate the composition and amount of waste generated. However, this latter method is rarely used as it is subject to high uncertainties related to the amount of materials and substances stocked in the anthroposphere (Brunner, Rechberger 2015). Moreover, both methods are time and resource consuming. Thus, data from previous sampling campaigns, previous studies or national/regional waste statistics are used, which can lead to use significantly different waste compositions than the real site or region-specific waste composition. MFA/SFA studies more frequently use specific waste compositions because they are more often conducted at process level and based on pilot or lab scale facilities for which an analysis of the input waste composition is performed.

Apart from the lack of available data, the composition of the studied waste stream is not thoroughly reported (Laurent et al. 2014b), which makes these studies difficult to compare with other ones, as some parameters (e.g. biogas composition) can be directly dependent of the waste composition. Note that in risk assessment studies, a description of waste composition could never be found. However, such data could be relevant to conduct a mass balance in the case of some prospective risk assessment studies for which future emissions need to be quantified, or to assess the risk of pollution depending of the type of input waste streams (e.g. as roughly applied in Rapti-Caputo et al. (2006) for 4 types of waste streams, i.e. inert, urban, industrial-non dangerous and dangerous).

Waste characterization at substance level

Apart from SFA studies, environmental evaluation and impact assessment studies rarely describe waste composition at substance level. This is due to the fact that while SFA studies conduct a consistent mass balance, other studies rarely conduct a full and complete substance balance. However, substance flow accounting is particularly appropriate in the SWM sector as the environmental and economic properties of products and emissions from waste are defined based on the chemical composition of these products/emissions. Not conducting a substance balance can lead to the modelling of an unrealistic scenario. This is particularly problematic in LCA studies (Laurent et al. 2014b). LCA practitioners usually collect data from various sources (site specific data, literature data etc.) without linking them to one another along the studied process chain. Data reconciliation is needed and currently rarely performed. Therefore, the lack of consistent substance flow accounting is an important gap in the inventory of LCA studies. This is also the case in RA studies, for which a mass balance is needed to evaluate any emission losses and related pathways to the environment that would otherwise be ignored (Pollard et al. 2006).

One major issue when accounting for substance flows is the completeness of the considered substances. Indeed, a large number of substances are emitted by SWM systems. The analysis of each of them is hardly possible and today studies focus on a limited number of substances without always justifying this choice or investigating if other harmful substances could be emitted. A list of substances emitted by specific waste treatment processes would help to account for substances in a more consistent way. Measurement campaigns could help drawing such a list.

Evolution of waste composition through the process chain

The composition of waste varies along the process chain. Pre-treatment steps such as waste collection and separation can modify the waste characteristics, e.g. by increasing its moisture content. Therefore, the evolution of waste composition through the process chain should be modelled. This point has been highlighted by Laurent et al. (2014b) in the case of LCA applied to SWM systems, but also concerns other assessment methods or analysis such as SFA and ExA when conducted at system level (e.g. Arena, Di Gregorio 2014). This modelling is rarely done today. This is due to a lack of awareness of its importance, but also to a lack of information on the effect of specific processes on waste composition.

Consistency of the collected data in the time frame of the study

Most studies found in literature compare several options in order to find the most sustainable treatment system or technology to implement in the future. However, the SWM sector is subject to annual changes, the main one being related to the amount and type of waste generated (e.g. due to changes in consumer behaviours). In practice, data inventoried to model the waste treatment system is mainly based on recent past data and few studies try to predict the evolution of waste composition and volume in the near future. This is particularly problematic in the case of waste management planning which aims at finding the most sustainable system for the next decades. It can also be an issue in the case of the assessment of some technologies whose efficiency depends on the volume of each waste fraction (e.g. MBT). Beigl et al. (2008) reviewed the models available to estimate the future generation of MSW, and several studies on the forecasting of waste generation using mathematical models and on the parameters affecting future waste composition and generation have recently been published (Intharathirat et al. 2015; Oribe-Garcia et al. 2015; Ferreira et al. 2014). However, such studies and models are rarely used as they require specific knowledge on mathematical modelling and need to be adapted to the local conditions of the studied system or technology.

2.2.1.2 Energy flow accounting

Several waste treatment technologies produce energy, e.g. electricity and heat from biogas produced by the anaerobic digestion of organic waste, or electricity recovered from waste incineration. Waste treatment technologies also consume energy, i.e. electricity, heat and fuels. In all studies, apart some MFA/SFA studies and studies on newly developed technologies such as gasification or pyrolysis, practitioners do not have difficulties to find data on energy consumption and production from waste treatment technologies and collected data is correctly reported. However, similarly to material and substance flows accounting, energy consumption and production can highly depend on waste composition.

2.2.2 Choice of the studied scenarios

Most of the time, the environmental impact assessment of SWM systems and technologies aims at comparing different waste treatment options and identifying the most sustainable one. The number of scenarios studied is often limited, and some choices in the definition of these scenarios (e.g. recycling rate, processes involved) can be arbitrary (Tascione, Raggi 2012). Therefore, some studies conclude on the most environmentally sustainable waste management system whereas not all scenarios have been studied. Some authors propose to use linear programming to overcome the limitation associated with the time and resources necessary to compare a large number of scenarios, which are anyway limited (Tascione et al. 2014; Solano 2012). Instead of defining a limited number of scenarios, linear programming evaluates the environmental sustainability of an unlimited number of scenarios by combining all the possible defined parameters and finding the optimum scenario. Multi-objective linear programming allows considering several objectives to optimize (e.g. to minimize the results for several impact categories in LCA), e.g. by assigning a weight to each of the parameters to optimize (Solano 2012), and allows finding a set of possible “best case” scenarios. The application of linear-programming could only be found in demonstrative studies aiming at testing linear-programming on simplified SWM systems (Solano 2012; Tascione et al. 2014).

2.2.3 Other inventory data and modeling choices specific to LCA and LCT-based analysis

LCT is “a concept that accounts for the upstream and downstream benefits and trade-offs” associated with a system or technology (JRC 2011a). It is based on the fact that the impacts of a product or service are not only caused by the process of primary interest but also by other steps upstream and downstream this process and that the production/delivery of the studied product/service can cause or avoid impacts in other industrial sectors. LCA is the tool that allows transposing the LCT concept in a quantitative framework (JRC 2011a). In addition to classic LCA, other types of LCA such as Energetic and Exergetic LCA allow transposing such concept.

Several guidance documents for LCA are available (e.g. ISO standards; see paragraph 2.3.5 and the **background document** “Current state in LCSA”) but a lot of freedom is still left in the field of data selection and methodological assumptions. Current practice in LCA applied to SWM systems was summarized by Laurent et al. (2014a, b). This paragraph is partly based on the outcomes of this study. Note that Laurent et al. (2014a, b) reported the lack of rigor of some practitioners in reporting these data and methodological choices in LCA reports, which makes LCA studies difficult to interpret and compare. This issue is not further reported in this review.

2.2.3.1 System boundaries

Laurent et al. (2014b) reported that some specific processes are rarely included in the scope of the analysis. Practitioners base this choice on the assumption that these processes do not have a significant contribution to the environmental footprint of the studied system. This is the case for infrastructure, the collection and transportation steps and the residuals and ashes final treatment. However, it has been shown in previous studies that for some technologies and systems, these steps could have an important contribution (Laurent et al. 2014b). Therefore, the specific processes included in the system boundaries should be carefully considered and justified (Laurent et al. 2014b). Note that the choice of the system boundaries also highly depends of the choice of the LCI modeling framework.

2.2.3.2 LCI substance framework

Most SWM systems and technologies are multi-functional systems, i.e. they are often multi-inputs and multi-outputs. Therefore, practitioners need to choose how the impacts of the studied system or technology will be allocated to the chosen functional unit. Three main approaches can be followed: allocation, cut-off or system expansion. In parallel, two types of LCA studies can be conducted: attributional LCA and consequential LCA. Attributional LCA quantifies the impacts of the incoming and outgoing flows related to a system or process without looking for impacts caused in other sectors. Consequential LCA looks for the consequences (displaced or avoided impacts) that a technology or system can have on other sectors or activities. In general, there is a lot of confusion concerning these terms, as system expansion is often only attributed to consequential LCA as a way to include the avoided processes in the system boundaries but can also be used in attributional LCA to avoid allocation.

In order to help choosing the most suitable LCI modeling framework, the ILCD Handbook proposes a decision tree based on the identification of the context situations of the study (Figure 2).

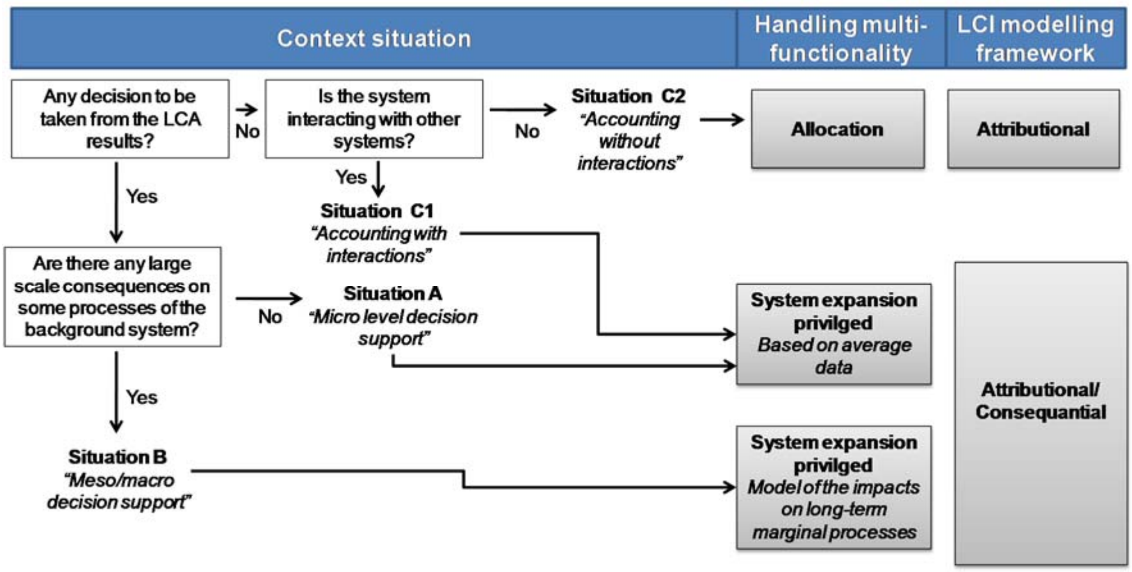

Figure 2: Identification of the LCI modeling framework based on the identified context situation of the study (retrieved from Laurent et al. 2014b based and JRC 2010).

First, Laurent et al. (2014b) highlighted the fact that none of the 222 reviewed LCA studies were referring to this framework. This leads to the fact that a lot of situations are assumed to correspond to the C1 and C2 situations whereas they actually aim at taking a decision. However, when practitioners identify that their situation does not correspond to the C situations, they need to make the difference between situations A and B (Laurent et al. (2014b). According to the ILCD Handbook, the decision on whether a study can be classified as a situation A or B should be based on whether or not "decision implies large-scale consequences in the installed equipment/capacity outside the foreground system of the analyzed system". By "large-scale consequences", the ILCD Handbook implies that the annual capacity of the studied system exceed the capacity of the annually replaced facilities. In the case of SWM, the studied systems and technologies often aim at replacing an actual system or technology in a defined area, and therefore the implementation is not conducted in parallel of an existing system but aims at completely replacing it. Thus, most LCA studies in the SWM sector belong to situation B and the consequences of the studied system on the other economic sectors should be carefully identified.

Secondly, Laurent et al. (2014b) reported a large variety of models applied to the waste management sector, from allocation based on a variety of parameters (mass, energy content etc), mixes between system expansion and allocation depending on the types of products (e.g. energy or material), to system expansion. This choice highly depends on the studied system. Therefore, the issue associated with this methodological choice is more related to the awareness to the impact of this choice on the results and the way the chosen model is handled, e.g. how the choices of the allocation factor or the avoided processes are made.

2.2.3.3 Data inventory to model the background system

The background system is the system delivering material and energy to the foreground system and providing waste treatment services (e.g. if the foreground system is a collection scheme, the solid waste treatment system is a background system). Usually, databases such as 18 substance (Frischknecht, Rebitzer 2006), Gabi (PE International 2013) or ELCD (JRC 2014a) are used to complete the data inventory of the background system. These databases are based on recent past data and are appropriate for the modeling of existing systems and technologies, e.g. to identify key factors or improvement potentials. However, they are less adapted to the modeling of the implementation of new SWM systems and technologies in the future. Some data included in the background systems might significantly change within the next decades and predictions should be made on data for background processes. This has already been done on electricity mixes in past studies but is not yet common practice in the SWM sector. Moreover, some region-specific data, which are rarely reported in LCA databases, can also have an impact on the LCA results. For example, electricity supply mixes can vary within one country and tools are missing to trace electricity flows from producer to consumer. Such tools would highly contribute to improve the outcome of waste LCA and LCT-based studies, for example by giving information on the most sustainable location for specific waste treatment technologies.

2.2.3.4 Avoided processes 18ubstance

Another issue in waste LCA and LCT-based studies is the choice of the processes avoided by the delivery of new products (material or energy) to the market. This applies in the case of studies belonging to situations A and B defined in the ILCD Handbook.

For the substitution of energy, one can choose between average (e.g. country heat production mix, situation A) and marginal data (e.g. most likely heat production source which will be replaced by the heat produced from the waste treatment, situation B). In most waste LCA studies, the choice made between marginal and average data for energy substitution is not justified (Laurent et al. 2014b). In theory, if the study has been identified as belonging to situation B, which should be the case for most LCA studies applied to SWM systems and technologies, the most likely substituted energy mix should be identified based on a national energy system analysis (Bernstad, la Cour Jansen 2012). Moreover, there is a choice between identifying the short-term (typically 5 years) and long-term marginal energy mix (Mathiesen et al. 2009). This choice mainly depends on the goal of the study, i.e. if the study is a prospective study or not. Such a work on energy system analysis can be considered as a study in itself and can hardly be seen as the responsibility of a single LCA practitioner. However, at least it should be tried to identify the marginal energy mix based on literature or expert interviews, as recommended by Finnveden et al. (2009). This approach is rarely found in literature. Some efforts in the past are worth citing as examples of good practice for further studies. Carlsson et al. (2015) used the energy planning model MARKAL-NORDIC to estimate the marginal electricity mix in Nordic countries, which was then used to model the marginal electricity mix in Sweden (Carlsson et al. 2015). Similarly, the authors used the NOVA model to estimate the marginal fuel for heat generation in Sweden (Carlsson et al. 2015). In a study on the factors influencing the environmental footprint of the recovery of energy from residual municipal waste in UK, Burnley et al. (2015) estimated both short-term and long-term marginal technologies for electricity production. The short-term marginal set of technologies was estimated from the analysis of data on running times of power plants in the studied area during a period of three months. The marginal technologies were those which were not running at full capacity during the studied time frame, i.e. the technologies flexible to electricity demand. The choice of the long-term marginal technology was based on the national waste policy. It was estimated to be combined cycle gas turbines as UK government is planning to build more facilities based on this technology to replace old facilities. These good practices are time consuming and should be supported by databases provided by the energy sector.

Similarly to energy, also materials substituted by by-products delivered by waste treatment facilities should be modeled and the choice between marginal and average data has to be made. This choice is rarely justified in waste LCA studies and in the majority of them, a substitution ratio of 1:1 with the substituted product is chosen and/or the estimation that the quality of the delivered material is similar to the substituted material is made (Laurent et al. 2014b). To identify the marginal product adapted to the context of the study, Laurent et al. (2014b) recommend to conduct a market analysis, and if the substitution ratio is not known, to conduct a sensitivity analysis on this parameter. Note that in product-LCA studies, several approaches are proposed by their authors and could also be applied in waste-LCA. Some experts suggest that the environmental credits of one unit of recycled material should be calculated as the weighted average of the impacts of producing the primary (i.e. virgin) and secondary (i.e. recycled) materials being used by the market as input materials for the production of new goods. In this respect Bala et al. (2015) have suggested the following formula for calculating credits for recycled materials considering both production mix and quality of the recycled materials:

$$\textbf{Environmental credit} = x \times \textbf{REC} + (1 - x) \times \textbf{Q} \times \textbf{VIR}$$

Where x is the proportion of recycled material in the average market mix, $(1-x)$ is the proportion of virgin material in the average market mix, Q is the quality factor of recycled material vs. virgin material (Q≤1), REC is the environmental load of the recycling process and VIR is the environmental load of the production process of the virgin material.

In addition to the choice of substituted material and the substitution ratio, the origin of the substituted material should be identified, i.e. to find the producers which will be affected by an increase of secondary material production, as done in Allegrini et al. (2015). Moreover, in this study, the authors identified the electricity production technology in China which would mostly be affected by a change in electricity consumption due to the production of secondary material in Europe. This type of approach to identify marginal technologies associated with the substituted product is rarely followed.

2.2.3.5 Modeling of long-term emissions

Laurent et al. (2014b) highlighted the fact that long-term emissions should be better quantified in waste LCA studies. There are three types of emissions: emissions due to the degradation of organic matter in landfills, emissions of pollutants in landfills and emissions associated with processed organic waste applied on the ground (e.g. digestate from anaerobic digestion). Today, there is no consensus on how to quantify these emissions, e.g. which time horizon should be used. Moreover, data is not always accessible and it can be time and resource consuming to understand the different existing models, leading practitioners to conduct a cradle-to-gate or gate-to-gate LCA. Comprehensive information is missing.

2.2.4 Conclusions on systems and technology modeling

Systems and technology modeling is a common step to all studies assessing the environmental aspects of sustainability. The use of specific and well-defined waste streams is rarely found, mainly because of a lack of data. Moreover, a complete substance balance is most of the time missing, which can lead to the analysis of unrealistic systems or the exclusion of some emissions from the scope of the study. Another main issue is the lack of data prediction in the studies. This is required for all studies analyzing SWM projects with a time frame of several decades, which concerns most waste treatment projects. For LCA and LCT-based studies, a consensus is missing on how to handle long-term emissions. Moreover, information on the diffusion of these emissions in the ground and the air is not always accessible for LCA experts. Another major issue is the choice of the avoided processes in LCA studies, which is most of the time chosen as the average production mix. However, in the SWM sector, the choice of marginal processes is most of the time more relevant.

2.3 Environmental impact assessment and conversion to indicators

2.3.1 Material and substance flow analysis

2.3.1.1 Applications and related indicators

Material and substance flow accounting are preliminary steps to impact assessments, but are also used as process and system efficiency studies on their own, i.e. to conduct material and substance flow analysis (MFA and SFA, respectively). MFA and SFA consist in a thorough analysis of the fate of materials or substances within the studied system and are used to calculate performance indicators.

In the SWM sector, MFA is mainly conducted at regional and sectorial levels, e.g. to assess the source and fate of waste streams within a defined region (e.g. Owens et al. 2011) or to optimize the waste management scheme of waste streams in a specific sector (e.g. Andarani, Goto 2014). Therefore, MFA is mostly conducted to have a macroscopic vision of waste management systems and mainly used in waste management planning. MFA is used to calculate recovery or recycling rates of specific materials, mass or volume of waste to landfill (Arena, Di Gregorio 2014) or stock of material in landfill (Bogucka et al. 2008). Similar indicators, called “resource efficiency indicators”, were used in the revision of the targets set by the EU Waste Framework Directive, i.e. recycling rates of materials, proportion of landfilled waste and amount of municipal material captured for recycling vs. amount material used in the EU[6].

SFA is used in the waste management sector to reach two goals (Brunner, Rechberger 2004): 1) ensure that a limited amount of hazardous substances is emitted to the environment during the final disposal of waste; 2) ensure that hazardous substances do not accumulate in recycled materials or that recycling or reuse processes are not associated with harmful emissions to the environment. Based on the new paradigm according to which waste should be considered as a resource, a third goal can be defined: identify where valuable substances accumulate in order to optimize their recovery. SFA is mainly used at micro scale, i.e. process level, for example to track precious “trace elements” from a specific type of waste (Chancerel et al. 2009), to compare possible treatment technologies for specific waste streams (Arena, Di Gregorio 2013; Cascarosa et al. 2013), or to evaluate the effect of waste separation on hazardous substances content of each separated fraction (Rotter et al. 2004). However, SFA has also been used to track substances at regional or sectorial level (e.g. in Arena, Di Gregorio 2014, Vyzinkarova et al. 2013). Several indicators based on SFA can be found

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[6] Proposal Directive for amending Directives 2008/98/EC on waste, 94/62/EC on packaging and packaging waste, 1999/31/EC on the landfill of waste, 2000/53/EC on end-of-life vehicles, 2006/66/EC on batteries and accumulators and waste batteries and accumulators, and 2012/19/EU on waste electrical and electronic equipment - COM(2014) 397

in literature: amount of a specific substance to landfill, amount of a specific substance in recycled product (Arena, Di Gregorio 2014; Vyzinkarova et al. 2013), velocity of the consumer stock evolution (Vyzinkarova et al. 2013) or carbon conversion efficiency (Arena et al. 2011).

2.3.1.2 Advantages and limitations

MFA and SFA are relatively easy analyses to understand. Moreover, trace elements are often the focus of the analysis whereas they are often neglected when other methods are applied. Another advantage is that MFA/SFA studies are easily comparable with one another.

Most of the limitations associated with MFA/SFA rely on their practical application (e.g. when studying a complex system, conducting a MFA/SFA in an excel file can be a real challenge and source of many errors), data availability (cf. paragraph 2.2.1 on material and substance flow accounting) and the interpretation of the results. Specific tools have been developed to facilitate the practical implementation of MFA/SFA and are detailed in paragraph 2.6.2. In SFA, the interpretation of the results can be a challenge for one who does not have a thorough understanding of the chemical and physical processes occurring within the studied system or process. For example, the recovery potential of metals after thermal treatments depends on which form they remain after the treatment: gasification allows recovering iron and copper under metallic form but not combustion after which metals are available in their oxidized form (Arena, Di Gregorio 2013). A simple mass balance without any further understanding of the process would lead to consider oxidized metals as recoverable as non-oxidized metals.

Moreover, MFA and SFA have an intrinsic limitation which is their inability to identify displacement of environmental burdens. They also do not consider energy aspects, e.g. when energy is consumed and/or produced.

2.3.2 Energy analysis

2.3.2.1 Applications and related indicators

An energy analysis (EA) is the analysis of all the energy flows going through and stocked within a system. There is no clear methodology defined to conduct energy analysis in the SWM sector. Different ways of accounting for energy consumption and generation from SWM systems have been found in literature, and most of them are gathered behind the common term “energy balance”. Some studies only evaluate the balance between the chemical energy embedded in the input waste (e.g. “feedstock energy” in Arena et al. 2011) and the output products, others calculate a ratio based on input energy from transportation and processing and output energy from the waste-treatment by-product (e.g. Comparetti et al. 2014) and some mix both (Cascarosa et al. 2013). Another approach converts all input sources of energy (electricity, gas, fuel etc) into primary energy and compares them to the energy embedded in the output products (Wallmann et al. 2008; Cimpan, Wenzel 2013). This highlights a lack of harmonization of the definition of the so called “energy balance”, which should be more specifically defined to enhance comparison between studies.

EA is mainly conducted to analyze the energy efficiency of systems based on waste-to-energy (WtE) technologies. EA is evenly used at system and technology levels. Many indicators based on energy balance can be found in literature: lost and available feedstock energy (Arena, Di Gregorio 2014), Primary Energy Input to Output (Pöschl, et al. 2010), electricity efficiency (De Meester et al. 2012), energy conversion efficiency (Nordlander et al. 2011), cold gas efficiency (Arena et al. 2011) etc. All these indicators are based on different energy flows and are not always adaptable to other types of treatment technologies.

The only energy efficiency formula that can be found in the EU legislation documents on waste is the so-called R1 formula, used to calculate the energy efficiency of waste incineration plants:

$$\text{Energy efficiency} = \frac{(Ep - (Ef + Ei))}{0.97 \times (Ew + Ef)}$$

Where E_p is the annual energy produced as heat or electricity converted in terms of primary energy, E_f is the annual energy input to the system from fuels contributing to the production of steam, E_w is the annual energy contained in the treated waste calculated using the net calorific value of the waste and E_i is the annual energy imported excluding E_w and E_f. The factor 0.97 accounts for energy losses due to bottom ashes and irradiation. This equation is not meant to be based on a full energy balance of WtE plants as some flows are excluded from the calculation (e.g. heat flows used by third parties) (EC 2011). This formula has been proposed by the European Commission to evaluate the energy efficiency of WtE plants as a basis for the classification as “disposal” or “recovery” facility. Its use cannot be found in scientific literature but it is the most commonly energy efficiency formula used by WtE plant operators in the EU and thus deserves special attention.

2.3.2.2 Advantages and limitations

Energy analysis is adapted for the evaluation of the energetic performances of WtE technologies. EA studies are easy to understand, accessible to non-experts and EA indicators allow comparing easily different improvement options concerning a specific WtE technology.

One first limitation concerns energy analysis based on the conversion of energy flows in terms of embedded energy (or feedstock energy) and primary energy. Both of them require the use of conversion factors or specific formulas which can have high impacts on the results of the study. As an example, the Low Heating Value (LHV) of a waste stream can be calculated based on the LHV of each waste fraction found in literature or based on the elemental composition of the whole waste stream. Both methods can lead to different results that may lead to different energy efficiencies. Thus, special attention should be paid to these factors and sensitivity analysis should be carried out to test the effect of this choice on the conclusions of the study.

Secondly, limitations appear when comparing different waste treatment options in order to find the most energy efficient one. Indeed, several practitioners end the EA of the studied system at the gate of the system. This is the case of studies conducting the EA of pre-treatment processes such as mechanical treatment or mechanical-biological treatment. Some pre-treatment systems are followed by other processes, which energy consumption and production depend on the output of the pre-treatment step. If the output products are not the same among the different studied processes, displacement of energy consumption/production can occur and is not taken into account. Therefore, conclusions on the most energy efficient treatment option should not be made on the basis of such short cut analysis and a LCT approach should be followed when conducting comparative EA. Such an approach is called Energetic Life Cycle Assessment (ELCA). An example of good practice of energy balance following a life cycle approach can be found in Cimpan, Wenzel 2013. Similarly, some practitioners use the system expansion approach to take into account the benefits of delivering products (e.g. electricity, heat) to the market (Cimpan, Wenzel 2013; Bonk et al. 2015). The choice of the marginal technology for energy production is particularly important in ELCA. Note that when downstream processes are included, upstream energy consumption should be allocated to the different products. However, such situation has not be found in literature as practitioners end the system boundaries at the gate of the studied system (with a risk of not considering energy consumption/production displacement), or include all the downstream processes in the studied system (which avoids allocation).

Thirdly, EA and ELCA have the intrinsic limitation that they are not suitable for comparison of waste management systems which do not deliver energy to the market, e.g. it does not allow comparing composting and anaerobic digestion of organic waste. The only energetic analysis allowing such comparisons are the ones based on feedstock energy flows alone, but with very limited outputs.

Specific limitations concerning the calculation of the energy efficiency of WtE plants based on the R1 formula can be highlighted. The R1 formula has been the subject of critics from several parties. Some critics are related to the fact that the equation is not consistent with the Reference Document on the Best Available Techniques for Waste Incineration (EC 2006) concerning the values chosen for equivalence factors used to convert thermal and electrical energy into primary energy (1.1 and 2.6, respectively, which promotes the production of electricity instead of heat), and concerning the efficiency to reach (0.6) (Ökopol 2006). Others are related to the fact that the equation does not highlight the difficulty of WtE plants to find end users for heat in some parts of Europe (e.g. in southern Europe) and that the equation does not consider differences of efficiencies related to the size of WtE plants (Grosso et al. 2010). Moreover, the scientific relevance of including the factor accounting for energy losses due to bottom ashes and irradiation in the denominator could be discussed. To overcome some of these issues, the use of exergy balance is an alternative recognized by the scientific community.

2.3.3 Exergy analysis

2.3.3.1 Applications and related indicators

Exergy is the maximum theoretical work that can be obtained from a system brought to equilibrium with the surrounding environment. The aim of exergy analysis is to identify exergy losses within a process or a system. It is based on a thorough material, substance and energy accounting. Each flow is then expressed in terms of exergy based on databases such as the one provided in Szargut (2005) or on calculations using the elemental composition of materials. Unlike energy analysis, exergy balance is a clear and harmonized term for accounting exergy flows within a process or system.

The suitability of exergy analysis to assess the efficiency of waste management systems has already been shown in the early 2000s (Dewulf, Van Langenhove 2002) and regularly highlighted by the scientific community (Hiraki, Akiyama 2009; Zhou et al. 2011; Brunner et al. 2015) but few practitioners are using this method. Compared to other analyses, exergy analyses found in literature cover a wider range of waste types, including waste solvent (Van der Vorst et al. 2010), waste cooking oil (Peiro et al. 2008), aluminium waste (Hiraki, Akiyama 2009), municipal organic waste (De Meester et al. 2012) and municipal solid waste (Zhou et al. 2011; Xydis et al. 2013). Most studies conduct exergy analysis at process level.

Two main types of exergy efficiencies can be calculated based on these flows: the functional exergy efficiency and the universal exergy efficiency. The functional exergy efficiency is the ratio between the exergy of the product of interest and the exergy inputs of the system. The universal exergy efficiency is the ratio between the output exergy flows and the input exergy flows. Both ratios are equally used by practitioners in the waste management sector.

2.3.3.2 Advantages and limitations

The main advantage of exergy compared to energy is that it takes into account both quantity of quality of different types of energy flows, e.g. electricity and heat produced by WtE plants (Grosso et al., 2010). Expressing the energy produced by a WtE plant in terms of exergy allows overcoming the issue related to the choice of equivalence factors in the R1 formula. It also allows taking into account climatic conditions by using ambient temperatures specific to the location of the WtE plants (Grosso et al. 2010). In general, for all types of waste treatment processes, exergy analysis is recognized to give more information than energy analysis.

The usefulness of exergy analysis compared to energy analysis has also been pointed out in the Best Available Techniques Reference Document (BREF) on energy efficiency were it is stated that “exergy analysis, although less used and more complex, is more useful because it points directly to where energy can be saved” (EC 2009). The main advantage of exergy compared to energy is its ability to translate both quantity and quality of energy. Moreover, it expresses all inventory flows (i.e. mass and energy flows) in the same unit, i.e. MJ_{exergy}.

The limited use of exergy analysis in the industry seems to be related to its seeming complexity and to the fact that additional data have to be collected (i.e. exergy content of inputs and outputs). In practice, exergy analysis is not more complex than converting the flows in term of primary energy.

Tables on exergy content are however less accessible due to the limited use of exergy analysis by industry. To facilitate the use of exergy analysis, some tools such as an online converter and a software tool (ExerCom) have been developed. Another limitation to the use of exergy analysis by industry is the lack of benchmark data that can be used to compare their own efficiency (EC 2009).

Similarly to EA, a LCT approach is necessary to account for all potential displacement of exergy destruction. This approach has been rarely applied in the SWM sector (for examples, see Hiraki, Akiyama 2009 and Van der Vorst et al. 2010), where exergy analysis is mostly applied at process level. However, exergy analysis has been integrated in the LCA framework through the development of impact assessment methods coupled with the 26ubstance database, i.e. the Cumulative Exergy Extraction from the Natural Environment (CEENE) method (Dewulf et al. 2007) and the Cumulative Exergy Demand (CexD) method (Bösch et al. 2007).

2.3.4 Emergy analysis

2.3.4.1 Applications and related indicators

Emergy accounts for all the original energy that has been consumed in the earlier steps of product or service making, i.e. solar energy, tidal energy and geothermal energy. Emergy was introduced by Odum (1996) based on the principle that the value of a resource depends on the amount of the three aforementioned energy types which were consumed to produce it. Emergy analysis is not often used in industry. However, it is subject to a growing interest in the USA, where a pilot project is running on its application in industry. The SWM sector counts few practitioners using emergy analysis as an evaluation method. Most studies on emergy analysis applied to the SWM sector found in literature were conducted in Asia, e.g. on waste exchanges within a sulfuric acid production system and a titanium dioxide production system in China (Zhang et al. 2011), to compare four treatment technologies for urban solid waste in China (Liu et al. 2013), on an e-waste treatment process in China (Song et al. 2012).

Several indicators based on emergy can be found in literature. The classical emergy indicators, also used in other sectors, are commonly used, e.g. the environmental loading ratio defined as the sum of all non-renewable emergy divided by the emergy of the impact of emissions to the renewable emergy, or the emergy yield ratio defined as the total emergy input by the total emergy purchased on the market (Zhang et al. 2011; Song et al. 2012). However, emergy-based indicators specifically developed in the context of SWM are also used, such as the emergy recovery ratio defined as the ratio of the recycled resources emergy over the initial waste emergy (applied on e-waste in Song et al. 2012) or the landfill to recycle ratio defined as the The ratio of emergy required for landfilling a material to the emergy required for recycling (Agostinho et al. 2013).

2.3.4.2 Advantages and limitations

One advantage of this method compared to other methods is that emergy analysis aims at accounting for the impact of a system on ecosystems services. It considers that emissions to air and water will be diluted by ecosystems services to reach an acceptable concentration. For example, emissions to air will be diluted by the action of wind, and emissions to water by the action of water flow. Therefore, impacts on ecosystems services are calculated based on the amount of emergy from nature necessary to dilute the pollutants. However, this approach is highly based on transformities values, i.e. the values used to convert flows in terms of original energy (geothermal, solar and tidal) consumed by the studied system, which have often been criticized by the scientific community for their associated lack of uncertainty quantification. Moreover, criticizes have been raised on several other methodological issues such as combining disparate time scales and allocation problems (Hau, Bakshi 2004b). Converting money into emergy terms introduces also a limitation. Transformity values converting money into emergy are calculated based on the amount of resources consumed by an economy, divided by a set of economic indicators such as GDP (Campbell, Lu 2009). First, this means that transformities are different from one country to another. Secondly, within a country, values used for economical transformities fluctuate over time. In the context of SWM, decision makers need to take decisions at a certain time and usually for implementation of the system or technology within the next decades. Therefore, a fluctuation of transformities which can change the ultimate results of the analysis can introduce a bias as the economic situation of a country at the date of implementation might not be the same as twenty years later.

Attempts have been made to couple emergy accounting with LCA. They are mainly based on applying the LCA principle to emergy accounting, i.e. conducting emergy accounting from a cradle-to-grave perspective, but without linking any process with LCI databases (Rugani, Benetto 2012). Note that Ingwersen (2011) followed such an approach in a gold mining case study, and tried to link the system to 27ubstance processes. No attempts in the SWM sector could be found.

2.3.5 Life cycle assessment

2.3.5.1 Applications and related indicators

Today, LCA is the most commonly used tool to assess the environmental sustainability of waste management systems and technologies (Allesch, Brunner 2014). Moreover, more than 65% of the LCA studies reviewed by Laurent et al. (2014a) were conducted in Europe. The interest of academia and decision makers for this tool has been increasing since the end of the 90s, but the release of the EU Waste Framework Directive in 2008 seems to have given a boost to the application of LCA to waste management systems and technologies in Europe (Laurent et al. 2014a). Treatment technologies (biological and thermal treatments, landfilling, recycling) are evenly studied, but not the different types of waste (Laurent et al. 2014a): most of the LCA studies treatment options for household waste but rarely for construction, mining and quarrying waste or manufacturing waste. Laurent et al. (2014a) give several explanations to this: the confidentiality of data on certain types of waste such as industrial waste, the perception of the environmental issues by the public which directly impacts the political agenda related to household waste management planning, and more generally the lack of specific data. In the SWM sector, LCA is used as an assessment tool to improve existing systems, or as a prospective tool to evaluate the implementation of new systems. It can be applied at different levels of a waste management scheme, i.e. from technology to system level.

Indicators reported in LCA studies of SWM systems and technologies are mainly life cycle impact assessment (LCIA) indicators, i.e. indicators directly obtained from the characterization of the impacts. They are mostly calculated using the CML and EDIP characterization methods, from which midpoint indicators are calculated, followed by the Ecoindicator 95 or 99 methods (Laurent et al. 2014b). Almost half of the LCA studies conduct normalization after characterization, and around one third conduct weighting, most of them by the mean of the application of the Ecoindicator 95 or 99 methods (Laurent et al. 2014b). A normalization and weighting method was proposed and applied by the JRC to calculate macro-scale monitoring indicators in the waste management sector at the EU-27 level (JRC 2012). Each result for a specific impact category is divided by the total impact for the same impact category within the studied region and during a reference year. Then, normalized results are weighted based on the JRC report from Huppes and van Oers (JRC 2011d). However, the authors suggest that such indicators should only be used for "demonstration purposes" and not as a recommendation. Another macro-scale indicator developed by the JRC is the Recycling Benefit Rate (RBR). This resource efficiency indicator was developed to compare different end-of-life scenarios of plastics. It expresses the potential environmental savings related to the recycling of a product over the environmental burdens of virgin production followed by disposal (Huysman et al. 2015), and combines the recyclability rate with LCA data. It has been recently applied to compare closed-loop and open-loop plastic recycling systems in Flanders, Belgium (Huysman et al. 2015).

2.3.5.2 Advantages and limitations

The main advantage of LCA is related to its LCT-based approach. It allows identifying the causes of the most impactful environmental burdens within the system or technology of primary interest but also those occurring in the upstream and downstream systems. It also allows identifying displacement of environmental burdens to other sectors, which is particularly appropriate in the case of waste management systems and technologies. Moreover, waste management systems and technologies emit a wide range of hazardous substances that might have different environmental impacts and LCA allows evaluating these impacts, i.e. impacts on resource depletion and of emissions into air, soil and water.

LCA has been shown to be an efficient tool to orientate decision makers towards improvement pathways or the implementation of new waste management schemes. However, several limitations of LCA applied to SWM have been highlighted in literature. In this review, these limitations are divided into two main categories: intrinsic limitations of LCA and practice-dependent limitations.

Intrinsic limitations of LCA

Intrinsic limitations of LCA can be defined as the limitations of the tool, as it is developed today, to assess the environmental burdens of a product or service, and which are independent of the methodological choices, modeled and selected data or inconsistencies introduced by the practitioner. The following limitations are of particular relevance when LCA is applied to SWM systems.

LCA does not allow characterizing the impacts geographically – Thus, it does not allow deciding where a waste facility should be built (Ekvall et al. 2007). Indeed, some local conditions have a direct effect on the impact of a specific compound released in the atmosphere. These conditions can affect pollution dispersion (e.g. wind, rainfall) or the reaction of the emitted pollutant with compounds already present in the atmosphere (e.g. the concentration of ammonia, which reacts with Nox to form nitric acid). This is also valid for emissions of compounds such as heavy metals in the ground, which depend on site-specific soil characteristics (e.g. porosity, composition).

The characterization of the impact of resource consumption in LCA is limited – Impact assessment methods characterizing impacts on the ecosystems and human health are much more developed that impact assessment methods characterizing the impact of resource consumption. Most studies use the Abiotic Depletion Potential method (ADP) (Guinée, Heijungs 1995), which is not always well understood (see **background document** “Current state in resource efficiency evaluation”). This is a large limitation when studying SWM systems and technologies in the framework of circular economy.

LCA does not consider important qualitative factors such as odor and noise nuisances – In the early 90s, Heijungs et al. (1992) proposed a methodology to account for odors in LCA. However, during 25 years, no major development has been made and practitioners have rarely considered odor emissions in LCA studies (Peters et al. 2014). The latest development is from Peters et al. (2014), who proposed characterization factors for 33 odorants used to perform an odor footprint. Noise has been regularly considered by LCA practitioners and some methods and approaches have been proposed, but mainly to assess the nuisances associated with road traffic (Cucurachi et al. 2012). Therefore, an impact category focusing on the impact of noise still needs to be developed.

LCA is still at an early stage of method development to characterize the impacts on biodiversity – If not properly handled, waste can have significant impacts on biodiversity, especially on marine and coastal species. Today no method is able to fully assess the impact on biodiversity in LCA. However, recent advancements have been made (Penman et al. 2010; Koellner et al. 2013; Verones et al. 2015), especially on impacts caused by land use change. However, very few improvements have been made on how to account for the impact of waste on biodiversity. This is lacking, for example to assess the benefits of designing new plastics with a different degradability or lower toxicity for marine ecosystems (Thompson et al. 2009).

Characterization factors are sometimes defined for a group of substances instead of single substances – Groups of substances can include substances having very different contributions to one specific impact category (Ekvall et al. 2007). For example, some impact assessment methods do not differentiate NO and NO_2 and gather them in the Nox family, whereas in CML 2013, these two compounds have different characterization factors. This could have a significant influence on the results of an LCA.

LCA does not allow characterizing the impacts of substances mixes – In contact of each other, some substances present in a mix of released substances react to produce other compounds which can have a specific impact on the environment. Today, LCA only considers the impact of substances without considering the mix in which they are emitted and the reactions that can occur within that mix.

No mature assessment method exists to characterize occupational health impacts – Occupational health impacts are the impact affecting specific stakeholders involved along the process chain (e.g. waste collectors) (Laurent et al. 2014b).

Practice-dependent limitations of LCA applied to waste management

Some important impact categories are not considered in LCA studies. Among all the LCA studies reviewed by Laurent et al. (2014b), more than 40% did not include the assessment of toxic impacts in the analysis whereas waste management activities have been shown to have a significant impact on human health and ecosystems (Laurent et al. 2014b). Moreover, less than 50% of the studies included the assessment of non-renewable resources whereas most of waste management systems have a direct impact on resource consumption through the production of by-products such as material and energy. This is related to the fact that no consensus exist on which non-renewable resource depletion indicators should be used (Laurent et al. 2014b) and that resource depletion is not seen by the LCA community as an impact on ecosystem quality but as an impact on ecosystems' capacity to provide resources to the economy (Dewulf et al. 2015). Note that land use and water use are not always relevant for all waste-LCA studies (Laurent et al. 2014b). Therefore, the exclusion of such indicators can be considered but should be justified.

Most LCA practitioners in the SWM sector do not perform a thorough interpretation of the results. First, they rarely take into account the impact of the variability and the uncertainty of the input data on the results (see paragraph 2.5). Secondly, they rarely discuss the impact of methodological choices such as the choice of the allocation factors or the choice between system expansion, cut-off or allocation, on the conclusions of their studies.

2.3.6 Risk assessment

2.3.6.1 Applications and related indicators

Risk assessment (RA) is a term which gathers several types of assessments. Finnveden et al. (2006) defines two types of risk assessments applied in the SWM sector: chemical risk assessment and accident risk assessment. The aim of chemical risk assessment is to quantify the exposure (magnitude and duration) of the environment surrounding an industrial site to chemicals. It is divided into two main assessments methods: human health risk assessment and ecological risk assessment. Accident risk assessment evaluates the potential impacts associated with accidents (e.g. due to explosions, extreme natural conditions etc.) on the studied site and is more related to safety measures.

RA studies are applied for two main purposes in the SWM sector: assessing the risk of exposure in actual or planned conditions of site management or plant operation at a steady state (e.g. Cangialosi et al. 2008; Davoli et al. 2010) or assess the risk of pollution in case of the modification of the actual or planned conditions of site management or plant operation (e.g. Rapti-Caputo et al. 2006; Ollson et al. 2014a, b). Two main RA studies can be found in literature: studies quantifying the amount and fate of the emissions from a waste treatment site (called RA1 studies in this report), and studies quantifying the amount and fate of the emissions as well as the associated impact of their receptors (called RA2). Some studies focus on few specific substances while others focus on specific environmental compartments such as the aquifer or the surrounding atmosphere. Note that most RA studies follow the principle of precaution, i.e. they use maximum data from measurement campaigns. This is not the case for all RA studies, as some choose average data reflecting the real situation rather than a risk of pollution.

However, all RA2 studies follow the principle of precaution by using the maximum exposure factor to assess the impact of emissions on the receptors.

Most RA studies conducted in Europe are conducted in the framework of the application to environmental permits and are not publicly available in the literature. Among RA studies found in the literature (14 studies, based on Allesch, Brunner (2014) and further research), 43% have been conducted in Europe where they equally focus on incineration plants and landfill sites (Cangialosi et al. 2008; Davoli et al. 2010; Schuhmacher et al. 2001; Rapti-Caputo et al. 2006). Most of these studies are human health assessment studies.

Indicators used in human health assessment studies are mostly the hazard index for non-carcinogenic pollutants (also called hazard quotient or hazard ratio) and the cancer risk for carcinogenic pollutants. The approaches followed by the few studies conducting an ecological risk assessment are more diverse. They consider potential impacts on different environmental compartments or receptors in the environment, e.g. aquifers (Rapti-Caputo et al. 2006), surrounding wildlife (Ollson et al. 2014b) or surrounding soils and vegetations (Wang et al. 2011; Wang et al. 2012c). The only ecological RA study taking into account the effect of pollutants on wildlife used a similar approach than for human health impacts, i.e. taking into account the average daily dose ingested and/or inhaled by mammals and birds to calculate an ecological hazard quotient (Ollson et al. 2014b).

2.3.6.2 Advantages and limitations

One main advantage of RA studies is that they evaluate the risk of impact under local specific conditions. The difference between human health risk assessment and endpoint life cycle impact assessment of direct carcinogenic and non-carcinogenic emissions in LCA is very narrow. Both evaluate the type, fate and effect of chemicals on human communities. However, in LCA the fate of chemicals is assessed at a larger scale (e.g. at continental scale in ReCiPe 2008; Goedkoop et al. 2009) than in risk assessment, and average data on atmospheric, lithospheric and hydrologic characteristics are used. In risk assessment, the dispersion and fate of chemicals is assessed based on the specific local conditions of the studied project or site. This is particularly relevant to help decision makers choose the location of a waste treatment facility (Rapti-Caputo et al. 2006).

One intrinsic limitation of RA is that it cannot evaluate global scale issues such as climate change. Similarly, it focuses on emissions and does not evaluate the risks that a site or plant consumes specific resources from the environment (Benetto et al. 2007). Moreover, emissions from waste treatment facilities are not continuous and emissions dispersion models used in RA might be inappropriate (Pollard et al. 2006). Another limitation is related to the fact that RA is hardly accessible to non-experts and requires involving experts having specific knowledge on pollutant dispersion in the aquifer, lithosphere and/or atmosphere.

2.3.7 Ecological footprint

2.3.7.1 Applications and related indicators

The Ecological Footprint is a method used to calculate the area needed to provide the resources consumed and to absorb the wastes generated by an activity (Wackernagel, Rees 1996). It converts input flows in the area needed to produce them and output wastes and emissions in the area needed to absorb them. Moreover, the benefits associated with the valorization of SWM products (e.g. recycled material or energy) can be accounted for by calculating the counter footprint, corresponding to the EF avoided by the delivery of products to the market (Herva, Roca 2013). EF is not widely used in the SWM sector. It has been applied to complete results obtained with other methods such as LCA (Cherubini et al. 2009), MCDA (Herva, Roca 2013) or MFA and EA (Herva et al. 2014). EF has been coupled with the 33ubstance database to calculate EF characterization factors for products and services extracted from the database, including SWM processes such as incineration, landfilling and recycling (Huijbregts et al. 2008).

Results of an EF are expressed as a single indicator expressed in area needed per product or service unit.

2.3.7.2 Advantages and limitations

EF has the advantage to be simple to understand. The unit (m^2, ha, etc) is understandable by everyone and the single score synthesizing the results makes EF a useful communication tool. However, EF fails in accounting for all the environmental burdens associated with a product or a service. It does not account for minerals and metals (Swart et al. 2015), which is a large limitation when studying SWM systems and technologies in the framework of the circular economy. Moreover, the single score is difficult to use when decision makers have a specific target to achieve. Therefore, EF should be more considered as a preliminary screening tool (Huijbregts et al. 2008) and used in parallel of other environmental sustainability assessment methods.

2.4 Integrated sustainability assessment of SWM systems and technologies

2.4.1 Cost-Benefit analysis

2.4.1.1 Applications

Waste management projects can often be classified as major projects eligible for EU funds. When an application to an EU grant is prepared, the EU regulation 1303/2013 requires a CBA to be performed. CBA is mainly used to comply with regulation, but also as a complement methods to LCA e.g. in the choice of new waste management schemes

(EEA 2006). Indeed, CBA aims at assessing the utility of a project for society by focusing on economic impacts, thus taking into account costs and benefits to society that are not taken into account in LCA, e.g. impact on employment or nuisance due to odors and noise. CBA is therefore not applied at technology level, but to evaluate the costs and benefits of a waste treatment site or system.

A variant of CBA, cost-effectiveness analysis (CEA), is also used in the SWM sector. Both tools are very similar but the aim of CEA is to evaluate the cost of a project to fulfill a specific goal. Therefore, the result of a CEA is expressed as the cost of the project per benefit gained or impact saved (e.g. €/t CO_2 eq; €/number of lives saved etc). This method is used in the waste management sector when specific targets have to be fulfilled (e.g. decrease of CO_2 emissions in a specific region; Schneider et al. 2012) or when not all the benefits associated with a project can be monetized.

One benefit of using CBA is that it strengthens the concept of rational behavior in the decision-making processes (Tol 2003). Rational behavior is the assumption made on the fact that decision-making processes look for the “greatest good for the greatest number” (Tol 2003). Thus, CBA allows increasing the objectivity of the decision-making process.

2.4.1.2 Integration of environmental indicators

One major limitation of CBA is the difficulty to monetize externalities, i.e. non-market outputs produced by the project. In the SWM sector, the main externalities that can be identified are the emission/reduction of Greenhouse Gases (GHG) and air pollutants, the emission/reduction of soil and water contaminants and visual disamenities, noise and odors (EC 2014). GHG emissions are the main emissions taken into account in CBA studies. Emissions to soil and water are usually not taken into account, mainly due to a lack of data on the economic impact of such emissions (EC 2014). In general, the monetization of non-market goods is a difficult task in CBA and one of its main limitation (Karmperis et al. 2013). However, for some externalities such as noise and odors, monetization methods exist (willingness-to-pay, valuation based on hedonic pricing, i.e. the devaluation of real-estate surrounding a waste treatment facility) and are rarely taken into account in literature.

Moreover, even if the EU Guide for CBA of investment projects (EC 2014) requires to identify factors influencing waste demand over the project time frame, these factors as well as other financial/economic variables (e.g. discount rate) can be difficult to forecast (Karmperis et al. 2013). Therefore, the evolution of emissions and resource consumption over time are rarely taken into account. Similarly, prices related to environmental benefits will increase due to resource scarcity (Norden 2007) and this should be quantified. Practitioners are missing tools and methods to account for these variations over time.

2.4.2 Multi-criteria decision analysis

2.4.2.1 Applications

Multi-criteria decision analysis (MCDA) is a method used to choose the preferable option based on the ranking of criteria. The criteria are chosen by a panel of experts and for each considered option, a score is given to the criteria. This score represents the performance of the option with regard to each criterion. Then, a weight is assigned to each criterion and the scores are multiplied by these different weights. Therefore, each option obtains a final score, used as a basis for comparison (Hanan et al. 2013).

Studies applying MCDA in the SWM sector mainly focus on municipal solid waste and are mainly applied to waste management systems (Allesch, Brunner 2014). A majority of them is conducted for public authorities (Allesch, Brunner 2014). Each study using MCDA in the SWM sector uses a specific method or focuses on specific criteria. Therefore, there is not a harmonized way of applying MCDA to SWM systems. The use of MCDA supporting tools contributes to harmonize some aspects of the method. The most used software tools in the SWM sector are AHP, ELECTRE and PROMETHEE (Achillas et al. 2013). Moreover, MCDA is used to help stakeholders choosing the best options among several waste treatment systems (Ekmekçioğlu et al., 2010; El Hanandeh et al. 2010; Hanan et al. 2013) or to help choosing the best site for building waste treatment facilities (Gómez-Delgado et al. 2006; Aragonés-Beltrán et al. 2010).

2.4.2.2 Integration of environmental indicators

The main advantage of MCDA is that it allows ranking different options based on a set of criteria that can include issues related to the three pillars of sustainable development, but also different criteria based on several environmental indicators such as emission-based or resource-based indicators. Moreover, it allows taking into account the concerns of different stakeholders on the considered criteria, e.g. considering that a waste facility manager and local authorities do not give the same weight to GHG emissions in the decision process (Gomes et al. 2008). This is particularly relevant for the SWM sector as waste management projects involve a wide range a stakeholders (e.g. municipalities, citizens, operators, researchers, government). Another strength of MCDA is its ability to consider both quantitative and qualitative criteria (Karmperis et al. 2013), which is of high relevance in the evaluation of waste management systems as they are associated with disamenities which can be difficult to quantify (e.g. visual disamenities).

One main limitation of MCDA is the subjectivity of the chosen environmental criteria and weighting values. The choice of the criteria themselves is often limited and not justified by practitioners. A more objective choice of criteria could be enhanced by involving sustainability experts, e.g. as the work conducted by Renn et al. (2006), who gathered the feedback of 52 European experts on the choice of social indicators for MCDA (Hanan et al. 2013). Similar work could be conducted on the choice of environmental indicators in MCDA applied to SWM systems and technologies. Concerning the choice of the weighting factors, Kemal Korucu and Erdagi (2012) highlighted the fact that in studies aiming at choosing a location for the disposal of municipal solid waste, MCDA was generally conducted by a small number of professionals or the authors themselves. Therefore, a wider range of stakeholders should be involved. This issue is however well acknowledged among MCDA practitioners, and sensitivity analysis is often used to test the impact of the chosen values on the results (Gómez-Delgado et al. 2006; Hanan et al. 2013).

The most often considered environmental criterion to evaluate SWM systems is the amount of GHG emissions. Indicators based on resource consumption are not commonly considered. An example of resource-based indicator found in literature is the amount of recovered waste, expressed in kg (Karagiannidis, Perkoulidis 2009; Ekmekçioğlu et al. 2010). Note that this indicator has a limited value as it does not give any information on the quality and the possible use of the recovered waste. In general, apart from studies using results from previous LCA studies (e.g. Hanan et al. 2013), there is a lack of transparency concerning the source of the environmental impacts considered.

2.5 Uncertainty and sensitivity analysis of environmental evaluation and impact assessment results in the SWM sector

Sensitivity and uncertainty analyses are key steps of the interpretation of the results. Sensitivity analysis is a method which identifies the parameters having the greatest effect on the results. Uncertainty analysis (rightly called “uncertainty propagation” in Laurent et al. (2014b) is a step that quantifies the uncertainty of the results due to uncertainties on input modeling data. Both analyses allow putting the data used to model the studied technology or system into perspective with the results of the study and are key steps in the interpretation of the results. Sometimes, the border between sensitivity analysis and uncertainty analysis is difficult to define as uncertainty analysis can be based on the variation of some parameters. However, in the case of uncertainty analysis, this variation is based on actual measured or estimated values, e.g. uncertainty of the results due to measurement tools. These terms bring some confusion in the interpretation of the results, as some practitioners draw conclusions on the best treatment technology based on sensitivity analysis, whereas the chosen variation of the input data might not represent the real uncertainty of this data. Moreover, some practitioners use the term “sensitivity analysis” for scenario analysis (Laurent et al. 2014b), whereas it is a way of dealing with uncertainty for other practitioners (Vyzinkarova, Brunner 2013). The difference between uncertainty analysis and sensitivity analysis is therefore not clear among practitioners.

2.5.1 Environmental impact assessment studies

Sensitivity and uncertainty analysis are rarely conducted in MFA, SFA, EA and ExA studies, in which they are mainly handled qualitatively. In some MFA and SFA studies, data uncertainty is handled by studying several scenarios, e.g. baseline, upper and lower scenario (Andarani, Goto 2014; Vyzinkarova, Brunner 2013). In some other MFA/SFA studies, uncertainty propagation based on mathematical modeling is applied (Andersen et al. 2011; Allegrini et al. 2015). Quantitative sensitivity and uncertainty analyses applied in EA and ExA studies could not be found in literature.

Sensitivity analysis and uncertainty analysis are more common practice in LCA studies, even if their implementation is still limited. Around 50% of LCA studies on SWM systems and technologies do not perform such analyses (Laurent et al. 2014b), which significantly limits the validity of their conclusions, e.g. on the ranking of waste treatment technologies. Most of the studies reviewed by Laurent et al. (2014b) and which perform sensitivity and/or uncertainty analysis perform a scenario analysis. The fact that a minority of practitioners perform uncertainty analysis is due to the fact that such analyses are time consuming and that there is a lack of framework and tools to conduct such analyses. Clavreul et al. (2012) proposed a methodology in four steps (scenario analysis, uncertainty propagation, sensitivity analysis and analysis of the shift of process/system ranking due to the variation of key parameters) to conduct sensitivity and uncertainty analysis in LCA studies in the SWM sector. This guidance is indeed recommended, but from step 2 (uncertainty propagation), the analysis starts to be highly time and resource consuming as practitioners need to define a probability function (e.g. normal, log-normal, uniform) for each of the parameters of the model. Providing to practitioners tables which summarize probability functions for the most common parameters in the SWM sector could help spreading this method to a wider number of studies and strengthen their conclusions. This idea was already suggested by Sonnemann et al. (2003) but never implemented. Note that other approaches to conduct uncertainty analysis have been found in literature, e.g. by using two different waste-LCA software (Burnley et al. 2015) or two different databases to model the studied processes (Pires et al. 2011b). These methods have the advantage to be accessible to non-LCA experts, but are not controlled procedures, i.e. they do not allow identifying specific highly uncertain or key parameters in the system.

Most RA practitioners do not conduct any sensitivity or uncertainty analysis. When it can be found, such analysis are conducted following different approaches, from scenario analysis (Cangialosi et al. 2007) and Monte Carlo analysis (Durmusoglu et al. 2010) to qualitative discussion (Ollson et al. 2014a, b). However, in several RA studies, input data is chosen to reasonably maximize exposure and effects (Ollson et al. 2014a, b), which is also a way of dealing with the uncertainty of data on these parameters.

2.5.2 Sustainability assessment studies

When CBA studies are conducted in the framework of the application to EU funds, a risk assessment is required as a sensitivity analysis method[7]. Note that the term "risk assessment" here is different from the risk assessment method detailed in this report and is more related to a sensitivity analysis as defined for LCA studies, e.g. the modification of parameters to evaluate their impact on the results. As few CBA studies on waste management systems are available in literature, it is difficult to draw conclusions on the implementation of uncertainty and sensitivity analysis in CBA studies which are conducted outside of this framework. Several CBA studies on paper recycling and disposal conducting sensitivity analysis could be found in literature, but have been published before 2006 (EEA 2006). Sensitivity analysis based on the variation of the discount rate and price of CO_2 emissions (Beattie 2014), as well as the application of Monte-Carlo simulation based on the uncertainty of three parameters (gate fee, waste volume and electricity prices; PWC 2014) were found in literature.

Because of the high subjectivity of the choice of weighting factors in MCDA studies, MCDA practitioners often conduct sensitivity analysis on this parameter. However, the objectivity of the choice of this specific parameter could also be improved by involving a larger group of stakeholders.

2.6 Initiatives to improve environmental evaluation and impact assessment practice in the SWM sector

2.6.1 Combining LCA and risk assessment

The comparison of RA and LCA shows possible complementarities of the two methods to account for both global and local impacts (Figure 3). Figure 3 shows the difference of emissions and impacted environment considered by both methods. RA considers emissions having an impact on the local environment, whereas LCA considers emissions having an impact on local and global environment, but using characterization factors defined for a standard environment, i.e. based on average data at continental level.

[7] Regulation (EU) No 1303/2013

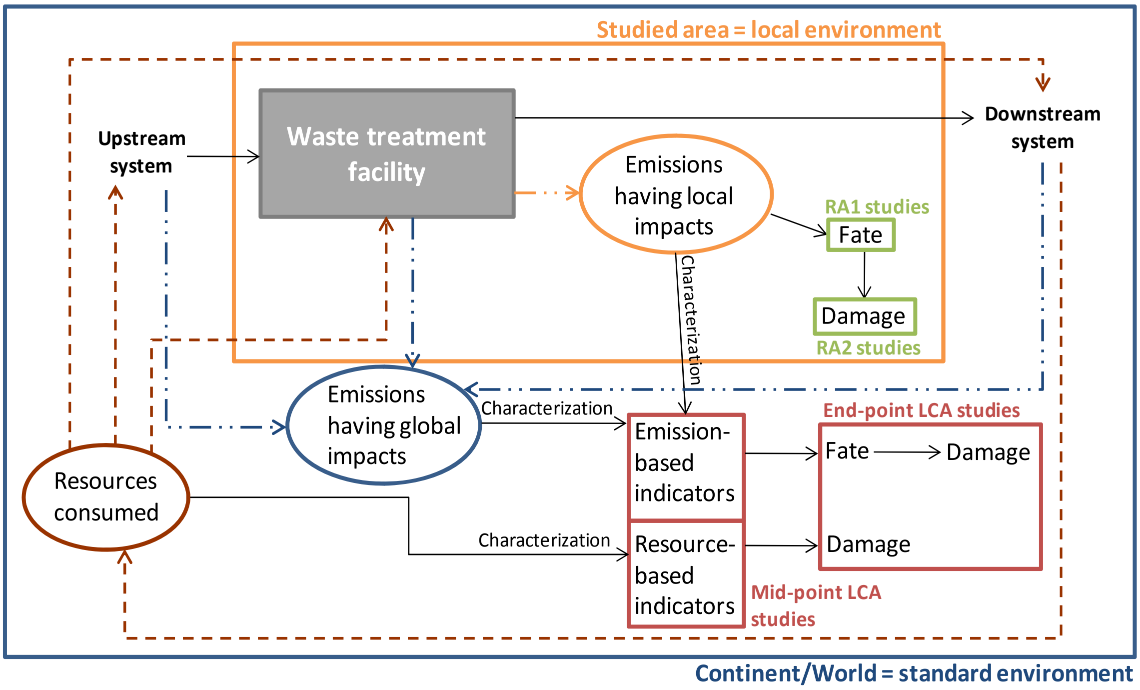

Figure 3: Comparison of the scales and types of emissions considered in RA and LCA studies.

Attempts to combine both methodologies have been published in the past, among which the three methodologies proposed by Benetto et al. (2007) and applied to the management of mineral waste. These three methodologies follow three different approaches to integrate both RA and LCA results. The first one (called “path 1” in Benetto et al. 2007) creates new impact indicators by multiplying the mid-point LCA ecotoxicity, acidification and eutrophication indicators by risk indexes related to each of the previously mentioned impact categories, and based on RA results (e.g. the risk index for acidification is calculated as the ratio of the concentration of substance i over the critical load of this substance in a specific compartment). The second method (path 2) proposes to substitute LCIA results for the three aforementioned impact categories by the related index risks. The third method (path 3) proposes to add new impact categories in addition to the LCA results, only based on the RA results. Then, a MCDA is conducted to compare the different scenarios. These three methodologies where tested on several case studies on the recycling of mineral, construction and demolition waste, showing that the integration of RA and LCA brings valuable insights, e.g. by taking into account emissions having a significant contribution on the local environment but which would not have had such a contribution on the standard environment considered in LCA. As mentioned by the authors, this methodology is a “ready to use” methodology for practitioners that already conducted a RA and an LCA separately. The idea of combining risk assessment and LCA has already been explored from the early 2000s until now in various sectors such as the construction (Nishioka et al. 2002), nanotechnology (Dhingra et al. 2010) and chemistry sectors (Olsen et al. 2001) as well as for application to industrial processes in general (Sonnemann et al. 2004) but without breakthrough of any proposed approach. One main limitation in improving environmental sustainability assessment by combining both methodologies is related to the large amount of resource and time required to conduct both studies at the same time, and a specific research project involving a wide range of experts from different fields as well as industry should be dedicated to this methodological improvement. However, the authors think that such a project should be encouraged and that combining LCA and RA would improve the quality and the value of environmental assessment studies in the SWM sector.

2.6.2 Existing tools to support evaluation and impact assessment in the SWM sector

2.6.2.1 Solid waste specific LCA tools

Several waste-LCA software tools have been developed to include the specificities of waste management systems which are not available in conventional LCA software tools such as Simapro and OpenLCA. These software tools provide waste specific processes and their associated databases. Moreover, they also propose a framework for system modeling, i.e. using assumptions to model key steps/processes in the studied system, such as collection schemes or substance transfer coefficients. Gentil et al. (2010) compared the technical assumptions of 9 waste-LCA software tools and highlighted the main differences between these tools.

Waste specific LCA software tools have a lot of advantages compared to conventional LCA software tools. However, some tools limit the freedom of practitioners to modify key data from the foreground system which contribute to conduct a geographic-specific analysis, e.g. the waste composition. Therefore, practitioners should pay attention to this default data that cannot be changed and compare them with geographic-specific data when available. This is also valid for the background system. In general, practitioners should be careful to clearly understand the assumptions behind each model.

Among the tools created over the past decades, very few are updated or subject to further developments in the long run. Among the 9 waste-LCA software tools reviewed by Gentil et al. (2010), only two are still under development: EASEWASTE, renamed as EASETECH after update by the Denmark Technical University, and MSW-DST, now released as SWOLF (Solid Waste Optimization Life-Cycle Framework) by the North Carolina State University (Clavreul et al. 2014). These two institutes are now co-developing the International Institute for Solid Waste Management Life-Cycle Modeling, which aims at developing consistent data and unify process models for the life cycle modeling of SWM systems. This initiative is particularly interesting knowing that both software tools have different features that could complement each other's, e.g. EASETECH is based on material flow modeling and SWOLF considers projected changes of the energy mix under various GHG mitigation policies in the next decades (e.g. 30 or 50 years).

2.6.2.2 STAN – a tool to support MFA and SFA

To facilitate the practical implementation of MFA and SFA, the Vienna University of Technology developed a free software, STAN (substance flow Analysis; Cencic, Rechberger 2008), which is now a recommended tool to perform MFA/SFA in the Austrian standard Önorm S 2096 (Austrian Standard Institute 2005). Practitioners can draw diagrams representing the material and substance flows entering, stocked and leaving the studied system. Moreover, STAN allows data reconciliation: when flows entered by the user are not fully consistent (i.e. the balance cannot be closed), the values are corrected to make contradictions disappear and unknown flows are calculated.

This software tool also allows entering uncertainties of the data, with the assumption that uncertainties are normally distributed. Based on this assumption, propagation of uncertainties along the system can be calculated following statistical equations. One limitation to this assumption is that uncertainties are only normally distributed for small uncertainties, as the higher the uncertainties are, the less symmetric the error intervals are (Vyzinkarova, Brunner 2013). In other words, the higher the uncertainties the user enters in STAN, the less accurate the results are. To overcome this issue, several scenarios can be analyzed separately and compared (e.g. in Vyzinkarova, Brunner 2013).

STAN has been used to conduct the SFA to analyze or improve specific processes (Di Gregorio, Zaccariello 2012; Andersen et al. 2011), to compare different waste treatment options (Arena, Di Gregorio 2013; Cascarosa et al. 2013) or to analyze waste management schemes at city or regional level (Vyzinkarova, Brunner 2013; Arena, Di Gregorio 2014). In these studies, STAN has proved to be a useful and reliable tool.

2.6.2.3 BIOMA – an engineering tool for data management

The software BIOMA was developed by the Technical University of Vienna to automatically calculate several parameters of input waste of WtE plants based on mass balance, e.g. amount of biogenic carbon and fossil carbon, moisture content or LHV/HHV. The original goal of this software was to quantify the amount of energy produced by WtE plants from biogenic sources. It uses measurement tools that are already implemented in WtE plants.

Results obtained from the use of BIOMA show a large variation of emissions during one year, highlighting the fact that measurement campaigns provide results with a very high uncertainty, and better data quality can be obtained by conducting mass balances based on routinely measured operating data from waste treatment facilities (Brunner, Rechberger 2015). Because of strict regulations on emissions of WtE plants to the environment, a wide range of measurement tools have already been implemented in these facilities. Similar tools could be implemented in other waste treatment facilities, allowing the use of BIOMA or a similar mass balance software tool. Therefore, the systematic use of engineer measurement tools to conduct mass balances in the SWM sector could be more developed and could highly participate to fill the data gaps and to improve data quality which both limit the sustainability assessment of waste management systems.

As a first step, a review of measurement tools already implemented in waste treatment facilities should be reviewed. Note that this is planned to be done during the revision of the BREF document on Waste Incineration and on Waste treatment (JRC 2014b; JRC 2015). Then, an analysis should be made to evaluate if data obtained by these tools would be enough to conduct mass balances at a chain level (e.g. waste processed in MBT facility, followed by incineration of RDF, recycling of plastic waste and composting of organic waste). If not, key measurement tools should be identified and their implementation in waste treatment facilities considered. In the far future, a centralized tool automatically gathering monthly data obtained from all the waste treatment plants on a defined territory could be implemented.

3 Outlook: key areas of further development

This study summarized the main tools and methodologies used in the SWM sector to assess the environmental sustainability of SWM systems and technologies. It highlighted the strengths and weaknesses of each of them. Several gaps related to their practical implementation and intrinsic characteristics were identified.

Specific waste composition should be used

To achieve this, an important work on data collection and measurement is needed. Engineering tools and methods to measure waste composition over time should be developed. For example, such tools could be based on the mass balance principle as done with the BIOMA software tool. Moreover, the uncertainty of the waste composition should be considered.

MFA and SFA should always be carried out

MFA/SFA is a key step to ensure that a realistic and consistent system is assessed. To facilitate the implementation of such analysis, the development and use of software tools such as STAN, allowing data reconciliation and filling the gaps of missing data. Generic data on transfer coefficient of substances in specific waste treatment processes should be gathered and more widely shared among practitioners.

In studies aiming at choosing the system or technology to be implemented within the next decades, practitioners should make predictions on the evolution of key factors within the project time frame

Sensitivity analysis should be conducted to identify these key factors, i.e. the ones for which a relatively slight change can modify the conclusions of the analysis (e.g. electricity mixes or waste composition). This would help optimizing the resource spent on the predictive model, i.e. to avoid predicting the evolution of all model parameters. Moreover, databases from other sectors such as the energy sector should be more widely used and made accessible to sustainability experts.

In the case of LCA studies, the modeling framework and the allocation method should be more systematically chosen

A review of the different modeling frameworks encountered in LCA studies on SWM systems and technologies could be performed and key choices which should be subject to sensitivity analysis identified and proposed to practitioners. This aims at prioritizing factors to be tested in the sensitivity analysis, which otherwise can be time consuming.

In the case of LCA studies, avoided products need to be much better identified

First, the choice of average or marginal electricity mix considered in the choice of the avoided processes should be better justified. If the marginal energy mix is chosen:

- LCA practitioners should conduct expert interviews and literature review to identify this energy mix or conduct sensitivity analysis to test the impact of their choice on the results

- Information provided by the energy sector should be shared and made accessible to LCA practitioners

Similarly, the choice of the avoided materials needs a thorough analysis to identify the possible substituted materials. Suggestions of databases, websites and reports should be made to make this task easier. Scenario analysis can also be conducted to test the choice of different substituted materials.

Research projects have to be encouraged to limit some intrinsic limitations of LCA

Environmental impacts should be better geographically characterized:

- For some specific waste treatment technologies such as incineration and landfill, practitioners should consider conducting a risk assessment in collaboration with experts from other disciplines (e.g. hydrology, climatology etc.)
- Projects on the regionalization of LCIA methods should be encouraged

Characterization factors should be as much as possible defined at substance level: a thorough analysis of substances emitted by waste treatment facilities could be carried out based on measurement tools which allow differentiating substances from a same family.

Impact on occupational health should be characterized:

- Research should be encouraged to characterize these impacts
- Combination of LCA and risk assessment can help filling this gap by taking into account direct health impacts on receptors such as workers.

Exergy-based analyses should be more widely used to evaluate processes efficiency

Exergy analysis is a well-recognized tool in the scientific community. However, its use within industry is limited. Efforts should be made to demonstrate the advantage of exergy analysis over energy analysis and to facilitate the calculation of exergy contents by developing more accessible databases and tools (e.g. see the software tool ExerCom compatible with the engineering tools Aspen Plus and Pro/II; study the possibility of integrating exergy calculation in waste-specific LCA software tools). Moreover, when evaluating processes efficiency, attention should be paid to potential impact shifting. This can be done by following a LCT approach.

The use and improvement of resource-based indicators should be encouraged

In available guidelines, resource-based indicators should be clearly differentiated from emission-based indicators and their specific significance clearly stated. Table 2 summarizes examples of indicators used in the SWM sector.

More meaningful indicators than “quantity of waste recovered” or “amount of waste sent to landfill” used in MCDA or MFA studies should be developed, e.g. by including the quality of these specific flows and putting them into perspective with the criticality of the recovered or disposed materials. For more information on resource efficiency indicators in process industries, see background document “Current state in resource efficiency evaluation”.

Table 2: Examples of indicators obtained from different methods and used by environmental sustainability experts and decisions makers in the SWM sector.

Method	Examples of indicators
MFA	Recycling and recovery rates Stock of material in landfill Amount of waste sent to landfill Amount of material recovered
SFA	Amount of substance to landfill Velocity of consumer stock evolution
Energy analysis	Lost and available feedstock energy Electricity efficiency Primary energy input to output
Exergy analysis	Rational exergy efficiency Functional exergy efficiency Universal exergy efficiency
LCA	Global warming potential Terrestrial acidification potential Resource depletion
Risk assessment	Hazard index for non-carcinogenic pollutants Cancer risk for carcinogenic pollutants Ecological hazard quotient

Work is necessary to turn sensitivity analysis and the calculation of uncertainty propagation into common practice

Journal reviewers have a key role concerning this particular issue. If such analyses are missing, reviewers have to report it to the authors. Moreover, supporting tools could be developed to help practitioners carry such analyses. A table gathering the probability functions of key parameters in SWM systems and technologies could be made available to practitioners. Indeed, because of the large number of data possibly available in the SWM sector, a project could be carried to collect data on each parameter within the EU and define associated probability functions.

The framework proposed by Clavreul et al. (2012) to conduct sensitivity analysis and calculate uncertainty propagation in LCA studies applied to SWM systems and technologies should be more widely tested by LCA practitioners.

To improve the outcomes of studies on the environmental sustainability of SWM systems and technologies, research projects should be encouraged to test the possibility to combine MFA/SFA, LCA and RA

As aforementioned, environmental evaluation and impact assessment studies (LCA studies in particular) should be based on material and substance flow accounting. This would highly improve the consistency of LCA studies. Moreover, it would help considering case specific waste composition, emissions to the environment and recoverable substances. RA can also help improving the characterization of the direct impacts of the studied technology or facility of primary interest by identifying the specific fate of the emissions under local conditions and the related potential impacts on the surroundings. Therefore, the possibility to integrate RA in LCA studies should be investigated.

Sustainability experts and decisions makers should better communicate to identify the type of study required to answer the decision makers' questions

Most of the time, decision makers order a sustainability assessment study to a research organization or consultancies. Decision makers should have a clear idea of the information they need, which should lead them to choose the best "method-expert" couple to answer to their question.

4 Abbreviations

ADP	Abiotic Depletion Potential
BREF	Best Available Techniques Reference Document
CBA	Cost-Benefit Analysis
CEENE	Cumulative Exergy Extraction from the Natural Environment
CexD	Cumulative Exergy Demand
EA	Energy Analysis
EEE	Electrical and Electronic Equipment
EF	Ecological Footprint
EIA	Environmental Impact Assessment
ELCA	Energetic Life Cycle Assessment
Ema	Emergy Analysis
EPD	Environmental Product Declaration
EU	European Union
ExA	Exergy Analysis
GHG	Greenhouse Gases
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LCSA	Life Cycle Sustainability Assessment
LCT	Life Cycle Thinking
MBT	Mechanical Biological Treatment
MCDA	Multi-Criteria Decision Analysis
MFA	Material flow analysis
RA	Risk Assessment
RBR	Recycling Benefit Rate
SFA	Substance Flow Analysis
SIA	Strategic Impact Assessment
SWM	Solid Waste Management
WtE	Waste-to-Energy

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