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Environmental and Economic Performance of Enhanced Landfill Mining

Maheshi Danthurebandara

Supervisors:
Prof. Dr. Karel Van Acker (KU Leuven)
Prof. Dr. Steven Van Passel (UHasselt)

Dissertation presented in partial fulfilment of the requirements for the joint degree of PhD in Engineering Science (KU Leuven) and PhD in Applied Economic Science (UHasselt)

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Supervisors:
Prof. Dr. Karel Van Acker (KU Leuven)
Prof. Dr. Steven Van Passel (UHasselt)
Members of the Examination Committee:
Prof. Dr. Adhemar Bultheel (KU Leuven)- Chair
Prof. Dr. Joakim Krook (Linköpings Universitet, Sweden)
Dr. Daneel Geysen (Group Machiels, Belgium)
Dr. Tom Kuppens (UHasselt)
Prof. Dr. Joost Duflou (KU Leuven)
Prof. Dr. Tom Van Gerven (KU Leuven)

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Uitgegeven in eigen beheer, MAHESHI DANTHUREBANDARA, KASTEELPARK ARENBERG 44, BUS 2450, 3001 HEVERLEE, BELGIUM.

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Summary

Enhanced Landfill Mining (ELFM) is an innovative concept developed to reintroduce historic waste streams present in the landfills to the material cycle either as energy or as materials. Besides, ELFM contributes to minimise the environmental and social impacts of landfills and to regain the large areas of land. The objective of this work was to investigate the environmental and economic performance of this ELFM concept. The research consisted of three parts.

In the first part, the general process flow diagram and a model to assess the environmental and economic performances of ELFM were developed. Traditional landfill mining case studies were considered in order to decide the major components to be in the process flow diagram of ELFM. The model was based on life cycle assessment and life cycle costing tools. The model allowed the assessment of the overall impact of the entire ELFM system, individual processes, and also of the trade-off between environmental and economic performances.

In the second part, the model was applied to the first ELFM case study to assess its environmental and economic feasibility. The study showed clear environmental benefits of ELFM against the landfill's existing situation. The study revealed that the thermal treatment process is the most important process both environmentally and economically. A cluster of environmental and economic drivers (recovery efficiencies of refuse derived fuel and metals, net electrical efficiency and investment cost of thermal treatment process, calorific value of refuse derived fuel, and price of electricity) that should be cautiously controlled in future ELFM studies, were identified. A separate analysis was conducted to identify the impact of use of thermal treatment residues to produce building materials such as aggregates and alternatives for products based on Portland cement. The study revealed that the quality of the products determines the net environmental and economic impacts of valorisation routes of thermal treatment residues. In addition, the study showed that the higher value applications of thermal treatment residues (alternatives for products based on Portland cement) are necessary to acquire the higher environmental benefits and economic profits. Furthermore, a detailed analysis was performed to identify the impact of plasma gasification in the context of ELFM. The study concluded that plasma gasification is a viable candidate in ELFM due to its combined energy and material valorisation capacity, which shows an overall better environmental and economic performance than a conventional incineration process. The analysis further confirmed that the environmental and economic performance of plasma gasification can be improved by using thermal treatment residues to produce building materials instead of landfilling.

In the last part, the ELFM concept was applied to reduce the environmental and socio-economic impacts due to open waste dumps in a developing country. It was found that the open waste dump mining largely eliminates the global warming potential caused by the emission from uncontrolled dumps. However, the study confirmed the need for the government's involvement in order to achieve the economic profits of open waste dump mining in a developing country.

Samenvatting

Stortplaatsontginning, beter bekend onder de Engelse term Enhanced Landfill Mining (ELFM), is een innovatief concept waarbij afval dat in stortplaatsen opgeborgen ligt terug in de materiaalkringloop gebracht wordt, hetzij als nieuwe grondstof, hetzij als energie. Daarnaast draagt ELFM ook bij tot het verkleinen van de ecologische en sociale impact van stortplaatsen en tot het herwinnen van land. De doelstelling van deze thesis is de ecologische en economische verdiensten van dit ELFM concept te onderzoeken. Het onderzoek bevat 3 onderdelen:

In het eerste gedeelte worden het algemeen stroomdiagram en het ontwikkelde model voor de beoordeling van de ecologische en economische impact van ELFM besproken. Praktijkvoorbeelden van voorlopers van stortplaatsontginning zijn bestudeerd om na te gaan wat de voornaamste onderdelen van een ELFM stroomdiagram zouden moeten zijn. Het opgebouwde model voor het beoordelen van de ecologische en economische impact is gebaseerd op levenscyclusanalyse en op life cycle costing. Het model laat ons toe de totale impact van het ELFM project, alsook de impact van de individuele processtappen, en de balans tussen de ecologische en economische verdiensten van ELFM te beoordelen.

In het tweede gedeelte is het model toegepast op de eerste ELFM implementatie in de praktijk, en werd daarmee de ecologische en economische haalbaarheid van het project beoordeeld. Deze studie toont een duidelijk ecologisch voordeel van ELFM aan in vergelijking met het continueren van de huidige stortplaats uitbating. De studie bracht verder aan het licht dat de thermische verwerkingsprocesstap de belangrijkste is zowel wat de economische als de ecologische impact van ELFM betreft. Er werd tevens een groep van ecologische en economische parameters geïdentificeerd, die beslissend zijn en nauwlettend gecontroleerd dienen te worden in toekomstige ELFM project, met name de efficiëntie van de nuttige toepassing van refuse derived fuel (brandstof uit afval) en metalen, het netto elektriciteitsrendement en de investeringskost van de thermische verwerking, de calorische waarde van refuse-derived fuel, en de marktprijs van elektriciteit. Een bijkomend onderzoek is gevoerd naar de impact van het valoriseren van thermische residuen in bouwmaterialen, zoals aggregaat en alternatieven voor de productie van Portland cement. Deze studie maakte duidelijk dat bovenal de kwaliteit van de bekomen producten de netto ecologische en economische impact van de valorisatieroutes voor thermische residuen bepaalt. Daarbij werd aangetoond dat toepassingen van thermische residuen met hogere toegevoegde waarde (zoals de alternatieven voor producten op basis van Portland cement) noodzakelijk is om meer ecologisch voordeel en economische winst te verkrijgen. Verder is er een gedetailleerde analyse van de impact van plasma vergassing als proces binnen ELFM uitgevoerd. De conclusie van deze analyse is dat plasma vergassing een haalbaar proces is in ELFM, omwille van de mogelijkheid tot gecombineerde energie- en materiaalvalorisatie en dat dit een globaal beter ecologisch en economisch profiel biedt dan conventionele afvalverbrandingsprocessen. De analyse bevestigt voorts dat het ecologisch en economisch profiel van plasma vergassing nog verder verbeterd kan worden door de

thermische residuen te gebruiken voor de productie van bouwmaterialen in plaats van ze te storten.

In het laatste gedeelte van de thesis is het ELFM concept vertaald naar de ontginning van open stortplaatsen in een ontwikkelingsland om de milieu en socio-economische impact van deze open storten te verkleinen. Er is vastgesteld dat ontginning van open stortplaatsen vooral de uitstoot van broeikasgassen grotendeels elimineert. Deze studie toont echter ook de nood van beleidsondersteuning om ontginning van open stortplaatsen in een ontwikkelingsland economische rendabel te maken.

Abbreviations

AIW	Assimilated industrial waste
APC	Air pollution control
BOD	Biological oxygen demand
$CaSO_4$	Calcium sulphate
CBA	Cost benefit analysis
CF	Cash flow
CH_4	Methane
CHP	Combined heat and power
CO_2	Carbon dioxide
COD	Chemical oxygen demand
CtC	Closing the circle
ELFM	Enhanced Landfill Mining
ELV	End of life vehicles
EWM	Enhanced waste management
GWP	Global warming potential
H_2O	Water
H_2S	Hydrogen sulphide
IRR	Internal rate of return
IW	Industrial waste
LCA	Life cycle assessment
LCC	Life cycle costing
LCI	Life cycle inventory
LFM	Landfill mining
MSW	Municipal solid waste
Na_2SiO_3	Sodium silicate
NaOH	Sodium hydroxide
NO_x	Nitrogen oxides
NPV	Net present value
OPC	Ordinary Portland cement
RDF	Refuse derived fuel
RQ	Research question
SCM	Supplementary cementitious materials
SO_2	Sulphur dioxide
TOC	Total organic carbon
V/V	Volume to volume
VOA	Volatile organic acid
VOC	Volatile organic carbon
W/W	Weight to weight
WtE	Waste to Energy
WtM	Waste to materials

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Chapter 1 : General Introduction

1.1. Background and scope

During the past 50 years, major paradigm shifts have occurred in management of both municipal solid waste (MSW) and industrial waste (IW) in Europe and worldwide. As shown in Figure 1.1, the first shift was to phase out uncontrolled landfilling. Since the 1980s, all landfills in Europe have been regulated to minimise environmental pollution and any threat to public health. However, regulated landfilling is also problematic for several reasons: (i) protection layers do not last, (ii) the after-care period is a maximum of 30 years and (iii) landfills require a large amount of space. To reduce materials to be landfilled, waste incineration was introduced, first without energy recovery and later with this feature. In many parts of the world, the quantity of MSW to be landfilled has been reduced due to the reuse and recycling approaches. According to the EU waste hierarchy outlined in the Waste Framework Directive (2008/98/EC), waste management has evolved to focus more strongly on the 3R concept: reduce, reuse and recycle.

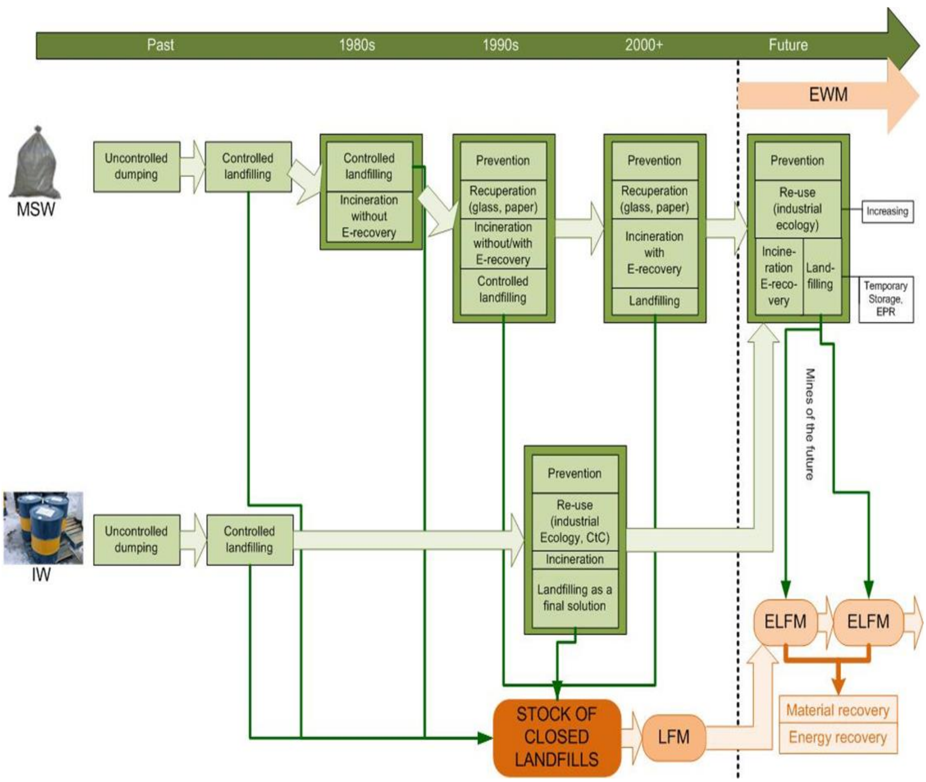


Figure 1.1: Schematic overview of the historic and future evolution of waste management (Jones et al. 2010)

Enhanced Waste Management (EWM) is an essential part of the broader concept of Sustainable Materials Management (SMM) which is defined as "an approach to promote sustainable materials use, integrating actions targeted at reducing negative environmental impacts and preserving natural capital throughout the life-cycle of materials, taking into account economic efficiency and social equity" (OECD 2005). EWM focuses on the waste phase of materials' life cycle. The difference with classical waste management is that waste is seen as a source of materials that needs to be prepared for a new application, instead of something that is simply to be discarded in an environmentally sound way (Wante 2010). In EWM prevention, reuse and recycling prevail over landfilling, while the latter is not accepted as 'a final solution' (Jones et al. 2010, Wante 2010). Under EWM, landfills are temporary storages or future mines of materials that cannot be recycled with existing technologies but have potential to be recycled in the near future. Today's practice of incineration eliminates the possibility of reusing materials and results in increased material costs. Therefore, 'temporary storage' is the first pillar of EWM because it ensures that more materials are recycled rather than incinerated or dumped.

In 2008, a transdisciplinary consortium of experts was established in Flanders (Belgium) in order to explore potential pathways to develop Enhanced Landfill Mining (ELFM) approach as the second pillar of EWM to integrate landfilling in a more sustainable waste management practice (Jones et al. 2010). As stated by Jones et al. (2013) the definition of ELFM is "the safe conditioning, excavation and integrated valorisation of (historic and/or future) landfilled waste streams as both materials (Waste-to-Material, WtM) and energy (Waste-to-Energy, WtE), using innovative transformation technologies and respecting the most stringent social and ecological criteria". Waste directed to temporary storages must be mined after a certain time period, and therefore, storage operations must be done so that mining and valorisation are efficient. For example, if high calorific values are required for energetic valorisation, the waste must be kept dry (Jones et al. 2010, Geysen 2013). Applicable to both new and old landfills, ELFM allows buried resource to be re-introduced into the material cycle and minimises the environmental burden of landfill emission. This option is denoted by 'stock of closed landfills' in Figure 1.1. ELFM is clearly distinct from traditional landfill mining (LFM), which is often limited to reclamation of landfill space, methane and a limited number of valuable metals such as copper or aluminium (van der Zee et al. 2004, Jones 2008, Prechthai et al. 2008). The concept of LFM has been introduced and practised around the world for over 50 years as a way of re-introducing buried resources into the material cycle and minimising the environmental burden caused by landfill emissions (Hogland 2002). However, LFM has not always been performed with a focus on resource recovery. The majority of the LFM studies have focused on conservation of landfill space and remediation, given the difficulty of obtaining permission to develop new landfills (Spencer 1990, Dickinson 1995, Cha et al. 1997, EPA 1997, van der Zee et al. 2004). Landfill mining has occasionally been used to simply restructure the landfill in more solid manner, due to landfill slope instability and inadequate landfill gas and leachate collection systems (Ayalon et al. 2006). However, the LFM history also includes some landfill mining projects whose main goal was materials and energy recovery (Cossu 1995, Cossu 1996, Canaleta and Ripoll 2012). There are a few exceptions in terms of projects that have explored the possibility of the recovery of specific valuable materials from waste deposits such as metals (Hino et al. 1998), foundry sand (Zanetti and Godio 2006) and waste fuel for energy generation (Rettenberger 1995, Obermeier 1997). In contrast to LFM, ELFM focuses on fully valorisation of all landfilled waste as both materials and energy and eventually regaining the land. As part of the sustainable approach, ELFM also incorporates the goal of preventing CO_2 emission during energy valorisation, such as by using carbon sequestration and storage techniques and using CO_2 as fertilizer in agricultural. The reclaimed land can be designated for nature, housing, agricultural or industrial purposes. As explained by Jones et al. (2013), in respect to the state-of-the-art, ELFM has a potential to generate several positive environmental effects: production of secondary raw materials through WtM not only saves energy but also land usage elsewhere, and energy generation through WtE avoids use of primary fossil fuels. ELFM also has a potential impact on local off-site gravel production as aggregates produced through ELFM over project life time can substitute for gravel in construction applications. In addition, ecosystem restoration after ELFM activities can have a positive impact on biodiversity (De Vocht and Descamps 2011).

The literature related to ELFM is limited due to the innovative nature of the concept. Jones et al. (2013) presented a constructive review by addressing ELFM as an opportunity for multiple resource recovery. Quaghebeur et al. (2013) presented a characterisation study in order to screen the potential of ELFM for a certain landfill, while Bosmans et al. (2013) performed a study on Waste-to-Energy technologies that could be used in ELFM. Moreover, Van Passel et al. (2013) addressed the carbon footprint and economics of ELFM considering the private and societal performance drivers. Although all of these aspects are definitely important, the knowledge about the critical factors of environmental performance of ELFM must also be developed in order to propel ELFM from the conceptual stage to the operational stage. Apart from technological improvements and breakthroughs, a multitude of socio-economic and environmental barriers need to be overcome. These barriers include: regulations, societal acceptance, environmental sustainability and economic uncertainty and feasibility. Although ELFM has already been identified as a productive way to increase resource autonomy in the coming years, it is still necessary to assess the most sustainable exploitation routes in order to maximise the economic return and minimise the environmental burden (Van Acker et al. 2010). This requires an integrated approach that addresses the complex interactions between economic costs and returns on one hand, and environmental considerations associated with ELFM on the other. In this background, the general objective of this research is to:

evaluate the environmental and economic performance of the novel concept of ELFM.

During the research, our focus is mainly given to a specific ELFM case study in Belgium: the Closing the Circle (CtC) project- the first ELFM project.

The methods and findings of the analyses performed for traditional landfill mining can be useful when assessing the environmental and economic performance of ELFM.

However the recent review done by Krook et al. (2012) highlighted that characterisation of deposited materials is the most studied main topic within landfill mining research in last two decades (Cossu 1995, Godio et al. 1999, Bernstone et al. 2000, Kurian et al. 2007). Most of those waste composition studies also address environmental and safety issues, but to a very limited extent. Emphasis has been on local risks related to the leaching of hazardous substances and formation of explosive and poisonous gas. The second most commonly addressed main topic is technology for excavation and materials processing (Cobb and Ruckstuhl 1988, Rettenberger 1995, Chang and Cramer 2003). According to Krook et al. theoretical discussions about possible benefits of landfill mining have also been performed briefly in many studies (Savage et al. 1993, Reith and Salerni 1997, Murphy 2000). Most of those studies have discussed the social benefits of landfill mining (Ayalon et al. 2006, Jain et al. 2013, Marella and Raga 2014, Zhou et al. 2015). Although many researches touch upon economic aspects, Krook et al. highlight that only two studies (Fisher and Findlay 1995, Van der Zee et al. 2004) strictly focus on this issue as their main topic. Fisher and Findlay mainly explain the costs associated with landfill mining processes while Van der Zee et al. present a method to explore the market for profitable landfill mining projects. This method includes a step-wise procedure for identifying high-potential landfills for mining in a region. Furthermore, Van der Zee et al. conclude land use as a major economic driver of landfill mining.

All these studies mainly explain a few processes of LFM especially excavation and separation while none of the studies draws the process flow of LFM combining, inputs, technologies and output products. Hence the necessity of a general process flow diagram for ELFM is foreseen and this can be introduced as the first step of addressing the main objective of this research. During this task, the focus should be placed on valorisation of all types of waste streams, which are in the landfill and also generated during different ELFM processes which will lead to design of general process flow diagram of ELFM including the inputs of ELFM, technologies, products of ELFM, final destinations of the products, and environmental and economic revenues. In all landfill mining cases, the associated processes are always the same but the technologies used are different. Thus, although the design process flow resembles the generic overview of ELFM, it can also be applied for specific landfills by choosing the relevant sub-processes and associated technologies. Hence the first research question of this study is:

RQ1. What is the generic process flow for landfill mining systems and how can it be applied to a specific landfill case?

As Krook et al. (2012) explained, there is no common framework for how landfill mining should actually be evaluated. For example, which economic aspects should be accounted for and how these different parameters should be calculated. In addition, environmental analysis is also lacking for both ELFM and LFM. Frändegärd et al. (2013) described an approach for environmental evaluation of landfill mining that combined the principles of life cycle assessment and Monte Carlo simulation. The authors focused only on three scenarios comprising landfill remediation and separation of landfilled waste either on-site or off-site; this leaves room for further research to address the full life cycle of landfill mining. Jain et al. (2014) provide operational data and enumeration of life cycle inventory for MSW landfill mining and identified the major environmental impact categories that are affected by two major cases: relocation of mined waste to a new landfill with modern environmental controls and the use of combustible materials in a WtE facility and recovery of recyclable metal components. Nevertheless, the authors mainly focused only on the emissions due to diesel fuel consumption within the landfill mining processes. In this context, Krook et al. (2012) highlight that standardised frameworks must be developed by applying a prominent systems approach, combining tools such as life cycle assessment and cost benefit analysis to evaluate the critical factors for economic and environmental performance of new landfill mining concepts. Within this framework, Van Passel et al. (2013) performed an integrated evaluation on carbon footprint and the economics of ELFM. However, their assessment is based only on greenhouse gas emission, which allows the possibility of further research on several other environmental impact categories. Furthermore, the authors divided the entire ELFM system only into two major parts: WtM and WtE, which means that identification of the relevance of each process associated with ELFM to the total environmental and economic impact is restricted.

Beyond this state-of-the-art, this research suggests an evaluation based on life cycle analysis (LCA) and life cycle costing (LCC) in order to identify the environmental and economic drivers of full scale ELFM project and then to evaluate the trade-off between the environmental and economic performances of ELFM. LCA and LCC are essential tools for consideration of both the direct and indirect impacts of waste management technologies and policies (Bogner et al. 2007). As explained in the previous paragraphs, analyses for LFM based on LCA and LCC are very limited. This provides room for us to perform a comprehensive LCA/LCC study for a full scale ELFM project. As the process flow of ELFM is a continuous chain, which is linked from a process to process with varying amounts of inputs and outputs, complex interactions can occur between the processes in terms of economic costs and returns and environmental impacts. Such complexity, should be addressed within the LCA/LCC modelling. Hence our second research question is as follows.

RQ2. How can the complexity of landfill mining exploitation be dealt with in LCA and LCC modelling?

Parallel to this research question, two other important aspects need to be considered. First, the sensitivity of different parameters on the environmental and economic feasibility should be identified. This will lead to the formation of a cluster of performance drivers which are needed to be considered in future ELFM projects. Secondly, knowing the effect of different scenarios comprising different technological processes is essential. In addition, benchmarking the ELFM scenarios against a reference scenario (i.e. a current landfill) is also important to identify how beneficial the ELFM scenarios are. This background resulted in two other research questions:

RQ3. What is the sensitivity of different model parameters on the environmental impact and the economic feasibility of ELFM?

RQ4. What are the effects of different scenarios for the landfill mining operations on the environmental and economic returns?

As explained in previous paragraphs, the initial studies on ELFM include a comparative technological analysis on thermal treatment processes that can be used in the valorisation of refuse derived fuel fraction obtained from landfills (Bosmans et al. 2013). The authors concluded that plasma gasification/vitrification is the most suitable candidate to be used within ELFM due to its combined energy and materials valorisation capacity. However, to realise the environmental goals of ELFM, only a technological analysis is not sufficient for a young technology such as plasma gasification. Detailed analyses from environmental and economic point of view are necessary for the processes specifically developed to be used in ELFM. Such analyses are not available yet for plasma gasification. Hence the next research question is:

RQ5. What are the environmental and economic impacts of new technologies (such as plasma gasification) developed specifically for landfill mining?

While the estimations for active, closed and abandoned landfills in the USA and in Europe are 100000 and 150000 (WBJ 2012, van Vossen 2005) respectively, the number of open waste dumps situated in the developing world are not yet estimated to an accurate value (Joseph et al. 2004). The threat to the environment of these open waste dumps are more severe than that of sanitary landfills as there are no gas and leachate management systems are installed in open waste dumps (Crowley et al. 2003). LFM concept has been applied in waste dump rehabilitation with the objective of upgrading the dump sites to sanitary landfills but not with the aim of resource recovery. In this situation, the following research question is drawn to explore the feasibility of applying novel landfill concepts such as ELFM in open waste dump mining.

RQ6. How can we transfer and apply the ELFM concept in open waste dump mining in developing countries?

Based on the above research questions this PhD thesis fills the need for an integrated environmental and economic assessment of ELFM based on LCA and LCC. To develop a LCA/LCC tool to understand the essential components of ELFM, we drew upon the first complete ELFM project, the so-called 'Closing the Circle' (CtC) project of Group Machiels, Belgium, which is described in the next section.

1.2. The Closing the Circle (CtC) ELFM project

Initiated in 2007, the CtC project investigated the opportunities and barriers of ELFM, using the REMO landfill site in Houthalen-Helchteren, Belgium as a case study. Valorisation tests were performed in 2009–2012. After 2013, the project entered a pilot phase. Full-scale WtE and WtM plants are planned, allowing resource recovery to start by 2017. The WtE and WtM plants will operate for 20 years, during which time the landfill site will be gradually developed into a sustainable nature park. For more details on the project time frame, see Jones et al. (2013) and Tielemans et al. (2010).

The REMO landfill contains more than 15 million tonnes of waste, both MSW and IW. The amount, type and location of various waste streams have been documented in a log book, which implies that separation and valorisation can be controlled, efficient and effective.

As Jones et al. (2013) explained, the CtC project mainly focuses on combined material and energy valorisation. CtC starts with the capturing and the valorisation of the landfill gas, and the processing of the leachate offering clean water to the site and its environment. After re-opening the landfill, waste is mined and fed to the material separation process. WtM targets to recuperate glass, ceramics, ferrous and non-ferrous metals, plastics, paper, wood, textiles, aggregate fractions and fines. The latter two are processed to ELFM building materials through a combination of processes. WtE valorises the recycling residue from the material recuperation process, the so called Refused Derived Fuel (RDF), containing mostly plastics, paper, wood and textiles. A characterisation study by Quaghebeur et al. (2013) found that the REMO landfill contained 23 to 50 percent combustibles and 40 to 60 percent fine-grained (<10mm) materials. Metals, glass, ceramics, stones and other inert materials in the landfill can be separated using advanced technologies. Because the fine-grained material contains high concentrations of heavy metals (Cu, Cr, Ni and Zn), there is great potential for metal extraction and recovery. For a detailed breakdown of metal composition, see Spooren et al. (2013). After screening several potential thermochemical conversion technologies (Chapman et al. 2010, Helsen and Bosmans 2011, Bosmans et al. 2013), the Gasplasma™/plasma gasification technology was selected for further trial runs due to its combined material and energy valorisation capacity. Residues of plasma gasification have a great potential and can be designed for use in diverse applications, mainly in the construction materials industry (Jones et al. 2013, Spooren et al. 2013).

In this context, CtC has the potential to generate several positive environmental effects mentioned in previous paragraphs. However to make ELFM more feasible, the environmental and economic benefits must be identified and realised, and therefore, an integrated decision tool to analyse the complex trade-off between economic, social and environmental aspects is needed.

1.3. Outline

To answer the above research questions, we visualised the outline in Figure 1.2.

Chapter 2 presents a detailed literature survey on the environmental and socio-economic impacts of landfills and their potential to be reservoirs for resources. The minimisation of environmental burdens of landfills through LFM and ELFM is also discussed.

Chapter 3, which partially addresses RQ1 and RQ2, describes the generic ELFM process flow diagram and development of the LCA/LCC model.

Application of the ELFM generic process flow to a specific landfill is explained in **Chapter 4**. Also, using a case study, LCA and LCC are used to assess the complex interactions of ELFM. Thus, Chapter 4 fully addresses RQ1 and RQ2. The chapter also answers RQ3 with a sensitivity analysis to identify which input parameters/variables significantly affect ELFM.

In **Chapter 5 and 6,** we assess the environmental and economic impact of various scenarios (RQ4). According to results discussed in chapter 4, the scenarios related to thermal treatment technology and the valorisation of thermal treatment residues are analysed in detail. The chapters also include discussions of the performance of new technologies and products developed specifically for ELFM, such as plasma gasification and building materials produced from residues of plasma gasification (RQ5). We also further address RQ3 with an assessment of the sensitivity of various parameters on the environmental and economic feasibility of the scenarios.

To explore the application of ELFM in the developing world, we applied the LCA/LCC model to assess the feasibility of open waste dump mining in Sri Lanka (RQ6), which is presented in **Chapter 7**.

Finally, an overall discussion, including conclusions and recommendations for further research, is given in **Chapter 8**.

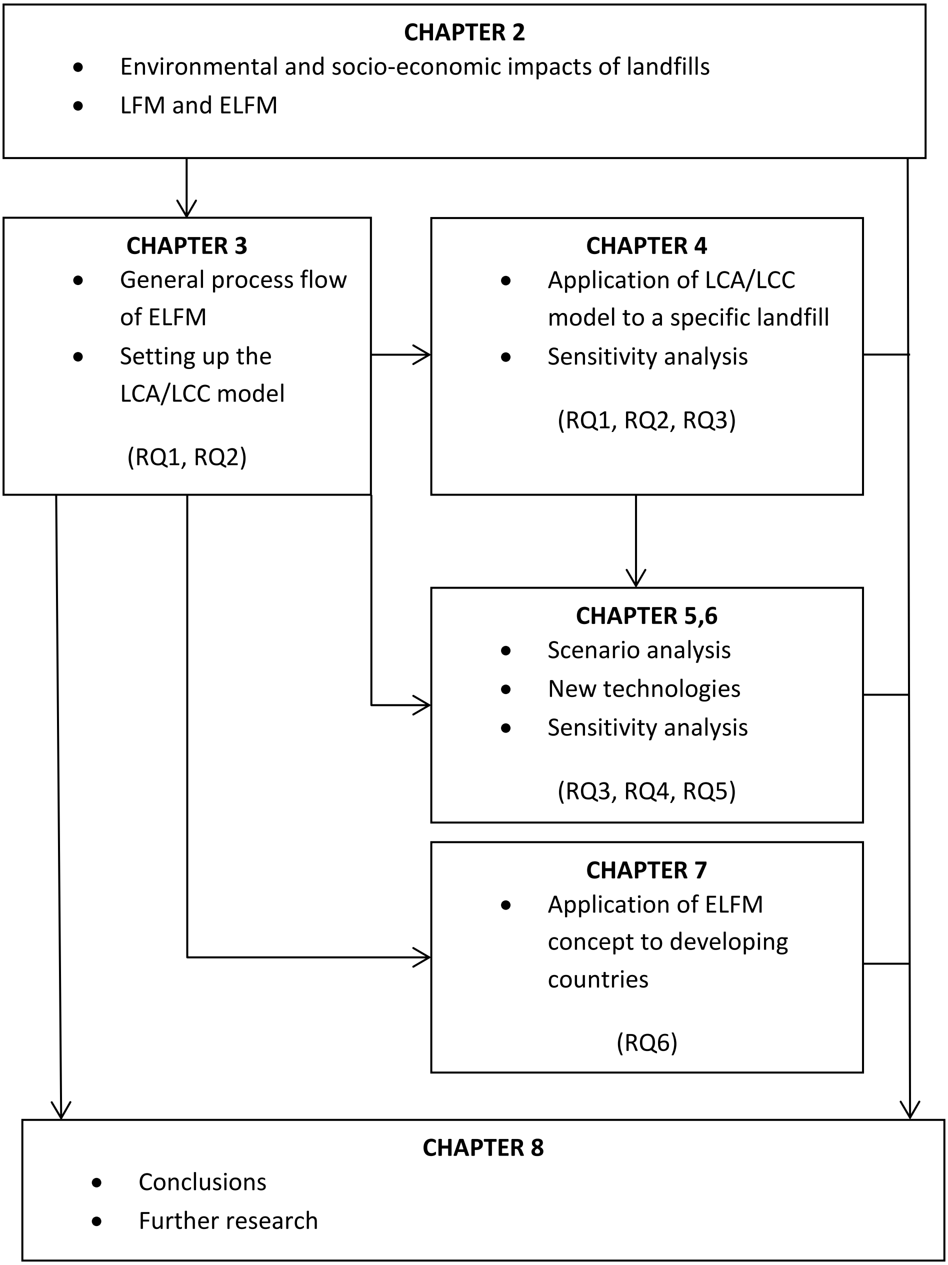


Figure 1.2: Outline to answer the research questions

Chapter 2 : Environmental and socio-economic impacts of landfills

This chapter is based on

Danthurebandara, M., Van Passel, S., Nelen, D., Tielemans, Y., Van Acker, K.. (2012). Environmental and socio-economic impacts of landfills. Linnaeus ECO-TECH-International conference on Natural Sciences and Environmental Technologies for Waste and Wastewater Treatment Remediation, Emissions Related to Climate, Environmental and Economic Effects. Kalmar, Sweden, November 26-28, 2012.

Abstract

A modern landfill is an engineered method for depositing waste in specially constructed and protected cells on the land surface or in excavations into the land surface. Despite the fact that an increasing amount of waste is reused, recycled or incinerated with energy recuperation, landfills still play an important role in waste management strategies. The degradation of wastes in the landfill results in the production of leachate and gases. These emissions are potential threats to human health and quality of the environment. Landfill gas consists mainly of methane and carbon dioxide, both important greenhouse gases. Leachate migrates to groundwater or even to surface water through the flaws in the liners, and causes water pollution. Socio-economic impacts of landfills include risks for public health derived from surface or groundwater contamination by leachate, the diffusion of litter into the wider environment and inadequate on-site recycling activities. Nuisances such as flies, odors, smoke and noise are frequently cited among the reasons why people do not want to reside close to landfills. Furthermore, the vast land requirement for landfills results in land scarcity in densely populated cities. This chapter further explains the structure of the landfills, and the environmental and socio-economic impacts related to landfill emissions. The study is also complemented with suggestions to minimise the environmental burden of landfills, and to re-introduce the buried resources to the material cycle.

2.1. Introduction

Despite the fact that the EU waste hierarchy, as set by the Waste Framework Directive (2008/98/EC) establishes the preference of reuse, recycling and recovery of waste above landfilling, a significant amount of waste still is landfilled (40 percent in EU-27 (Eurostat, 2011 #81)). It is a well-known fact that landfilling has environmental effects, mainly due to the long term methane emission and leachate production. Landfill gas consists mainly of methane and carbon dioxide and it can also contain a large number of other gases at low concentrations some of which are toxic (Crowley et al. 2003). The substances that are present in landfill gas are known to contribute to several environmental problems such as global warming, acidification, depletion of the quality of ecosystem as well as social issues like human health (Krupa 1996, EEA

2000, Ready 2005, Emery et al. 2007, Akinjare et al. 2011, Damgaard et al. 2011). Leachate production is also a major concern as leachate can migrate to surface and groundwater. This is more serious than river pollution because aquifers require extensive time for rehabilitation (Crowley et al. 2003). Landfill leachate may present significant concentrations of trace metals, nutrients such as nitrates and phosphates, ammonia and chlorides (Kjeldsen et al. 2002). Apart from the environmental burdens, occupation and requirement of the enormous space for landfills generates the issue of land scarcity for the development of human society and eco systems. Moreover, landfills decrease the market value of the surrounding area (Ready 2005, Akinjare et al. 2011). These impacts have been quantified by using various modeling approaches.

The purpose of this chapter is to review the existing literature on environmental and socio-economic impacts of landfills. Prior to describing the impacts of landfills, the structure of the landfills and their major emissions are briefly sketched. Evolving landfill concepts which are identified as solutions to minimise the potential environmental and socio-economic burdens of landfills are presented at the end of this chapter.

2.2. Landfills

2.2.1. Classification of landfills

A landfill is a location for depositing solid waste on the land surface or in excavations into the land surface. Landfilling is the term used to describe the process in which solid waste residuals are placed in a landfill. During the past 50 years, major paradigm shifts have occurred in landfilling in Europe as well as in the rest of the world. The first shift was the phasing out of uncontrolled landfills due to introducing a number of regulations. Then controlled landfilling has been further developed with specially constructed and protected cells for waste deposition. In Europe, landfills are classified in one of three categories according to the European Council Directive 1999/31/EC:

- Landfills for hazardous waste: A waste stream is considered hazardous when it displays characteristics listed in Directive 91/689/EEC (eg. anatomical, pharmaceutical substances, biocides, etc.) or when it contains one of the constituents listed in the same directive (eg. Beryllium, Vanadium, etc.)
- Landfills for non-hazardous waste: These landfills are used for municipal solid waste and non-hazardous waste of any other origin.
- Landfills for inerts: Inert waste that does not undergo any significant physical, chemical or biological transformation are deposited in this type of landfills.

Since each landfill can only be classified in one category, co-disposal is no longer possible in Europe. Nevertheless, the possibility of exploiting more than one category of landfills on the same site exists. As mentioned in the European Council Directive (1999/31/EC), liquid waste, flammable waste, explosive waste, oxidizing waste, corrosive waste, hospital and other clinical waste arising from medical or veterinary establishments and tyres are not accepted in landfills. Special regulations exist for the exploitation of monofills (landfills where only one specific waste stream which is produced in large amounts is deposited). Depending on the characteristics of the waste stream, they are classified in one of the above categories.

2.2.2. Landfill structure

In the past, the term 'sanitary landfill' was used to denote a landfill in which the waste was covered at the end of each day's operation. Today, sanitary landfill refers to an engineered facility for the disposal of waste designed and operated to minimise public health and environmental impacts. Within the landfill, biological, chemical and physical processes occur and they promote the degradation of waste and result in the production of leachate and gases. Collection, treatment and disposal of leachate and gas are described in detail underneath. The landfill design and construction must include the elements that permit control of leachate and gas. The major design components of a landfill, as shown in Figure 2.1, include the (i) bottom liner, (ii) leachate collection and management system, (iii) gas management facilities, (iv) storm water management, (v) intermediate covers and (vi) final cover (Tchobanoglous et al. 1993, Vesilind et al. 2002).

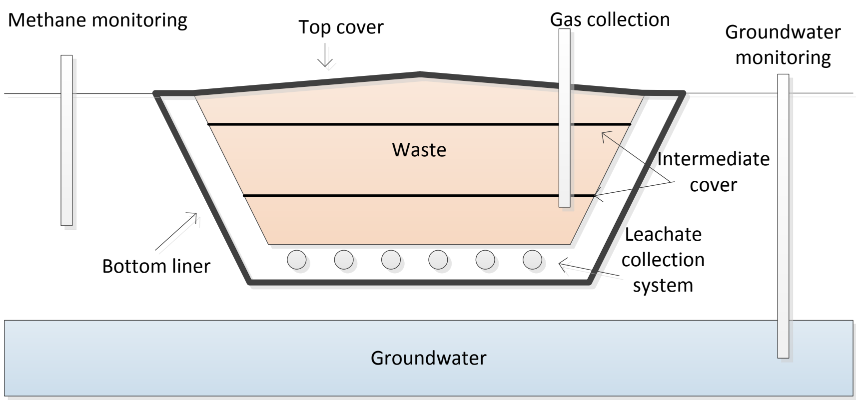


Figure 2.1: The major design components of a landfill based on Vesilind et al. (2002)

The bottom liner system is required to prevent migration of leachate from the landfill and to facilitate removal of leachate. It generally consists of multiple layers of natural materials and/or geomembranes selected for their low permeability. Landfills are designed with single, composite or double liners depending on the local geology and environmental requirements of the landfill site. For example, in locations where there is no direct contact to groundwater, a single compacted clay liner could be sufficient. In locations where both leachate and gas migration must be controlled, the use of composite liner composed of a clay liner and a geosynthetic liner with an appropriate drainage and soil protection layer is necessary. Collection of the leachate that accumulates in the bottom of a landfill is usually accomplished by using a series of sloped terraces and a system of collection pipes. The collected leachate is removed for treatment or re-application to the surface of the landfill.

A landfill gas management system comprises a gas collection system and a gas utilization system. The gas collection system links collection wells with piping and extracts the gas under vacuum created by a central blower. Collection wells may be vertical or horizontal wells, although vertical wells are more frequently employed. Landfill gas can be flared on site, however this is not a beneficial application of this resource. Beneficial energy recovery systems include electricity and heat generation and conversion to chemicals or fuels.

A storm water management system is required in order to minimize the production of leachate, erosion and contamination of surface water. Storm water runoff and runon is prevented by diverting storm water from active areas of the landfill. Typical measures to control the storm water include constructing ditches, dikes or culverts to divert the flow. Intermediate cover layers are used to cover the wastes placed each day to enhance the aesthetic appearance of the landfill site, to limit the amount of surface infiltration and to eliminate the harboring of disease vectors. The types of materials that have been used as intermediate landfill cover include a variety of native soils, composted municipal solid waste (MSW), composted yard waste, agricultural residues, synthetic foam, geomembranes, construction and demolition waste, incineration bottom ashes, etc.. Once the landfill reaches the design height, a final cap is placed to minimise infiltration of rain water, to minimise dispersal of wastes and to facilitate long-term maintenance of the landfill. A modern landfill cover is made up of a series of layers, each of which has a special function. The final cover consists, from top to bottom, of vegetation and supporting soil, a filler and drainage layer, a hydraulic barrier, foundation for the hydraulic barrier and a gas control layer (Vesilind et al. 2002).

Landfill monitoring is critical to the operation of a landfill. Most commonly, the landfill monitoring comprises the following: (i) leachate head on the liner, (ii) leakage through the landfill liner, (iii) groundwater quality, (iv) ambient air quality, (v) gas in the surrounding soil, (vi) leachate quality and quantity, (vii) landfill gas quality and quantity, and (viii) stability of the final cover. Once a landfill is closed, the land can be reused for many purposes such as natural areas, recreation parks, golf courses, parking lots etc..

2.2.3. Landfill emissions

Landfill gas and leachate are known as the major emissions of landfills. In addition, wind-blown litter, vermin and insects are identified as the minor emissions. However, the following discussion is limited to the landfill gas and leachate as they are the most important causes for a number of environmental and socio economic impacts.

The landfill ecosystem is quite diverse due to the heterogeneous nature of waste and the variety of landfill operating characteristics. The diversity of the ecosystem promotes stability; however the system is strongly influenced by environmental conditions such as temperature, pH, the presence of toxins, moisture content and the oxidation reduction potential.

Phases of landfill

A number of studies have suggested that the stabilisation of waste proceeds in five sequential and distinct phases (Vesilind et al. 2002). During these phases, the rate and characteristics of leachate produced and gas generated from a landfill are dissimilar which reflects the different microbial processes taking place inside the landfill. Figure 2.2 shows these different phases and their respective processes and products. As explained by Tchobanoglous et al. (1993) and Vesilind et al. (2002), the phases illustrated in Figure 2.2 can be described as follows.

- Phase I (Initial adjustment phase): This phase is associated with the initial deposition of solid waste and accumulation of moisture within landfills. The decomposition occurs under aerobic conditions because a certain amount of air is trapped in the landfill and generates the compounds of CO_2 and H_2O. Phase I decomposition can last for days or months, depending on how much oxygen is present when the waste is disposed of in the landfill.
- Phase II (Transition phase): In the transition phase transformation from aerobic to anaerobic environment occurs. Hydrolysis and fermentation processes take place and measurable concentrations of chemical oxygen demand (COD) and volatile organic acids (VOAs) can be detected in the leachate by the end of this phase.
- Phase III (Acid formation phase): The continuous hydrolysis of solid waste followed by the microbial conversion of biodegradable organic content results in the production of intermediate volatile organic acids at high concentrations throughout this phase.
- Phase IV (Methane fermentation phase): Intermediate acids are consumed by methanogenic bacteria and converted into CH_4 and CO_2.
- Phase V (Maturation phase): During the final state of landfill stabilisation, nutrients and available substrate become limited, gas production dramatically drops and leachate strength stays steady at much lower concentrations. Reappearance of oxygen and oxidized species may slowly be observed.

The time scale of these phases varies with the physical, chemical and biological factors within the landfill environment, the age and characteristics of landfilled waste, the operational and management controls applied as well as the site-specific external conditions.

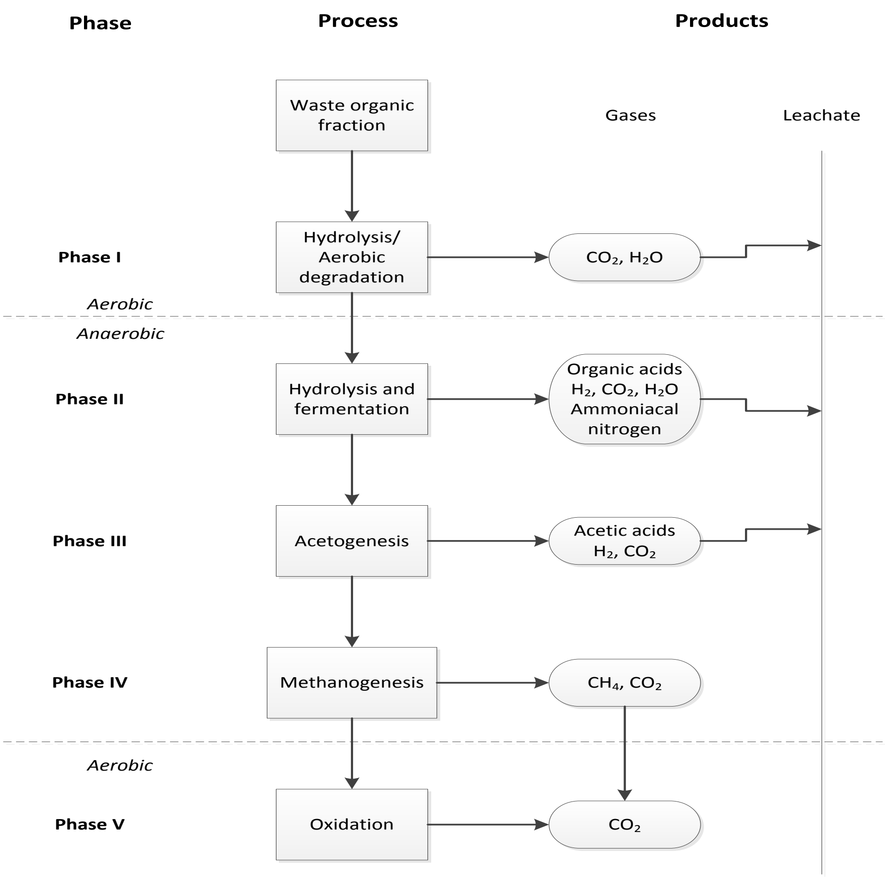


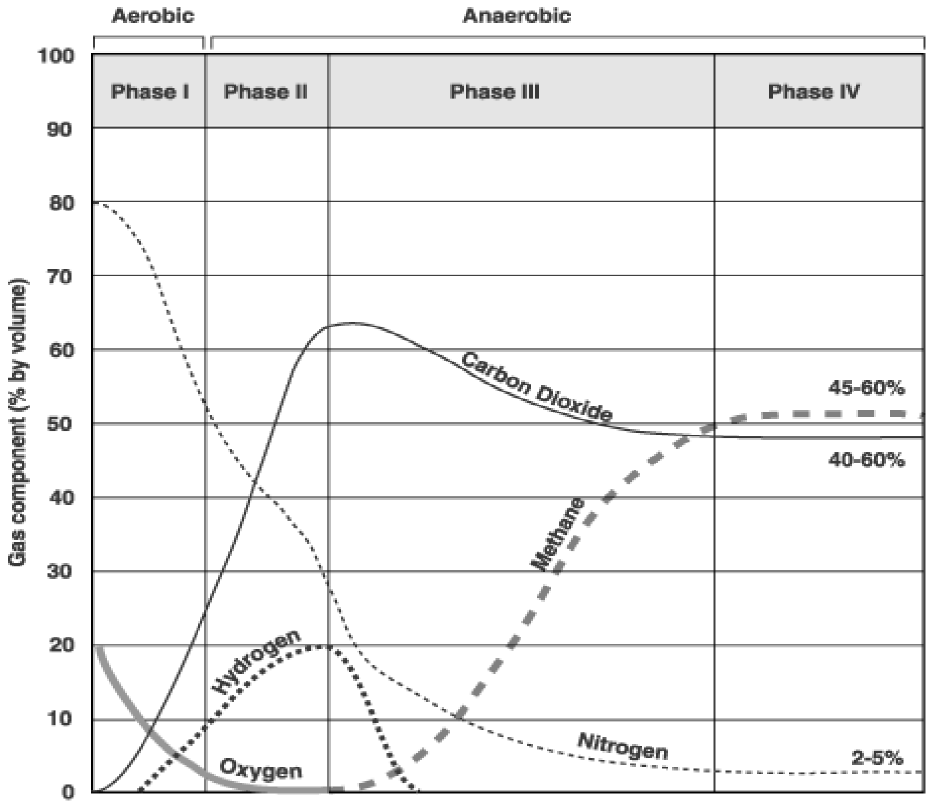
Figure 2.2: The major stages of waste degradation in a landfill (DOE 1995)

Landfill gas

Landfill gas production at different phases are illustrated in Figure 2.3. As explained in earlier paragraphs, oxygen depletes during the first two phases. CO_2 and CH_4 emissions increase gradually and reach a steady state. In theory the biological decomposition of one tonne of MSW produces 442 cubic meters of landfill gas containing 55 percent methane and a calorific value of 15 - 21 MJ/m^3 (EPA 2000), which is approximately half that of natural gas. The major components of landfill gas are CH_4 and CO_2, with a large number of other constituents at low concentrations such as ammonia, sulfide and non-methane volatile organic compounds (VOCs) (Crowley et al. 2003). Moreover, the US EPA (1991) listed ninety four non-methane organic compounds found in air emissions from MSW landfills, which included benzene, toluene, chloroform, vinyl chloride, carbon tetrachloride, and 1,1,1trichloroethane. Forty one are halogenated compounds. Toluene, xylenes, propylbenzenes, vinyl chloride, tetrachloroethylene, methanethiol and methanol have been reported from landfills that received both municipal and industrial wastes (O'Leary and Tansel 1986). In addition, inclusion of large amounts of particular types of industrial waste in a landfill can generate high quantities of other gaseous compounds. For example, a very large proportion of plasterboard (i.e. gypsum, $CaSO_4$) may cause the emission of H_2S (Westlake 1995). As explained in previous paragraphs, landfill gas is generally controlled by installing vertical or horizontal wells within the landfill. These wells are either vented to the atmosphere or connected to a central blower system that pulls gas to a flare or to a treatment process. The Intergovernmental Panel on Climate Change (IPCC) reported that the landfill gas collection efficiencies range from 9-90 percent and estimates an average of 20 percent (IPCC, 2006). The uncaptured gas can pose an environmental threat because methane is a greenhouse gas and many of the VOCs are odorous and toxic. Landfill sites contribute 20 percent of the total global anthropogenic methane emission (Hutchinson et al. 1997).

Leachate

Leachate is defined as any liquid percolating through the deposited waste and emitted from or contained within a landfill. As it percolates through the waste, it picks up suspended and soluble materials that originate from the degradation of the waste. The principal organic contents of leachate are formed during the breakdown process described above and its organic strength is normally measured in terms of biochemical oxygen demand (BOD), chemical oxygen demand (COD), or total organic carbon (TOC) (Crowley et al. 2003). Leachate characteristics during the waste degradation phases are summarized in Table 2.1 as presented in Vesilind et al. (2002). In addition, the MSW leachate contains a wide variety of hazardous, toxic or carcinogenic chemical contaminants (EEA 2000). Mining wastes, sewage sludge and air pollution control residues deposited in landfills contain high concentrations of trace metals, a range of acids and even radioactive material. Under the acidic conditions hazardous trace metals such as copper, cadmium, zinc and lead dissolve and travel with leachate (Crowley et al. 2003). The characteristics of leachate produced are highly variable depending on the composition of the waste, precipitation rates, site hydrology, compaction, cover design, waste age, sampling procedures and interaction of leachate with the environment and landfill design and operation. It is important to control and manage the leachate production and discharge due to the potential threat of it to both the environment, particularly groundwater, and human health. An effective leachate collection and removal system is a prerequisite for all non-hazardous and hazardous landfill sites and it must function over the landfill’s design lifetime.



Note: Phase duration time varies with landfill conditions (Source: EPA 1997)

Figure 2.3: Production phases of landfill gas (EPA 1997)

Table 2.1: Leachate composition in different phases of landfill stabilisation (Vesilind et al. 2002)

Parameter	Phase II	Phase III	Phase IV	Phase V
COD (mg/l)	480-18000	1500-71000	580-9760	31-900
Total volatile acids (mg/l as acetic acid)	100-3000	3000-18800	250-4000	0
Ammonia (mg/l-N)	120-125	2-1030	6-430	6-430
pH	6.7	4.7-7.7	6.3-8.8	7.1-8.8
Conductivity (µS/cm)	2450-3310	1600-17100	2900-7700	1400-4500

2.3. Impacts of landfills

As with any waste management activity, landfilling is also a potential threat to the quality of the environment due to its gaseous and leachate emissions as well as wind-blown litter and dust. There are also substantial environmental effects associated with waste transport and collection. Apart from the environmental impacts, significant socio-economic impacts due to landfills can also be identified. In order to calculate these impacts quantification of landfill emission is essential. Modeling landfill emissions and their impacts already exists for several decades. Many researchers have conducted studies to evaluate the landfill emission management. Most of the studies are mainly about landfill gas and leachate and a few of them address nuisances like odor, dust and noise. The section 2.3.1 underneath summarizes the different modeling approaches available to evaluate and quantify the landfill emission and their environmental and socio-economic impacts. In the sections 2.3.2 and 2.3.3, the environmental impacts, and health and socio-economic impacts of landfilling are discussed, making use of the results of the modeling approaches discussed in 2.3.1.

2.3.1. Modelling approaches to assess the landfill emission and their impacts

The first landfill gas formation models were made to help determine the size of landfill gas recovery projects. They estimate the amount of formation and include future expectation and gas recovery. More recent models quantify methane emission. As described in the review of Oonk (2010), modeling of methane emission generally requires modeling of methane generation, measuring landfill gas recovery and assuming some methane oxidation. The emission equals the gas generation minus the gas recovery minus the gas oxidation.

According to Oonk, the major issue when modeling methane emissions is the modeling of the methane or landfill gas formation. Most of the models are based on a first order decay model (first order decay models have one half-time of biodegradation) or a multi-phase model (multi-phase models consider 3 fractions: fast, moderate and slow degradation of waste, each with their own half-time of biodegradation). Modeling oxidation has received less attention: in most cases 10 percent of the methane flux through the top layer simply is assumed to be oxidized. Nevertheless, more recent models are being developed for the evaluation of methane oxidation as well. The most widely applied generation models are the IPCC model, the TNO model, GasSim Lite, Landgem, the Afvalzorg-model, the French E-PRTR-model and the Finnish E-PRTR-model (Oonk 2010). The IPCC model is intended to give guidance to national authorities in the quantification of methane emissions from all landfills in a country. But the model itself can also be used for individual landfills. The choices exist between a first order decay model and a multi-phase model. The IPCC model accommodates four different climate regions (IPCC 2010). TNO is the first model in which model parameters were based on real data of landfill gas generation in a larger group of landfills. Both a first order and a multi-phase model were made, that describe landfill gas generation as a function of amount of waste deposited from different origin (Oonk et al. 1994, Oonk and Boom 1995). GasSim Lite quantifies all landfill gas problems of a landfill, ranging from methane emissions, effects of utilization of landfill gas on local air quality to landfill gas migration via the subsoil to adjacent buildings (Oonk 2010). Landgem is a first order decay model, with separate default values for the rate constant of biodegradation for conventional and arid regions (EPA 2000). The Afvalzorg model itself is a multi-phase model and is intended to give a more realistic prognosis of methane generation at landfills with little or no household waste deposited. The French E-PRTR-model is a simplified first order decay model and the Finnish E-PRTR-model is a multi-phase model with model parameters for different climatic regions (Oonk 2010). In addition to these models, 3D models have been developed for transport and reaction of gaseous mixtures in a landfill (Hashemi et al. 2002, Sanchez et al. 2006, Sanchez et al. 2007, Sanchez et al. 2010, Li et al. 2011).

Successful prediction of the amount of landfill leachate generated and its composition is a highly complex and difficult task. As discussed in previous sections, the amount of leachate generated is primarily a function of water availability, waste characteristics and landfill surface conditions. Similar to landfill gas, numerous leachate generation and transport models have been developed. These models can be classified into two types: (i) models that emphasize only the quantity of leachate generated; and (ii) models that combine both quantity and composition (El-Fadel et al. 1997). Among these models that can estimate the volume of leachate generated from a landfill, the Water Balance Method (WBM) is the most commonly used one (Baccini et al. 1987, Gee 1987, El-Fadel et al. 1997). The WBM simply states that water infiltrating through the landfill cover and past the depth influenced by evapotranspiration will eventually emanate from the landfill as leachate. This is valid after the solid waste reaches absorptive capacity for holding water, which may take several years. Although this method is theoretically correct and simple, a great degree of uncertainty is associated with estimating its variables (El-Fadel and Khoury 2000). Demetracopoulos et al. (1986) built up a mathematical model for the generation and transport of solute contaminants through a solid waste landfill. A 3D mathematical model has been developed by Demirekler et al. (1999) to estimate the quality and quantity of the leachate produced. The model takes the effects of changing hydraulic conductivity with overburden pressure and time dependent landfill development into consideration. Laner et al. (2011) suggested a methodology to estimate future emission levels, mainly leachate, for a closed MSW landfill. The approach is based on an assessment of the state of the landfill including detailed analysis of landfill monitoring data, investigations of the landfill waste and an evaluation of engineered landfill facilities.

Apart from these gas and leachate generation models, many modeling approaches have been developed for assessing the environmental and socio-economic impacts of landfills. Landfill modeling in life cycle analysis (LCA) is the most common approach and a few of them are as follows. Obersteiner et al. (2007) introduced and discussed the different approaches concerning time horizon and life cycle inventory data for landfills in Central Europe. Damgaard et al. (2011) performed an economic and environmental evaluation of landfill leachate and gas technologies by using waste LCA model EASEWASTE. A methodology to estimate future emission rates and evaluate the response of the affected environment based on the current state of the landfill and its surroundings has been introduced by Laner et al. (2011). They present a modeling approach to evaluate residual environmental impacts in view of different post closure management strategies. In addition, numerous LCA studies have been conducted to compare the environmental impact of landfills with that of other waste treatment technologies (Clift et al. 2000, Finnveden et al. 2005, Moberg et al. 2005). Furthermore, Úbeda et al. (2010) developed a Gaussian dispersion model to evaluate the odor impact from a landfill area. Apart from environmental modeling a few studies report for economic models of landfills. Similar to the environmental modeling, landfilling has been compared with the other waste management systems from an economic point of view (Reich 2005, Emery et al. 2007). Some studies have been performed to assess the social impacts of landfills. Assessing the impact of landfills on residential property values is an example (Reichert et al. 1992, Ready 2005, Akinjare et al. 2011).

2.3.2. Environmental impacts of landfills

Impacts during the construction phase

Site selection of waste management facilities can be a major issue as all infrastructural projects have the capacity to damage the ecology of the site on which they are developed, causing landscape changes, loss of habitats and displacement of fauna. Such impacts are generally site specific and need to be assessed on a case by case basis (EPA 1995a, EPA 1995b, Treweek 1999, Crowley et al. 2003). The selected sites tend to suffer from high levels of disturbances and their chemical and physical properties differ from those of the surrounding areas due to the general removal of topsoil as well as specific process related changes. Soil is an important resource, which supports a variety of ecological, economic and cultural functions. The factors like porosity, density, water holding capacity and aggregate strength that operates the soil quality are best developed in the top soil fraction. This quality can be disturbed during the construction activities. The movements of heavy machinery can lead to excessive compaction of topsoil and subsoil, and in deeper soil this may only be reversible over relatively longer time periods. There is a considerable impact on flora and fauna during the construction phase of landfills due to the removal of existing vegetation. But this damage could be recovered after the closing phase of the landfills. The studies have shown that landfills are capable of supporting a rich and varied fauna including exotic species during the operational and closing phase of landfills (Mellanby 1992).

Impacts due to landfill gas

The environmental impact of gaseous emission from landfills, which are of global or regional significance, can be mainly grouped as contribution to the greenhouse effect and damage to the eco system. Apart from that, risk of explosion and odor problem due to some trace gases can also be identified as significant impacts.

CO_2 and CH_4 present in the landfill gas act as greenhouse gases of global significance, with CH_4 being the most active but CO_2 being produced in the greatest quantities

(Krupa 1996). The LCA modeling performed by Damgaard, et al.(2011) shows that landfills are one of the main contributors for global warming when they are not facilitated with proper gas collection technologies. Landfills emit 1.3 tonnes CO_2 equivalent per tonne of landfilled waste without any gas collection facility while this value reduces down to 0.6 when the landfill gas is used to produce electricity (Cherubini et al. 2009). According to Clarke (1986), O'Neill (1993) and Wellburn (1994), CH_4 reacts with hydroxyl radicals and oxygen in the atmosphere to generate CO_2 within a period of days to a few years, thereby losing some of its greenhouse gas potential. Small amounts of methane are also consumed after absorption by soil (Leggett 1990). Nevertheless, control of these emissions at the source is necessary from an environmental protection viewpoint and to address the obligations under the Kyoto protocol.

Gaseous pollutants have significant effects on plants, animals and the entire eco system. The lateral migration of gas through soil beyond landfill boundaries causes the displacement of oxygen from soil. This results in a decline in soil faunal populations and burrowing animals and causes vegetation dieback. Mainly the vegetation around the landfill and the newly planted vegetation on a closed landfill can be damaged due to the suppression of air around the roots by migrated landfill gas (Crowley et al. 2003). The acidic gaseous constituents contribute to the phenomenon of acid rains and its secondary effects on the acidification of soils and ecosystems. Ammonia is a major acidic constituent, which can be found in the landfill gas. It is a secondary acidifying agent following its atmospheric oxidation to nitric acid. It has effects on plants, causing a loss of stomatal control, a reduction in photosynthesis, enzyme inhibition, changes in synthetic pathways, and depressed growth and yield. Hydrogen sulfide is also having an extensive impact on ecosystem. It is an extremely biotoxic gas, effective at a few parts per billion in mammals. Plants are far less sensitive to direct toxicity effects but have a threshold of 1μg/g (Finnecy and Pearce 1986, Wellburn 1994). The most severe impact on plants is inhibition and destruction of root growth and vegetation cover due to the anaerobic soil conditions created by high concentration of sulfides which laterally seeps from landfill sites. VOCs play a significant role in formation of ground level ozone. High concentrations of ground level ozone tend to inhibit the photosynthesis, reduce growth and depress the agricultural yields (Yunus and Iqbal 1996, Agrawal and Agrawal 1999).

Gendebien et al.(1992) say that the lateral migration of gas through soil has been the cause of a number of hazardous explosions as methane is inflammable and explosive when it mixes with sufficient amount of air. Moreover, an unpleasant odor can be caused by the series of trace elements present in the landfill gas especially organic fatty acids from the acid phase and H_2S and other sulfur containing compounds. These impacts are discussed further under the section of socio- economic impacts of landfills.

Impacts due to leachate

The leachate production decreases very slowly and some parameters might be of environmental relevance for many decades to centuries (Kjeldsen et al. 2002). The main constituents of landfill leachate are dissolved methane, fatty acids, sulfate, nitrate, nitrite, phosphates, calcium, sodium, chloride, magnesium, potassium and trace metals like chromium, manganese, iron, nickel, copper, zinc, cadmium, mercury and lead (Kjeldsen et al. 2002). Leachate can migrate through the soil to groundwater or even to surface water due to the absence of proper liner system or damages of the liners, and this results in a serious problem as aquifers require extensive time periods for rehabilitation. Moreover, soil can retain the constituents of the leachate like metals and nutrients and can cause adverse impacts on the eco system.

The metals retained by the soil are taken up by plants and thereby provide a key route for metals entering into the food chain. Deposition of trace metals in plants can affect crop growth and productivity and also poses a greater threat to animal health. Those metals such as lead, zinc and cadmium show differential mobility through the vegetation and invertebrate trophic levels and must be assessed on a case by case basis (Crowley et al. 2003). Uptake by plants is affected by soil pH and salinity. Cadmium and lead uptake is enhanced by the chloride complexation of the metals present in the leachate (Alloway 1995). Eutrophication is the most extensive threat when the leachate is mixed with surface water with higher concentrations of nitrate and phosphates (Lehane 1999). Eutrophic conditions invariably cause excessive production of planktonic algae and cyanobacteria in the open sectors of lakes. This excessive production of algae results adverse impacts on fish species in the lake by limiting the light penetration into the lake. Ammonia generated from leachate within landfills migrates through the soil horizons where it is progressively nitrified to nitrite and nitrate and causes eutrophication problem. A number of chemicals can disrupt the reproductive behavior in a range of species by acting as oestrogen mimics. Dempsey and Costello (1998) found landfill leachate as a potential source for these substances.

Above mentioned metals can be present in the leachate either in large or small concentrations depending on the waste categories disposed of in the landfills. Mercury is one of the best studied contaminants. It is one of the most toxic metals within the food chain, being readily absorbed by animals, fish and shellfish. Landfills are potential mercury emitters to the eco system due to the disposal of batteries and paint residues in the landfills (Crowley et al. 2003). Alloway (1995) revealed that the chromide to chromate conversion in the landfills is environmentally significant as chromate is more toxic to plants than chromide.

2.3.3. Health and socio-economic impacts of landfills

Apart from the environmental impacts, landfills are sources for several socio-economic impacts like public health issues due to the exposure to landfill gas and to the ground and surface water contaminated by landfill leachate. Although modern landfill sites are well designed to reduce emissions, the emissions from landfills still continue to give rise to concerns about the health effects of living and working near these sites, both new and old. The exposure to contaminants and emissions can be via direct contact, inhalation or ingestion of contaminated food and water. Drinking water contamination has been identified as the source of exposure to harmful substances in many studies (Griffith et al. 1989, Berry and Bove 1997, Adami et al. 2001). Those studies revealed that congenital malformations, birth weight deficiency, prematurity and child growth and cancers have a significant impact due to landfill emissions. In a multi- site study of residents of New York State, a 12% increased risk of congenital malformations in children born to families within one mile of hazardous waste sites were reported (Geschwind et al. 1992). Fielder et al. (2000) and Vrijheid et al. (2002) also found an increased risk of congenital malformations in populations live near landfill sites. A multi-site European study called EUROHAZCON discovered a 33 percent increase in non- chromosomal birth defects among the residents living within three kilometers of the 21 hazardous waste landfill sites studied (Dolk et al. 1998). This conclusion was confirmed by the study conducted by Elliott et al.(2001). A number of studies revealed that there is a higher risk of developing cancer among the people near landfill sites and the elevated risks were observed for cancers of the stomach, liver and intrahepatic bile ducts and trachea, bronchus, lung, cervix and prostate (Goldberg et al. 1995, Goldberg et al. 1999).

In addition to the health issues, landfills yield substantial impacts on land value, land degradation and land availability. Various researches conclude that landfills likely have an adverse negative impact upon housing values depending upon the actual distance from the landfill (Reichert et al. 1992, Ready 2005, Akinjare et al. 2011). Reichert et al. (1992) revealed that 40 percent of participants to their survey reported odor and unattractiveness as the most severe nuisance while 35 percent reported about the toxic water runoff and methane gas emission. Their study concluded that landfills have a negative impact of 5.5-7.3 percent of market value depending on the distance to landfills. Akinjare et al. (2011) found that all residential property values increased with the distance away from landfill sites at an average of 6 percent. Ready (2005) performed a meta-analysis that included all available hedonic price studies of the impact of landfills on nearby property values. It showed that landfills that accept high volumes of waste (500 tons per day or more) depress the value of an adjacent property by 12.9 percent while a low volume landfill depresses this value only by 2.5 percent. Furthermore, occupation and requirement of the enormous space for landfills contribute to land scarcity in highly populated areas.

2.4. Evolving landfill concepts

Despite the landfilling has become the final option of the waste hierarchy defined by the EU waste directive (2008/98/EC), it is still expected to be applied in several cases because of the growing amount of solid waste and a lack of suitable techniques to treat all kinds of waste. However, it is very clear that the landfill concept should evolve to minimise the environmental and socio-economic burdens explained above. On the other hand, new concepts should be brought forward in order to re-introduce the buried resources to the material cycle. One approach is represented by engineered bioreactor landfills in which a controlled degradation is allowed in order to guarantee the long term stability of the landfill (Warith 2003). Another approach is the concepts of landfill mining (LFM) and Enhanced Landfill Mining (ELFM) that reduces the emission and potential hazard of landfills and valorises the resources enclosed in landfills.

2.4.1. Landfill bioreactors

The waste decomposing period of a MSW landfill is estimated as over fifty years. There is a considerable interest in techniques for shortening this time because it has the potential of reducing overall costs and risks. One method is considering a landfill as a bioreactor in which the degradation processes are provocatively accelerated (Crowley et al. 2003). A bioreactor landfill is a sanitary landfill site that uses enhanced microbiological processes to transform and stabilize the readily and moderately decomposable organic waste constituents within 5 to 8 years of bioreactor process implementation (Warith 2003). According to Warith's study, the bioreactor landfill significantly increases the extent of organic waste decomposition, conversion rates and process effectiveness over those occurring within the traditional landfill sites. The environmental performance measurement parameters (landfill gas composition and generation rate, and leachate constituent concentrations) remain at steady levels. A bioreactor landfill site requires effective operation of liquid addition and management. Next to this, waste shredding, pH adjustment, nutrient addition and balance, waste pre-disposal and post-disposal conditioning, and temperature management may also serve to optimize the bioreactor process. There are three different general types of bioreactor landfill configurations (EPA):

- Aerobic: In an aerobic bioreactor landfill, leachate is removed from the bottom layer, piped to liquids storage tanks, and re-circulated into the landfill in a controlled manner. Air is injected into the waste mass, using vertical or horizontal wells, to promote aerobic activity and accelerate waste stabilization.
- Anaerobic: In an anaerobic bioreactor landfill, moisture is added to the waste mass in the form of re-circulated leachate and other sources to obtain optimal moisture levels. Biodegradation occurs in the absence of oxygen (anaerobically) and produces landfill gas. Landfill gas, primarily methane, can be captured and valorised to minimize the greenhouse gas emission.
- Hybrid (Aerobic-Anaerobic): The hybrid bioreactor landfill accelerates waste degradation by employing a sequential aerobic-anaerobic treatment to rapidly degrade organics in the upper sections of the landfill and collect gas from lower sections. Operation as a hybrid results in the earlier onset of methanogenesis compared to aerobic landfills.

Decomposition and biological stabilization of the waste in a bioreactor landfill can occur in a much shorter time frame than occurs in a traditional landfill providing a potential decrease in long-term environmental risks and burdens and landfill operating and post-closure costs. The potential advantages of bioreactors include: (i) decomposition and biological stabilization in years vs. decades in traditional landfills, (ii) lower waste toxicity and mobility due to both aerobic and anaerobic conditions, (iii) reduced leachate disposal costs, (iv) a 15 to 30 percent gain in landfill space due to a decrease in density of waste mass, (v) significant increased landfill gas generation allows energy production and (vi) reduced post-closure care.

2.4.2. Landfill mining (LFM) and enhanced landfill mining (ELFM)

As described in the previous sections, landfills contribute to several environmental and socio-economic impacts. Examples include global warming, acidification, depletion of the quality of ecosystem and pollution of surface and groundwater mainly due to the long term landfill emissions (EEA 2000, Crowley et al. 2003, Mor et al. 2006, Emery et al. 2007, Sormunen et al. 2008, Damgaard et al. 2011) and degradation of land value and land availability due to the occupation and requirement of vast land surface (Ready 2005, Akinjare et al. 2011).

Landfills, however, could also be regarded as potential reservoirs of resources. In many regions of the world, massive amounts of important materials such as metals (Lifset et al. 2002, Kapur and Graedel 2006, Muller et al. 2006), waste fuels (Krook et al. 2012) and significant amounts of fines that can be used as construction materials (Hogland et al. 2004, Kurian et al. 2007, Quaghebeur et al. 2013) are accumulated in landfills.

As explained in the chapter 1 of this thesis, the concept of landfill mining has been introduced and practised around the world for over 5 decades. The major objective of this traditional landfill mining was one or more of the followings: (i) conservation of landfill space (ii) installation of bottom liners (iii) installation of gas or/and leachate collection systems (iii) recovery of metals such as copper or aluminium. Krook et al. (2012) highlighted that the accumulation of massive amounts of strategically important materials in the landfills challenges to change the current view on landfill mining towards a new perspective based on a strategy for extracting valuable material and energy resources. Related to this new perspective, the ELFM concept has been introduced with a strong focus on material and energy recuperation and recycling, and eventually regaining the land (Hogland et al. 2010, Jones et al. 2013). ELFM differs from traditional landfill mining in that it targets the integrated optimisation of materials and energy recovery, while using various techniques to mitigate CO_2 emissions. This concept also emphasizes the temporary storage of currently non-recyclable material and energy resources with the objective of future valorisation. In that approach, landfills become future mines for materials which could not be recycled with existing technologies or show a clear potential to be recycled in a more effective way in near future (Hogland et al. 2010, Jones et al. 2010). The feasibility of ELFM is studied by synthesizing the research on the Closing the Circle project, the first ELFM project targeting more than 15 million ton of waste present in the REMO landfill in Houthalen-Helchteren in the East of Belgium (Jones et al. 2013). The initial studies of this project highlighted the worldwide potential of ELFM in terms of climate gains, materials and energy utilization, job creation and land reclamation.

2.5. Conclusions

Landfills mainly emit gas and contaminated water as well as wind-blown litter and dust. Landfills are a potential threat to the quality of the environment, although the full extent of this threat has not always been scientifically validated. The main potential impacts are due to landfill gas and leachate. Both are highly complex mixtures and vary from site to site and with waste composition and age of the landfill. Available literature highlights that the landfills create significant impacts on global warming, eco system, ground and surface water, human health, land value and land availability. These impacts become severe when the proper gas and leachate collection systems are not installed. Landfills could also be considered as reservoirs for resources as a large amount of metals, combustibles and construction residues are buried there. In order to minimise the environmental and socio-economic burdens of landfills and on the other hand to re-introduce the buried resources to the material cycle, innovative landfill concepts should be brought forward. Landfill bioreactors and ELFM can be seen as promising approaches in this regard. Although the landfill bioreactors have a number of advantages, they currently do not address the valorisation of the waste present in the old landfill sites. In contrast, ELFM concept targets both new and old landfill sites with the aspects of temporary storage and mining of old waste. A few studies have been performed to identify the quality and quantity of the wastes present in the landfills, possible waste valorisation techniques and the economic drivers of ELFM. However, detailed studies are necessary to assess the environmental sustainability and economic feasibility of ELFM in order to compare the benefits and burdens of this concept with the existing situations of targeted landfills.

Chapter 3 : Model to assess the environmental effect and economic feasibility of Enhanced Landfill Mining

This chapter is based on

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And

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Abstract

Enhanced Landfill Mining (ELFM) is an innovative concept which allows the recovery of land, reintroduction of materials back to the material cycles and recovery of energy from a considerably large stock of resources held in landfills. The knowledge about the critical factors for environmental and economic performance of ELFM is necessary in order to drive ELFM from the conceptual to the operational stage. This chapter presents the initial step of a study which includes comprehensive environmental and economic assessment of ELFM system. The development of a model based on life cycle assessment and life cycle costing to evaluate the performance of ELFM, is described in this chapter.

3.1. Introduction

This chapter introduces an integrated approach for environmental and economic evaluation of ELFM, combining the principles of life cycle assessment (LCA) and life cycle costing (LCC). The structure of this chapter is organised in the following way. Sections 3.2 and 3.3 describe respectively the available modelling tools for waste management systems and the necessity of a modelling tool for ELFM. In the 3.4 section, the methodology of setting up a model for ELFM is explained. Finally the limitations of the developed approach are described in the 3.5 section. In order to demonstrate its validity and usability, the developed approach was applied to the ELFM of the REMO landfill which is the case study landfill of the first, comprehensive ELFM project (the so-called Closing the Circle project of Group Machiels, Belgium) and the results are described in chapter 4 of this thesis.

3.2. Modeling tools for waste management systems

In order to assess the economic and environmental sustainability of the waste management systems, a number of models have been gradually developed since early 1990s by a range of environmental protection agencies, universities and consultancies (Gentil et al. 2010). The models based on life cycle costing, life cycle assessment and cost benefit analysis are widely used in sustainability assessment of waste management systems (Hoogmartens et al. 2014).

- Models based on life cycle costing (LCC)

LCC is a tool to determine the most cost-effective option among different competing products, when each is equally appropriate to be implemented on technical grounds. Apart from the initial investment cost, LCC takes into account all costs related to operating, future periodic maintenance and rehabilitation of a project over a defined period of time. Payback time, internal rate of return (IRR) and net present value (NPV) are the most important economic indicators associated with LCC to verify whether or not investing in a project is worthwhile financially (Brealey et al. 2010). In order to deal with financial, environmental and social concerns, four LCC types have been introduced: fLCC, eLCC, feLCC and sLCC (Finkbeiner et al., 2010). LCC assessments that only focus on private investments are categorized as financial LCC (fLCC) (Ness et al. 2007, Rorarius 2007) . An Environmental LCC (eLCC) builds upon data of fLCC and extends it to environmental costs such as CO_2 taxes (Carlsson Reich, 2005). The full environmental LCC (feLCC) is not a commonly accepted concept in the world of sustainability assessment tools. feLCC extends eLCC with monetized, non-internalized environmental costs that can be identified by an environmental assessment method such as eLCA (explained underneath). In Societal LCC (sLCC) all costs borne by anyone in society, whether today or in the future, and associated with the life cycle of a product are taken into account.

- Models based on life cycle assessment (LCA)

LCA is a technique to quantify the environmental and health impacts associated with producing a product or carrying out a process or activity from raw material extraction through materials processing, products manufacturing, distribution, use, repair and maintenance and disposal or recycling. The LCA methodology consists of 4 phases: goal and scope definition, life cycle inventory (LCI) analysis, impact assessment and interpretation (ISO14040 , ISO14044). During the LCI phase identifying and quantifying the materials and emissions crossing the system boundaries is necessary. The primary objective of the impact assessment phase is to transform the long list of LCI results, into a limited number of indicator scores. These indicator scores express the relative severity on an environmental impact category. For example, an emission of SO_2 could result in increased acidity. Increased acidity can cause changes in the soil that result in dying trees. By using several environmental mechanisms, the LCI result can then be translated into a number of impact categories such as acidification, climate change, etc. In a final step, these impacts can eventually be aggregated in one single ‘environmental impact’ score. In order to deal with different pillars of sustainability, two LCA types are developed that can be applied separately or in combination: eLCA and sLCA (Hauschild et al. 2008, Ramirez and Petti 2011, UNEP 2009). An environmental LCA (eLCA) is the conventional type of LCA that assesses environmental impacts such as material, energy and waste flows of a product from cradle to grave (Hunt et al., 1992). Social LCA (sLCA) is recently developed based on the already well-grounded eLCA that permit to make decisions about social impacts throughout the full life cycle of products (Hauschild et al. 2008, Jorgensen et al. 2008, UNEP2009).

- Models based on cost benefit analysis (CBA)

CBA is a methodology that aims to select projects and policies, which are efficient in terms of resource use. All the positive and negative effects of a proposed project or policy are valued in monetary terms, providing a list of benefits and costs. This means that impacts which do not have a monetary value, such as environmental impacts, must be estimated in monetary terms (Morrissey and Browne 2004). This method is very similar to what is called an environmental LCC (Reich 2005), which is LCC extended with the monetarised effects such as the impact of emissions and resource use typically described in LCA. Financial, environmental and social concerns have led to three CBA types: fCBA, eCBA and sCBA (Rorarius 2007). Financial CBA (fCBA) is a tool for private profitability assessment while eCBA is to assess the external costs caused by environmental impacts. Social CBA (sCBA) evaluates a project from the viewpoint of society as a whole.

Most models for integrated waste management combine the elements of LCA with LCC. Some examples of mostly used software packages for integrated waste management are ORWARE (Eriksson et al. 2002, Assefa et al. 2005), WISARD (De Feo and Malvano 2009) and EASEWASTE (Kirkeby et al. 2007, Manfredi and Christenen 2009). Apart from the above three models several other models are described in the review of Gentil et al. (2010).

3.3. The necessity of a modeling tool for ELFM

It is evident that there are some essential differences between landfill mining and integrated waste management. The impact of transport of waste is important in an integrated waste management system while excavation is imperative in landfill mining. Hence the operational costs of landfill mining are different from waste management systems. With regard to landfill mining, the inputs are only originating from the landfill itself and not from different waste sources at different sites. In most cases the input streams are extremely mixed and can also be degraded. This leads to a requirement of optimized separation techniques. While present waste management has to deal with the currently available waste streams, landfill mining is exploiting a large stock of historical waste. In order to obtain more homogeneous waste streams in landfill mining, it can select sectors of the landfill for mining. Other aspects of landfill mining which do not play a role in waste management systems are: regaining land, lowering the maintenance and/or reclamation costs and risks of landfills, site sanitation, recovering soil materials and better control of hazardous wastes when uncovered during the landfill mining of old sites (RenoSam 2009). Due to evolving technologies and changing consumption patterns, current waste management systems have to deal with different waste compositions than landfill mining. All these facts show the requirement of a flexible system that can optimise inputs and outputs in function of variations in time due to economic (market prices, regulation) and operational (quality of inputs, availability and capacity of processing units) reasons.

All existing models for integrated waste management describe input flows of waste in terms of waste fractions, with only minor differences in characterisation (mainly organics, metal, glass, plastic, paper, incineration ashes), which is very different from the mixed waste input of landfill mining. In addition, the level of details differs from model to model. Some of these models have been developed for different regional characteristics. They do not allow easy adjustments to other regions or other situations like landfill mining (Eriksson et al. 2002). This reveals that use of existing waste management models for ELFM is not the best option. Nevertheless, Emery et al. (2007) remark that no waste management tool currently integrates all sustainability aspects (ecological, economic and social). Hence progress can be made by integrating more sustainability dimensions in order to address the complex interactions between economic costs and returns and environmental considerations associated with ELFM.

3.4. Methodology of setting up the model for ELFM

Conventional LCA or eLCA has been chosen as one of the implementing modelling tool of this study to develop the environmental model. As LCA and fLCC can be applied in a parallel way, the model used fLCC or conventional LCC to assess the economic performance. Another reason to use eLCA and fLCC is that they provide information on different aspects without double counting (Hoogmartens et al. 2014). This approach allows (i) to calculate environmental and economic performances and their performance drivers separately and (ii) to analyse the trade-off between two aspects.

For the easiness of setting up the model, the work was divided into several sub steps: setting up the (i). general process flow diagram, (ii). input and output parameters (related to materials, energy, costs and revenues) of ELFM activities (iii). environmental model (LCA) and (iv). economic model (LCC). The following sections describe each step in detail.

3.4.1. Setting up the general process flow diagram

Traditional landfill mining comprises excavation, processing, treatment and/or recycling of deposited materials (Frändegård et al. 2013). Novel ELFM consists of the same activities but broader attention is given to the valorisation of all types of waste streams such as waste present in the landfill and even the waste generated during processing of the landfilled waste. Figure 3.1 is the general and simplified process flow diagram based on these premises. The flow sheet contains six major sections: (i) landfill, (ii) inputs, (iii) technology, (iv) products, (v) final destinations and (vi) revenues. The ‘landfill’ section represents the existing situation of the landfill with three types of waste: (i) municipal solid waste and assimilated industrial waste (MSW & AIW), (ii) industrial waste (Gunawardana et al.), and (iii) mixed waste (which is applicable when the landfilled waste cannot be distinguished clearly either as MSW & AIW or IW). The ‘technology’ section denotes the main activities identified in ELFM, which are: (i) vegetation and top soil removal, (ii) conditioning, (iii) waste excavation, (iv) separation, (v) transformation of intermediate products and (vi) land reclamation. The processes described in the previous landfill mining and landfill reclamation studies have been considered prior to defining the above processes in the general process flow diagram (RenoSam 2009, Canaleta and Ripoll 2012, Ritzkowski and Stegmann 2012, Frändegård et al. 2013, Raga and Cossu 2014). The ‘Inputs’ block includes all inputs needed for the technological processes and mainly consists of energy, water and additives, apart from the waste to be processed. In addition, all investment and operational costs of the ELFM activities are also denoted by the input block. All outputs are included in the ‘Products’ section. The products can be categorised as (i) intermediate products, (ii) end products (materials, energy, land), (iii) waste products (solid waste and wastewater) and (iv) emissions. The intermediate products can be transformed on-site into valuable material or energy or can be sold to an external company. They can also be temporarily re-stored if the available technologies for further processing are not sufficient at the moment. These destinations are indicated by the ‘Final destination’ block. The disposal category signifies both the temporary disposal of intermediate products and the permanent disposal of waste products. The products of ELFM processes substitute the virgin material production somewhere else or in other words the environmental impact of virgin material production is avoided by the use of recovered materials derived from ELFM. This is known as environmental revenue and it can be either benefit or burden. Furthermore, by assigning appropriate selling prices to the products, ELFM can also acquire economic revenues. The ‘Revenues’ component shows these two types of revenues of ELFM.

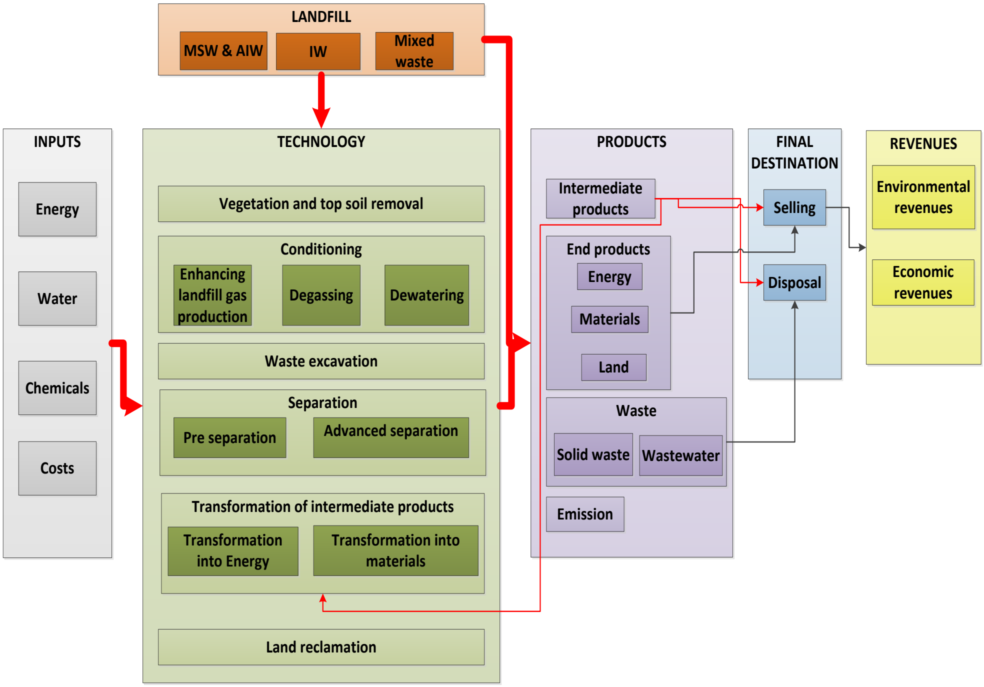


Figure 3.1: General process flow diagram of ELFM

The suggested technological processes for ELFM are further detailed and explained as follows.

Vegetation and top soil removal. This is the first process of ELFM prior to opening the landfill for waste excavation which could cause an significant impact on biodiversity (De Vocht and Descamps 2011).

Conditioning. A conditioning step is necessary in ELFM since it is required to decide how to deal with the landfill gas produced by the landfill prior to opening it for waste recovery. The proper handling of landfill gas during excavation avoids the environmental and health impacts that can be raised due to gas leakages. The decisions to be made here are directly related to the landfill gas production curve of the landfill. If a considerable gas production is still expected, it might be considered to enhance the production. In that case the area related to the gas production can be mined in a later stage for waste recovery. If the gas production potential is very low then the microbial activity can be immobilised in order to stop the gas production (degassing). Apart from the landfill gas, water accumulated in the landfills due to infiltration and the biological processes occurring within the landfill needs to be removed prior to excavation in order to minimise the difficulties that can arise during waste excavation and transportation towards the separation premises. According to the above considerations, the 'conditioning' process has been decomposed into three major parts: enhancing the landfill gas production, degassing and dewatering.

Waste excavation. Waste excavation is a basic process of ELFM. Both selective and unselective excavations are addressed by the term 'excavation'. The fraction subjected to selective excavation includes materials situated within the landfill at distinct layers of depth or locations. They are not mixed with the other materials. These materials can also be available in a large quantity. By selective excavation, material separation and treatment steps can be performed more efficiently. Unselective excavation takes place when the landfilled waste are mixed. The developed model facilitates both types of excavations. During the excavation, oversized waste is identified and removed by the crane operators in order to avoid them entering the separation process and causing unnecessary obstructions to the process. Selectively excavated waste are subjected to separate valorisation schemes with special treatment techniques depending on the unique characteristics of each type of waste. The waste obtained from unselective excavation is further separated prior to materials and energy valorisation.

Separation. After excavation, the waste should be directed to a proper separation process. As shown in Figure 3.1, the 'separation' process is decomposed into two sub processes: pre-separation and advanced separation. The content of the separation process is decided based on the flow diagrams provided by Jones et al. (2013), the characterisation study performed by Quaghebeur et al.(2013) and the oral communications with industrial experts of separation technology. Pre-separation comprises visual separation, manual/drum screening, shredding and on-site sieving. The purpose of the visual separation and manual/drum screening activity firstly is to recognize the hazardous waste and secondly to further identify the oversized wastes present in the excavated waste. Then the oversized parts obtained in visual separation and manual/drum screening enter a shredding step. Finally, a mechanical sieving activity is suggested in order to remove fines. 'Fines' denotes the material fraction below a certain particle size, which has to be removed prior to or during the material separation processes (Spooren et al. 2013). This sieving process is one of the most essential steps as this avoids a large quantity of fines entering the advanced separation process. The rest of the portion is then ready to enter the advanced separation process.

In the developed model, advanced separation is included in a very generic way. Depending on the moisture content of the pre-separated waste, composition and the quantity of the waste it has to be decided which type of advanced separation technology is going to be used. The model comprises common major sub components of separation (air separation, dense media separation, magnetic separation and eddy current separation) and their concomitant free parameters are included. The general process flow diagram of the separation phase is shown in Figure 3.2. Water, chemicals and electricity are identified as the major input parameters, in addition to the mixed waste input. The previous studies in landfill mining reveal that recycling of plastic, paper/cardboard wood and textile fractions is not possible since the level of contamination is too high to allow high quality material recycling (Quaghebeur et al. 2013). As those fractions have the potential to generate energy, in this model, they are assigned to one refuse derived fuel (RDF) fraction instead of addressing them individually. The separation process mainly generates fines, RDF, ferrous metals, non-ferrous metals, stones/aggregates and glass. Apart from that, some undefined materials, solid waste and wastewater can also be generated. Fines and RDF fractions generated here are considered as intermediate products. Ferrous metals, non-ferrous metals, stones/aggregates and glass fractions are known as end products and can be further characterised according to the type and the quality.

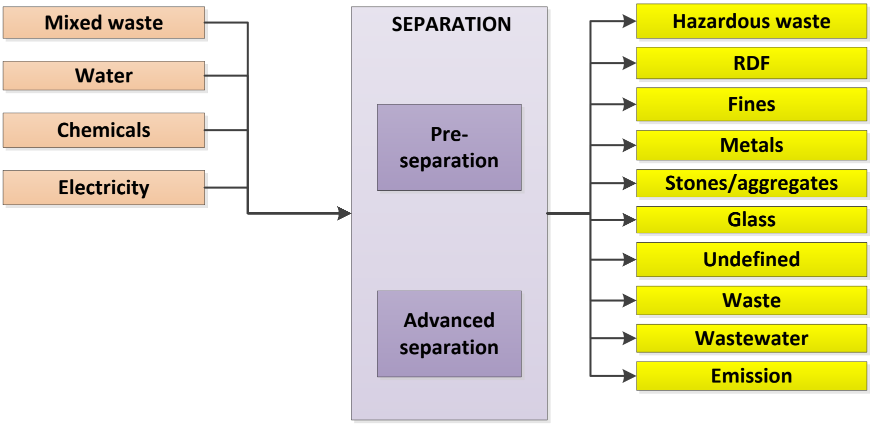


Figure 3.2: General process flow diagram for separation

Transformation of intermediate products. As described above, the separation process results in two major fractions: intermediate products and end products. It is necessary now to decide the destination of the intermediate products. These products can be sold to an outside company potentially at a low price. Alternatively, as explained in previous sections, they can be re-landfilled if technology or economics impede the valorisation at the moment. It is also possible that they can be transformed into valuable products in order to get higher revenues. The model foresees that this transformation can be performed in two ways. One is transformation into energy and the other is transformation into materials.

With regards to 'transformation into energy', the RDF produced mainly during separation processes and also in some other transformation processes, can be thermally treated with energy recovery. Although many existing thermal treatment technologies can be used in processing RDF, it is an objective of the novel ELFM concept to find integrated technologies aiming at 'zero waste' processes incorporating recycling, recovery and upgrade of (residue) materials, besides energy production (Spooren et al. 2013). Bosmans et al. (2013) recently analysed and compared several thermal treatment technologies including incineration, gasification, pyrolysis, plasma technologies and combinations for their suitability in ELFM. One of their conclusions is that plasma gasification/vitrification is a viable candidate for combined energy and material valorisation in the framework of ELFM. Hence, plasma gasification is included in this model as one of the thermal treatment processes. Plasma gasification offers a number of advantages such as high heat and reactant transfer rates, forming cleaner and high energy synthesis gas containing mainly hydrogen and carbon monoxide and allowing the use of low-energy fuels such as household and industrial waste (Chapman et al. 2011, Ray et al. 2012, Bosmans et al. 2013, Taylor et al. 2013). Taylor et al. (2013) highlight that the plasma gasification technology is able to efficiently produce a clean syngas and an environmentally stable vitrified product (plasmastone) from historically landfilled materials. The synthesis gas can be used for production of chemicals (methanol, hydrogen), electricity and/or heat or as second generation liquid fuels. In addition, several valorisation possibilities are put forward for plasmastone (Chapman et al. 2010, Iacobescu et al. 2013, Pontikes et al. 2013, Spooren et al. 2013, Taylor et al. 2013, Machiels et al. 2014). Plasma gasification encompasses four main stages: gasifier and plasma convertor, waste heat boiler, gas cleaning unit and syngas valorisation. As incineration is commonly used in treatment of a very wide range of wastes, it is also included in the model under thermal treatment technologies. Combustion chamber, waste heat boiler, steam turbine and gas cleaning unit are the main components of grate incineration. More detailed process descriptions of incineration can be found elsewhere (Limerick 2005, BREF 2006, BREF 2010). Other than above two special technologies, the model also allows to consider the thermal treatment as a black box. In that way, the user has the ability to change a set of common parameters depending on the technology applied.

The category, 'transformation into materials' covers three major parts: (i) valorisation of fines, (ii) valorisation of materials obtained from selective excavation and (iii) valorisation of residues of thermal treatment. The fines can be further separated as construction sand, aggregates, soil, RDF and metals (Quaghebeur et al. 2013). The RDF can be directed to thermal transformation and the other products can be further categorised according to the quality. The heterogeneous composition that can be expected in the fines fraction highlights the requirement of profound knowledge of the valorisation technologies. The valorisation routes for fines considered in this model consist of direct use without treatment, removal of organic matter, soil remediation, magnetic separation and metal leaching.

During selective excavation, many materials can be identified that should not enter the separation process. These materials can also be disposed, sold at lower prices or transformed into valuable products. Transformation techniques of those materials depend on their type and the quality.

Thermal treatment residue is foreseen to be transformed into building materials. As plasma gasification and incineration are the selected thermal treatment methods in this model, valorisation routes for the plasmastone and incineration ash (bottom ash) were studied. Transformation of plasmastone can lead to the production of inorganic polymers, blended cement and aggregates (Ray et al. 2012, Iacobescu et al. 2013, Spooren et al. 2013, Machiels et al. 2014). After recovering valuable metals in the bottom ash, the rest can be potentially used as aggregates.

Land reclamation. Land reclamation is the last part of the technology block and forms another essential part of ELFM. It has been identified that the reclaimed land after landfill mining can be used in four different ways: (i) as a nature reserve, (ii) land for industrial activities, (iii) land for housing and (iv) land for agricultural activities (Van Passel et al. 2013).

3.4.2. Setting up the model parameters

Having identified the possible ELFM activities and developing the general process flow diagram of ELFM, the next challenge is to recognize the relevant input and output parameters of each of these activities. Determining the inputs and outputs of the ELFM system is extremely important in order to build up a robust environmental and economic model. In our model, these parameter inventories are worked out in Excel and divided into different categories depending on the type of parameter, that is, (i). waste composition of landfill (ii). materials and energy in and out of processes (iii). costs and revenues of the processes and products.

Firstly, the parameters associated to the category of 'waste composition of landfill' describe the existing situation of the landfill. Parameters related to major waste fractions (MSW, IW, mixed waste as explained in section 3.4.1), landfill characteristics such as corresponding surface areas of different waste fractions, depth of landfill, depth of top cover, type of top and bottom covers and situation of the landfill emission control have to be included. Parameter values related to quantities of different waste fractions are given as 'as is' mass. This 'as is' mass of the waste fractions is the first important parameter of the entire model. It varies along the processes and is one of the key factors required for determining the dimensions of the process steps and materials and energy requirement to be provided externally. It is essential to perform necessary corrections for the degradation and moisture content for the waste quantities mentioned in the log books of landfills prior to feeding them into the model. If the log books or sufficient information on waste composition and quantity are not available, necessary characterisation studies should be performed to obtain more realistic data.

Secondly, the parameters related to the category of 'materials and energy in and out' mainly address the 'technology' block of Figure 3.1. Under each activity, all possible inputs and outputs associated to materials, energy and water have to be listed. In addition, possible emission parameters of each activity also have to be included. For the purpose of providing information that can easily be scaled to any application, all the values are requested to be quoted per 1 m^2 of surface area for vegetation and top soil removal, conditioning and land reclamation processes and per 1 tonne of waste for the other processes. These values can then be readily used to calculate the environmental and economic impacts of valorisation of a given amount of landfilled waste.

The 'Costs and revenues' group is committed to the economic parameters like inflation rates, discount rates, depreciation rates, residual values, investment and operational costs of the suggested processes and the expected selling prices of the ELFM products. Similar to the parameters of 'materials and energy in and out', cost parameters are also set up per 1m^2 of surface area and per 1 tonne of waste/product.

These three major parameter inventories provide the foundation for the environmental and economic models. The illustration of the structure and the data flow of the developed model are shown in Figure 3.3. The number of input parameters defined in each division is mentioned in the figure and the use of these parameters can be varied from case to case. The idea is not to use all defined parameters but to identify the most relevant parameters according to the landfill characteristics, waste fractions and the technologies to be used.

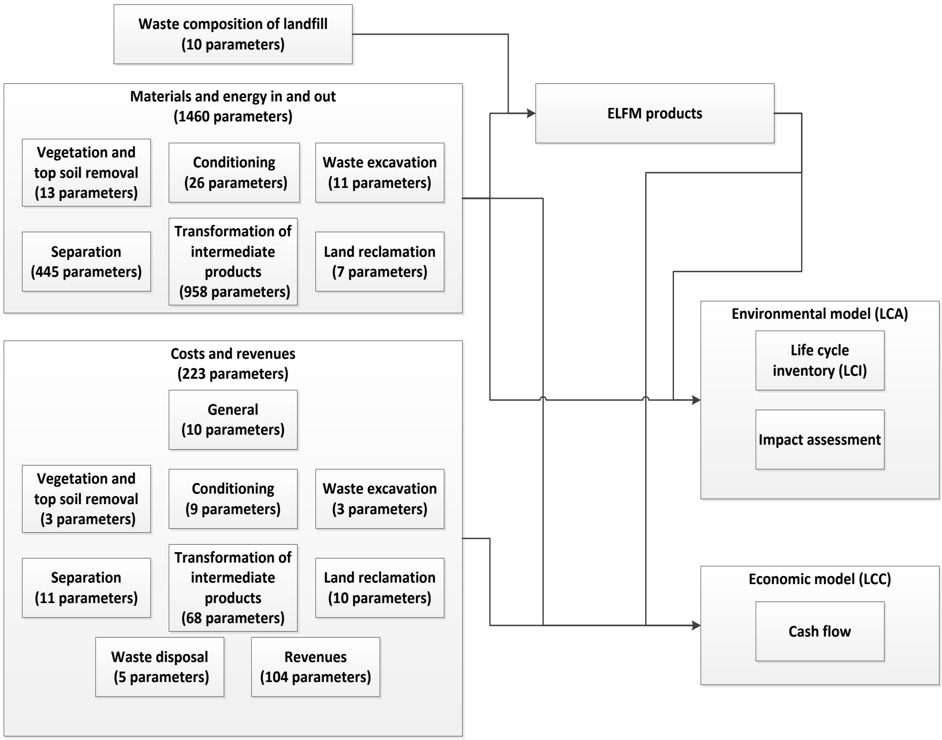


Figure 3.3: Illustration of the structure and data flow of ELFM model

3.4.3. Setting up the environmental model

As explained in the previous sections, the environmental model consists of a comprehensive LCA model. The developed inventories on landfilled waste composition, and materials and energy inputs and outputs create the structure of the environmental model as shown in Figure 3.3. International standards for LCA (ISO14040 2006, ISO14044 2006) were used as guidance and SimaPro 7 (PRéConsultants 2010) has been used as the LCA software tool in order to set up the LCA model. The inventories mentioned in the section 3.4.2 deliver the basis to build up the life cycle inventory (LCI) in the selected software tool. LCI comprises individual building blocks (unit processes) for each activity described in 3.4.1 with all possible inputs and outputs and also the relevant substitution of the virgin material/energy production (avoided impact) due to ELFM products. These inputs, outputs and avoided impacts can be changed from case to case as the LCI is not the same for all case studies. The user can choose appropriate building blocks in order to build up a unique LCA model for a certain landfill. This approach enables assessing individual process steps (chapter 5 and 6) as well as the different scenarios by combining several individual processes (chapter 4 and 7). The model set up is described in detail with the data of a case study in the next chapters of this thesis. Although, SimaPro was chosen as the LCA modeling tool of this work, one can feed inventories of parameters and calculations developed in Excel interface into another compatible LCA software tool to build up a particular LCA model. The environmental impact assessment can be performed by selecting a suitable impact assessment method present in the software. The different stages of life cycle impact assessment are briefly explained underneath as explained by Baumann and Tillman 2004.

- Classification

The inventory result of an LCA usually contains hundreds of different emissions and resource extraction parameters. Once the relevant impact categories have been determined, the LCI results must be assigned to these impact categories. For example CO_2 and CH_4 are both assigned to the impact category "Global warming", while SO_2 and NH_3 are both assigned to the impact category "Acidification". It is possible to assign emissions to more than one impact category at the same time; for example SO_2 may also be assigned to an impact category like "Human health", or "Respiratory diseases".

- Characterisation

Once the impact categories are defined and the LCI results are assigned to these impact categories, it is necessary to define characterisation factors. These factors should reflect the relative contribution of an LCI result to the impact category. For example, on a time scale of 100 years the contribution of 1 kg CH_4 to global warming is 25 times as high as the emission of 1 kg CO_2. This means that if the characterisation factor of CO_2 is 1, the characterisation factor of CH_4 is 25. Thus, the impact category indicator result for global warming can be calculated by multiplying the LCI results with their respective characterisation factor and summing them up. There are two types of impact categories: "endpoint" and "midpoint". Endpoints are to be understood as issues of environmental concern, like human health, extinction of species, availability of resources for future generations etc. The ISO standard allows the use of impact category indicators that are somewhere between the inventory result (i.e. emission) and the "endpoint". Indicators that are chosen between the inventory results and the "endpoints" are referred to as indicators at "midpoint level". In general, indicators that are chosen close to the inventory result have a lower uncertainty, as only a small part of the environmental mechanism needs to be modelled, while indicators near endpoint level can have significant uncertainties. However, indicators at endpoint level are easier to understand and interpret by decision makers than indicators at midpoint.

- Normalisation

Normalisation is a procedure needed to show to what extent an impact category has a significant contribution to the overall environmental problem. This is done by dividing the impact category indicators by a reference value. There are different ways to determine the reference value. The most common procedure is to determine the impact category indicators for a region during a year and, if desired, divide this result by the number of inhabitants in that area. After normalization the impact category indicators all have the same unit, which makes it easier to compare them. In addition, impact categories that contribute only a small amount compared to other impact categories could be left out of consideration, thus reducing the number of issues that need to be evaluated.

- Damage assessment

The purpose of damage assessment is to combine a number of impact category indicators into a damage category (also called area of protection). In the damage assessment step, impact category indicators with a common unit can be added. For example, in the ReCiPe method, all impact categories that refer to human health are expressed in DALY (disability adjusted life years). This allows interpreting reduced set of indicators.

- Weighting

This means the impact (or damage) category indicator results are multiplied by weighting factors, and are added to create a total or single score. Weighting can be applied on normalized or non-normalised scores. In SimaPro, there are often alternative weighting sets available, always in combination with a normalization set.

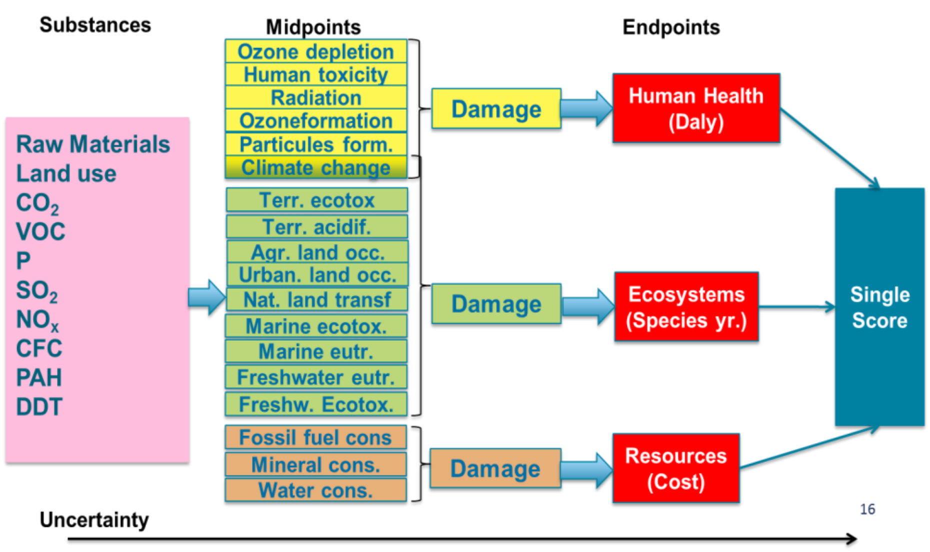


Figure 3.4: Illustration of the ReCiPe method based on Goedkoop et al. (2013)

In this study we mainly used two impact assessment methods: (i) ReCiPe (Goedkoop et al. 2013) and (ii) IPCC 2007 GWP 100a method (PRéConsultants 2008). The main reason to use ReCiPe was that ReCiPe is the successor of the methods Eco-indicator 99 and CML-IA that integrated the 'problem oriented approach' (midpoint level) of CML-IA and the 'damage oriented approach' (end point level) of Eco-indicator 99. Hence, ReCiPe addresses wide range of both midpoint (problem oriented) and endpoint (damage oriented) impact categories as shown in Figure 3.3. ReCiPe endpoint method was used when the aggregated results in one common unit (environmental points) are needed to compare them with overall economic performance of ELFM scenarios. European normalisation and weighting factors were used throughout the study. The characterisation factors, normal values and weighting factors can be found in Goedkoop et al. (2013) and PRéConsultants 2014. IPCC 2007 GWP 100a method was used where global warming potential (GWP) becomes the priority impact category. Normalization and weighting are not a part of this method.

3.4.4. Setting up the economic model

A LCC model was developed to assess the economic performance of ELFM (Figure 3.3) by combining the defined set of inventories. The LCC model consists of a detailed cash flow with all relevant investment costs, operational costs and revenues for a certain time period. The economic model delivers values for two economic indicators: net present value (NPV) and internal rate of return (IRR). The NPV is calculated by subtracting the investment cost from the sum of the discounted cash flow and can be considered as the expected profit of the investment (Brealey et al. 2010). It takes the time value of money and all the relevant cash flow elements over a pre-defined period into account. Equation 3.1 shows the mathematical representation of NPV.

$$\mathrm{NPV} = \sum_{t=1}^{T} \frac{\mathrm{CF}_t}{(1+x)^{t-1}} \tag{3.1}$$

Where, CF_t is the cash flow including investment cost in year t, T is the time horizon and x is the discount rate.

The IRR is the discount rate at which the NPV is zero (Brealey et al. 2010). It gives an idea about the relative return of the investment but does not take the scale of the project into account. The user can choose one or both of these economic indicators according to the objectives of the analysis. In this study we have two major objectives: (i) compare different scenarios and (ii) analyse the impact of performance drivers. While the IRR of two projects can be the same, the NPV of one project can be larger than the NPV of the other. Furthermore, positive NPV indicates addition to shareholder's wealth and negative NPV represents the opposite situation. This rule of thumb cannot be applied to IRR. In addition, the IRR method cannot be used to evaluate the projects where there are changing cash flows. Hence in this study we used NPV as the economic indicator. We used a general rule of thumb to select a discount factor in order to calculate NPV: i.e. 15% for private investment and 4% for a social cost benefit analysis. As this study includes private costs and benefits 15% discount rate was chosen. Recent economic analysis performed for ELFM has also been used the same discount rate (Van Passel et al. 2013).

A sensitivity analysis was used to evaluate the variation of NPV when the values of uncertain assumptions are modified. A Monte Carlo Simulation approach has been used with Oracle's Crystal Ball software. When performing a Monte Carlo sensitivity analysis, probability distributions are specified for uncertain values of input parameters. In this study, the probabilities that were used to express uncertainty were assumed to have triangular distributions. The normal distribution $N(\mu,\sigma^2)$ cannot be used because the standard deviation (σ) of the distribution is often unknown. When only literature data or expert judgments and no large datasets or historical data are available, only the lowest value, the highest value and the most likely value of the input variables can be assessed. The triangular distribution is an adequate solution when literature is insufficient for deriving probabilities (Haimes 2004, Kuppens 2012). Maximums and minimums of the triangular distributions considered in this study are mainly decided based on literature data and personal communication with the experts in the relevant industries. After the distributions are established, a large number of trials (for instance 100000), taking each time a random sample from the distribution for each and every uncertain parameter, are executed in order to produce a large number of NPVs and their empirical joint distribution. The results of the model not only incorporate the uncertainties of the input parameters, they also provide an idea about their importance by showing each parameter's contribution to the variance of NPV. This contribution to variance of the input parameters is determined by considering the uncertainties of the relevant parameters all together.

3.6.Conclusions

This chapter describes an approach based on LCA and LCC to evaluate the environmental and economic feasibility of ELFM. The model consists of a number of input parameters to cover all possible activities that can be found in the context of ELFM. These parameters include waste composition of a landfill, landfill characteristics, materials and energy inputs and outputs of different ELFM processes, investment and operational costs and finally the market prices of the ELFM products. The LCA model delivers the environmental impact of individual ELFM processes as well as of the total ELFM project. Furthermore, the model facilitates assessing different scenarios containing different exploitation technologies. This approach allows identifying the optimal scenarios/technologies that can be applied in ELFM. The economic model permits comparing the same scenarios used in the LCA model in order to measure their economic impact. In order to demonstrate the usability of the developed model, it was applied to the first ELFM case study (Closing the Circle project of Group Machiels, Belgium) and is described in the chapter 4 of this document. In addition, the studies presented in the chapters 5, 6 and 7 are also based on the different building blocks of this model.

Chapter 4 : Assessment of environmental and economic feasibility of Enhanced Landfill Mining

This chapter is based on

Danthurebandara, M., Van Passel, S., Van Acker, K. (2013). Life cycle analysis of Enhanced Landfill Mining: Case study for the Remo Landfill. In Jones, P. (Ed.), Geysen, D. (Ed.),. International Academic Symposium on Enhanced Landfill Mining (ELFM). Houthalen-Helchteren, Belgium, 14-16 October 2013 (pp. 275-296).

And

Danthurebandara, M., Van Passel, S., Vanderreydt, I., Van Acker, K.. (2015). Assessment of environmental and economic feasibility of Enhanced Landfill Mining. *Waste Management*. DOI: 10.1016/j.wasman.2015.01.041

Abstract

This chapter evaluates the environmental and economic performance of Enhanced Landfill Mining (ELFM). The model described in chapter 3 is applied to a case study, that is the first ELFM project in Belgium. The environmental and economic analysis is performed in order to study the valorisation of different waste types in the landfill, such as municipal solid waste, industrial waste and total waste. We found that ELFM is promising for the case study landfill as greater environmental benefits are foreseen in several impact categories compared to the landfill's current situation (the 'Do-nothing' scenario). Among the considered processes, the thermal treatment process dominates both the environmental and economic performances of ELFM. Improvements in the electrical efficiency of thermal treatment process, the calorific value of refuse derived fuel and recovery efficiencies of different waste fractions lead the performance of ELFM towards an environmentally sustainable and economically feasible direction. Although the environmental and economic profiles of ELFM will differ from case to case, the results of this analysis can be used as a benchmark for future ELFM projects.

4.1. Introduction

In order to fill the absence of a proper sustainability evaluation tool for ELFM, an integrated environmental and economic model based on life cycle analysis (LCA) and life cycle costing (LCC) was developed and discussed in the chapter 3 of this thesis. In this chapter 4, the developed model was applied to the REMO landfill which is the case study landfill of the first, comprehensive ELFM project (the so-called Closing the Circle project of Group Machiels, Belgium). The Closing the Circle (CtC) project (Tielemans and Laevers 2010, Jones et al. 2013) is the first case study for the ELFM Consortium to investigate the opportunities and barriers for ELFM. The project's focus is on the REMO landfill site, which is located in Houthalen-Helchteren in the province of Limburg in Belgium.

This paper presents a broad evaluation of the various ELFM activities conceived for the case study landfill and the major environmental and economic drivers of ELFM. The evaluation includes comprehensive LCA and LCC studies and illustrates the importance, contribution and sensitivity of certain ELFM activities included in a basic scenario to the total environmental and economic impact of the project. The background of the case study landfill and the related processes are described in the section 4.2 and 4.3 respectively. In section 4.4, the LCA/LCC approach is explained. Finally the 4.5 section presents the results obtained by the model.

4.2. Background of the REMO landfill

The REMO landfill site has been in operation since the early 1970s and covers an area of 130 hectares (Jones et al. 2013). More than 15 million tonnes of waste has been stored in the landfill. Approximately half of the waste is MSW and the other half consists of IW such as shredder material from the recycling of end-of-life-vehicles (ELV), metallurgical slags, pyrite containing slags, dried sludge, etc. (Quaghebeur et al. 2013). Quaghebeur et al.'s (2013) characterisation study for this landfill suggests that, for the plastics, paper/cardboard, wood and textile present in this site, thermal valorisation or feedstock recycling is the most suitable valorisation route since the level of contamination is too high to allow high-quality material recycling. Several other studies have also highlighted the usability of combustible MSW in advanced thermal treatment plants (Malkow 2004, Zolezzi et al. 2004, Consonni et al. 2005, Al-Salem et al. 2009, Xiao et al. 2009). Hence, plastics, paper/cardboard, wood and textiles are considered in the present study as refuse derived fuel (RDF). The amount of such combustibles varies between 23 percent and 50 percent (w/w). The same study proposes that for metals, glass, ceramics, stones and other inerts in the landfill, material valorisation is possible when the materials can adequately be separated. The fine-grained (<10mm) materials present in the REMO site vary between 40 percent and 60 percent (w/w). Furthermore, the characterisation study revealed that, especially for IW (mainly shredder from ELV), the fines fraction contains higher concentrations of heavy metals (Cu, Cr, Ni and Zn) and offers opportunities for metal extraction and recovery. A detailed waste composition can be found in Appendix A.

4.3. ELFM activities and scenarios

Possible ELFM activities related to the REMO landfill were identified according to the technological processes described in previous chapter 3. Figure 4.1 provides an overview of the processes that can take place in the case study landfill. As shown in the figure, several decisions need to be taken on which technologies are going to be used, whether or not the certain waste fractions will be treated or temporary disposed of, etc. These decisions were taken by the expert panels associated to CtC project according to the previous characterisation and validation studies performed for REMO landfill. After performing vegetation and top soil removal, the landfill is ready for excavation. Stainless steel slag, pyrite ashes and industrial sludge were identified as the materials that required selective excavation. It should be determined whether these fractions are sold at a lower price, treated further or temporarily disposed of. Stainless steel slag contains on average 5 percent of stainless steel particles and recovery of this steel is economically interesting. However, crushing or grinding of the slag is necessary in order to recover this steel. The resulting fine-grained material is difficult to reuse due to the elevated leaching of chromium and molybdenum, as well as the swelling behaviour of the slag material (Spooren et al. 2013). Therefore, it is necessary to treat or stabilise the slag materials to lower the leachability of heavy metals prior to use as a construction material. As these stabilisation methods have not yet been well confirmed, the stainless steel slag fraction was considered to be disposed of temporarily, as shown in Figure 4.1. Similar

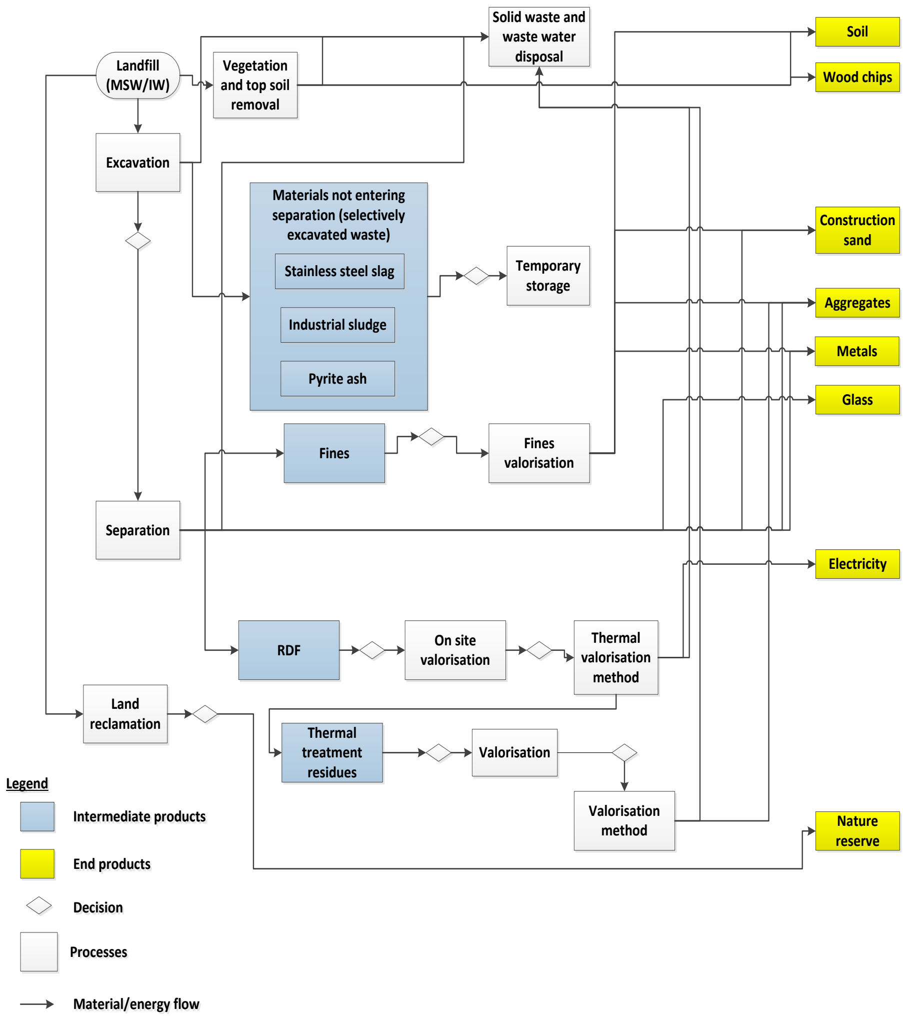


Figure 4.1: Overview of the ELFM processes of REMO landfill

to stainless steel slag, pyrite ashes and industrial sludge present in the REMO landfill site contain heavy metals such as Cd, Cu and Zn, in amounts that are considerably above the limit values of the Flemish legislation for use of the materials in or as a construction material (Spooren et al. 2013). Temporary disposal was also applied to pyrite ash and industrial sludge fractions until proper valorisation routes have been identified. More details on implementation of temporary storage at the REMO landfill site can be found in Geysen (2013).

The fraction of waste obtained from unselective excavation (mixed waste fraction) is sent directly to the separation process. This process starts with pre-separation, which identifies the hazardous materials to be disposed of and fine grained materials that should not enter the advanced components of the separation process. After removing hazardous materials and fine-grained materials, the residual fraction is subjected to an advanced separation technology for further separation. The separation technology depends on the characteristics of the excavated waste; that is, the moisture content, particle size distribution, etc. The detailed separation process applied to the case study landfill is not described further here due to confidentiality reasons. However, the considered separation process comprises air separation, dense media separation, magnetic separation and eddy current separation as explained in chapter 3. The destination of the fines fraction resulting from the separation process was supposed to be further valorisation instead of selling for a lower price or temporary disposal. A decision was made to treat the RDF fraction thermally on-site. According to the concept of ELFM, the destination of the residues created during the thermal treatment should also be defined. For the case study landfill, these residues are intended to be further treated on site instead of selling or landfilling. All types of wastewater and solid waste generated during the treatment processes were directed to suitable treatment systems. As a final component of ELFM system, land reclamation was addressed by reclaiming the land as a nature reserve for the case study landfill site.

The variety of possible choices for several processes meant that a number of scenarios for the ELFM system of REMO landfill are possible. Each of these scenarios contains the processes of vegetation and top soil removal, waste excavation, separation, thermal treatment of RDF, valorisation of thermal treatment residues, valorisation of fines and land reclamation. The scenarios distinguish between (i) the waste type (MSW versus IW), (ii) the applied separation technology (depending on the characteristics of the excavated waste), (iii) the thermal treatment technology for RDF and (iv) the valorisation route of the thermal treatment residues. For the basic ELFM scenario discussed in this study, plasma gasification was chosen as the thermal treatment method. This decision was made based on Bosmans et al.'s (2013) recent study, which concluded that plasma gasification is a viable candidate for combined energy and material valorisation in the framework of ELFM. This method has a high efficiency and the flexibility to valorise the resulting syngas in many ways, such as the production of electricity and/or heat, as a feedstock for chemical industry (hydrogen, methanol) or as a second-generation liquid fuel (Chapman et al. 2010, Chapman et al. 2011, Ray et al. 2012, Taylor et al. 2013). The present study only considered electricity production as the syngas valorisation route. The produced heat is used internally (for example, for boilers, drying of RDF, wastewater treatment, etc.) Besides energy production, plasma gasification also delivers an environmentally stable vitrified residue called plasmastone, which can be converted into building materials. Although many methods have been identified recently for the valorisation of plasmastone (Iacobescu et al. 2013, Machiels et al. 2014), only the most obvious valorisation route, aggregate production (Chapman et al. 2011, Ray et al. 2012, Taylor et al. 2013), was considered in the basic ELFM scenario. Plasma gasification was coupled with plasmastone valorisation and is denoted in the rest of this chapter as 'thermal treatment'.

Apart from the basic ELFM scenario mentioned above, a 'Do-nothing' scenario is used as reference scenario. The Do-nothing scenario supposes that no landfill mining activities are undertaken. Landfill gases and leachate are managed as mandated by the corresponding regulatory framework, applying common practices and ensuring adequate periodic maintenance and/or replacement of existing infrastructure. As explained in the previous chapter 2, landfill gas collection efficiencies range from 9-90 percent (IPCC 2006). However, the landfill gas collection efficiency of REMO site is assumed as 50 percent and the collected gas is used for combined heat and power (CHP) generation. This value is based on the personal communication with the expert panel of the CtC case and the sensitivity of this value is discussed in the discussion section of this chapter. The gas production curve shows that this landfill enters its long-term landfill phases (maturation phase) in which the methane production continuously decreases while the CO_2 concentration increases, as explained by Kjeldsen etal. (2002). Due to the drop of methane production, the CHP generation is only foreseen for the next five years (until 2019). The leachate collection and treatment systems are in place and comply with the Flemish and European legislation.

4.4. LCA and LCC methodology

The goal of this LCA study is to evaluate the environmental impacts of the valorisation of landfilled waste in the context of ELFM. The methodology is in accordance with the International Standards for LCA (ISO14040 , ISO14044) as explained in the chapter 3 of this thesis. The developed model structure as described in the chapter 3 was used to build up the LCA model for the case study landfill. The appropriate building blocks developed in SimaPro 7 have been used with all possible inputs and outputs (refer appendix A) and also the relevant substitution of the virgin material and energy production (avoided impact).

Figure 4.2 presents the general structure of the ELFM and Do-nothing scenarios and the system boundary of the assessment. The quality of the ELFM products that substitutes the virgin energy and material production are as follows. The metals recovered from separation, fines valorisation and thermal treatment processes are able to substitute the corresponding scrap metals. Sand, aggregates and soil recovered from fines valorisation processes and the aggregates obtained from valorisation of plasmastone have the quality of gravel that can be used in construction activities (Jones et al. 2013). The produced electricity from plasma gasification replaces the base load of electricity production in Belgium, being the Belgian electricity mix, which includes 53 percent nuclear energy, 40 percent conventional thermal energy (of which 25 percent is from natural gas, 11 percent from coal and 2 percent from oil, according to the ecoinvent database version 2.2), 2 percent hydro energy and 3 percent wind energy (Eurostat). A 47 percent share of biogenic carbon dioxide emission from plasma gasification process was used in this study. This value is similar to the fixed share of renewable energy fraction in waste incineration used in Flanders, Belgium (Van Passel et al. 2013, OVAM 2011). Recovered land is converted into a nature reserve, as this is a specific feature of the Closing the Circle project in Belgium (De Vocht and Descamps 2011).

The input data was based on measured data, data obtained from published sources, calculated and estimated data (refer Appendix A). For the background processes, such as the production of electricity and raw materials, the life cycle inventory data with European averages were used, which have been published mainly in the ecoinvent database (version 2.2) present in the software. In this study we used a reference flow instead of a functional unit as explained in the ILCD handbook (2010). Using a reference flow instead of a functional unit is very common in LCAs of waste treatment (Consonni et al. 2005, Frändegård et al. 2013). Hence, the reference flow was defined as the valorisation of a certain mass of landfilled waste. Based on this reference flow, the environmental impact was calculated for valorisation of (i) 1 tonne of MSW, (ii) 1 tonne of IW and (iii) total waste present in the landfill. In the third case, the environmental impact of ELFM was compared with that of the Do-nothing scenario. The environmental performance of the Do-nothing scenario was calculated as follows.

In order to determine whether the ELFM is environmentally beneficial compared to the existing situation (the Do-nothing scenario), the residual impact of the landfill should be determined. The extrapolated landfill gas production curve revealed that production would last for the next 50 years in very low concentrations. Hence all inputs and outputs were calculated for 50 years in order to determine the respective environmental impact. The CO_2 emissions in this scenario were considered to be CO_2-neutral because of their biogenic origin. The data of the effluent of the leachate treatment plant of the REMO site was used to determine the emission to water for the considered 50 year period. This analysis does not consider the long-term releases that can occur after the considered period.

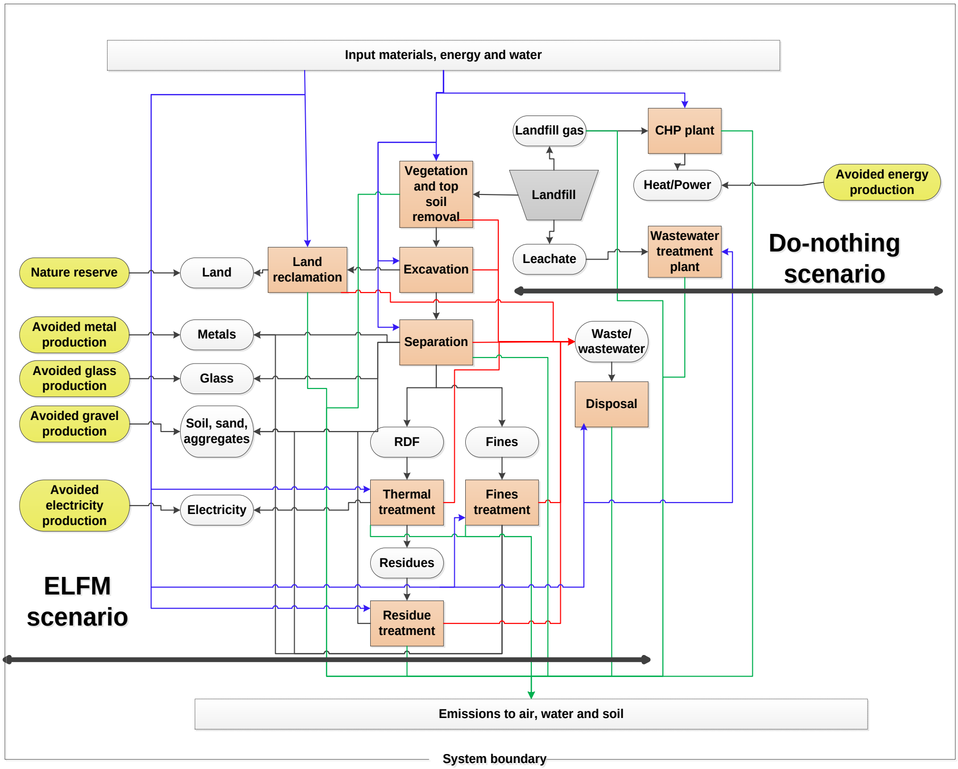


Figure 4.2: System boundary of ELFM and Do-nothing scenarios

For the environmental impact assessment of this study, as explained in the previous chapter 3, the ReCiPe endpoint method (Hierarchist version, H/A) was selected because it includes a variety of impact categories. These are: (i) climate change on human health, (ii) climate change on ecosystems, (iii) ozone depletion, (iv) terrestrial acidification, (v) freshwater eutrophication, (vi) human toxicity, (vii) photochemical oxidant formation, (viii) particulate matter formation, (ix) terrestrial ecotoxicity, (x) freshwater ecotoxicity, (xi) ionising radiation, (xii) agricultural land occupation, (xiii) urban land occupation, (xiv) natural land transformation, (xv) metal depletion, and (xvi) fossil fuel depletion (Goedkoop et al. 2013). The characterisation and normalisation stages of LCA were both considered when presenting the results. The results of the characterisation stage are presented in order to show the relative contribution of each waste type to each impact category. Next, normalisation is used to make the impacts on different impact categories comparable with each other and to show the extent to which an impact category makes a significant contribution to the overall environmental problem (PRéConsultants 2010). For this study, normalisation was performed on a European level. Finally, sensitivity analyses are performed for the most relevant parameters in order to determine the influence of a change in the inventory data on the results of the impact assessment.
The goal of the LCC study was to identify the economic drivers of ELFM. The developed LCC model was used to assess the economic performance of ELFM, by combining the defined set of inventories of relevant process as explained in the

chapter 3. The LCC model consists of a detailed cash flow with all relevant investment costs, operational costs and revenues for 20 years of period (refer appendix A). When building up the cash flow for the ELFM of the REMO landfill, it was assumed that waste processing capacity during the first two years of the project life time would be 30 percent, after which the project would run at full capacity (100 percent). The net present value (NPV) was used as the major economic indicator in order to determine the major economic drivers of ELFM. For this assessment, a 15 percent discount factor was applied (Van Passel et al. 2013) as the study uses only the private costs and benefits. To examine how the NPV varies when the values of uncertain assumptions are modified, a Monte Carlo simulation approach was used, as explained by Van Passel et al. (2013). This approach was explained in detail in the previous chapter 3.

4.5. Results and discussion

4.5.1. Environmental performance of waste valorisation

This section discusses the individual environmental profiles of the basic scenarios for the two types of wastes (MSW and IW). This discussion provides an insight into the processes that contribute most to the environmental impact of ELFM. The environmental burdens are expressed as positive values and the benefits are indicated as negative values.

Environmental performance of basic ELFM scenarios

Figure 4.3 represents the environmental profiles of valorisation of the extremely mixed fraction (waste subjected to unselective excavation) of MSW and IW, respectively. The figure illustrates the comparison of the valorisation of two types of wastes for the basic scenarios. The net impact (= burdens minus benefits) of the valorisation of 1 tonne of MSW and IW is shown for each impact category. It can be concluded from Figure 4.3 that none of the waste types has the highest or the lowest environmental score for all impact categories considered in the context of ELFM. For example, valorisation of MSW is the most favourable in the fossil depletion, ionising radiation and urban land occupation impact categories. However, it also makes the highest contribution to climate change impact. On the other hand, valorisation of IW delivers the highest benefit in metal depletion. Its influence on the ozone depletion impact category is considerably higher than that of MSW valorisation. However, the valorisation of both types of waste yields burdens in only three impact categories: climate change on human health, climate change on eco systems, and ozone depletion.

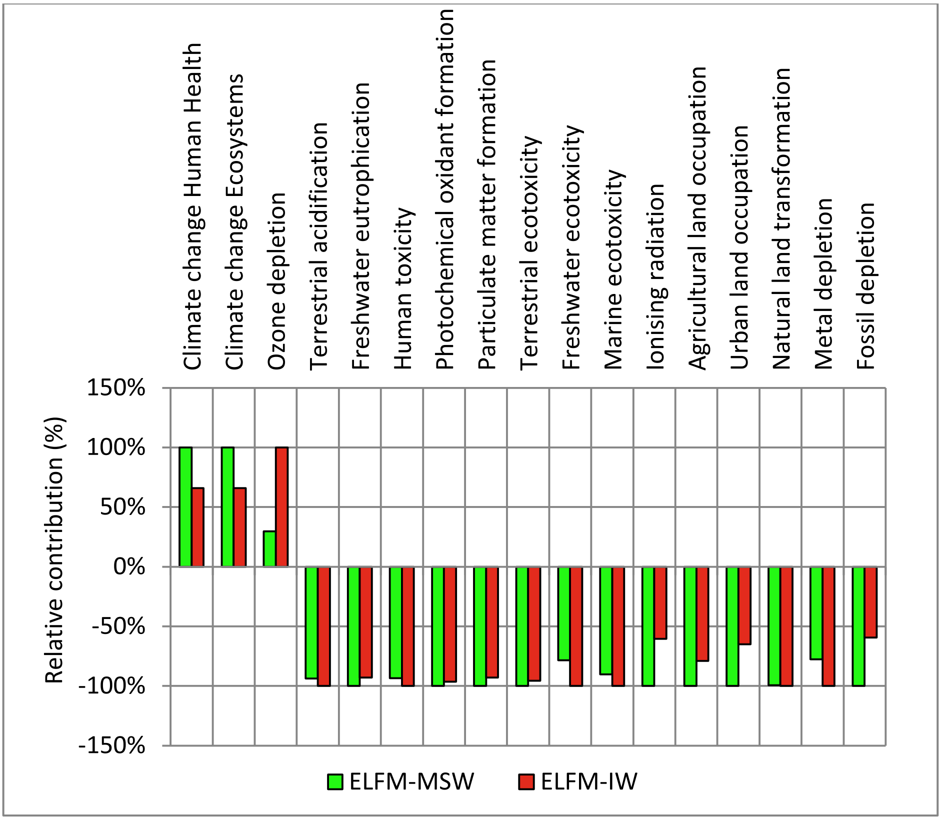


Figure 4.3: Comparison between two types of wastes- net environmental impact of valorisation of 1 tonne of MSW/IW (basic scenario)

Figure 4.4 compares the different impact categories with each other after normalisation. The contribution to climate change, fossil depletion, metal depletion and natural land transformation is important for both waste types. In addition, the contribution to human toxicity and particular matter formation cannot be neglected. It is important to note the insignificance of the other impact categories. Based on Figures 4.3 and 4.4 alone, however, we are not yet able to provide a straightforward explanation for the differences of impacts in each impact category. Therefore, the normalised environmental profiles that illustrate the contribution of the different stages or processes of ELFM to the total environmental impact are evaluated in more detail (Figure 4.5). Only the significant impact categories (threshold ±0.0005) are indicated in the figure.

From figure 4.5, it can be deduced that, in the valorisation of both waste types, the thermal treatment process dominates most impact categories. In both waste types, the thermal treatment process induces an environmental burden only on the climate change impact category. As illustrated in UCL (2014), flue gas emissions and oxygen usage within the process generate this burden. Next to the burdens on climate change impact category, fossil depletion and metal depletion impact categories are significantly credited by the thermal treatment process due to the electricity production and metal recovery from the plasma convertor. These burdens and benefits caused by thermal treatment process are higher in the MSW valorisation than in the IW waste valorisation. These differences are directly linked with the RDF content in the landfill and the recovery efficiency of the separation technology. In this study, we applied the same RDF recovery efficiency (80%) for both MSW and IW. Thus, the differences in burdens and benefits of this case are mainly caused by the RDF content. The characterisation studies conducted for the case study landfill found that the RDF content per tonne of waste in the case study landfill is higher in MSW than in IW (Appendix A).

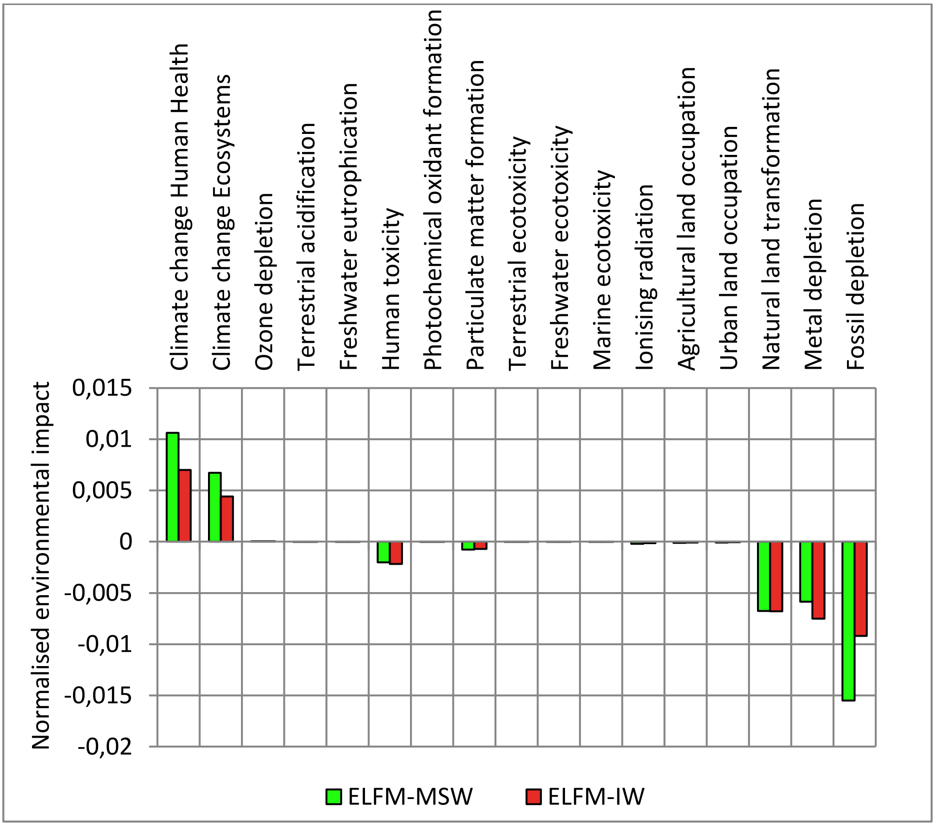


Figure 4.4: Normalised environmental profile of valorisation of 1 tonne of MSW/IW (basic scenario)

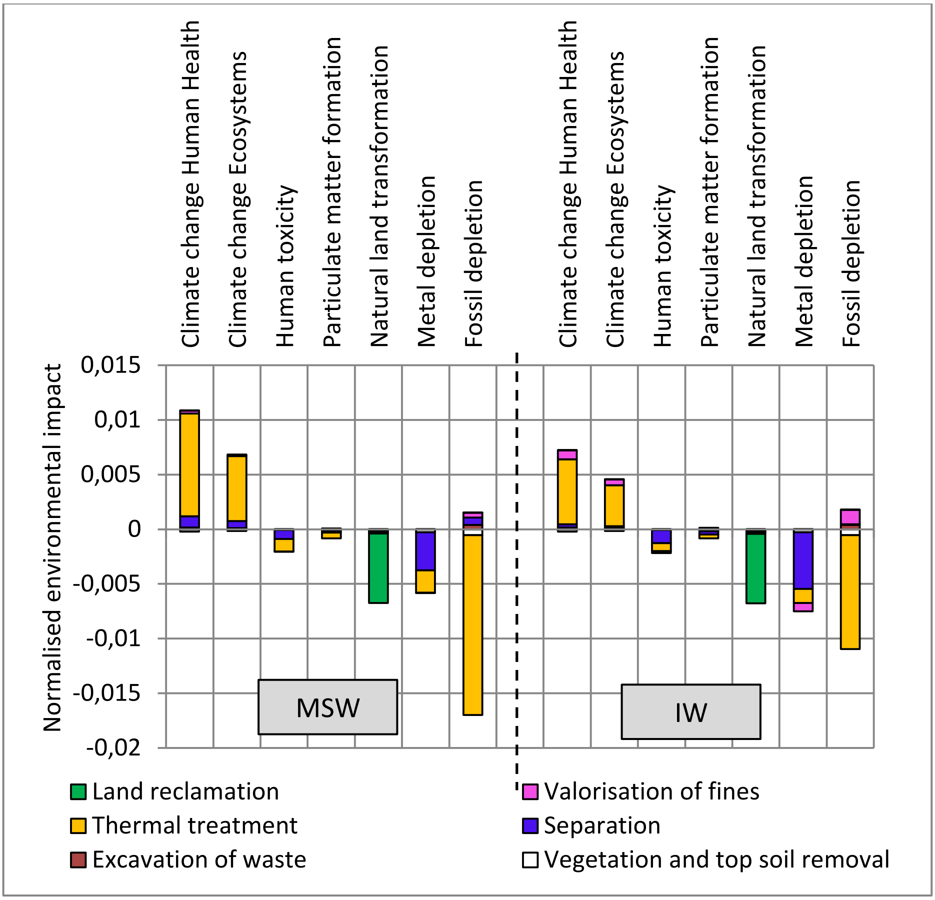


Figure 4.5: Contribution of different ELFM processes- Normalised environmental profile of valorisation of 1 tonne of MSW/IW (basic scenario)

As suggested by Jones et al. (2013), the burden due to flue gas could be reduced considerably by using the flue gas (rich in CO_2) and low temperature waste heat in local horticulture. The CO_2 acts as a fertiliser for the plants, while the residual heat warms the greenhouses, avoiding the use of primary fossil fuels. Use of an alternative energy source for oxygen production can also further reduce the impact of plasma gasification. Further research is necessary to investigate the possibility of using air in plasma gasification instead of pure oxygen. The net electrical efficiency of the system and the calorific value of the RDF fraction play important roles in determining the credits due to avoided electricity production. Obtaining higher environmental benefits in the fossil depletion impact category seems to be possible as the net electrical efficiency of plasma gasification can be improved up to 30 percent or more (Taylor et al. 2013). In addition, the existence of higher calorific values such as 20-26 MJ/kg is also possible for RDF (Arina and Orupe 2012, Spooren et al. 2013). The environmental performance of plasma gasification can be further improved by using different plasmastone valorisation options. In this study, plasmastone is converted into aggregates. Nonetheless, the avoided environmental burden created by aggregate production is insignificant compared to the environmental impact of plasma gasification itself. Machiels et al. (2014) and Iacobescu et al. (2013) elucidated the possibility of developing binding materials from plasmastone that can be used as low-carbon alternatives for ordinary Portland cement (OPC) in construction applications. Application of those methods in ELFM is discussed in detail in the next chapter 5. Note, however, that these impacts of thermal treatment process have been derived from a comparison with the Belgian electricity generation mix, in which nuclear energy has a share of 53 percent (Eurostat). According to Ecoinvent, the environmental impact of the production of nuclear energy is lower than the other energy production methods in most impact categories. Hence, the replaced impact is also lower when the Belgian electricity mix is used as the substituted product of the thermal treatment process. The use of different energy mixes as the substituted product leads to significant changes in environmental impact of thermal treatment process. The environmental burden of thermal treatment processes increased by 65 percent for the electricity mix of France, which includes 79 percent of nuclear energy. The environmental burden decreased by 87 percent for the electricity mix of the Netherlands, which consisted of only 4 percent of nuclear energy and 92 percent of conventional thermal energy. Moreover, 78% reduction in the environmental burden of the thermal treatment process can be observed when the Belgian marginal electric energy source natural gas (European Commission 2007) is considered as the substituted product.

Next to the thermal treatment process, Figure 4.5 shows that the separation process yields environmental benefits on metal depletion and human toxicity impact categories, mainly due to the metal recovery. The benefits from the recovery of stones and glass are very insignificant compared to the benefits due to metal recovery. The metal composition in the landfilled waste is imperative for the environmental profile of the process. Appendix A indicates that the ferrous metal content is higher in MSW than that in IW. In contrast, IW contains more non-ferrous metals than MSW does. According to the individual environmental profile of the separation process, non-ferrous metals give rise to a higher avoided environmental burden than ferrous metals. The above facts largely explain the higher benefit in metal depletion and human toxicity impact categories of the separation process in IW valorisation compared to MSW valorisation (Figure 4.5).

Valorisation of fines affects the environmental profiles differently in MSW and IW. According to Figure 4.5, the contribution of valorisation of fines is less important for MSW, while it becomes significant for IW. This difference is due to the different products that can be derived from the fines. Fines, in the case of IW, contain more metals than those of the MSW case. As Appendix A shows, 24 percent ferrous metals, 2 percent non-ferrous metals, 9 percent RDF, and 65 percent of other fraction that can be used as construction sand, soil or aggregates are present in IW fines. However, the metal percentage is very low in MSW fines and more than 80 percent of the total MSW fines have the quality of sand, soil and aggregates. Individual environmental profiles of fines valorisation show that the benefits due to recovery of sand, soil and aggregates are smaller, despite their high mass proportion, than the benefits due to metal recovery. This is the reason why fines valorisation becomes insignificant in MSW valorisation.

Vegetation and top soil removal can result in a partial (and temporary) loss of ecosystems. As Figure 4.5 shows, however, the impact of that process is negligible compared to other activities. As explained by De Vocht and Descamps (2011), gradual restoration is possible after the landfill mining activities; this point is corroborated by the activity of land reclamation. As shown in Figure 4.5, the contribution of the land reclamation to the total impact is only beneficial in the natural land transformation impact category and this benefit is very significant. The land area to be reclaimed is considered to be the same for MSW and IW in this study, providing the density of both waste types and the landfill depth are equal. Therefore, the impact of land reclamation is also the same in both cases.

Environmental performance of ELFM vs Do-nothing scenario

We have discussed the environmental impact of valorisation of 1 tonne of MSW and IW separately. When transforming ELFM from conceptual to implementation phase, it is necessary to know whether the ELFM is beneficial compared to the Do-nothing scenario. Figure 4.6 shows the environmental impact of valorisation of total waste (MSW+IW) present in the landfill with their actual amounts. In addition, those impacts were compared with the impact of the Do-nothing scenario for the total amount of waste. Figure 4.6 only shows the significant impact categories.

The net environmental impact of the valorisation of the total waste is very significant in all impact categories compared to the Do-nothing scenario. The impact of the Do-nothing scenario is negligible compared to the impact of ELFM scenarios, assuming that the landfill stays well controlled and maintained in the future. In the Do-nothing scenario, the burdens are mainly found in the impacts on climate change on human health and climate change on ecosystems. However, these burdens are much smaller. According to the gas production curve of the case study landfill, the REMO landfill is in the stage at which the methane production continuously decreases. Methane leakage to the environment is comparatively lower in this phase. Hence, in this situation, the Do-nothing scenario does not produce higher burdens towards the environment. Although this scenario creates a benefit on fossil depletion due to methane recuperation, it is less pronounced than the benefit created by the ELFM scenario (as shown in Figure 4.6). This is because only a small amount of energy from methane is produced due to the lower methane production, and no material is recuperated at all. Nevertheless, for a 10 percent change in the landfill gas collection efficiency leads to a 17 percent change in the impact on climate change.

The carbon footprint analysis performed by Van Passel et al. (2013) for the same landfill stated that the estimated CO_2 equivalent emission for the ELFM scenario is 5.3 million tonnes, compared to 6.3 million tonnes for the Do-nothing scenario; this suggests that ELFM is more beneficial. The present study is not fully comparable with the study of Van Passel et al., as the two studies used different methodologies.

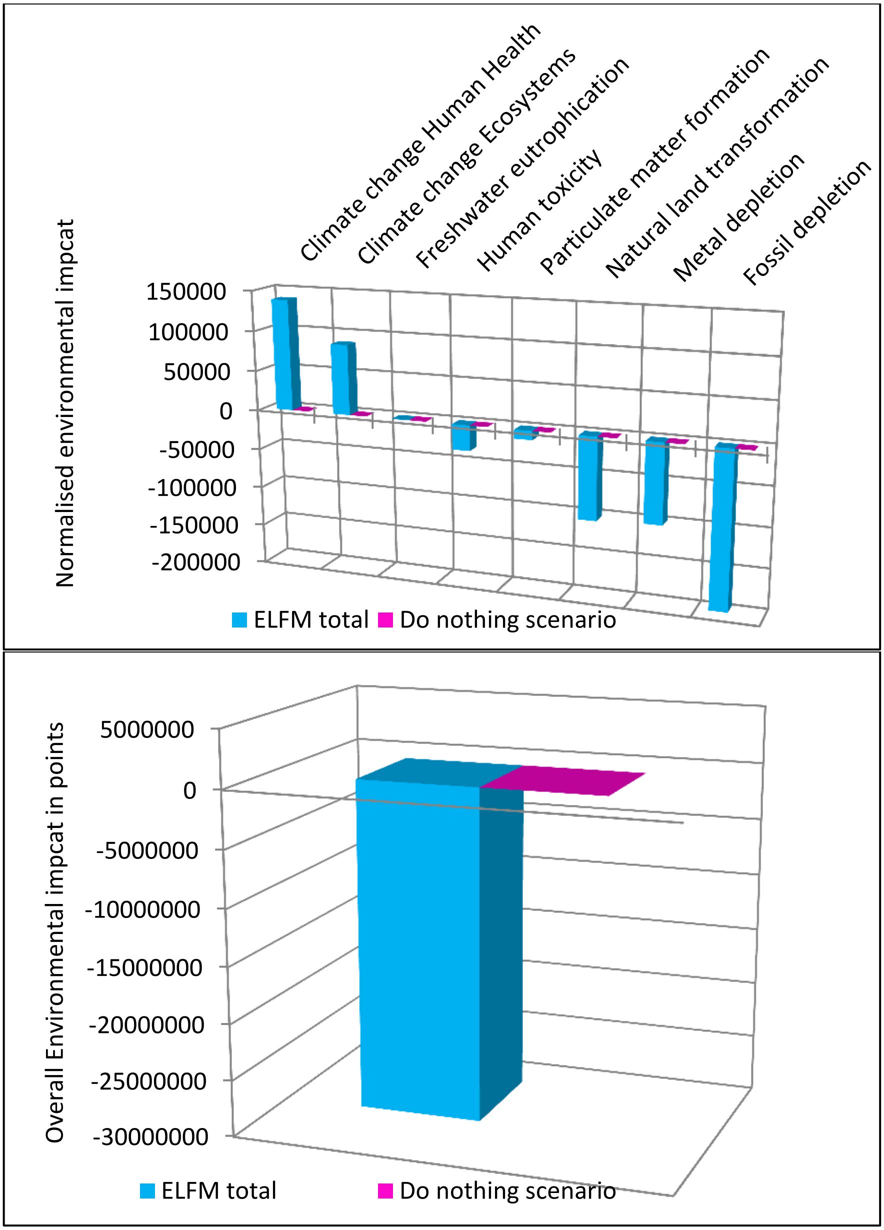


Figure 4.6: Environmental profile of valorisation of total waste present in the landfill compared to Do Nothing scenario with normalised data per impact category (top panel) and single score data (bottom panel)

However, the impact on climate change impact category can be approximately compared as it is directly linked with the CO_2 equivalent emission. The present study contrasts with that of Van Passel et al. in that this impact is lower in the Do-nothing scenario than that in ELFM scenario. However, Van Passel et al. considered that the energy recovery from methane would last for approximately 15 years, which is not the case in the present study (energy recovery will take place for only five more years). Moreover, the authors considered the materials and energy to have been purchased on the market in the case of the Do-nothing scenario. Thus, the associated emissions of conventional market production methods were accounted for when estimating the emissions of the Do-nothing scenario. As this fact is not considered in this study (but as an avoided burden in the ELFM scenarios), it resulted in a lower impact in the Do-nothing scenario. Nevertheless, because of the lower environmental impact on climate change impact category, it should not be concluded that ELFM is not favourable compared to the Do-nothing scenario. As explained in the previous sections, all impact categories should be taken into account in decision making of ELFM from an environmental point of view. Nevertheless, the bottom panel of Figure 4.6 shows that the overall environmental impact of ELFM scenario is significantly beneficial compared to that of Do-nothing scenario.

4.5.2. Sensitivity analysis in environmental profiles

From the above analysis, it was identified that the separation, thermal treatment and IW fines valorisation processes are the most influencing processes in ELFM. Furthermore, metal recovery in the separation and IW fines valorisation processes, along with electricity production in the thermal treatment process, were recognised as the main factors that dominate the environmental profiles. Table 4.1 summarises the parameters for which the sensitivity analyses are performed. Table 4.2 illustrates the results of the sensitivity analysis. The table shows the type of the net impact (that is, benefit or burden) of the basic scenario in each impact category. The other columns of the table indicate how this net impact deviates with the scenarios in the sensitivity analysis. Increments and decrements are denoted by (+) and (-) signs, respectively.

Table 4.1: Overview of the sensitivity analyses

Parameter	Scenario	Value
Metal recovery efficiency in separation process	basic	80%
	best case	90%
	worst case	70%
RDF recovery efficiency in separation process	basic	80%
	best case	90%
	worst case	70%
Calorific value of RDF	basic	20 MJ/kg
	best case	22 MJ/kg
	worst case	18MJ/kg
Net electrical efficiency of plasma gasification system	basic	27%
	best case	30%
	worst case	24%
Metal recovery efficiency in IW fines valorisation process	basic	10%
	sensitivity analysis	30%
	sensitivity analysis	50%
	sensitivity analysis	70%
	sensitivity analysis	90%

Table 4.2 shows that the climate change and fossil depletion impact categories are sensitive to the changes in RDF recovery efficiency, the calorific value of RDF and the electrical efficiency of thermal treatment process. A 10 percent increment in RDF recovery efficiency increases the burden on climate change impact category by 9-10 percent. However, the same change yields a 12 percent improvement in the benefit on fossil depletion impact category. The reason for this is that processing more RDF generates more flue gas, although it also produces more electricity. The benefit on fossil depletion impact category increases by 15-17 percent when the calorific value of RDF and the net electrical efficiency of plasma gasification system are improved by 10 percent. These changes significantly affect the particulate matter formation impact category by increasing its benefit by 18-29 percent. The metal depletion impact category is clearly sensitive to changes in the metal recovery efficiency of the separation process and the IW fines valorisation process. The metal recovery efficiency of IW fines valorisation was set to 10 percent for the basic scenario with respect to the available valorisation technologies. Robust hydrometallurgical treatments are needed that can selectively recover valuable metals and produce a residue with improved environmental properties, so that it can be used as a secondary raw material (Spooren et al. 2013). Evidently, the environmental contribution escalates when higher metal recovery efficiencies are applied (Table 4.2). This improvement is especially pronounced in the metal depletion impact category. When applying different metal recovery efficiencies in this sensitivity analysis, the other process inputs, such as energy and chemicals, were kept constant as applied in the basic scenario. This enabled the environmental profiles to change according to the changes in energy and chemical inputs that need to be performed in order to obtain higher recovery efficiencies.

4.5.3. Economic performance of waste valorisation

Using the cash flow model described in the section 4.4, the sensitivity of NPV to a wide range of parameters was investigated for MSW valorisation, IW valorisation and total waste (MSW+IW) valorisation. The following major parameters were considered: (i) key waste fractions, (ii) recovery efficiencies, (iii) the amount of different input materials to various processes, (iv) calorific value of RDF, (v) efficiencies of thermal treatment systems, (vi) investment and operational costs of the different valorisation processes, and (vii) the selling prices of the different products. Monte Carlo simulations show that the following parameters have an important impact on the economic performance for the basic scenarios: (i) net electrical efficiency of thermal treatment system, (ii) calorific value of RDF, (iii) price of electricity, (iv) price of green certificates, (v) green energy fraction (vi) investment cost of thermal treatment system, and (vii) operational costs of thermal treatment system. Table 4.3 illustrates the contribution of the different parameters in explaining the variation in NPV and their direction of influence obtained from Monte Carlo simulations using triangular distributions.

Table 4.2: Percentage changes in net impact of basic scenario, for the scenarios in the sensitivity analysis (colored cells represent the IW valorisation)

Impact categories	Climate change Human Health	Climate change Ecosystems	Human toxicity	Particulate matter formation	Natural land transformation	Metal depletion	Fossil depletion
Scenarios							
*Basic scenario**	Burden	Burden	Benefit	Benefit	Benefit	Benefit	Benefit
	Burden	Burden	Benefit	Benefit	Benefit	Benefit	Benefit
Metal recovery efficiency of separation process							
Best case (90%)	-1%	-1%	+5%	+5%	0%	+7%	+1%
	-1%	-1%	+7%	+8%	0%	+9%	+2%
Worst case (70%)	+1%	+1%	-5%	-5%	0%	-7%	-1%
	+1%	+1%	-7%	-8%	0%	-9%	-2%
RDF recovery efficiency of separation process							
Best case (90%)	+10%	+10%	+6%	+8%	0%	+4%	+12%
	+9%	+9%	+3%	+5%	0%	+2%	+12%
Worst case (70%)	-10%	-10%	-6%	-8%	0%	-4%	-12%
	-8%	-8%	-3%	-5%	0%	-2%	-11%
Calorific value of RDF							
Best case (22 MJ/kg)	-10%	-10%	+7%	+26%	0%	+1%	+15%
	-9%	-9%	+4%	+18%	0%	0%	+16%
Worst case (18 MJ/kg)	+10%	+10%	-7%	-26%	0%	-1%	-15%
	+9%	+9%	-4%	-18%	0%	0%	-16%
Net electrical efficiency of plasma gasification system							
Best case (30%)	-11%	-11%	+8%	+29%	0%	+1%	+16%
	-10%	-10%	+4%	+20%	0%	0%	+17%
Worst case (24%)	+11%	+11%	-8%	-29%	0%	-1%	-16%
	+10%	+10%	-4%	-20%	0%	0%	-17%
Metal recovery efficiency of IW fines valorisation process							
30%	-4%	-4%	+16%	+23%	0%	+20%	+6%
50%	-7%	-7%	+33%	+46%	0%	+39%	+12%
70%	-11%	-11%	+49%	+68%	0%	+59%	+18%
90%	-15%	-15%	+66%	+91%	0%	+78%	+25%

*basic scenario comprises 80% of metals and RDF recovery efficiency of the separation process, 20 MJ/kg calorific value of RDF, 27% net electrical efficiency of plasma gasification system and 10% metal recovery efficiency in IW fines valorisation process (Appendix A)

Table 4.3: Net Present Value sensitivity analysis using Monte Carlo simulations

Parameter	Minimum value	Maximum value	Contribution to variance of NPV (%)		
			MSW valorisation	IW valorisation	Total waste valorisation
Net electrical efficiency of thermal treatment process (%)	24	30	27.5 (+)	27.7 (+)	29.7 (+)
Calorific value of RDF (MJ/kg)	18	22	18.8 (+)	17.8 (+)	14.0 (+)
Price of electricity (€/MWh)	60	76	12.4 (+)	13.4 (+)	11.3 (+)
Price of green certificates (€/MWh)	110	124	5.3 (+)	4.7 (+)	5.4 (+)
Green energy fraction (%)	42	52	5.4 (+)	4.2 (+)	4.9 (+)
Investment cost of thermal treatment process (€/t RDF)	45	55	26.2 (-)	27.9 (-)	29.4 (-)
Operational cost of thermal treatment process (€/t RDF)	57	77	3.5 (-)	3.9 (-)	3.5 (-)

According to Table 4.3, all highly sensitive parameters belong to the thermal treatment process. The table highlights that the thermal treatment process dominates not only the environmental performance of ELFM but also the economic performance. This is the same in MSW valorisation or IW valorisation or total waste (MSW+IW) valorisation. The total variation in the NPV can be explained for 27-30 percent by the variation in net electrical efficiency of thermal treatment process. Higher efficiency logically results in a higher NPV. This shows that the efficiency of the thermal treatment process is of key importance for the economic feasibility of ELFM. Improvements in electrical efficiency may lead to higher investment costs. It appears that higher investment costs have a negative effect on NPV (26-30 percent). Therefore, these two parameters should be carefully controlled in order to reach the optimal profit of ELFM. The next important parameter is the calorific value of RDF. As the calorific value of the organics decreases over time due to degradation (Quaghebeur et al. 2013), starting ELFM activities before the landfill reaches its final stages of waste degradation makes it possible to treat waste with a high calorific value and obtain a higher energy output. Because plastics dominate the RDF fraction in the case study landfill and the landfill is already in its final stages of waste degradation, the calorific value cannot increase further. However, this finding could be considered in other future ELFM projects. Along with the calorific value, the price of electricity, the price of green certificates, the green energy fraction and the operational cost of thermal treatment process also contribute to the NPV considerably. The impact of other parameters, such as recovery efficiencies and prices of recovered materials, is negligible. Importantly, the LCA study identified that the metal recovery is highly beneficial in the metal depletion impact category, but its impact on the economic profile is insignificant. However, the range definitions of the various parameters strongly influence the final impact of the different parameters on the NPV. In this study, for most of the parameters, a 10 percent margin from the average value was set to the maximums and minimums of the range. For the cost parameters, such as green certificates and investment and operational costs, these ranges are defined after communication with the experts in the relevant industries. These results are in line with those obtained by Van Passel et al. (2013). Unlike that study, however, the present study includes all the possible activities that can be conceived within a ELFM project. However, the economic drivers identified by Van Passel et al. remain the same for this study as well, despite the large range of parameters considered. As explained in the previous study, technology (efficiency, investment cost), markets (electricity price) and regulations (price of green certificates, green energy fraction) determine the economic performance of ELFM to a large extent.

4.6. Conclusions

This chapter has presented a full LCA and LCC of ELFM based on the REMO landfill as a case study. The results show that the total environmental impact of ELFM depends on the type and composition of the waste and the chosen process technologies. Apart from that, the net environmental impact of ELFM depends heavily on the quality and the quantity of the output products. In this research, we assumed that all metals recovered from the landfill have the quality of the secondary metals. This leads to large energy savings and helps avoid many kinds of environmental pollution caused by the replacement of primary material. The recovered soil, sand and aggregates, with the quality level of gravel, avoid a large area of arable land that has to be converted for gravel extraction. The produced electricity substitutes the Belgian electricity mix, which contains 40 percent of conventional thermal energy. In this respect, it is necessary to perform ELFM with the maximum product quality.

We conclude that the waste types and different processes of ELFM behave differently on the considered impact categories. None of the waste types or processes has the highest or lowest environmental score for all impact categories. On the other hand, ELFM does not yield only the benefits on all impact categories. In this case study, the impact categories related to climate change are always influenced adversely by ELFM, while human toxicity, particulate matter formation, natural land transformation, metal depletion and fossil depletion impact categories are positively affected.

The environmental impact (both benefits and burdens) of valorisation of total waste (IW+MSW) in all impact categories is highly significant compared to the Do-nothing scenario. However, the level of this impact differs depending on the type and phase or average age of the landfill. This suggests that the actual situation of the landfill is important in decision making in ELFM.

We found that the thermal treatment (plasma gasification) is the process that has the greatest influence, both from an environmental and an economic point of view. Therefore, it is necessary to explore the possibility of using other possible thermal treatment technologies as well. Essentially plasma gasification must be benchmarked against conventional incineration, a commonly used thermal treatment method in waste processing, with the purpose of proving that plasma gasification is one of the efficient technologies for achieving the goals of ELFM concept. In addition, it is important to know how the by-products of plasma gasification (plasmastone) contribute to the performance of ELFM. Apart from the use of plasmastone in aggregate production, its higher added value applications should also be analysed in order to investigate how the environmental and economic impacts of ELFM vary along the different product qualities.

Importantly, this study shows that the impact of some parameters and processes are negligible from an economic perspective, but become key drivers from an environmental point of view, and vice versa. Examples include the higher influence of metal recovery in environmental profiles and the insignificant impact of metal prices in economic profiles. The study further confirms that the technology, regulations and markets have a clear impact on the economic feasibility of ELFM. Finally, it can be concluded that the environmental and economic profiles of ELFM vary from case to case depending on landfill characteristics, compositions, technologies used and products of ELFM. In fact, the results obtained for this case study landfill suggest a cluster of parameters (waste composition, recovery efficiencies, thermal treatment technology, net electrical efficiency of thermal treatment process, calorific value of RDF, price of electricity, investment cost of thermal treatment process) that need to be considered in future ELFM projects in order to minimise their environmental burden and to maximise the economic return.

Chapter 5 : Valorisation of thermal treatment residues in Enhanced Landfill Mining: Environmental and economic evaluation

This chapter is based on

Danthurebandara, M., Van Passel, S., Machiels, L., Van Acker, K.. (2015). Valorisation of thermal treatment residues in Enhanced Landfill Mining: Environmental and economic evaluation. *Journal of Cleaner Production* 99: 275-285

Abstract

This chapter presents an environmental and economic evaluation of the valorisation of thermal treatment residues in the context of Enhanced Landfill Mining (ELFM). The thermal treatment residues discussed in this work include plasmastone, generated by the plasma gasification process. The most common valorisation route, that is the treatment of plasmastone via production of aggregates, is compared with two other possible, higher added value applications, which are inorganic polymer production and blended cement production. The evaluation is based on a life cycle assessment and life cycle costing. The environmental and economic impacts are expressed in global warming potential and net present value, respectively. The study suggests that the environmental and economic performances of the valorisation routes depend mainly on the quality and quantity of the final products produced from a certain amount of plasmastone. The materials with the greatest contribution to potential global warming and to the net present value of the valorisation scenarios are the process input materials of sodium silicate, sodium hydroxide and cement. The study reveals that the plasmastone valorisation via inorganic polymer production yields higher environmental benefits, while the blended cement production provides higher economic profits. Plasmastone valorisation via aggregates production yields neither economic nor environmental benefits.

5.1. Introduction

Chapter 3 describes the major process steps of ELFM, including vegetation and topsoil removal, conditioning, excavation, separation, transformation of intermediate products, and land reclamation. Amongst these processes, the separation process can be considered as the backbone process of ELFM as it results in many waste fractions that can be sold directly. In addition, intermediate products also are sorted out in the separation process. Refuse derived fuel (RDF) is an important intermediate product that can be valorised in a thermal treatment with energy recovery (Quaghebeur et al. 2013). Previous chapters highlight the usability of plasma gasification in ELFM to valorise RDF due to its combined energy and material valorisation capacity, and a number of other advantages (Chapman et al. 2010, Ray et al. 2012, Bosmans et al.

2013). Plasma gasification technology is able to efficiently produce a clean synthesis gas and an environmentally stable vitrified product (plasmastone) from historically landfilled materials (Taylor et al. 2013). The synthesis gas can be used for production of electricity and/or heat or as second-generation liquid fuels. As ELFM focuses broader attention on the valorisation of all types of landfill waste, even waste and by-products generated during processing of landfill waste, the obtained plasmastone should also be treated to obtain valuable products. In this context, several valorisation possibilities have been proposed for plasmastone (Iacobescu et al. 2013, Pontikes et al. 2013, Machiels et al. 2014).

The residues (bottom ash) produced in traditional thermal treatment processes like incineration are disposed of directly to landfills in many cases. This material needs to be pretreated if it is to be utilized as a secondary aggregate. In contrast, plasmastone has a great potential and can be designed for use in rather diverse applications, mainly in the construction materials industry (Jones et al. 2013, Spooren et al. 2013). Leaching tests have indicated that plasmastone may be safely used as an aggregate/gravel replacement (Chapman et al. 2011). Hence, the most evident valorisation route is the use of plasmastone as an aggregate for road construction or building blocks. Nevertheless, ELFM targets higher value applications. Jones et al. (2013) highlighted that depending on the RDF chemistry and the cooling method applied, the following higher added value products can be developed from plasmastone: glass-ceramic monoliths for use as building materials or glass-ceramic aggregates for use in high-strength concrete; hydraulic binders, pozzolanic binders, or inorganic polymer precursors.

According to the International Energy Agency's (IEA) Greenhouse Gas R&D Program (Hendriks et al. 2000), ordinary Portland cement (OPC) production generates an average world carbon emission of 0.81 kilogram of CO_2 per kilogram of cement produced. On average, one tonne of concrete is produced each year for every human being in the world (Lippiatt and Ahmad 2004). Production of alternatives for cement can mitigate this heavy CO_2 burden. So far, fly ash and other by-products of the energy and materials industry, currently disposed of as waste, have been used to produce these alternative products (Huntzinger and Eatmon 2009, Turgut 2012, Van den Heede and De Belie 2012). Machiels et al. (2014) and Iacobescu et al. (2013) explained the possibility of developing binding materials from plasmastone to be used as low-carbon alternatives for OPC in construction applications.

Based on these premises, several valorisation routes for plasmastone have been tested at KU Leuven, Belgium, in the framework of the first comprehensive ELFM project ('Closing the Circle' project by Group Machiels, Belgium). These valorisation routes mainly include production of inorganic polymer and blended cement products out of plasmastone. To bring ELFM from the conceptual to the operational stage, knowledge about the critical factors of environmental and economic performance of the associated technologies is important. In addition, one of the conclusions of chapter 4 is that, apart from use of plasmastone in aggregate production, its higher added value applications should also be analysed in order to investigate how the environmental and economic impacts of ELFM vary along the different product qualities. Nonetheless, because of the novelty of the ELFM concept, such evaluations for plasmastone valorisation in ELFM have not yet been reported, although several other studies have evaluated the products based on waste materials and by-products. For example, Weil et al. (2009) conducted a detailed life cycle analysis of geopolymers produced both from resource-intensive materials like metakaolin and less resource-intensive materials like fly ash, and McLellan et al. (2011) examined the environmental and economic impacts of the life cycle of geopolymers produced from fly ash. In addition, several studies have discussed the environmental performance of blast furnace slag used in geopolymer production (Habert et al. 2011, Van den Heede and De Belie 2012). Although these studies explain the possible environmental impacts of transformation of waste materials into alternatives to OPC, a more detailed evaluation is required for plasmastone valorisation to identify its usability in ELFM.

This chapter addresses the current lack of environmental and economic evaluation for valorisation of thermal treatment residues in ELFM. The study comprises life cycle assessment (LCA) and life cycle costing (LCC). The most common valorisation route, aggregate production, was compared with two other higher added value applications, inorganic polymer production and blended cement production. This chapter identifies and discusses the environmental and economic drivers of plasmastone valorisation, analyzes the relative advantages and disadvantages of different scenarios, and suggests possible improvements in design and operating parameters. In addition, a trade-off analysis indicates the most beneficial valorisation options to be used in ELFM.

5.2. The system studied

As explained in the previous chapters 3 and 4, the excavated landfill waste is subjected to a series of separation processes to sort different waste fractions. The RDF fraction obtained by the separation process is directed to a thermal treatment process (plasma gasification) as shown in Figure 5.1. The main products identified were synthesis gas and plasmastone. Synthesis gas can be used mainly for energy production (electricity and/or heat), although other valorisation options include production of liquid fuels. Plasmastone, which is recovered from the plasma convertor, is fully vitrified, mechanically strong and environmentally stable. The obtained plasmastone is exposed to additional various treatments in order to obtain valuable products. Although the entire ELFM system presents a multitude of complex interactions, the focus in this study was only on the subsystem of plasmastone valorisation, which is the step highlighted by the dotted line in Figure 5.1. Different valorisation options were compared to identify the best option, according to environmental and economic considerations.

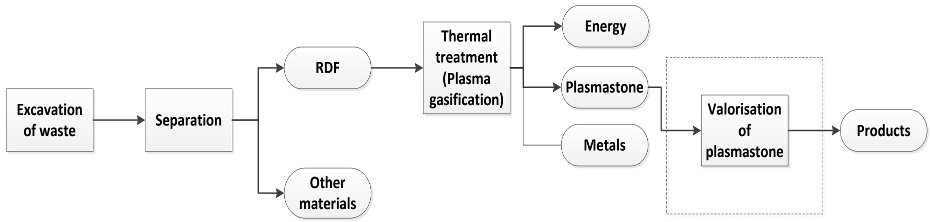


Figure 5.1: Interactions of ELFM and the system studied (focus of the current study is outlined by the dotted line)

5.3. Valorisation/treatment routes

This study included three main scenarios: Valorisation of plasmastone via (i) production of inorganic polymer cement/block, (ii) production of blended cement/block, and (iii) production of aggregates. The process flow diagrams were developed according to the literature data and lab scale experiments conducted at KU Leuven.

Scenario 1 – Valorisation of plasmastone via production of inorganic polymer cement/ block

Inorganic polymers, alternatively termed geopolymers, are a class of materials formed by the reaction between an alkaline solution and a reactive precursor material, rich in silica and commonly alumina (Provis and Deventer 2009, Deventer et al. 2010, Davidovits 2011). Inorganic polymers display outstanding technical properties, such as high strength, high acid resistance, and high temperature resistance. These materials form a hard, durable body that can be used as an alternative to OPC for standard or more demanding applications, including in environments with high temperature or acid conditions, as well as for the encapsulation and disposal of hazardous wastes (Davidovits 2011). Well-performing inorganic polymers also can be obtained using secondary raw materials, such as industrial by-products like fly ash or slag. The use of these materials as input for inorganic polymer production not only could solve the waste problem, but also reduce consumption of primary raw materials.

As explained above, this study focused on inorganic polymer production from plasmastone. Currently, inorganic polymer cement and a precast product are being developed at the Department of Materials Engineering, KU Leuven. Plasmastone is an ideal precursor for a inorganic polymer cement because it can be designed to be composed of more than 90 percent glass, which ensures a very high reactivity in an inorganic polymer cement (Pontikes et al. 2013). Work is in progress on the optimization of the plasmastone chemistry and of the inorganic polymer cement blend composition, and partial results have been published (Machiels et al. 2014). This study utilized data from Machiels et al. (2014) regarding the optimal blend composition for cement production, which is a blend composed of more than 90 percent plasmastone that can deliver superior properties, such as higher compressive strength, compared to traditional OPC. This plasmastone composition can be obtained by treatment of a mixture of industrial and household waste RDF, derived from ELFM, in a plasma gasification system (Spooren et al. 2013). Figure 5.2 is the process flow diagram for Scenario 1, depicting the following major steps (see also Machiels et al. 2014).

- Milling: The size of the plasmastone received from the plasma gasification process after application of water quenching to obtain the required reactivity is approximately 0.5 centimeters on average (Pontikes et al. 2013). Milling of the plasmastone is the first step of the treatment process and is needed to obtain the required uniform grain size.
- Pre-mixing: Milled plasmastone is mixed with an alkali source (NaOH) and a silicate solution (Na_2SiO_3) to produce an inorganic polymer cement.
- Mixing: The resulting mixture of plasmastone and alkali and silicate solutions (inorganic polymer cement) is then mixed with water and aggregates to obtain an inorganic polymer mortar or concrete that can be shaped to blocks as an alternative to OPC-based concrete blocks and bricks.
- Curing: In this study, curing was done at room temperature (20°C) because inorganic polymers based on reactive materials activated with a Na-silicate solution can achieve the desired technical properties within a few hours or days at room temperature without any heat curing (Bakharev et al. 1999, Duxson et al. 2007).

Two sub-scenarios emerged from the process shown in Figure 5.2: Scenario 1a – valorisation of plasmastone via inorganic polymer cement production; and Scenario 1b – valorisation of plasmastone via inorganic polymer block production.

Scenario 2 –Valorisation of plasmastone via production of blended cement/ block

Cement and concrete terminology defines blended cement as hydraulic cements that are consisting of an intimate and uniform blend of a number of constituent materials, generally termed supplementary cementitious materials (SCM) (ACI 2005, Snellings et al. 2012). To produce blended cements, Portland cement clinker may be either intergrinded or blended with the SCM, or a combination of both. Blended cements have the advantages of improved workability, improved resistance to sulfate attack and chloride penetration, improved resistance to alkali-silica reactions, and improved long-term strength development. Plasmastone has good potential as SCM because it has a high glass content and an appropriate chemistry (high silica content, and substantial calcium and aluminium content), which enables it to react with OPC clinker to form calcium silicate hydrate binding phases (Iacobescu et al. 2013). Figure 5.3 depicts the process flow for treatment of plasmastone via blended cement production. The milled plasmastone and cement were blended together to produce blended cement. In addition, water and aggregates can be added to the blend to produce a pre-cast product or blended cement blocks. As in Scenario 1, curing was done at room temperature. As defined by Iacobescu et al. (2013), we considered a mixture of 20 percent plasmastone and 80 percent OPC in this study.

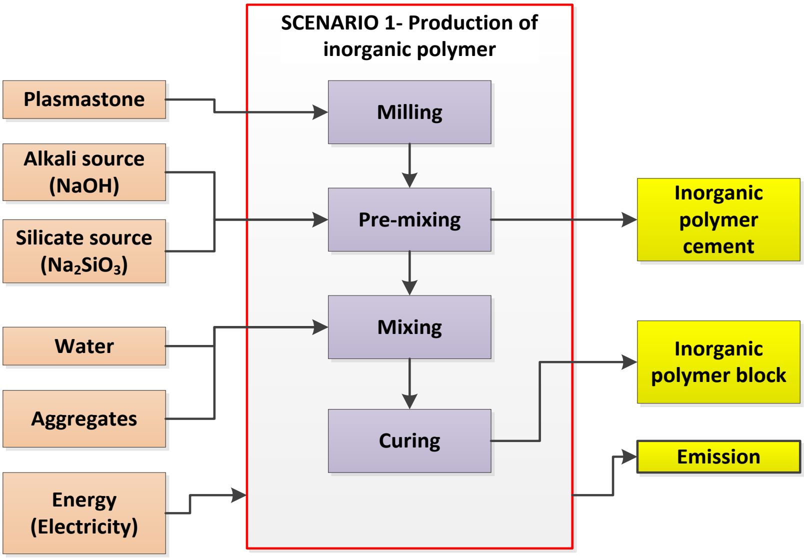


Figure 5.2: Scenario 1 for the valorisation of plasmastone

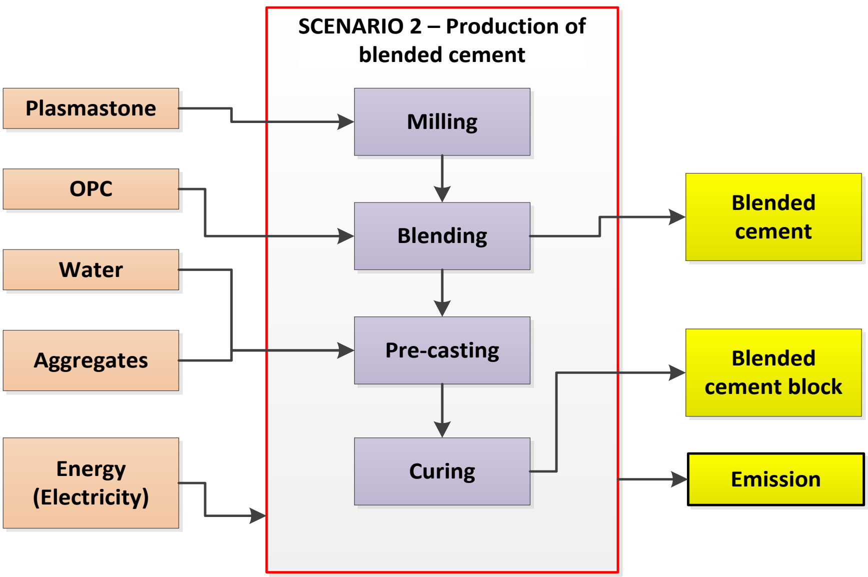


Figure 5.3: Scenario 2 for the valorisation of plasmastone

Two sub-scenarios emerged from the process shown in Figure 5.3: Scenario 2a – valorisation of plasmastone via production of blended cement; and Scenario 2b – valorisation of plasmastone via production of blended cement block.

Scenario 3 – Valorisation of plasmastone via production of aggregates

Most materials for aggregate production come from bedrock or from unconsolidated deposits. The vast majority of materials used in the mineral aggregate industry are obtained from surface-mined stone quarries or from sand and gravel pits. In addition, increasing amounts of recycled materials are being used to supplement natural aggregates (Ray et al. 2012, Taylor et al. 2013). Plasmastone's unique combination of high mechanical strength and hardness, as well as its extremely high resistance to chemical leaching, make it a perfect secondary aggregate material for use in road paving and pipe bedding (Ray et al. 2012, Taylor et al. 2013). Use of secondary raw materials as aggregates avoids the long production process of natural aggregates, including extraction or mining, transportation to the processing plants, separation, crushing, scrubbing, and screening (Kellenberger et al. 2007). As shown in Figure 5.4, Scenario 3 comprised only two simple processes: crushing and sieving.

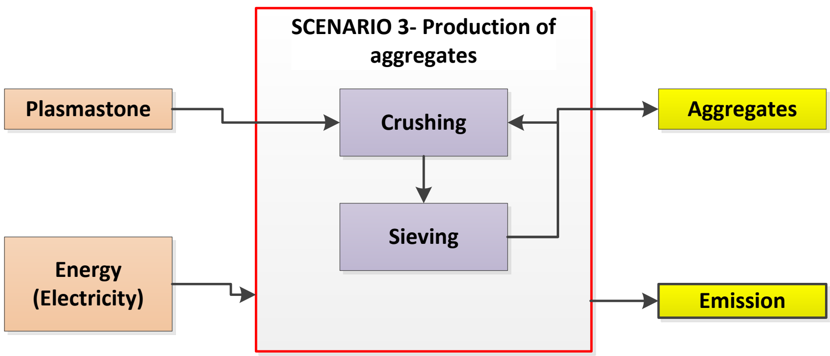


Figure 5.4: Scenario 3 for the valorisation of plasmastone

5.4. LCA methodology

The respective building blocks in the model described in chapter 3 were used to elaborate the environmental assessment of this study. The goal of this LCA study was to use the scenarios mentioned in section 2.2 to evaluate the environmental impacts of the valorisation of a certain mass of thermal treatment residues (plasmastone) obtained in the plasma gasification process, and thus, to identify the most beneficial treatment option to be used in ELFM. As described in chapter 3, the study followed the international standard for LCA (ISO 14040, 14044) and used SimaPro 7 (PRéConsultants 2010) as the software tool to set up the LCA model. Figures 5.1 to 5.4 show the system boundaries, and the material inputs and outputs under the study's different scenarios. Similar to chapter 4, in this study also we used a reference flow instead of a functional unit as explained in the ILCD handbook (2010). To provide information that could readily be scaled to any application, we used a reference flow of 1 tonne of plasmastone for the analysis, since the study's objective was to evaluate the treatment process rather than the product. In this way, we could identify the best valorisation route to process all plasmastone generated in a certain ELFM project.

The results could be different if the focus was on production of 1 tonne of product. But the key objective of ELFM is to use innovative transformation technologies to valorise the total waste in a landfill while minimizing the environmental burden and maximizing the economic return. We did not include the production phase of plasmastone because the amount of plasmastone to be treated in all scenarios was equal.

All values were quoted per tonne of plasmastone, and these values could be readily used to calculate the environmental impacts of valorisation for a requested functional unit of plasmastone. Table 5.1 shows the process inputs used in all scenarios and the variation of the quantities of the output products of different scenarios, as follows: Treatment of 1 tonne of plasmastone produces 1.142 tonnes inorganic polymer cement (Scenario 1a), 4.412 tonnes inorganic polymer blocks (Scenario 1b), 5 tonnes blended cement (Scenario 2a), 22.250 tonnes blended cement blocks (Scenario 2b), and 1 tonne of aggregates (Scenario 3). The uncertainty of the input data is low as they were mainly based on measured data during the lab scale experiments.For the background processes, the life cycle inventory data published in the ecoinvent database was used (Ecoinvent 2010). Transportation of input materials was not included because we assumed all input materials to be manufactured within Belgium where the study took place.

As described previously, the alternative products obtained in plasmastone valorisation can mitigate the heavy CO_2 burden caused by OPC-based products. Therefore, we selected global warming potential (GWP) as the priority impact category for the environmental impact assessment of this study because it directly relates to the CO_2 burden. IPCC 2007 GWP 100a method (PRéConsultants 2008) was used as the assessment method. On the other hand, the valorisation methods can affect several other impact categories. To investigate this influence, we used the ReCiPe midpoint method (Goedkoop et al. 2013) to assess the impact categories of (i) climate change, (ii) ozone depletion, (iii) terrestrial acidification, (iv) freshwater eutrophication, (v) marine eutrophication, (vi) human toxicity, (vii) photochemical oxidant formation, (viii) particulate matter formation, (ix) terrestrial ecotoxicity, (x) freshwater ecotoxicity, (xi) marine ecotoxicity, (xii) ionising radiation, (xiii) agricultural land occupation, (xiv) urban land occupation, (xv) natural land transformation, (xvi) water depletion, (xvii) metal depletion, and (xviii) fossil fuel depletion.

Table 5.1: Inputs and outputs of the different treatment scenarios

Inputs and outputs	Scenario 1a	Scenario 1b	Scenario 2a	Scenario 2b	Scenario 3
Inputs					
Plasmastone (t)	1.000	1.000	1.000	1.000	1.000
NaOH (t)	0.064[a]	0.064[a]			
Na_2SiO_3 (t)	0.078[a]	0.078[a]			
Water (t)		0.308[a]		2.500[b]	
Aggregates (t)		3.000[a]		15.000[b]	
OPC (t)			4.000[b]	4.000[b]	
Electricity (kWh)	20.000[c]	55.000[c]	20.000[c]	55.000[c]	10[c]
Output products					
Inorganic polymer cement (t)	1.142[a]				
Inorganic polymer block (t)		4.142[a]			
Blended cement (t)			5.000[b]		
Blended cement block (t)				22.250[b]	
Aggregates (t)					1.000[c]

[a]Machiels et al. (2014) , [b]Iacobescu et al. (2013), [c]measured value

5.5. LCC methodology

A typical cement production plant is 500 000 or 1 000 000 tonne/year and a production line is typically 100 000 tonne (so one plant is 5-10 production lines). To perform LCC for the study's selected scenarios (Section 5.3), one production line of a hypothetical treatment plant was considered with a line capacity to treat a 100,000 tonnes of plasmastone per year and having a lifetime of 20 years. The cash-flow model described in chapter 3 was further developed for the 20-year period, including all costs and revenues associated with the scenarios (see Table 5.2, the uncertainty ranges are based on literature data, personal communication and own assumptions). The cost advantages due to size, output, or scale of operation (economies of scale) were considered to determine the investment and operational costs of the scenarios. Current investment costs of cement production were used to determine the investment costs under Scenarios 1a, 1b, 2a, and 2b. Unlike in the manufacture of Portland cement, a kiln and other infrastructure are not needed to produce inorganic polymer or blended cement; consequently, the investment cost of Scenarios 1a and 2a were assumed to be 30 percent of the reported investment cost of a cement production plant (€263 per tonne cement/year for a plant with a capacity of 1 million tonnes/year and a 20-year lifetime) (ETSAP 2010). The ratios of plasmastone to inorganic polymer cement and plasmastone to blended cement were used to convert the units into euros per tonne of plasmastone. The investment costs of Scenarios 1b and 2b were assumed to be twice those of Scenarios 1a and 1b. Materials and energy costs were estimated according to the market prices in Belgium where all materials were to be produced. However, a sensitivity analysis was performed for all these assumptions and estimations as a higher uncertainty can be expected for the data related to these young technologies. Similar to the LCA study, the LCC study did not consider the production phase of plasmastone; therefore, a value is not given for the price of plasmastone. The next challenge was to determine the selling prices of the products, according to their physical and chemical characteristics. Because inorganic polymer cement offers higher compressive strength than traditional Portland cement, the selling price of inorganic polymer cement was estimated at €150/t. Considering the proportion of cement and aggregates of inorganic polymer block, we calculated the selling price of inorganic polymer block at €70/t. In the same way, €100/t and €50/t were assigned for blended cement and blended cement block. According to their quality, aggregates were priced at €5/t. A sensitivity analysis was performed to address the high uncertainty that is foreseen for the selling prices of above relatively new products.

Because our main objective was to compare the different scenarios, we used NPV as the major economic indicator, which we calculated by subtracting the investment cost from the discounted cash flows. For this private assessment, we applied a 15 percent discount factor (Van Passel et al. 2013). For the simplicity, we used a zero percent inflation rate.

Similar to chapter 4, Monte Carlo simulation approach was used, to examine how the NPV varies when the value of uncertain assumptions is modified (Van Passel et al. 2013).

In general, private investors consider projects with an IRR of 15 percent to be profitable (Van Passel et al. 2013), a fact used to calculate the minimum selling prices of the products and maximum buying price of plasmastone. As IRR is the discount rate at which the NPV is zero, we used Equation 5.1 to calculate the minimum selling prices of the products and Equation 5.2 to calculate the maximum buying price of plasmastone.

$$NPV = \sum_{t=0}^{20} \frac{(costs - (x * amount\ of\ product\ per\ year))}{(1 + Discount\ rate)^t} = 0 \qquad (5.1)$$

Where t is time and x indicates the unit selling price of a product.

$$NPV = \sum_{t=0}^{20} \frac{(((y * annual\ treatment\ capacity) + other\ costs) - Revenues))}{(1 + Discount\ rate)^t} = 0 \qquad (5.2)$$

Where t is time and y indicates the unit price of plasmastone.

Table 5.2: Input values used in cash flow model

Description	value	Sources
General data		
Plasmastone treatment capacity (t/y)	100,000	case study
Life time (y)	20	case study
Discount factor (%)	15	case study
Scenario 1		
Investment cost (€/t plasmastone)		
Scenario 1a	5 ± 3	calculated value (mentioned in the text)
Scenario 1b	10 ± 6	calculated value (mentioned in the text)
Materials prices		
Price of plasmastone (€/t)	0	case study
Price of water (€/t)	4	industrial reference (SWDE, Belgium), Chiara (2008)
Price of NaOH (€/t)	275±25	ICIS (2010)
Price of Na_2SiO_3 (€/t)	405±225	ICIS (2008)
Price of aggregates (€/t)	10 ± 5	Gardiner & Theobald (2012)
Energy price		
Price of electricity (€/MWh)	68	Europe's Energy Portal (2013)
Maintenance and repair cost (% from investment cost)	10	Industrial reference (CRH, Holcim)
Other cost (labour+ other unforeseen costs) (€/t plasmastone)		
Scenario 1a	17	Industrial reference (CRH, Holcim)
Scenario 1b	19	Industrial reference (CRH, Holcim)
Revenues		
Selling price of inorganic polymer cement (€/t)	150 ± 15	case study
Selling price of inorganic polymer block (€/t)	70 ± 7	case study
Scenario 2		
Investment cost (€/t plasmastone)		
Scenario 2a	20 ± 12	calculated value(mentioned in the text)
Scenario 2b	40 ± 24	calculated value(mentioned in the text)
Materials prices		
Price of plasmastone (€/t)	0	case study
Price of water (€/t)	4	industrial reference (SWDE, Belgium), Chiara (2008)
Price of cement (€/t)	77 ± 7	industrial reference (Hubo, Holcim)
Price of aggregates (€/t)	10 ± 5	Gardiner & Theobald (2012)
Energy price		
Price of electricity (€/MWh)	68	Europe's Energy Portal (2013)
Maintenance and repair cost (% from investment cost)	10	Industrial reference (CRH, Holcim)
Other cost (labour+ other unforeseen costs) (€/t plasmastone)		
Scenario 2a	30	Industrial reference (CRH, Holcim)
Scenario 2b	39	Industrial reference (CRH, Holcim)
Revenues		
Selling price of blended cement (€/t)	100 ± 10	case study
Selling price of blended cement block (€/t)	50 ± 5	case study
Scenario 3		
Investment cost (€/t plasmastone)	3 ± 2	industrial reference (UEPG)
Operational +maintenance+ other costs(€/t plasmastone)	2	industrial reference (UEPG)
Revenues		
Selling price of aggregates (€/t)	5 ± 2	case study

5.6. Results and discussion

5.6.1. Environmental evaluation of valorisation of plasmastone

Influence of the process inputs

Apart from the processing conditions, the input raw materials selected are important parameters that determine the final products' setting behavior, workability, and chemical and physical properties. In addition, they largely define the environmental profiles of the treatment processes.

Figure 5.5 shows the influence of process inputs on the impact category of GWP. The graph presents the greenhouse gas emission in terms of kilograms of CO_2 equivalent per valorisation of 1 tonne of plasmastone. Clearly, there are significant differences between resource intensive and less resource intensive primary raw materials. In Scenario 1b, aggregates contribute only a little to GWP, despite their high mass proportion compared to NaOH and Na_2SiO_3. The provision of water also does not noticeably contribute to GWP. NaOH and Na_2SiO_3 significantly contribute to the GWP, and Na_2SiO_3 dominates the environmental profile of both sub-scenarios of Scenario 1. These results agree with other studies on environmental evaluation of inorganic polymer production (Weil et al. 2009, Habert et al. 2011).

The impact of aggregates used in Scenario 2b is only 1 percent of the total impact, although the quantity of aggregates used was more than three times higher than the quantity of OPC. Compared with Scenario 1, Scenario 2 shows significantly higher greenhouse gas emission due to process inputs, 216 to 223 kilograms of CO_2 equivalent compared with 3350 to 3387 kilograms of CO_2 equivalent, because Scenario 2 used OPC. To treat 1 tonne of plasmastone, 4 tonnes of OPC must be used, and OPC production generates an average greenhouse gas emission of 0.81 kilograms of CO_2 per kilograms of cement produced (Hendriks et al. 2000). Although NaOH and Na_2SiO_3 production have higher emissions, like 1.1 and 1.59 kilograms of CO_2 per kilogram (Althaus et al. 2007), the need for these chemicals is comparatively low to treat 1 tonne of plasmastone (0.064 tonne NaOH and 0.078 tonne Na_2SiO_3). Compared with Scenarios 1 and 2, the greenhouse gas emission is very low in Scenario 3 because electricity was the sole input to this process, and the energy requirement was comparatively low.

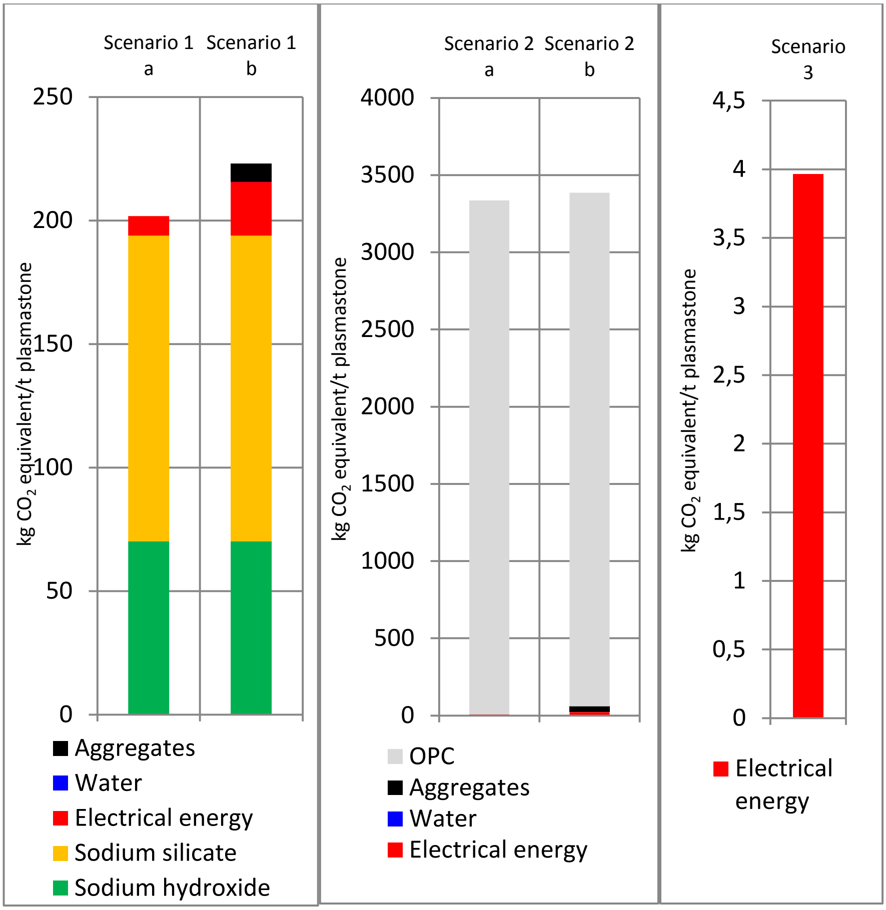


Figure 5.5: Greenhouse gas emission of process inputs of plasmastone valorisation scenarios

Influence of the substituted products (avoided environmental burden)

One key objective of the novel ELFM concept is to avoid primary material production to a certain extent by reintroducing buried resources in the material cycle. Hence, the quality of the products obtained in ELFM must be identified in order to determine clearly which materials can be replaced and to what extent. ELFM products have an 'avoided burden', meaning that these recycled materials avoid the impact of virgin material production. Hence, the overall environmental impact of the valorisation scenarios can be calculated by subtracting the avoided environmental impact from the environmental impact of process inputs shown in Figure 5.5.

To define the substituted products, we analyzed the ELFM products to determining their compressive strengths, porosity, consistence upon water immersion, and so on. The inorganic polymer cement obtained from Scenario 1a was found to exceed the quality of traditional OPC (a higher 28-day compressive strength). Hence, the substituted product of Scenario 1a was OPC, strength class 52.5 (CEM I, 52.5) (Kellenberger et al. 2007). Similarly, concrete blocks and bricks were the substituted products of Scenario 1b. Note that OPC is used in concrete production, but not in brick production. The quality checks indicated that the resulting blended cement in Scenario 2a had a compressive strength of 32.5 MPa (Iacobescu et al. 2013); therefore, we used CEM II 32.5 (Kellenberger et al. 2007) to calculate the avoided environmental impact under Scenario 2a. The selected substituted products to determine the avoided burden of Scenario 2b are OPC-based concrete blocks and non-OPC-based sand-lime bricks. The production methods and emissions of these products are clearly explained in ecoinvent report number 7 (Kellenberger et al. 2007). Finally, we assumed that Scenario 3 could replace the conventional gravel production. Figure 5.6 represents the net environmental profiles, including the avoided environmental burden of all scenarios. The red lines indicate the net environmental impact (environmental impact of process inputs- avoided environmental burden) of the scenarios.

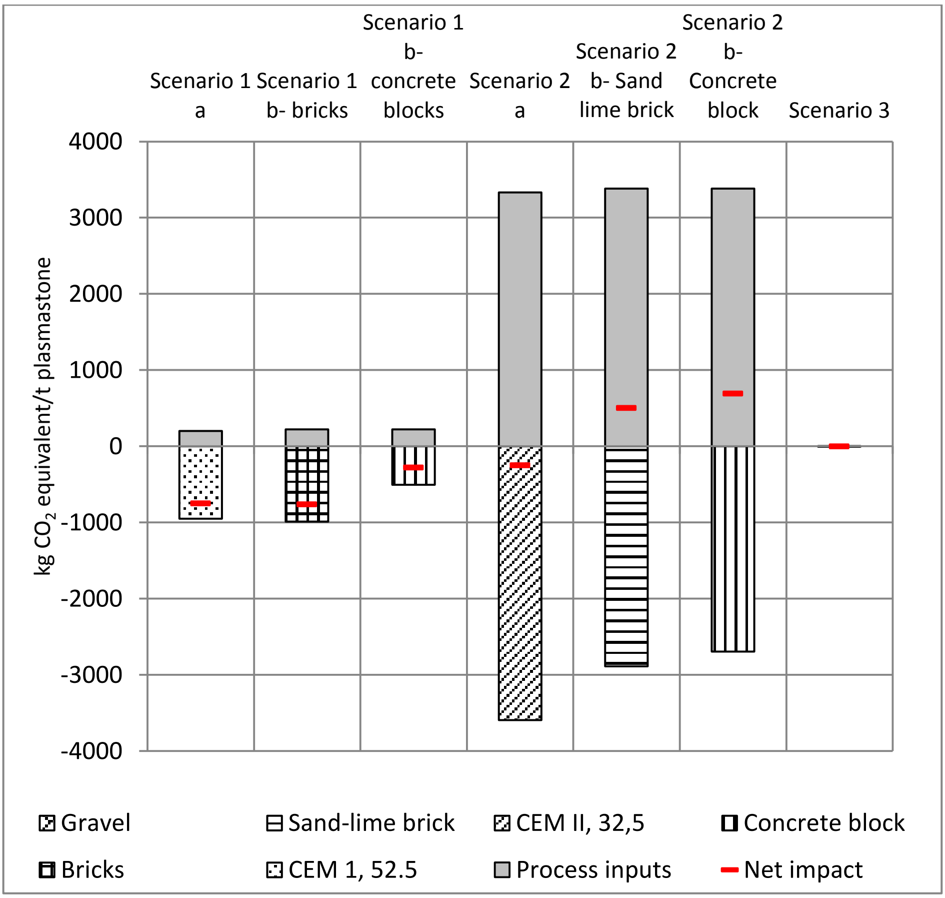


Figure 5.6: Net environmental profile of plasmastone valorisation scenarios

In Figure 5.6, the dotted column indicates the environmental impact that could be avoided under Scenario 1a. In this scenario, valorisation of 1 tonne of plasmastone produces 1.142 tonnes of inorganic polymer cement. This amount of inorganic polymer cement avoids the production of the same amount of OPC, strength class 52.5 (CEM I, 52.5) or avoids greenhouse gas emission of 950 kilograms of CO_2 equivalent; hence, the net savings of valorisation of 1 tonne of plasmastone via inorganic polymer cement production is 748 kilograms of CO_2 equivalent. In Scenario 1b, 1 tonne of plasmastone resulted in 4.142 tonnes of inorganic polymer blocks, which avoid the production of an equal amount of bricks or concrete blocks. The checkered and vertically striped columns in Figure 5.6 illustrate the avoided environmental impacts under Scenario 1b when bricks and concrete blocks are the substituted products. These substitutions prevent an emission of 985 and 501 kilograms of CO_2 equivalent, respectively. The net CO_2 equivalent savings under Scenario 1b is 762 kilograms for the substitution of bricks, and 278 kilograms for the substitution of concrete blocks. The overall environmental impact of Scenarios 1a and 1b do not show a significant difference when bricks are the substituted product under Scenario 1b. On the other hand, when concrete blocks are replaced, the net impact of Scenario 1b is 2.5 times less than that of Scenario 1a. Nevertheless, all described sub-scenarios are credited with the GWP impact category.

The diagonally striped column in Figure 5.6 indicates the avoided environmental burden under Scenario 2a. This avoided emission of 3591 kilograms of CO_2 equivalent resulted from replacing the production of 5 tonnes of CEM II 32.5 with the same amount of blended cement via valorisation of 1 tonne of plasmastone. In this way, 249 kilograms of net CO_2 savings is possible under Scenario 2a. In Scenario 2b, valorisation of 1 tonne of plasmastone produced 22.25 tonnes of blended cement blocks. This amount can replace the same amount of either sand-lime bricks or concrete blocks. The higher environmental burden from the process inputs under Scenario 2b is largely offset by the avoided environmental burden, although the net environmental impact still remains as a burden. In this study, we used a standard cement-to-aggregate ratio of 1:3 for blocks produced, according to EN-196 (1994), without any optimization of the cement-to-aggregate ratio and the aggregate particle-size distribution. In commercial concrete, this optimization is made, which results in much lower cement content and a lower environmental impact. Hence, the replaced impact also is lower when commercial concrete blocks are used as the substituted product under Scenario 2b, which explains that scenario's burden level.

Despite the high mass production, Scenario 2 has less environmental benefits than Scenario 1 mainly because of the higher environmental burden from the OPC used Scenario 2. The environmental profiles of Scenario 1 suggested that all its sub-scenarios are favorable for the valorisation of plasmastone with respect to the environmental impact. In contrast, only Scenario 2a shows environmental friendly conditions in plasmastone valorisation.

As shown in Figure 5.6, the overall environmental impact of scenario 3 is a burden for two main reasons. First, unlike Scenarios 1 and 2, in Scenario 3, 1 tonne of plasmastone produces only 1 tonne of aggregates. Second, the greenhouse gas emission of conventional gravel production is comparatively low, and hence, the avoided environmental burden is also low. Nevertheless, the net burden of Scenario 3 is significantly smaller than that of Scenario 2b.

As defined by Jones et al. (2013), ELFM implies that landfilled waste should be processed using innovative transformation technologies respecting the most stringent social and ecological criteria. For that reason, the best option for valorisation of plasmastone obtained in ELFM activities must be identified. Based on the analysis of this study's scenarios, Scenario 1 is clearly the most favorable treatment option for obtaining the maximum environmental benefit; however, this analysis was based on only one impact category and not all potential environmental impacts were included in this impact assessment method. Another environmental analysis was performed to identify the effect of the valorisation scenarios on other impact categories.

Table 5.3 displays results obtained from an environmental impact assessment using the ReCiPe midpoint method (hierarchist version) with European normalization. Similar to the previous assessment, plus values represent burdens and minus values indicate benefits. The highest burdens and benefits of each impact category are highlighted in the table. Among all scenarios, Scenario 1a is responsible for the highest negative impact on freshwater eutrophication, human toxicity, terrestrial eco toxicity, freshwater eco toxicity, marine eco toxicity, and metal depletion. Scenario 1b (with bricks replacement) creates the highest positive impact on climate change, terrestrial acidification, marine eutrophication, photochemical oxidant formation, and particulate matter formation. The highest burden on the ozone depletion impact category is attributed to Scenarios 1a and 1b, with concrete block replacement, and Scenario 2b (with sand-lime brick replacement) has the maximum positive impact on the same impact category. Ionising radiation and fossil depletion impact categories are influenced negatively, mainly because of Scenario 1b with concrete block substitution. Moreover, Scenario 2b (with sand-lime brick substitution) is responsible for the maximum negative impact on terrestrial acidification, marine eutrophication, and the formation of photochemical oxidant and particulate matter, and for the highest negative impact on ozone depletion, terrestrial eco toxicity, agricultural land occupation, natural land transformation, and fossil depletion impact categories. In addition, Scenario 2b (with concrete block replacement) has a high positive influence on freshwater eutrophication, human toxicity, freshwater eco toxicity, marine eco toxicity, urban land occupation, and metal depletion impact categories. Notably, Scenarios 2a and 3 are not responsible for any of the highest positive or negative impacts on any impact category.

This preliminary analysis suggests that the studied scenarios influence not only the impact category of GWP, but also several other impact categories. Hence, a detailed study is necessary to identify the reasons for the impact distribution.

Table 5.3: Normalised environmental impact of different scenarios on different impact categories -ReCiPe midpoint method with European normalisation (The highest burden and benefit on each impact category are highlighted by bold italic and underlined text respectively)

	Scenario						
Impact category	1a	1b-bricks	1b-concrete blocks	2a	2b-sand lime bricks	2b-concrete blocks	3
Climate change	-0.0667	<u>**-0.0680**</u>	-0.0248	-0.0228	0.0449	***0.0618***	0.0001
Ozone depletion	-0.0003	-0.0019	***0.0006***	-0.0001	<u>**-0.0068**</u>	***0,0006***	0.0001
Terrestrial acidification	-0.0194	<u>**-0.0406**</u>	-0.0069	-0.0111	***0.0160***	-0.0111	-0.0002
Freshwater eutrophication	***0.2308***	-0.0156	0.1851	-0.0135	-0.2420	<u>**-0.4169**</u>	-0.0022
Marine eutrophication	-0.0015	<u>**-0.0052**</u>	0.0001	-0.0018	***0.0046***	-0.0039	-0.0001
Human toxicity	***0.1395***	0.0316	0.1067	-0.0254	-0,1738	<u>**-0,3653**</u>	-0.0016
Photochemical oxidant formation	-0.0224	<u>**-0.0415**</u>	-0.0128	-0.0097	***0.0151***	-0.0080	-0.0003
Particulate matter formation	-0.0210	<u>**-0.0365**</u>	-0.0123	-0.0108	***0.0012***	-0.0260	-0.0004
Terrestrial ecotoxicity	***0.0008***	-0.0010	0.0007	-0.0006	<u>**-0.0215**</u>	-0.0043	0.0000
Freshwater ecotoxicity	***0.1433***	0.0064	0.0860	-0.0133	-0.3517	<u>**-0.4392**</u>	-0.0018
Marine ecotoxicity	***0.1898***	0.0088	0.1118	-0.0195	-0.5663	<u>**-0.5922**</u>	-0.0026
Ionising radiation	0.0003	0.0055	***0.0082***	0.0039	0.0009	0.0070	0.0000
Agricultural land occupation	0.0000	-0.0045	-0.0046	-0.0004	<u>**-0.0521**</u>	-0.0269	0.0000
Urban land occupation	-0.0010	-0.0042	-0.0056	-0.0020	-0.0295	<u>**-0.0332**</u>	-0.0011
Natural land transformation	-0.0220	-0.5823	-0.1034	-0.0931	<u>**-2.2125**</u>	-0.9278	-0.0565
Water depletion	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
Metal depletion	***0.0158***	-0.0037	-0.0441	-0.0042	-0.3073	<u>**-0.3403**</u>	-0.0008
Fossil depletion	-0.0104	-0.0960	***0.0060***	-0.0129	<u>**-0.1725**</u>	-0.0120	0.0000

5.6.2. Economic evaluation of valorisation of plasmastone

Using the cash-flow model, we investigated the sensitivity of NPV to a wide range of parameters, including the amount of different input materials, the investment and operational costs of the different valorisation scenarios, and the revenues of different products. Table 5.4 illustrates the parameters' influence on the variation in NPV obtained from Monte Carlo simulations. Positive contributions indicate that an increase in the parameter yields an increase in the economic indicator. Negative contributions imply the opposite situation. We calculated the maximum and minimum values of the considered ranges as follows: For the material prices, we decided the price ranges based on published literature (ICIS 2008, ICIS 2010, Gardiner&Theobald 2012) and communication with experts in the relevant industries. 10 percent and 50 percent from the investment cost of OPC industry was considered to set the minimum and maximum values of the investment costs of the scenarios (The respective calculation is explained in detail in Section 5.5). A 10 percent margin from the average value was set to the maximums and minimums of the quantities of process inputs and product prices.

Table 5.4: Most sensitive parameters to the NPV

Parameter	Minimum value	Maximum value	Contribution to variance of NPV (%)
Scenario 1a			
Price of Na_2SiO_3 (€/t)	180	630	45.6 (-)
Price of NaOH (€/t)	250	300	0.2 (-)
Amount of Na_2SiO_3 (t/t plasmastone)	0.07	0.086	0.5 (-)
Amount of NaOH (t/t plasmastone)	0.058	0.07	0.1 (-)
Selling price of inorganic polymer cement (€/t)	135	165	43.9 (+)
Investment cost (€/t plasmastone)	3	8	9.7 (-)
Scenario 1b			
Price of Na_2SiO3 (€/t)	180	630	14.4 (-)
Price of NaOH (€/t)	250	300	0.1 (-)
Price of aggregates (€/t)	5	15	11.3 (-)
Amount of Na_2SiO_3 (t/t plasmastone)	0.07	0.086	0.6 (-)
Amount of NaOH (t/t plasmastone)	0.058	0.07	0.2 (-)
Amount of aggregates (t/t plasmastone)	2.7	3.3	17.2 (+)
Selling price of inorganic polymer block (€/t)	63	77	42.7 (+)
Investment cost (€/t plasmastone)	6	16	13.3 (-)
Scenario 2a			
Price of OPC (€/t)	70	84	13.7 (-)
Amount of OPC (t/t plasmastone)	3.6	4.4	1.0 (+)
Selling price of blended cement (€/t)	90	110	52.3 (+)
Investment cost (€/t plasmastone)	7	32	32.9 (-)
Scenario 2b			
Price of OPC (€/t)	70	84	2.9 (-)
Price of aggregates (€/t)	5	15	18.4 (-)
Amount of OPC (t/t plasmastone)	3.6	4.4	0.4 (+)
Amount of aggregates (t/t plasmastone)	13.5	16.5	11.7 (+)
Selling price of blended cement block (€/t)	45	55	43.9 (+)
Investment cost (€/t plasmastone)	14	64	22.6 (-)
Scenario 3			
Selling price of aggregates (€/t)	3	7	7.6 (+)
Investment cost (€/t plasmastone)	1	5	90.7 (-)
Operational+maintenance+othercosts (€/t plasmastone)	1	3	1.8 (-)

As described in the LCA study, the quality of the products of the valorisation scenarios essentially contributes to the net environmental and economic impact, which the substituted products in each scenario explained. In the LCC study, the selling price of the product measured this contribution, which logically results in a higher NPV when selling prices are higher. Table 5.4 shows that the selling price of the products is of key importance for the economic feasibility of the treatment scenarios. In addition, the investment costs, and operational costs such as materials prices (Na_2SiO_3, aggregates and OPC) also have an important impact on NPV. Nevertheless, the range definitions mentioned in Table 5.4 affects the final impact of the different parameters on the NPV.

As shown in Table 5.4, the input materials with the highest economic impact are Na_2SiO_3 in Scenarios 1a and 1b and OPC in Scenario 2a. Remarkably, these materials also cause the highest environmental impact (Figure 5.5). Although the amounts of aggregates used in Scenarios 1b and 2b do not show a significant environmental impact in the LCA study, their price makes 11 percent and 18 percent contribution to NPV in these respective scenarios. When calculating the net environmental impact, the total amount of products has a very important influence. In Scenario 1b and 2b, this influence is largely caused by the amount of aggregates. Similarly, in the economic evaluation, the amount of aggregates has a very significant contribution on the variation of the NPV.

Varying the amount of aggregates yields an impact in two different ways. On one hand, it changes the quality of the product, which in turn leads to changes in selling price. On the other hand, it generates variations in the quantity of product that can be generated from 1 tonne of plasmastone. Eventually, both these changes would influence the NPV. Figure 5.7 illustrates the variation in NPV for different selling prices of the products for three different amounts of aggregates in Scenarios 1b and 2b. The different lines represent the variation of NPV according to the changes in selling prices for different levels of aggregates. A 10 percent increase in the selling price leads to a 18 to 20 percent gain in NPV under Scenario 1b, and of 22 to 25 percent under Scenario 2b. A 10 percent increase in the amount of aggregates results in increments of 12 percent and 13 percent in NPV for Scenarios 1b and 2b, respectively.

To make the valorisation routes more profitable, it is important to know the product's lowest possible selling price. Equation 5.1 (Section 5.5) provided minimum selling prices of €77/t for inorganic polymer cement (Scenario 1a), €33/t for inorganic polymer block (Scenario 1b), €82/t for blended cement (Scenario 2a), €30/t for blended cement block (Scenario 2b), and €12/t for aggregates (Scenario 3). Because these values lay below the existing market prices of OPC and OPC-based concrete blocks, we could assume these products would be economical alternatives to OPC-based products.

In this study, we omitted the cost of plasmastone because plasma gasification and plasmastone valorisation take place at the same premises and are components of one project. If plasmastone is purchased in order to produce suggested products, it would be important to know the maximum purchase price to keep the project at the lowest profit margin. Using equation 5.2, (Section 2.4), we calculated the maximum purchase price for Scenarios 1a, 1b, 2a, and 2b to be 83, 152, 91, and 468 €/t plasmastone, respectively. However, these numbers are valid only with the data used in Tables 1 and 2, and depend not only on the selling price of the products, but also the quantity of the products. If the company or the authority responsible for the ELFM project wants to sell the produced plasmastone to an outside company to produce suggested building materials, then these values can be considered as maximum selling prices.

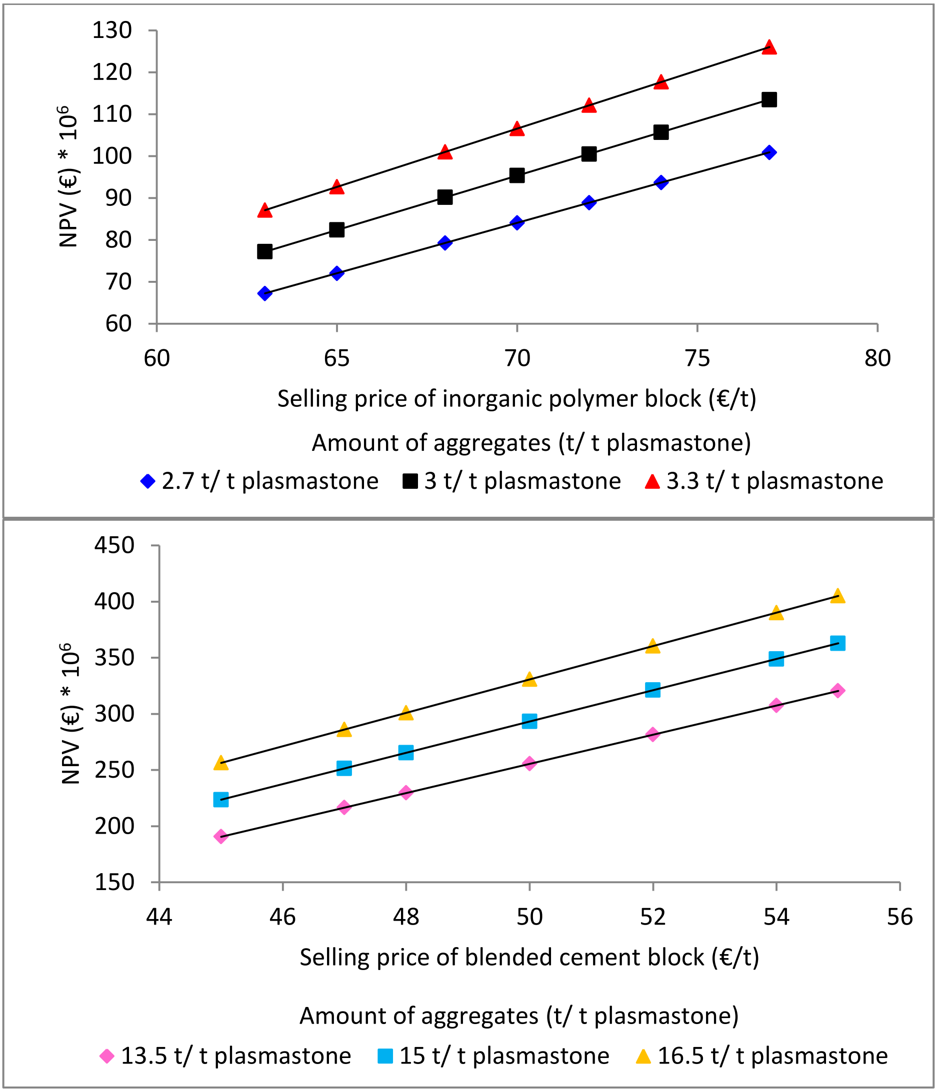


Figure 5.7: The impact of variations in selling prices of the products on NPV in scenario 1b (top) and scenario 2b (bottom)

A trade-off analysis was performed to find the relationship between environmental and economic performances of the scenarios. For Scenario 1b, the same product price (€70/t) was used for both types of substituted products (bricks and concrete blocks). The situation was the same for Scenario 2b (product price was €50/t). Figure 5.8 shows the relationship between economic and environmental impact of different valorisation scenarios. The total environmental impact in terms of kilograms of CO_2 equivalent was calculated for the hypothetical treatment plant used in the economic analysis (100,000 tonnes of plasmastone per year of treatment capacity and a 20-year lifetime). In fact, the positive values of NPV imply economic profits, while the negative values of environmental impact indicate environmental benefits; therefore, Scenarios 1a, 1b, and 2a are viable both environmentally and economically. Scenario 2b achieves the highest economic benefit and the lowest environmental benefit. Scenario 1b shows the highest environmental benefit when bricks are chosen as the substituted product, a benefit that is slightly higher than that of Scenario 1a. In addition, Scenario 1a marks the lowest NPV among the economically and environmentally viable scenarios. Scenario 3 gives neither economic nor environmental profit; however, the environmental burden caused by Scenario 3 is considerably lower than that of Scenario 2b. The arrow lines in Figure 5.8 illustrate the trade-off line of plasmastone valorisation scenarios, or a way to give up economic benefit to obtain environmental benefit. The trade-off line starts from Scenario 2b with concrete blocks as substituted product, passes scenario 2b with sand-lime brick as substituted product, and then reaches Scenario 1b with concrete blocks and bricks as substituted products, respectively. These results suggest that Scenario 1, valorisation of plasmastone via production of inorganic polymer cement/block is the most worthwhile candidate for yielding environmental and economic profits in ELFM.

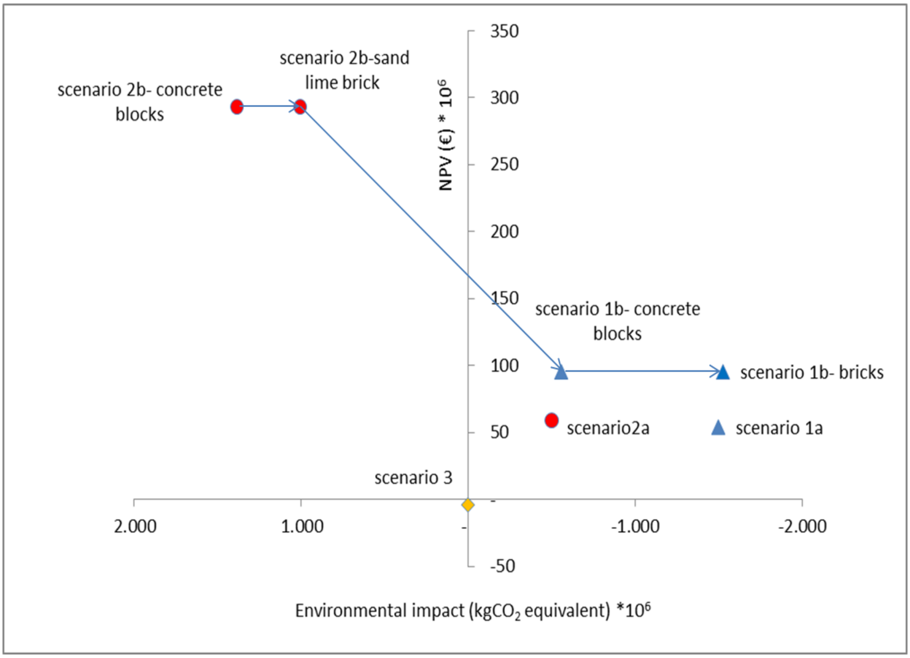


Figure 5.8: Trade off analysis of plasmastone valorisation scenarios

5.7. Conclusions

This chapter includes the results of an environmental and economic evaluation performed to identify the best scenarios for valorisation of plasmastone in the framework of the novel concept of ELFM. Based on the LCA study, we found that, despite the high mass production of products, valorisation of plasmastone via production of blended cement/block yields fewer environmental benefits compared with valorisation of plasmastone via the production of inorganic polymer cement/block. In addition, the LCA study further shows that Na_2SiO_3, NaOH, and OPC inputs produce the greatest environmental impact in the scenarios. The study illustrates that the impact of some parameters are negligible environmentally, but become key economic drivers, and vice versa. Examples include the negligible influence of aggregates to GWP and its substantial contribution to the NPV. Both the LCA and LCC studies reveal that the quantity of products obtained from a certain amount of plasmastone are also important when calculating the net impact of the scenarios. Apart from the quantity, the product quality also creates an essential impact on environmental and economic performance of plasmastone valorisation. A careful choice of the quantity and quality of input materials is needed to obtain a high-quality product and the product quality determines the avoided environmental burden and the selling prices of the products. In the trade-off analysis, we conclude that decisions regarding of the appropriate valorisation routes should be made cautiously because when economic profits are at the maximum, environmental benefits become the lowest in some scenarios. Nevertheless, valorisation of plasmastone via (i) production of inorganic polymer cement, (ii) production of inorganic polymer block, and (iii) production of blended cement are the most viable scenarios both environmentally and economically. However, the environmental impact discussed in this chapter is based mainly on one impact category (GWP), although a preliminary analysis suggests that the studied scenarios would influence several other impact categories as well. Hence, potential exists for further research to thoroughly examine other impact categories.

Chapter 6 : Environmental and economic performance of plasma gasification in Enhanced Landfill Mining

This chapter is based on

Danthurebandara, M., Vanderreydt, I., Van Acker, K.. The environmental performance of plasma gasification within the framework of Enhanced Landfill Mining: A life cycle assessment study. 5th International Symposium on Energy from Biomass and Waste –Venice 2014. 17-20 November 2014. Venice, Italy.

And

Danthurebandara, M., Van Passel, S., Van Acker, K.. Environmental and economic performance of plasma gasification in Enhanced Landfill Mining. (Revised version under review)

Abstract

This chapter describes an environmental and economic assessment of plasma gasification, one of the viable candidates for the valorisation of refuse derived fuel from ELFM. The study is based on life cycle assessment and life cycle costing. Plasma gasification is benchmarked against conventional incineration, and the study indicates that the process could have significant impact on climate change, human toxicity, particulate matter formation, metal depletion and fossil depletion. Flue gas emission, oxygen usage and disposal of residues (plasmastone) are the major environmental burdens, while electricity production and metal recovery represent the major benefits. Reductions in burdens and improvements in benefits are found when the plasmastone is valorised in building materials instead of landfilling. The study indicates that the overall environmental performance of plasma gasification is clearly better than incineration. The LCC study confirms a clear trade-off between the environmental and economic performance of the discussed scenarios. Net electrical efficiency and investment cost of the plasma gasification process and the selling price of the products are the major economic drivers.

6.1. Introduction

Refuse derived fuel (RDF) is one of the important intermediate products of Enhanced Landfill Mining (ELFM), which can be further valorised with the objective of recovering energy. Incineration, pyrolysis, gasification and plasma technologies are the main thermochemical treatment technologies for MSW and/or RDF. Selecting a suitable thermal treatment technology for ELFM, should be done according to their combined energy and material valorisation capacity as the major objective of ELFM is to realise ‘zero waste’.

MSW incinerators offer a large potential source of heat and electricity, especially when combined heat and power (CHP) is applied (Limerick 2005, BREF 2006, BREF

2010). Solid waste incinerators can obtain a significant waste reduction of about 90 percent, but because of the risk of leaching heavy metals, a substantial volume of residues must be disposed of mostly in landfills (Cheeseman et al. 2003) and cannot be recuperated as material. These facts prove that incinerators have considerable Waste to Energy (WtE) potential, but not a promising Waste to Material (WtM) potential.

Pyrolysis produces a combustible gas that can be used in steam turbines, gas turbines, gas engines and even in fuel cells, but is feasible only for specific homogeneous feed materials, such as tires and electronic waste, and does not offer a complete alternative to MSW incineration (Bosmans et al. 2013). Pyrolysis also has the major environmental disadvantage of requiring disposal of solid residues in landfills (Young 2010).

Gasification has several advantages over traditional combustion of MSW: Only a fraction of the stoichiometric amount of oxygen necessary for combustion is required, and the formation of dioxins, SO_2 and NO_x is limited, which results in smaller, less expensive gas cleaning equipment. The syngas generated by gasification can be used in combined cycle turbines, gas engines and potentially in fuel cells for electricity and heat generation, or as a chemical compound to produce methanol. Gasification also offers WtM potential if a slagging gasifier is used (Hirschfelder and Olschar 2010).

Although the application of plasma-based systems for waste management is a relatively new concept, many studies reveal that plasma technology is an attractive waste treatment option in ELFM compared with other processes. Plasma-based systems offer flexibility, fast process control and more options in process chemistry, including the possibility of generating valuable products (Ray et al. 2012, Bosmans et al. 2013, Taylor et al. 2013).

Previous chapters described the advantages and applications of plasma gasification in ELFM. In chapter 4, we concluded that among the ELFM processes, plasma gasification has the greatest influence on environmental and economic factors, and suggested that a detailed assessment of plasma gasification is needed to identify possible improvements in the technology. Therefore, in this chapter, we discuss a comprehensive study of life cycle analysis (LCA) and life cycle costing (LCC) with the results of previous chapter 5, to address the environmental and economic performance of plasma gasification in ELFM. In addition, plasma gasification is benchmarked against incineration, a commonly used thermal treatment in municipal solid waste (MSW) processing.

6.2. Process description and system boundaries

Plasma is known as the fourth state of matter. The presence of charged gaseous species make the plasma highly reactive and cause it to behave significantly differently from other gases, solids or liquids. Plasma is generated when gaseous molecules are forced into high-energy collisions with charged electrons, which generated charged particles. The energy required to create a plasma can be thermal or carried by either an electric current or electromagnetic radiations (Bosmans et al. 2013). More details on main groups of plasmas can be found in Huang and Tang (2007) and Tendero et al. (2006).

Plasma offers a number of advantages to waste treatment processes (Heberlein and Murphy 2008). The high-energy densities and temperatures that can be achieved in plasma processes enable high heat and reactant transfer rates, which can reduce the size of the installation for a given waste throughput and can melt materials at high temperature, increasing the overall waste volume reduction. Plasma-based systems also have the important advantage of being able to crack tars and chars, and therefore, the efficiency of conversion to high-quality syngas is much higher compared with non-plasma systems (Spooren et al. 2013). Since electricity is used as the energy source, heat generation is decoupled from process chemistry, which increases process controllability and flexibility (Bosmans et al. 2013).

Heberlein and Murphy (2008) described the categories of plasma technologies for waste treatment: plasma pyrolysis, plasma gasification, plasma compaction and vitrification of solid wastes, and the combinations of these three. Plasma pyrolysis installations treat polymer, medical waste and low-level radioactive waste (Guddeti et al. 2000, Nema and Ganeshprasad 2002, HTTC 2009); however, no information is available on industrial plasma pyrolysis facilities for processing MSW or RDF, the type of solid waste that is the focus of this study (Bosmans et al. 2013). Hence, Bosmans et al. (2013) noted that plasma gasification and vitrification is the preferred plasma-based technology for solid waste treatment.

More often plasma gasification is combined with vitrification to treat solid waste containing high amounts of organics. Plasma gasification systems may be either single- or two-stage. In the single-stage design, waste is directly treated with plasma jets; the two-stage design adds plasma cleaning of the produced synthesis gas. High temperatures are reached in plasma gasification, forming high-energy synthesis gas consisting mainly of hydrogen and carbon monoxide. The energy contained in plasma allows low-energy fuels to be treated, such as household waste and industrial waste, which often cannot sustain their own gasification without additional fuel. The resulting synthesis gas is cleaner than conventional gasification process because tar, char and dioxins are broken down. In addition, any inorganic components (glass, metals, silicates) are melted and converted into a dense, inert, non-leaching vitrified slag. The synthesis gas produced can be used for electricity/heat generation and for conversion to second-generation liquid fuel.

Residues from the synthesis gas cleaning process (metals, fly ash) can be recycled internally and captured in the slag, which is vitrified to avoid leaching risks. This vitrified slag (plasmastone) has great potential for rather diverse applications, mainly in the construction materials industry (Jones et al. 2013, Spooren et al. 2013). More details regarding plasmastone valorisation can be found in Chapter 5.

Figure 6.1 shows how plasma gasification fits in an ELFM system. The RDF fraction obtained by the separation process is directed to plasma gasification. Although the resulted syngas can be valorised in many ways as described, this study examined only electricity production. Plasma gasification offers an intrinsic advantage because of metal recuperation, as shown in Figure 6.1. The input RDF contains a minor fraction of metals that cannot be recovered during the separation process and that are melted during the thermal treatment process. In plasma gasification, metals can be separated from the plasmastone when the molten material is discharged into a quenching bath. Produced plasmastone is valorised to achieve the zero-waste goal of ELFM. This study focused only on the subsystem of thermal treatment with ELFM, as highlighted by the dotted line in Figure 6.1.

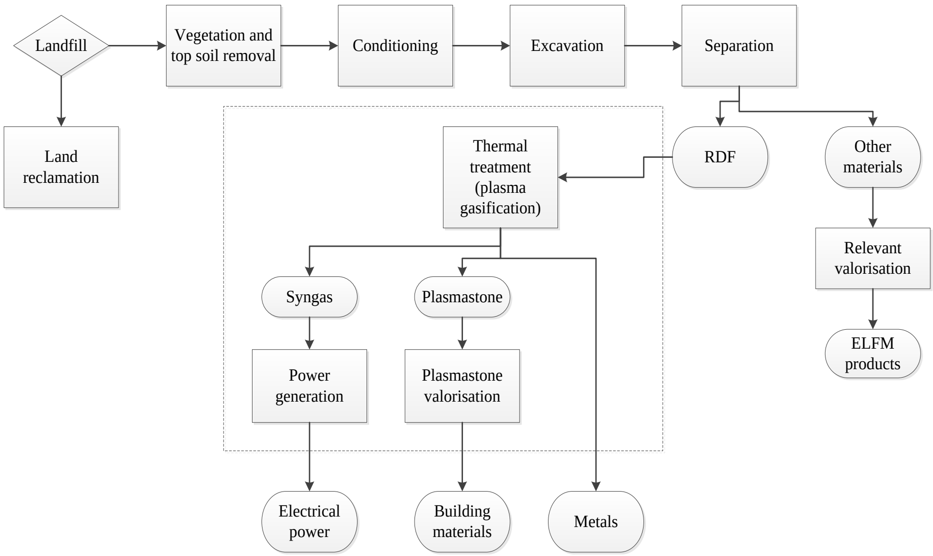


Figure 6.1: Interactions of ELFM and the system studied (indicated by the dotted line)

6.3. LCA and LCC methodology

The goal of this LCA study was to evaluate the environmental impact of plasma gasification in ELFM. A comparative LCA model was developed using the relevant building blocks of the model described in chapter 3, according to the international Standard for LCA (ISO 14040, 14044). Plasma gasification was coupled with plasmastone valorisation to achieve ELFM's unique objective: valorising not only landfill waste, but also waste and by-products generated within various ELFM processes. The system function was to reduce environmental impact and realise potential energy and material resource recovery from landfills. The reference flow was defined as the valorisation of 1 tonne of RDF obtained from landfilled waste. Also, plasma gasification was benchmarked against incineration, a commonly used thermal treatment method. The simplified comparative LCA model is shown in Figure 6.2.

Eight scenarios were proposed depending on the plasmastone valorisation method. The first scenario consisted of no valorisation, and the plasmastone is landfilled immediately after the plasma gasification process. Based on the applications of plasmastone described in chapter 5, five additional scenarios were designed in which plasmastone is used in aggregate production, inorganic polymer cement production, inorganic polymer block production, blended cement production and blended cement block production. In addition, two scenarios were designed to include incineration, one that valorised bottom ash as aggregate in the construction industry and one that did not. Table 6.1 summarizes the scenarios analysed in this study.

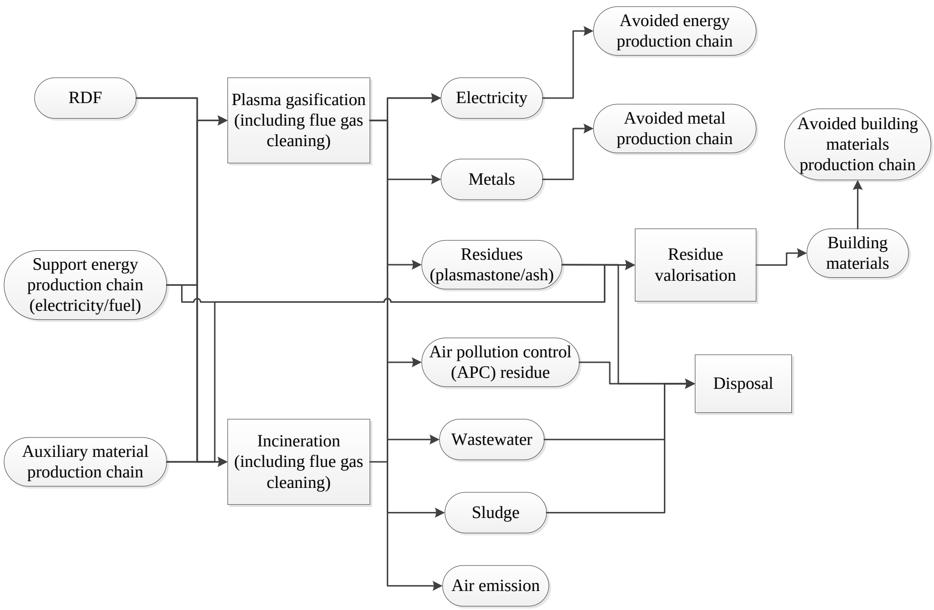


Figure 6.2: Overview of the LCA model

Table 6.1: Summary of the scenarios

Scenario	Description
Scenario 1	Plasma gasification with landfilling of plasmastone
Scenario 2	Incineration with landfilling of bottom ash
Scenario 3	Incineration with aggregate production out of bottom ash
Scenario 4	Plasma gasification with aggregate production out of plasmastone
Scenario 5	Plasma gasification with inorganic polymer cement production out of plasmastone
Scenario 6	Plasma gasification with inorganic polymer block production out of plasmastone
Scenario 7	Plasma gasification with blended cement production out of plasmastone
Scenario 8	Plasma gasification with blended cement block production out of plasmastone

An input-output inventory was made for the plasma gasification process. The input data included the RDF to be processed, energy consumption and auxiliary materials; output data included emissions (to air, water, soil), wastes and products. Auxiliary materials for plasmastone valorisation are described in Machiels et al. (2014), Iacobescu et al. (2013) and previous chapter 5. Transportation of recovered RDF was not considered in the model because we assumed all processing plants to be situated on the landfill premises. The environmental impact of landfill and processing plant personnel were not considered, and wastewater generated from all processes was directed to relevant treatment methods. we also assume the produced heat to be used in the process itself (for example, for boilers). The qualities of various other products were:

- Metals recovered have the quality that enables substituting corresponding scrap metals;
- The produced electricity replaces the Belgian electricity mix, which includes 53 percent nuclear, 40 percent conventional thermal, 2 percent hydro and 3 percent wind energy (Eurostat);
- Aggregates produced in scenarios 3 and 4 have the quality of gravel that can be used in construction activities (chapter 5);
- Produced inorganic polymer cement in scenario 5 has the quality of OPC, strength class CEM I 52.5 (chapter 5);
- Produced blended cement in scenario 7 has the quality of OPC, strength class CEM II, 32.5 (chapter 5);
- Produced inorganic polymer blocks and blended cement blocks in scenarios 6 and 8 have the quality of commercially available concrete blocks (chapter 5).

Table 6.2 shows inflow-outflow energy and solid waste from the processes considered in the life-cycle inventory (Troschinetz and Mihelcic). The start-up energy required for the system was taken from Belgium's average country electricity grid. The data for plasma gasification were based mainly on a pilot experiment performed for RDF obtained from the case study landfill of the first comprehensive ELFM project in Belgium (Tielemans and Laevers 2010). Indaver (2012) and BREF (2006, 2010) provided necessary data for the incineration process. The inventory of auxiliary materials required, mainly for flue gas cleaning, are illustrated in Table 6.3. Note that plasma gasification requires a considerable amount of pure oxygen, in contrast to incineration. All emission data (see Table 6.4) used in plasma gasification was obtained from the previously mentioned pilot experiment; incineration emission data was obtained from Indaver (2012) and Zaman (2013). Because carbon dioxide is a combination of biogenic and fossil carbon, the proportion of each had to be identified to calculate the environmental burden. Although the Ecoinvent database showed that MSW contributes 60.5 percent of biogenic carbon, a 47 percent share of biogenic carbon dioxide was used in this study because that is the fixed share used in Flanders, Belgium (Van Passel et al. 2013).

Table 6.2: Energy and residues data used in LCI

Parameter	Scenario 1 (plasma gasification)	Scenario 2 (incineration)
Start-up energy (kWh/t RDF)	269[a,b]	78[f]
Calorific value of RDF (MJ/kg RDF)	20[c]	20[c]
Net electrical efficiency (%)	27[a,b,d]	22[d,g]
Solid residue generation (apart from APC residues) (t/t RDF)	0.17[e] (valorisation of this fraction is considered in scenario 4-8)	0.228[f] (this fraction is considered to be landfilled in scenario 2 and to be valorised in scenario 3)
APC residues (t/t RDF)	0.024[e]	0.043[f]
Metal recuperation (t/t RDF)	0.01[e]	-

[a] Chapmen et al. (2010),[b] Bosmans et al. (2013) [c] industrial reference (Group Machiels), [d]UCL (2014), [e] industrial reference (Advance Plasma Power) [f]Indaver (2012), [g]BREF (2006, 2010),

Table 6.3: Auxiliary material data used in LCI

Material	Scenario 1 (plasma gasification)	Scenario 2 (incineration)
Oxygen (t/t RDF)	0.55[a]	-
$NaHCO_3$ (kg/t RDF)	4[a]	-
Activated carbon (kg/t RDF)	0.2[a]	0.5[b]
NaOH (kg/t RDF)	0.8[a]	-
H_2O_2 (kg/t RDF)	0.4[a]	-
Urea (kg/t RDF)	1.2[a]	3.5[b]
Limestone (kg/t RDF)	-	6.7[b]
Quicklime (kg/t RDF)	-	4.4[b]

[a] industrial reference (Advance Plasma Power), [b]Indaver (2012)

Table 6.4: Emission to air used in LCI

Substance	Scenario 1 (plasma gasification)	Scenario 2 (incineration)
Carbon dioxide (kg/t RDF)	1465[a]	1678[b]
biogenic	689[c]	789[c]
fossil	776[c]	889[c]
Carbon monoxide (kg/t RDF)	0.02[a]	0.09[b]
Particulates (kg/t RDF)	0.2[a]	0.014[b]
Nitrogen oxides (kg/t RDF)	0.43[a]	1.49[b]
Sulphur dioxide (kg/t RDF)	0.08[a]	0.019[b]
Hydrogen chloride (kg/t RDF)	0.02[a]	0.003[b]
Dioxins (kg/t RDF)	-	8×10^{-8}[b]
Mercury (kg/t RDF)	-	1.6×10^{-6}[b]
Heavy metals (kg/t RDF)	-	0.052[b]

[a] industrial reference (Advance Plasma Power), [b]Indaver (2012), [c] calculated by using 47% biogenic fraction

We selected the ReCiPe endpoint method (Hierarchist version, H/A) as the impact assessment method, with following categories: (i) climate change on human health, (ii) climate change on ecosystems, (iii) ozone depletion, (iv) terrestrial acidification, (v) freshwater eutrophication, (vi) human toxicity, (vii) photochemical oxidant formation, (viii) particulate matter formation, (ix) terrestrial ecotoxicity, (x) freshwater ecotoxicity, (xi) ionising radiation, (xii) agricultural land occupation, (xiii) urban land occupation, (xiv) natural land transformation, (xv) metal depletion and (xvi) fossil fuel depletion (Goedkoop et al. 2013).

To perform LCC for the selected scenarios in Table 6.1, we considered a hypothetical processing plant with a processing capacity of 100,000 tonnes RDF/year and a 20-year lifetime. we used the cash-flow model described in chapter 3, with necessary costs and revenues associated with the scenarios. The cash-flow model was extended for 20 years with a 15 percent discount factor. For the incineration process, we used 40 €/t RDF in investment costs and 60 €/t RDF in operational cost (Ducharme 2010); all the other costs and revenues were similar to those applied in Chapters 4 and 5. The net present value (NPV) was used as the major economic indicator. Monte Carlo sensitivity analysis is also performed as described in the previous chapters.

6.4. Results and discussion

6.4.1. Environmental performance

Figure 6.3 indicates the characterisation data for the study's basic scenario: plasma gasification of 1 tonne of RDF with landfilling of plasmastone. To get a better view of the relative contributions, the total environmental impact for each scenario was set at 100 percent. As explained in previous chapters, the environmental burdens were expressed as positive values and the benefits as negative values.

As the figure indicates, electricity production yields an environmental benefit in all impact categories. Next to that, a significant benefit can be seen from metal recovery on several impact categories, especially metal depletion. Major burdens are caused by flue gas emission, oxygen usage and disposal of solid residues. As a result, flue gas emission dominates the climate change impact category. The graph indicates that half of the burden in this impact category could be compensated by the avoided burden due to electricity production. However, the impact of the thermal treatment process was derived from a comparison with the Belgian electricity generation mix. Hence, the replaced impact is low as the environmental impact of nuclear energy production is lower than other energy production methods. Chapter 4 discusses how this avoided burden varies when different electricity mixes and the Belgian marginal electric energy source natural gas are used as the substituted product.

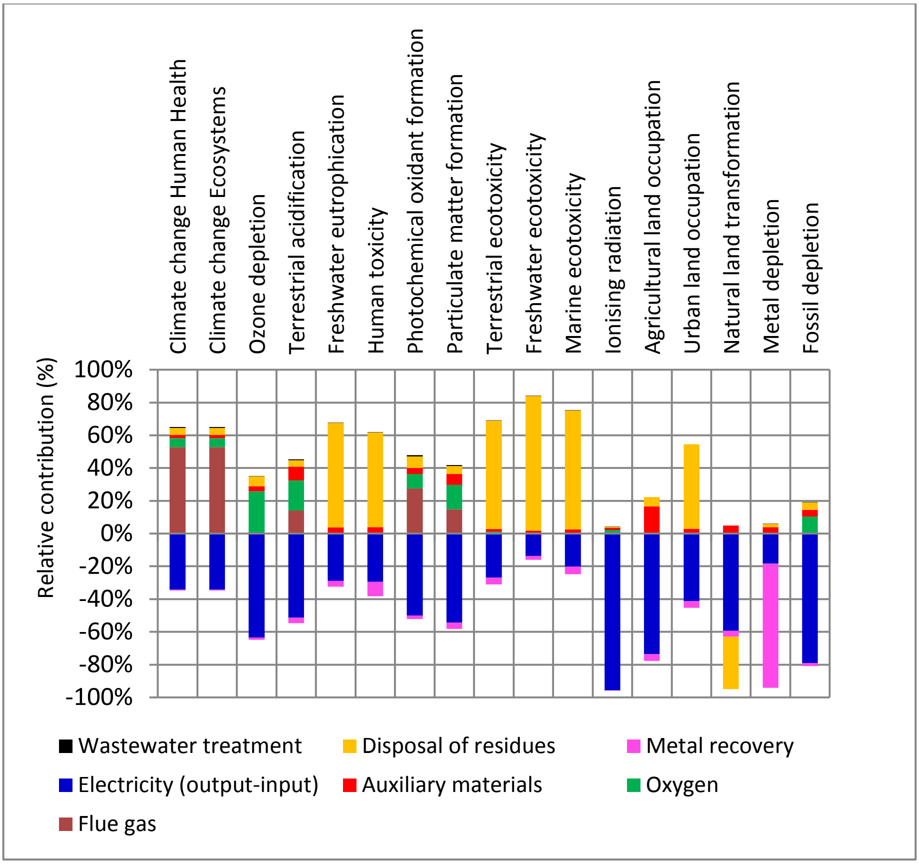


Figure 6.3: Environmental profile of scenario 1 (characterisation results)

The process can be extended through the use of flue gas in local horticulture, because flue gas contains higher amounts of CO_2 and lower temperature waste heat. CO_2 acts as a plant fertiliser, while the residual heat warms greenhouses, avoiding the use of primary fossil fuels; therefore, the burden of flue gas could be mitigated (Jones et al. 2013). The burden of oxygen mainly results because of the energy source used in the entire production process. Using renewable energy sources for oxygen production can further reduce the impact of plasma gasification. In addition, investigating the possibility of using air instead of pure oxygen in plasma gasification is worthwhile from an environmental point of view.

Disposal of solid residues contributes considerably to the burden yield in the impact categories of freshwater eutrophication, human toxicity, terrestrial ecotoxicity, freshwater ecotoxicity, marine ecotoxicity and urban land occupation. Scenario 1 assumed that solid residues (plasmastone) are landfilled, and therefore, higher volumes of inert residues impose a higher burden on the environment. However, various studies have shown that inert residues could be used as construction material (Iacobescu et al. 2013, Pontikes et al. 2013, Spooren et al. 2013, Machiels et al. 2014), thus eliminating the burden of solid residue disposal. This possibility is discussed in the other scenarios.

Normalisation was performed on a European level in order to make the impacts in different categories comparable with each other (PRéConsultants 2010). Figure 6.4 presents the normalised comparison of the environmental profile of valorisation of 1 tonne of RDF (scenario 1). In the same graph, the normalised results of the incineration process (scenario 2) are presented for better comparison with the plasma gasification process.

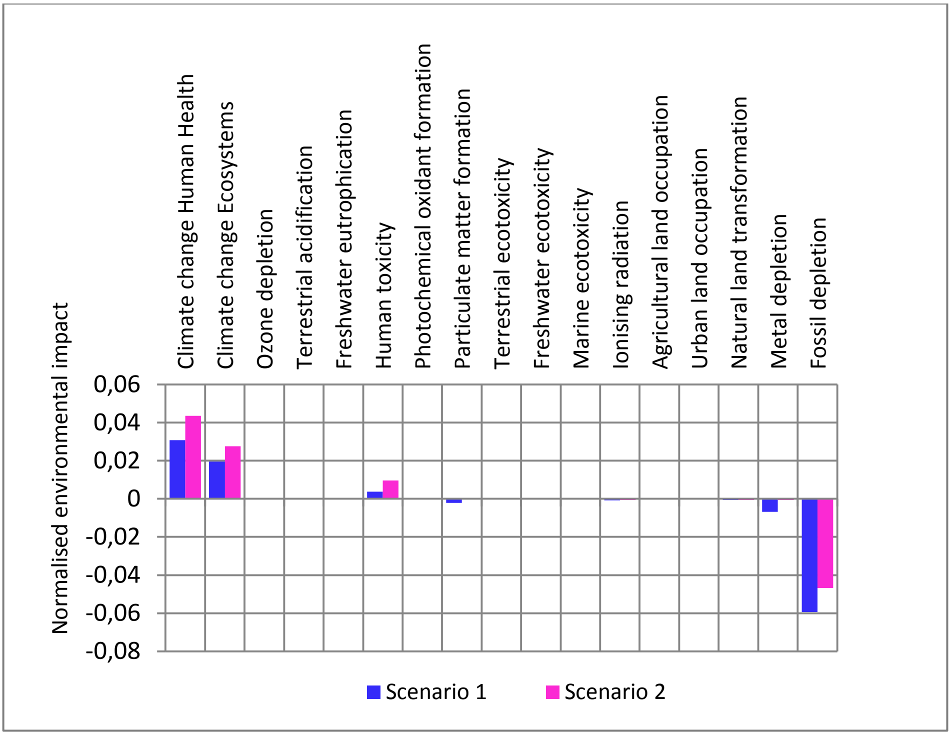


Figure 6.4: Normalised environmental profile of scenario 1 and 2

Normalisation shows that for both scenarios the contribution to climate change and fossil depletion is very important. In addition, the contribution to human toxicity also is substantial. The impact on metal depletion impact category is significant only for plasma gasification because the process offers intrinsic advantage for the recovery of minor amounts of metals in the RDF. Particulate matter formation is also slightly significant only in plasma gasification. Important to notice is the insignificance of other impact categories for both plasma gasification and incineration.

At first glance, the impact of plasma gasification on climate change and human toxicity is remarkably lower than that of incineration. On the other hand, plasma gasification creates a higher benefit in the fossil depletion impact category compared with incineration, mainly because of the higher electrical efficiency of plasma gasification (27 percent) compared with that of incineration (22 percent). The treatment principle of the two technologies also is a reason for these differences. Incineration is done under uncontrolled airflow that produces higher CO_2 emissions into the atmosphere, whereas plasma gasification is performed under a controlled volume of oxygen, and therefore, CO_2 emissions are comparatively lower, as shown in Table 6.4. According to Figure 6.3, disposal of solid residues is the main contributor to human toxicity impact category. Emissions to air and groundwater during the disposal activity cause this burden. Because plasma gasification generates fewer solid residues than incineration, the respective burden is also less.

To investigate the sensitivity of the environmental profile of plasma gasification coupled with various plasmastone valorisation techniques, we replaced landfilling of plasmastone in scenario 1 with higher added-value applications, as shown in Table 6.1. Figure 6.5 illustrates the comparative environmental profiles of the scenarios applying various plasmastone valorisation methods. In addition, the impact of incineration, with valorisation of bottom ash as aggregates (scenario 3), is also shown. Only the most significant impact categories were included in the figure.

Figure 6.5 indicates that the environmental burden of plasma gasification on the climate change impact category varied considerably when plasmastone is valorised. The impact on both climate change on human health and on ecosystems decreased by 12 percent, 41 percent, 23 percent and 24 percent respectively when plasmastone is used in aggregate production, inorganic polymer cement production, inorganic polymer block production and blended cement production. These decrements are due to replacement of conventional gravel production and OPC-based products, which have higher greenhouse gas emissions.

In contrast, the use of plasmastone in blended cement block production results in a 12 percent increased burden in the same impact categories. As previously explained, in both inorganic polymer cement/block production and blended cement/block production processes, traditional Portland cement/concrete with a heavy CO_2 burden is replaced. However, in blended cement production, Portland cement is used as an input material, which explains the difference in environmental impact between the two valorisation techniques. The impact of these valorisation technologies on global warming potential (GWP) can be found in detail in chapter 5.

Importantly, the burden on human toxicity impact category become a benefit when the above higher added value valorisation applications are introduced to the process. Moreover, benefits in the categories of particulate matter formation, metal depletion and fossil depletion also increased with various valorisation technologies. As shown in Figure 6.5, blended cement production is more beneficial for metal depletion than other valorisation methods. Based on this method’s individual environmental profile, this benefit can be attributed to avoidance of steel usage in the construction of the concrete production plant. Also, this benefit is significantly higher in blended cement production than in inorganic polymer block production because five times more blended cement blocks than inorganic polymer blocks can be produced from 1 tonne of plasmastone (Iacobescu et al. 2013, Machiels et al. 2014).

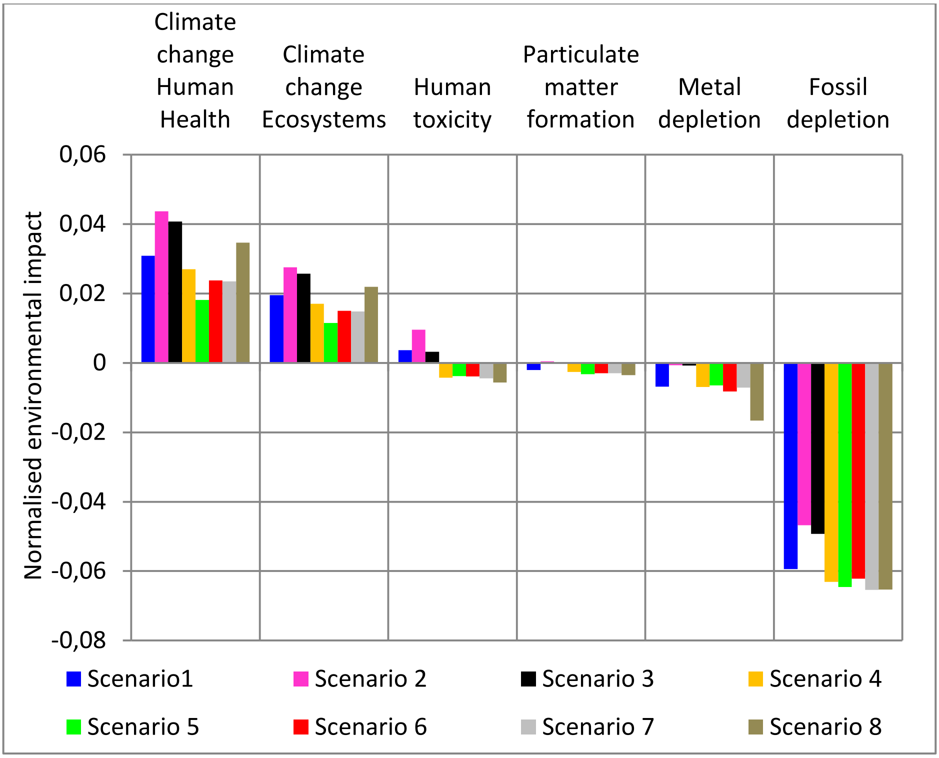


Figure 6.5: Normalised environmental profile of different scenarios

When the impact assessment method of IPCC 2007 GWP 100a (Goedkoop et al. 2013) is used, the net CO_2 equivalent emission of plasma gasification process with the least value plasmastone valorisation method (aggregate production) is 0.5 t CO_2 equivalent/ t RDF. This finding is nearly two times less than the CO_2 equivalent emission of traditional incineration with bottom ash valorisation (scenario 3), and aligns with UCL's (2014) recent comparative LCA study performed for plasma gasification and traditional incineration.

Finally, the results obtained for the various scenarios in the impact assessment phase were aggregated to a single environmental score (single score) in order to have an indication of the overall environmental performance of each (see Figure 6.6). This methodology is explained in detail in the chapter 3.

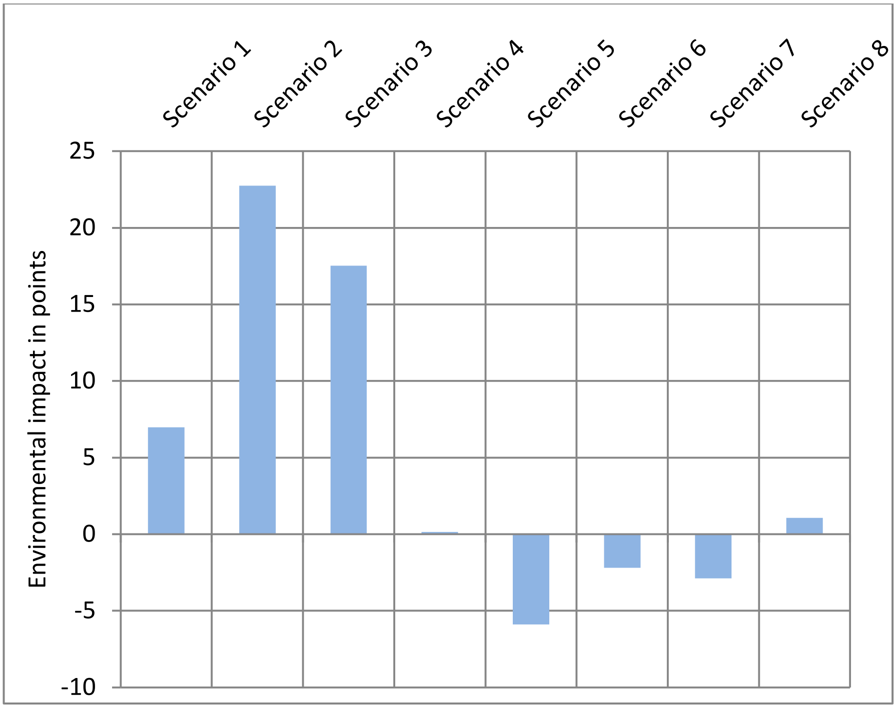


Figure 6.6: Environmental profile of different scenarios in single environmental score

As shown in Figure 6.6, the basic scenario of plasma gasification (scenario 1) shows a better environmental performance compared with incineration process (scenario 2). The overall environmental impact (burden) of incineration is more than three times higher than that of the basic scenario of plasma gasification. When various plasmastone valorisation methods are introduced, the total impact of plasma gasification is reduced further. For scenario 4 (aggregate production out of plasmastone), the net impact is almost neutral, although it becomes beneficial for the scenarios that employed inorganic polymer cement/ block and blended cement production. For the scenario using blended cement block production (scenario 8), the environmental impact is more than six times less than the basic scenario of plasma gasification. In fact, as the graph suggests, plasma gasification is not only a WtE process but also a WtM process that addresses the key objective of ELFM: a combined valorisation of landfilled waste as both WtM and WtE that maximises economic returns and minimises environmental burdens. According to the review of Bosmans et al. (2013), plasma gasification/vitrification is a viable candidate to achieve ELFM's technological goals. In addition, this LCA study indicates that plasma gasification is capable of realizing the environmental goals of ELFM through both WtE and WtM.

6.4.2. Economic performance

Figures 6.7 and 6.8 show the relationship between the economic and environmental impact of the various scenarios formulated for RDF valorisation, including plasma gasification and incineration. The total environmental impact was calculated for the hypothetical treatment plant described in the section 6.3 (100,000 t RDF/year of treatment capacity and a 20-year lifetime). In Figure 6.7 the environmental impact is indicated in points: the output of ReCiPe endpoint method delivered as a single environmental score. Because both thermal treatment methods and the developed residue valorisation options directly contribute to global warming potential, Figure 6.8 illustrates the environmental impact in kilograms CO_2 equivalent. In both graphs, the economic impact is expressed in NPVs, with positive values of NPV implying economic profits and negative values of environmental impact indicating environmental benefits. At first glance, none of the scenarios appear viable both environmentally and economically. In chapter 5, we concluded that plasmastone valorisation by inorganic polymer cement production, inorganic polymer block production and blended cement production yield both environmental and economic profits. Nevertheless, when plasma gasification is coupled with those plasmastone valorisation scenarios, the overall economic impact is negative. The benefit on CO_2 equivalent observed in chapter 5 for these same three scenarios also is a burden when plasma gasification and plasmastone valorisation processes are combined (Figure 6.8). However, when a variety (ReCiPe method) instead of a single impact category (GWP method) is considered, the net environmental impact of the scenarios remain beneficial (Figure 6.7). Only plasma gasification with blended cement block production out of plasmastone (scenario 8) yields an economic benefit. The NPV is five times less when blended cement block production (scenario 8) is substituted with inorganic polymer block production (scenario 6). Aggregate production (scenario 4) results in a 45 percent decrease in NPV compared with scenarios with blended cement and inorganic polymer cement production (scenarios 7 and 5). Economic analysis reveals that the NPV of plasma gasification is higher when the higher added-value applications for plasmastone are considered. In addition, as Figures 6.7 and 6.8 imply, blended cement is the economic driver of the plasma gasification process. It is important to notice that the lowest environmental and economic profit is obtained when the scenario includes incineration and landfilling of residues (scenario 2). In contrast to this study, the recent analysis of Winterstetter et al. (2015) shows better environmental performance in incineration than in plasma gasification. However, the percentage difference of efficiencies of incineration and plasma gasification that Winterstetter et al. used is only 7% (30% for incineration and 32% for plasma gasification) while the present study used a 23% difference (22% for incineration and 27% for plasma gasification). Residue valorisation and emission levels for incineration and plasma gasification are also not reported in Winterstetter et al.. Furthermore, Winterstetter et al. considered the substitution of marginal Belgian electricity produced by natural gas instead of Belgian average energy mix used in this study. Nevertheless, Both studies highlight that none of the scenarios offers CO2 equivalent saving as well as positive NPVs.

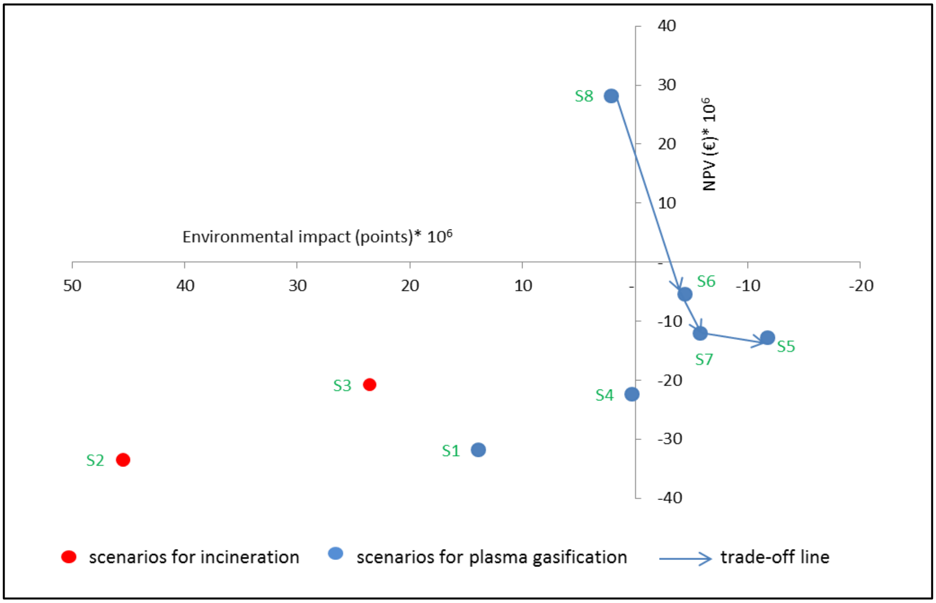


Figure 6.7: Trade-off analysis between NPV and overall environmental impact of different RDF valorisation scenarios

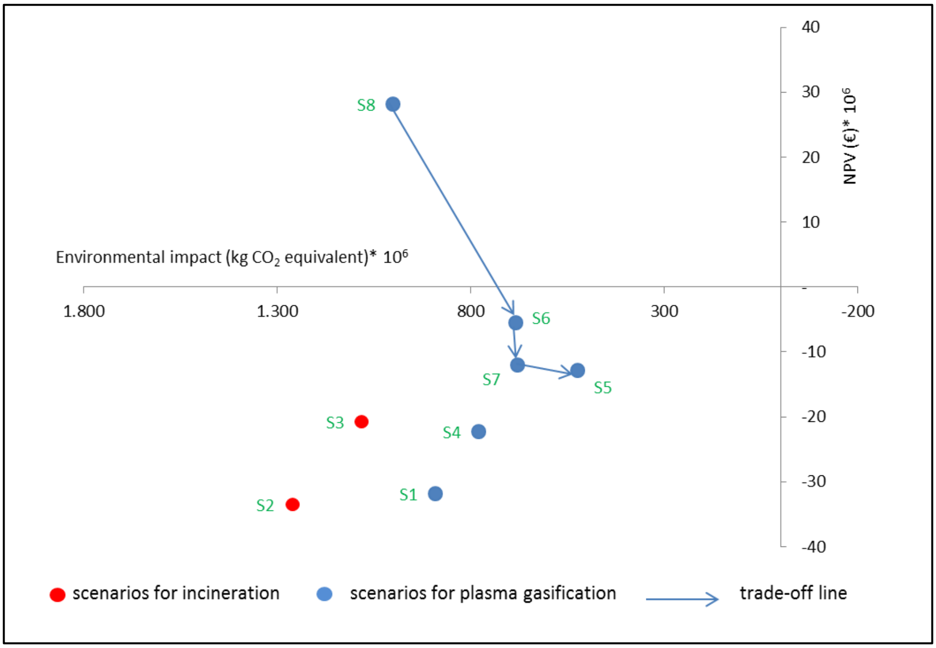


Figure 6.8: Trade-off analysis between NPV and GWP of different RDF valorisation scenarios

The trade-off line of the RDF valorisation scenarios indicates the economic benefit that must be forfeited to obtain the environmental benefit. In both Figures 6.7 and 6.8, the trade-off line starts from plasma gasification with blended cement block production (scenario 8), passes plasma gasification with inorganic polymer block production (scenario 6), then reaches plasma gasification with blended cement production (scenario 7), and ends at plasma gasification with inorganic polymer cement production (scenario 5).

Monte Carlo simulations show that the following parameters have an important impact on economic performance: (i) the net electrical efficiency of the thermal treatment system, (ii) the calorific value of RDF, (iii) the price of electricity, (iv) the price of green certificates, (v) the green energy fraction and (vi) the investment and the operational costs of the thermal treatment system. Prominently, these parameters were the most influencing parameters of a full-scale ELFM project, as discussed in chapter 4.

Table 6.5 illustrates the contribution of the different parameters to explain the variation in NPV obtained from Monte Carlo simulations using triangular distributions. The table also indicates the direction of influence. An increase in the NPV with an increase of a parameter is indicated as a positive contribution and the opposite situation is presented as a negative contribution. Only the most sensitive parameters are included in Table 6.5. The table shows that the increments of the net electrical efficiency yield higher NPVs; however, the improvement of efficiency may conceive significant growths in investment cost, which affects the NPV negatively. Hence, as concluded in chapter 4, a careful control of these two parameters is necessary to make plasma gasification an economically viable candidate in RDF valorisation. In addition to these parameters, the selling price of the higher-value products obtained from plasmastone becomes an important parameter for scenarios 5-8. This positive contribution of the product's selling price is significantly higher for scenarios associated with blended cement production; an average of 41.2 percent and 15.8 percent for product price ranges of 90–110 €/t and 45–55 €/t respectively. This finding suggests that high-quality products will increase the economic performance of the plasma gasification because higher quality eventually yields a higher selling price.

Table 6.5: Most sensitive parameters to the NPV

Parameter	Min. value	Max. value	Contribution to variance of NPV (%) in different scenarios							
			S1	*S2*	*S3*	*S4*	*S5*	*S6*	*S7*	*S8*
Net electrical efficiency of the plasma gasification system(%)	24 (+)	30 (+)	26.6 (+)			26.5 (+)	25.7 (+)	25.8 (+)	19.5 (+)	22.0 (+)
Net electrical efficiency of the incineration system(%)	20 (+)	24 (+)		22.9 (+)	23.5 (+)					
Calorific value of RDF (MJ/kg)	18 (+)	22 (+)	25.8 (+)	22.8 (+)	22.5 (+)	25.7 (+)	24.5 (+)	25.4 (+)	17.9 (+)	21.9 (+)
Price of electricity (€/MWh)	60 (+)	76 (+)	10.0 (+)	7.4 (+)	7.8 (+)	9.4 (+)	9.9 (+)	9.4 (+)	4.8 (+)	7.7 (+)
Price of green certificates (€/MWh)	110 (+)	124 (+)	4.4 (+)	4.6 (+)	4.5 (+)	5.3 (+)	4.7 (+)	4.7 (+)	3.2 (+)	4.4 (+)
Green energy fraction (%)	42 (+)	52 (+)	4.2 (+)	4.2 (+)	4.0 (+)	5.1 (+)	5.2 (+)	4.6 (+)	2.5 (+)	4.6 (+)
Investment cost of plasma gasification system (€/t RDF)	45 (+)	55 (+)	25.7 (-)			24.4 (-)	24.0 (-)	25.5 (-)	17.2 (-)	21.1 (-)
Operational cost of plasma gasification system (€/t RDF)	57 (+)	77 (+)	3.0 (-)			3.4 (-)	3.4 (-)	3.6 (-)	2.2 (-)	2.8 (-)
Investment cost of incineration system (€/t RDF)	35 (+)	45 (+)		35.1 (-)	34.1 (-)					
Operational cost of incineration system (€/t RDF)	50 (+)	70 (+)		2.6 (-)	2.3 (-)					
Selling price of inorganic polymer cement (€/t)	135 (+)	165 (+)					2.0 (+)			
Selling price of inorganic polymer cement block(€/t)	63 (+)	77 (+)						0.7 (+)		
Selling price of blended cement (€/t)	90 (+)	110 (+)							41.2 (+)	
Selling price of blended cement block(€/t)	45 (+)	55 (+)								15.8 (+)

6.6. Conclusions

This chapter includes the results of an environmental and economic evaluation performed to identify the capability of plasma gasification process to be used in the novel concept of ELFM. Besides energy production, plasma gasification is capable of producing building materials out of its residues, which contributes to the WtM component of ELFM. The study shows that the environmental burdens created by the process decrease when the plasmastone is subjected to various valorisation methods instead of landfilling. Similarly, environmental benefits also increase, because of the replacement of OPC-based products, which are associated with higher greenhouse gas emissions.

The economic analysis supports the production of blended cement as the economic driver of plasma gasification, and further analysis confirms that net electrical efficiency and investment costs of the plasma gasification system should be controlled carefully to obtain positive NPVs. In addition, the selling prices of the higher-value products obtained from plasmastone positively affect the NPV. Because the product's quality directly determines its selling price, improvements in product quality could expand the economic benefits of the process.

Among the discussed scenarios, traditional incineration obtains the lowest economic and environmental profit. A clear trade-off exists between economic and environmental performances of the scenarios; nevertheless, the results indicate that plasma gasification has a great potential to maximise the economic and environmental profits with cautious handling of certain parameters (net electrical efficiency, investment cost, product quality). Although this study considered only power generation, the produced syngas from plasma gasification could be used in many other applications. Finally, a detailed analysis of all possible syngas valorisation technologies would be required to have a straightforward and broad knowledge of the maximum contribution of plasma gasification within ELFM.

Chapter 7 : Feasibility of 'open waste dump' mining in Sri Lanka

This chapter is based on

Danthurebandara, M., Van Passel, S., Van Acker, K.. Feasibility of 'open waste dump' mining in Sri Lanka. (Revised version under review)

Abstract

Open waste dumps in Sri Lanka generate adverse environmental and socio-economic impacts due to inadequate maintenance. In this study, a concept of 'open waste dump mining' is suggested in order to minimise the environmental and socio-economic impacts, together with resource recovery. A model based on life cycle assessment and life cycle costing has been used to assess the environmental and economic feasibility of the suggested open waste dump mining. Two scenarios have been defined, dependent on the destination of the refuse derived fuel (RDF) fraction. Scenario 1 comprises direct selling of RDF as an alternative fuel to replace coal usage in the cement industry, while scenario 2 consists of thermal treatment of RDF with the objective of producing electricity. The study shows that both scenarios are beneficial from an environmental point of view, but not from an economic one. However, economic profits can be obtained by adjusting waste transport distances and the price of electricity. The environmental analysis further reveals that the higher global warming potential of open waste dumps can be eliminated to a large extent by applying suggested mining and waste valorisation scenarios.

7.1. Introduction

Sri Lanka is an island in the Indian Ocean located in the southern coast of the Indian subcontinent. It has a surface area of 65,610 km^2 and the current population is 20.7 million. The shift from an agricultural to an industrial economy has resulted in increasing urbanization and a rural-urban population shift. The rapid growth of the urban population has placed increasing pressures on different urban infrastructure services such as electricity, water supply, and solid waste and wastewater management. Municipal solid waste (MSW) management is now a growing problem in Sri Lanka due to rapid urban growth, recovery after a long civil war, low income, and an extremely fragmented solid waste management approach. Moreover, the tsunami attack of 2004 added millions of tonnes of additional waste to the already overloaded waste management system. MSW-related matters are more serious in the cities and urbanized areas than in the rural areas; Colombo, Moratuwa, Kandy, Matale, Gampaha, and Negombo are some of the municipalities that are suffering from increased MSW pollution due to a lack of proper disposal or recycling methods.

MSW generated in Sri Lanka has been dumped in open yards for several decades. This resulted in heterogeneous waste mountains which yield adverse impacts on environment and human health. These waste yards are known as ‘open waste dumps’. The purpose of this chapter is to introduce a concept to minimise the environmental and socio-economic threat of open waste dumps and to assess the feasibility of the suggested concept from an environmental and economic point of view, by using life cycle assessment (LCA) and life cycle costing (LCC) tools. The suggested concept is ‘open waste dump mining’, which can be categorized as a method for waste-dump rehabilitation. This concept is based on the novel concept of Enhanced Landfill Mining (ELFM), which is the core element of this thesis. This study encompasses (i) a concise literature review on waste management in Sri Lanka; (ii) environmental, health, and socio-economic impacts of open waste dumps; and (iii) their rehabilitation, in sections 7.2, 7.3, and 7.4 respectively. After that, in section 7.5 the methodology of applying ELFM concept in open waste dump mining is described. The results obtained in this study are presented and discussed in section 7.6.

7.2. Waste management in Sri Lanka

With changes in consumption patterns, the quantity of solid waste in Sri Lanka has increased over the years. In 1999, the average MSW generation per capita was 0.89 kg/cap/day, and it has been predicted to reach 1.0 kg/cap/day by 2025 (WorldBank 1999, Vidanaarachchi et al. 2006, Menikpura et al. 2012). In Sri Lanka, MSW contains a high organic matter fraction, moderate plastic and paper content, and low metal and glass fraction (Gunawardana et al. 2009). Generally, a higher moisture content is associated with this MSW, ranging from 60 percent to 80 percent on a wet basis (Vidanaarachchi et al. 2006, Menikpura et al. 2007). The average composition of the MSW in Sri Lanka compared to the MSW composition in other parts of the world is shown in Figure 7.1, as described in Visvanarthan et al. (2003).

Like in many other Asian countries, solid waste collection and disposal have been an issue in Sri Lanka for the past decades, where burning and dumping garbage into collection yards are the most common modes of disposal. The problem is not severe in rural areas, since the residents have sufficiently large plots of land where they can dispose of their daily waste collections. In urban areas, the local authorities perform house-to-house, communal and curbside collection of waste. After collection and transportation, approximately 85 percent of the total MSW generated is disposed of in ‘open dumps’, without any pre-treatment, cover, or compaction (Visvanathan et al. 2004). An open dump site is (i) a land disposal site at which solid wastes are disposed of without considering the environmental protection, (ii) susceptible to open burning, and (iii) exposed to the elements, disease vectors and scavengers. These dumps are located in environmentally sensitive areas such as wetlands, marshes, beaches and areas adjacent to water bodies or close to residential houses or public institutions (Gunawardana et al. 2009). As the waste separation is not well developed in Sri Lanka, the dump sites contain heterogeneous waste piles. The continuous dumping of waste in open areas eventually resulted in a number of garbage mountains in several municipalities in the country.

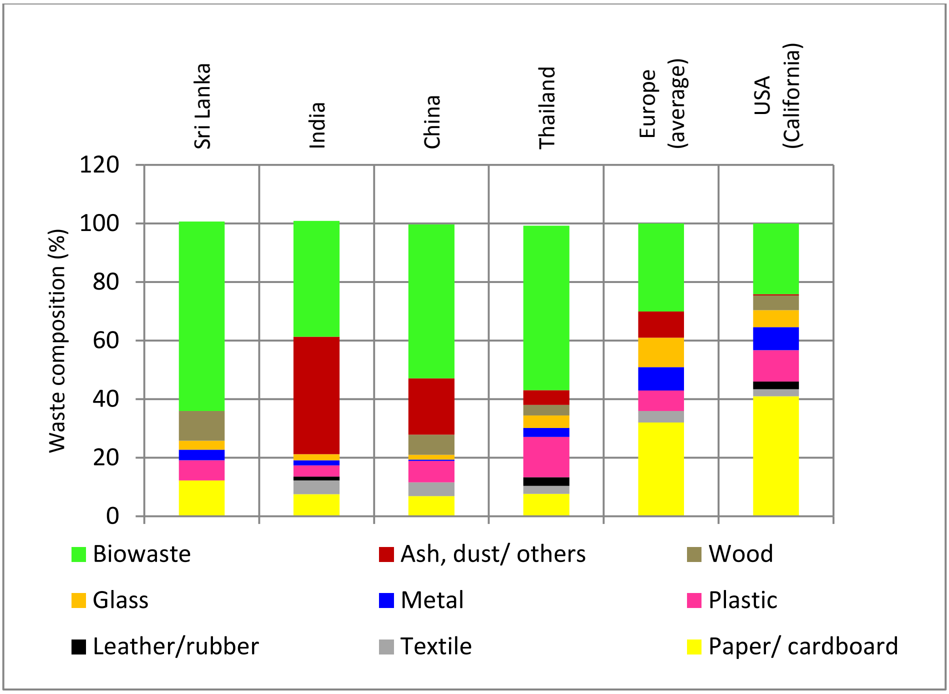


Figure 7.1: Composition of MSW of different countries (Visvanathan et al. 2003)

Figure 7.2: left: Bloemendhal waste dumping site, Colombo, Sri Lanka (DailyNews 2007), right: Gohagoda waste dumping site, Kandy, Sri Lanka (Weerakoon 2010)

The 'Bloemendhal' dump site, located in Colombo, Sri Lanka's commercial capital city, is an example: it occupies an area of 6.5 ha, goes to an average height of 30 meters and contains about 1.5–2.5 million tonnes of garbage (Sathees et al. 2014). For many years Bloemendhal has been an eyesore for nearby residents, including the poorest people of around 350 shanty dwellings (Sathees et al. 2014). The daily average waste collection in Colombo city is about 650 tonnes (APO 2007), and such waste is directly dumped into the Bloemendhal site. In addition to this landmarked dump site, many other small dump sites exist in the same municipal area. However, the quantities of waste dumped in these yards are not yet known. 'Gohagoda' is another well-known dump site located three kilometres away from Kandy, one of the culturally valued cities of Sri Lanka. Up to 1960 Gohagoda was used as an isolated area to dump hospital waste, then as a sewage dump site, and finally as the place for dumping all the waste generated by the Kandy municipal council. At present, 100 tonnes of MSW collected in the city are dumped at this site per day (Menikpura et al. 2008). Figure 7.2 shows the picture of both Bloemendhal and Gohagoda dump sites.

As a solution for the growing garbage problem in Sri Lanka, the Ministry of Environment developed a national strategy on solid waste management; implementation started in the recent past. The strategy has addressed the following activities: (i) waste avoidance, (ii) reduction, (iii) reuse, (iv) recycling, (v) composting, (vi) energy recovery, (vii) biogas utilization, and (viii) final disposal in an environmentally friendly manner (JICA 2008). Under this strategy, the possibility of initiating many solid waste treatment projects such as composting, anaerobic digestion, and material recycling (plastic/polythene and electronic and electrical waste) have been investigated, and some of them are already implemented throughout the island. Moreover, the construction of the first sanitary landfill in Sri Lanka was recently completed in Maligawaththa, Dompe, and four other sanitary landfills are proposed to be built in four other districts (CEA 2014). Although the national strategy on solid waste management addresses the present and future improvements of the solid waste management in Sri Lanka, it does not discuss the future management of the historical open waste dumps that appeared in the past decades due to improper waste handling. In order to realize the necessity of a strategy for the future management of open waste dumps, firstly the relevant environmental and social impacts of waste dumps and secondly the possibilities of waste dump rehabilitation should be investigated. Sections 7.3 and 7.4 of this chapter describe those components in detail.

7.3. Environmental, health and social impacts of open waste dumps

Waste dumping in open areas is a common practice not only in Sri Lanka, but also in many other Asian countries. In Thailand and India, 70 to 90 percent of landfills are just open dump sites (Visvanathan et al. 2003). Joseph et al. (2004) highlighted the following as the characteristics of a typical dump site: (i) no planning, (ii) no one on site who can exercise authority, (iii) no access control or control over the type of waste entering the site, (iv) no control of waste deposition, (v) no confinement of the waste body, and (vi) uncontrolled burning of waste. Moreover, the authors described that no proper site investigation and no engineering design are done for these sites; therefore, they have no gas collection system, groundwater protection, and drainage controls. As explained by Joseph et al. (2002), this situation accelerates the ground water and air pollution. Similar to landfill gas, CO_2 and CH_4 are the important constituents of gaseous emission of open waste dump sites. The absence of gas collection and utilization systems in open waste dumps results in a severe contribution to global warming potential, as CO_2 and CH_4 act as greenhouse gases.

The International Solid Waste Association (ISWA) explains the most important potential impacts of open dumping on the environment and to public safety and health as follows: ground and surface water pollution can occur as the leachate from the waste dumps which contains dissolved methane, fatty acids, sulphate, nitrate, nitrite, phosphates, calcium, sodium, chloride, magnesium, potassium, and trace metals (chromium, manganese, iron, nickel, copper, zinc, cadmium, mercury, and lead) migrates to the water table and surface water. This situation is very serious as it yields a severe pollution in the aquifers and serious eutrophication conditions in surface waters (Han et al. 2014). However, the impacts of open waste dumps depend on a number of site-specific factors, the most important being (i) location, waterway, geological/hydrogeological and climatic conditions, (ii) solid waste composition and quantity, (iii) the physical extent of the operation, and (iv) age of the dumpsite (ISWA 2007).

The burning of waste on open waste dumps often creates gaseous emission, which contains particulates, carbon monoxide, and other contaminant gases including low levels of dioxins. The exposure to this gaseous emission can result in respiratory issues, dizziness, and headaches in the short term, as well as potentially more serious diseases such as cancers and heart disease in the long term (Crowley et al. 2003). In addition to the environmental and health issues, open waste dumps create considerable impacts on land value, land degradation, and land availability. Chapter 2 of this thesis presents a detailed discussion on the environmental, health, and socioeconomic impacts of both controlled and uncontrolled landfills. The circumstances described in chapter 2 are at least the same in the surrounding environments of open waste dumps as qualitatively open dumps are worse than landfills. These unplanned heaps of uncovered wastes, often burning and surrounded by pools of stagnated polluted water, rats and fly infestations, with domestic animals roaming freely and families of scavengers picking through the waste highlight the environmental and societal degradation of a country (Joseph et al. 2002). Figure 7.3 shows “the scavengers”: both people and animals that can commonly be seen in open waste dumps.

Figure 7.3: Scavengers in open waste dumps in Sri Lanka (Thilakarathna 2008, Fazlulhaq 2012, Somawardhana 2014)

Data on environmental, health, and social impact assessments of open waste dump sites in Sri Lanka are very limited and not publicly available. However, a few studies are available that were performed for two landmarked dump sites: the Bloemendhal dump site, in Colombo, and the Gohagoda dump site, in Kandy. The study conducted by Sathees et al. (2014) revealed that the soil of the Bloemendhal dump site is sandy, and therefore the percolation is high through the deep layers; hence, contamination of the ground water can be expected. The leachate and soil within 150- and 400-meter radius from the centre of the waste pile contain high amounts of nitrate, phosphate, organic matter, heavy metals, and coliform bacteria. These values always exceed the standard levels set by the Sri Lanka Standards Institute. The data on this site's gaseous emission has not been reported yet. The characterisation study of leachate and groundwater of the Gohagoda dump site, performed by Dharmarathne et al. (2013), showed that the levels of pH, sulphate, nitrate, nitrites, and heavy metals (Pb, Zn, Ni, Cr, Co, Fe, Mn, Cu) are above the standards required by the World Health Organization for drinking water. This dump site exists at a distance of about 50 meters from the Gohagoda water intake plant. Furthermore, Menikpura et al. (2008) proved that the predicted leachate emission rate from this dump site is 30304 m^3/year and that it is highly polluted, with 15,000–20,000 mg/l of biological oxygen demand (BOD) value. The same study discovered that the predicted amount of greenhouse gas emission of this site is 2.61Gg/year.

7.4. Open waste dump rehabilitation

Dump site closure would help moderate the environmental and health impacts described in the above paragraphs. APO (2007) described that to have a properly closed landfill, two basic goals must be considered: (i) minimising the need for further maintenance of the landfill site, and (ii) placing the landfill in a condition that will minimise future environmental impact. These conditions must also be applied for open waste dump closing. Currently, dump site closure is done in several ways. The simplest way is stop dumping at the dump sites that have reached their maximum capacity or are overloaded. In this case, no extra closing methods and post-closure care are followed; the dump sites are still open to scavengers and the only difference that there is no waste inflow. Another way of closure is through applying a cover layer (such as soil) on top of the dump site. This reduces the unpleasant aesthetic appearance and the access to scavengers to a certain extent. However, the second option is not possible when the dump sites are several meters high. Applying a top layer in such dump sites covers only the top, but the sides of those large heaps are still uncovered and open for scavengers. Although a limitation of leaching can be expected to a certain extent in both methods, the damage to the environment is still high and the goals highlighted by APO (2007) are not achieved. Another option to mitigate the threat towards the environment and human health is upgrading and rehabilitating dump sites to sanitary landfills. This transformation has to be achieved by a step-by-step approach, starting from open dump to controlled dump, to engineered landfill, and eventually to sustainable landfill (APO 2007). Another main option to rehabilitate existing open dumps is through landfill mining where resource recovery is encouraged. Landfill mining is an option of exhuming existing or closed dump sites and landfills and sorting the exhumed materials for recycling, processing, or other deposition (Joseph et al. 2002). As described in the previous chapters of this thesis, the objective of landfill mining could be one or more of the following: redevelopment of landfill sites; conservation of landfill space; reduction in landfill area; elimination of potential contamination source; energy recovery from recovered wastes; and reuse of recovered materials (van der Zee et al. 2004, Jones 2008, Prechthai et al. 2008). More details on landfill mining projects in the Asian region can be found in APO (2007).

Although the above opportunities have been suggested and practiced all over the world, the dump site's own characteristics must be analysed and considered to determine which option is going to be applied. In Sri Lanka, the main landmarked dump sites are situated in the centre of urban areas. Dump site closure with or without applying a top cover does not result in remarkable changes. It is challenging to control the odour and leachate discharge only by stopping the dumping of waste and by covering the top of the waste heap. Applying the concept of landfill mining is comparatively better: under this concept, the materials can be recovered as much as possible by applying appropriate separation technologies, and the residual materials can be deposited in the protected cells created at the same site or off-site. When the residuals are deposited on site, the site becomes a closed sanitary landfill, the environmental burden due to open dump reduces, and the social and health impacts towards the nearby residents also decrease. However, the applications of the upgraded land are still limited. The land can be converted into a nature reserve or a recreational park, but use as land for housing, agriculture, or industry is restricted. Disposal of the residuals off-site especially in the newly constructed sanitary landfills far from urban areas will give a higher value for the recovered land. Considering all these facts, the novel ELFM concept can be applied for dump site rehabilitation in Sri Lanka as ELFM includes the combined valorisation of the historic waste streams as both materials and energy (or in other words Waste-to Materials (WtM) and Waste-to Energy (WtE)) and finally regaining the land as explained in the previous chapters. The possibility of applying the ELFM concept in open waste dump rehabilitation or to initiate open dump mining for resource recovery and land regaining is further described in section 7.5.

7.5. Methodology of applying ELFM to open dump mining

With the objective of removing the landmarked open waste dumps from urban areas in a sustainable way, the application of the above concept of ELFM appears to be possible. However, assessing the environmental and economic feasibility of this concept is obligatory prior to transferring open waste dump mining from the conceptual to the operational stage. The LCA/LCC model described in chapter 3 can be used in assessing the feasibility of open waste dump mining.

As the real open dump mining cases do not yet exist in Sri Lanka, the above model is applied to a hypothetical case. A process flow is developed for open waste dump mining that is moderately similar to the process flow of ELFM as illustrated in chapter 3. This hypothetical case and methodology of applying the ELFM model are described in detail in sections 7.5.1 and 7.5.2, respectively.

7.5.1. Hypothetical case

Considering the characteristics and situation of Sri Lanka's major landmarked dump sites, Bloemendhal and Gohagoda, a hypothetical case has been drawn. The basic outline for the hypothetical case is an open waste dump site which contains approximately 1,000,000 tonnes of waste and occupies an urban land of five ha (50,000 m^2) within Colombo's city limits. It is assumed that the waste dump was open for the past 30 years, with daily waste dumping of 100 tonnes/day. There is currently no waste inflow. Similar to typical waste dumps in Sri Lanka, no gas or leachate collection systems are installed in the considered dump site. The dump site mining scenario is organized as illustrated in Figure 7.4, which is moderately similar to the process flow of ELFM described in chapter 3. Waste excavation is performed by excavators, bulldozers, cranes, and other suitable equipment. The oversized waste (chairs, tyres, wooden pieces, etc.) identified during the excavation, are sorted out shredded, , and added to the relevant end-product category. After excavation, the waste is directed to a proper separation process. Pre-separation takes place at the dump site right after the excavation to separate the hazardous waste and fines.

Advanced separation can be done on site or off-site. As the considered dump sites are situated in highly populated areas, performing advanced processes on site seems difficult. Therefore, in this study it is assumed that the necessary processes after pre-separation are carried out in separate premises outside the city limits. Thus, the pre-separated waste is transported to the required premises. The advanced separation technology to be used depend on the moisture content of the pre-separated waste, composition, and the quantity of the waste. It has to be decided which type of advanced separation technology is going to be used. In this study air separation, dense media separation, magnetic separation, and eddy current separation are presumed to be in the advanced separation process. According to the conclusions of previous studies in landfill mining, in this study also, plastic, paper/cardboard, wood, and textile fractions are considered as one refuse derived fuel (RDF) fraction due to their high level of contamination (Quaghebeur et al. 2013). The major outputs of the advanced separation process include fines, RDF, ferrous metals, non-ferrous metals, stones/aggregates, and glass. RDF fraction is used as an alternative fuel in the cement industry or is incinerated to generate energy. After excavating and processing the entire dump site, the land can be reclaimed either as land for nature reserve, housing, agriculture, or industry. Considering the destination of produced RDF, two major scenarios have been developed for the analysis.

- Scenario 1 includes the processes of excavation, transportation, separation, fines treatment, and land reclamation. In this scenario, RDF is considered as an end-product of open dump mining and it is sold to the cement industry to be used as an alternative fuel.
- In scenario 2, RDF is treated as an intermediate product and is subjected to incineration in order to produce energy. Scenario 2 comprises the processes of excavation, transportation, separation, fines treatment, thermal treatment of RDF, and land reclamation.

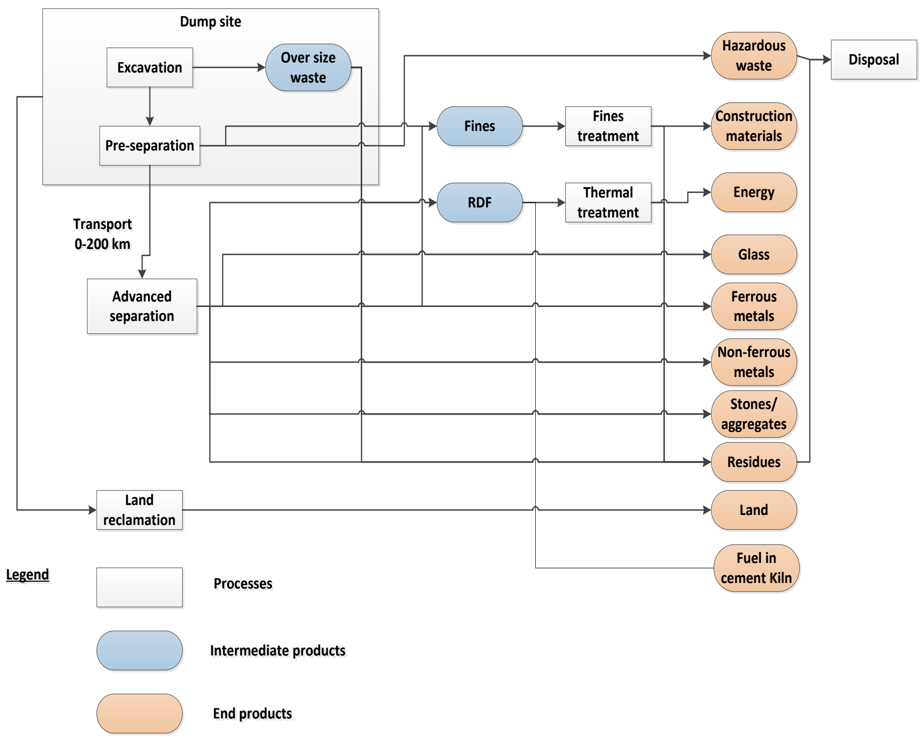


Figure 7.4: Open waste dump mining scenario

7.5.2. Environmental and economic assessment

As explained in the previous sections of this chapter, the environmental and economic assessments are based on LCA and LCC, respectively. The model developed for ELFM explained in chapter 3 was used to perform the LCA and LCC studies. The goal of this LCA study was to evaluate the environmental impacts of the open waste dump mining for resources and land recovery. The relevant building blocks of the ELFM model were used in order to build up the life cycle inventory (Troschinetz and Mihelcic) of each activity illustrated in Figure 7.4. In addition to the existing building blocks in the original ELFM model, an extra block for 'transportation' has been added, as transportation becomes an essential component in the considered case study. All the LCA guidelines considered in this work are similar to the ones explained in the previous chapters. The impact assessment was done in SimaPro 7 software.

The input data of this study comprise the data obtained from published sources, calculated data, and estimated data. The LCI data published mainly in the ecoinvent database (version 2.2) were used for the background processes with appropriate modification according to the Sri Lankan standards. Relevant processes were combined to estimate the overall impact of open waste dump mining with respect to the reference flow of one tonne of waste present in the dump site. It is assumed that with the exception of the pre-separation equipment, all other processing plants are situated in a specific ground, which is 150 kilometres away from the studied dump site. The quality of the products of open waste dump mining considered in this study are as follows:

- The metals recovered from separation processes have the quality which enables substituting the corresponding scrap metals.
- Stones and the other construction materials (sand, aggregates, etc.) recovered from separation and fines treatment processes have the quality of gravel that can be used in construction activities.
- When the RDF is used as an alternative fuel in the cement industry, one tonne of RDF replaces the production of 0.6 tonnes of coal. Furthermore, it avoids transportation of the same amount of coal from Indonesia to Sri Lanka. The calculation is based on the average calorific values of RDF and coal (20 MJ/kg vs. 33 MJ/kg (Fisher, 2003)).
- When the RDF is incinerated in order to produce energy, the produced electricity replaces the base load of electricity production in Sri Lanka, which includes 70 percent conventional thermal energy, 23 percent hydro energy, and 6 percent wind energy (SLSEA 2012). The produced heat is assumed to be used in the process itself.
- Recovered land is converted into land for a nature reserve.

For the environmental impact assessment of this study, the ReCiPe endpoint method (Hierarchist version, H/A) was selected, as it addresses several impact categories such as (i) climate change on human health; (ii) climate change on ecosystems; (iii) ozone depletion; (iv) terrestrial acidification; (v) freshwater eutrophication; (vi) human toxicity; (vii) photochemical oxidant formation; (viii) particulate matter formation; (ix) terrestrial ecotoxicity; (x) freshwater ecotoxicity; (xi) ionising radiation; (xii) agricultural land occupation; (xiii) urban land occupation; (xiv) natural land transformation; (xv) metal depletion; and (xvi) fossil fuel depletion (Goedkoop et al. 2013).

The goal of the LCC study was to evaluate the economic drivers of open waste dump mining. The developed LCC model for ELFM was used for this assessment. The cash flow model was set up for the period of five years including all costs and revenues associated with the different processes. The waste processing is completed within five years and the depreciation rate is assumed to be five percent. As a result, all processing plants remain with a residual value after five years. These remaining processing plants are considered to be used in future waste separation and processing under developing national solid waste management strategy or in other open waste dump mining cases. Hence, the cash flow was facilitated with a residual value for the processing plants. Net present value (NPV) was used as the major economic indicator in order to determine the major economic drivers of open waste dump mining. The Monte Carlo simulation approach was used to examine the sensitivity of different parameters on NPV, as explained by Van Passel et al. (2013).

Table 7.1 shows the composition of dumpsite mined waste presented by Menikpura et al., (2008) which was used in this study with necessary modifications. It is important to notice here that the metal and glass percentages are comparatively lower than that of waste composition shown in Figure 7.1 due to the scavenger effect. To elaborate our study in detail, metal fraction is further divided into ferrous and non-ferrous metals with equal percentages. Biodegradables, polyethylene, coconut comb and husk, textile, wood, rubber and leather, and paper fractions are combined into one RDF fraction. The previous landfill mining and open dump mining case studies in Europe and Asia revealed that a considerable amount of fines can be present in the mined waste (Joseph et al. 2004). Therefore, we assumed that 25 percent of the RDF fraction goes to fines, together with 50 percent of the fraction of construction demolitions. The adjusted composition is illustrated in Table 7.2 with the recovery efficiencies applied for all waste fractions in the separation process. Unrecovered portions of each waste fraction are considered as residues to be disposed of in a sanitary landfill. Energy, materials, and emission data of the incineration plant of Scenario 2 is presented in Table 7.3. As incineration plants are not yet existed in Sri Lanka, the data of a well-established, large scale incineration plant in Europe (Indaver) were used. It is assumed 50 percent of biogenic fraction in order to calculate the biogenic and fossil CO_2 emission. Costs and product selling prices used in the economic analysis are illustrated in Table 7.4.

Table 7.1: Average composition of dump site mined waste in Sri Lanka (Menikpura et al. 2008)

Waste fraction	Percentage (%)	Waste fraction	Percentage (%)
Biodegradable	59.69	Paper	0.92
Polyethylene	24.66	Glass	1.28
Coconut comb and husk	4.74	Metal	0.02
Textile	3.66	Stones	1.35
Wood	1.29	Construction demolitions	0.10
Rubber and leather	1.20	Undefined	1.09

Table 7.2: Adjusted waste composition of dump site mined waste and separation efficiencies of the separation process

Waste fraction	Percentage (%)	Recovery efficiency (%)
RDF	72.12	80
Fines	24.09	80
Ferrous metals	0.01	80
Non-ferrous metals	0.01	80
Glass	1.28	80
Stones	1.35	80
Construction demolitions	0.05	80
Undefined	1.09	

Table 7.3: Energy, materials and emission data of incineration

Parameter	Value	Source
Calorific value of RDF (MJ/kg RDF)	20	Menikpura et al. (2008)
Start-up energy (kWh/t RDF)	78	Indaver (2012)
Net electrical efficiency (%)	22	BREF (2006, 2010)
Bottom ash generation (t/t RDF)	0.228 (to be landfilled)	Indaver (2012)
Air pollution control (APC) residues (t/t RDF)	0.043 (to be landfilled)	Indaver (2012)
Auxiliary materials		Indaver (2012)
Activated carbon (kg/t RDF)	0.5	
Urea (kg/t RDF)	3.5	
Limestone (kg/t RDF)	6.7	
Quicklime (kg/t RDF)	4.4	
Emission		Indaver (2012)
Carbon dioxide (kg/t RDF)		
biogenic	839	
fossil	839	
Carbon monoxide (kg/t RDF)	0.09	
Particulates (kg/t RDF)	0.014	
Nitrogen oxides (kg/t RDF)	1.49	
Sulphur dioxide (kg/t RDF)	0.019	
Hydrogen chloride (kg/t RDF)	0.003	
Dioxins (kg/t RDF)	8×10^{-8}	
Mercury (kg/t RDF)	1.6×10^{-6}	
Heavy metals (kg/t RDF)	0.052	

Table 7.4: Data used in the economic analysis

Parameter	Value	Source
Time length (years)	5	Case study
Depreciation rate (%)	5	Case study
Excavation cost (€/t)	1.60	Industrial reference (United Tractor and Equipment)
Transport cost (€/tkm)	0.13	Rathi (2007)
Investment cost of separation (€/t)	5	Industrial reference (BERNS, ENVIROMECH)
Operational cost of separation (€/t)	7	Industrial reference (BERNS, ENVIROMECH)
Investment cost of incineration (€/t)	60	Ducharme (2010)
Operational cost of incineration (€/t)	40	Ducharme (2010)
Electricity price (€/MWh)	125	PUCSL (2012)
Disposal cost of residues (€/t)	90	Central Environmental Authority
Price of metals (€/t)	800	Commercial reference (Ceylon steel, Recycleinme.com)
Price of RDF (€/t)	33	Calculated*
Price of glass (€/t)	6	Commercial reference (Recycleinme.com)
Price of aggregates (€/t)	10	Commercial reference (Recycleinme.com)
Price of land (€/m^2)	25	Central Environmental Authority

* This value was calculated considering the calorific value of RDF and average price and calorific value of coal. Price of coal: 55 €/t (Infomine 2015) , calorific value of coal: 33 MJ/kg (Fisher 2003)

7.6. Results and Discussion

7.6.1. Environmental performance of open waste dump mining

Figures 7.5 and 7.6 illustrate the environmental impact of valorisation of one tonne of waste present in the dump site for Scenarios 1 and 2, respectively. Figure 7.5 confirms that the separation and the transportation processes dominate the most impact categories. The significant benefits of the separation process on several impact categories are due to the avoided burdens caused by different end-products produced during separation. The individual environmental profile of the separation process reveals that the major benefit is due to the replacement of coal production and transportation by using RDF as an alternative fuel in the cement industry. In this study, one tonne of waste present in the dump site is responsible for reducing production and transportation of 0.348 tonnes of coal. Although the recovery of metals, aggregates, and glass also yield environmental benefits, its importance is lower than the benefits due to RDF. In Scenario 2, thermal treatment of RDF dominates the environmental profile, and separation and transportation become the next important processes. In Scenario 2, the RDF obtained from the separation process is treated in an incinerator in order to produce energy instead of direct-selling to the cement industry. One tonne of waste present in the landfill contributes for a production of 710 kWh of electricity. In this way, the influence of the separation process in different impact categories is different in the two scenarios. Figures 7.5 and 7.6 illustrate the contribution of each process in open dump mining relative to the different impact categories. As the total environmental impact in each impact category is set to 100 percent, the figures do not conclude to what extent an impact category has a significant contribution and which scenario performs better.

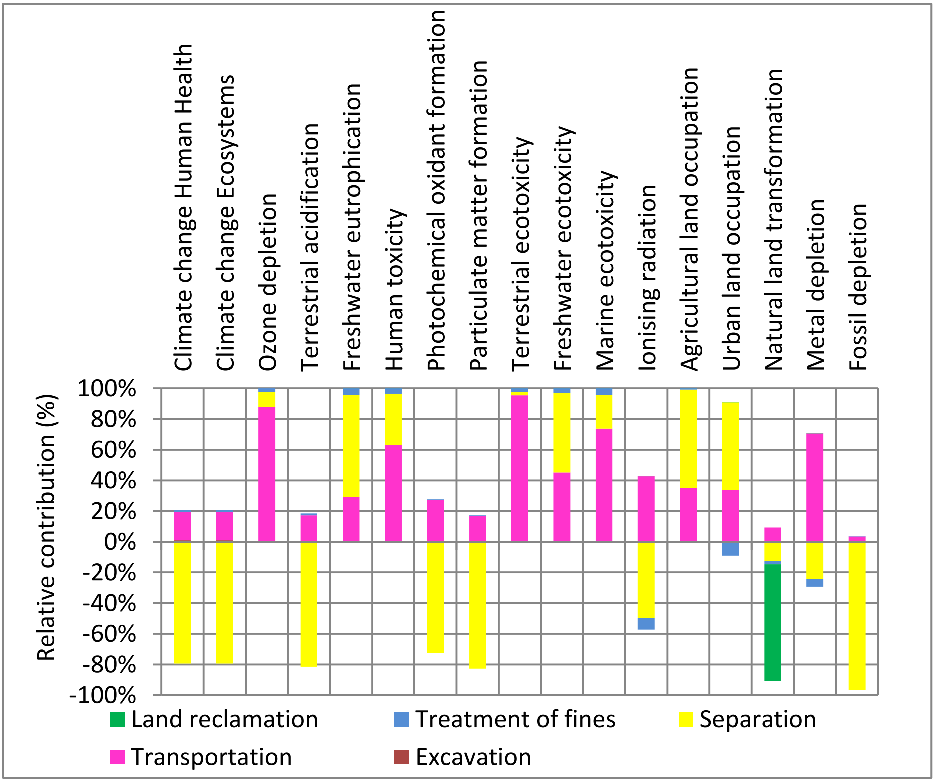


Figure 7.5: Environmental profile of scenario 1 (reference flow- 1 tonne of waste in the dump site)

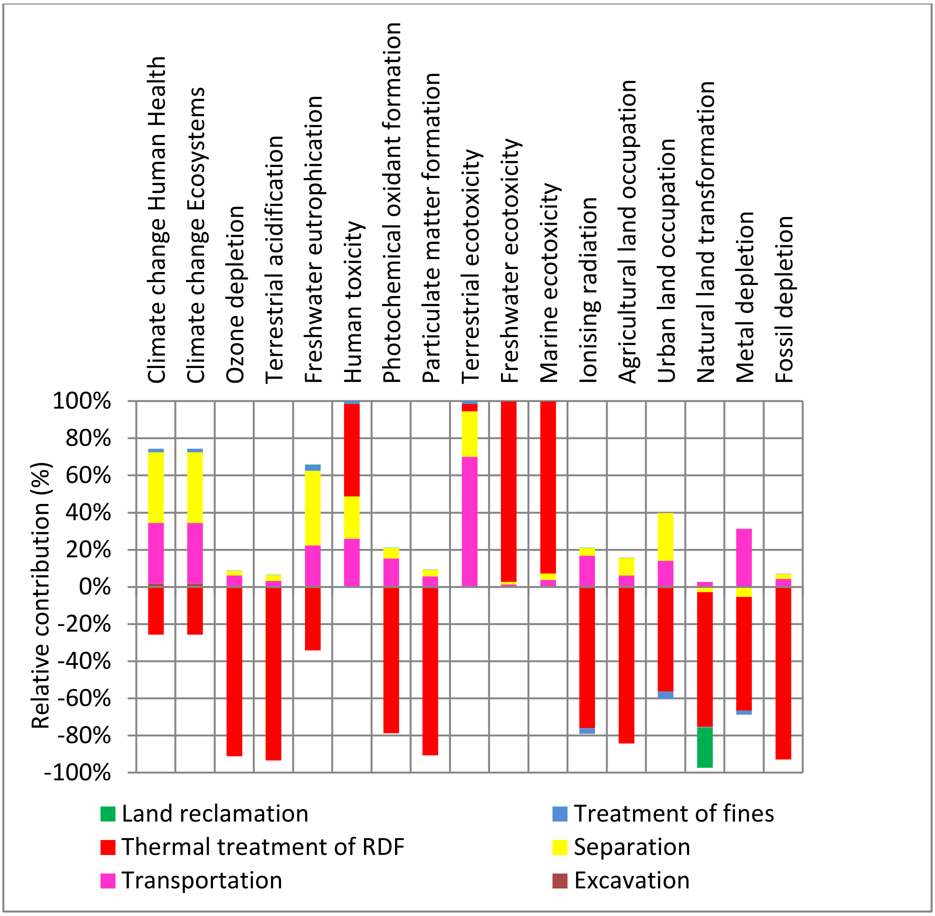


Figure 7.6: Environmental profile of scenario 2 (reference flow- 1 tonne of waste in the dump site)

Figure 7.7 clarifies the overall performance of the scenarios and the mostly influenced impact categories. Figure 7.7 shows that in both scenarios, impact in fossil depletion is very significant. Next to that, the contributions to particulate matter formation and climate change on human health are also important. The impact on other categories is insignificant. The benefit in fossil depletion is higher when the RDF is used as an alternative to coal fuel in the cement industry (Scenario 1) than when it is thermally treated in order to replace the conventional electricity production in the country (Scenario 2). Contrastingly, the environmental credits in particulate matter formation are higher in Scenario 2. Scenario 1 is beneficial in the climate change impact category, while Scenario 2 is not; the flue-gas emission with high CO_2 concentration in the thermal treatment process is a reason for this difference. However, both scenarios yield a net environmental benefit, as shown in Figure 7.7. Furthermore, Scenario 1 is 30 percent more beneficial than scenario 2. We discussed above the environmental impact of the valorisation of one tonne of waste present in the dump site. To bring the open dump mining concept to the operational phase, it is necessary to know whether it is beneficial compared to the Do-nothing scenario or reference situation. The Do-nothing scenario or reference situation supposes that no mining activities are undertaken; the dump site remains as it is, without any maintenance or environmental protection activity. No collection or treatment takes place for the gases and leachate.

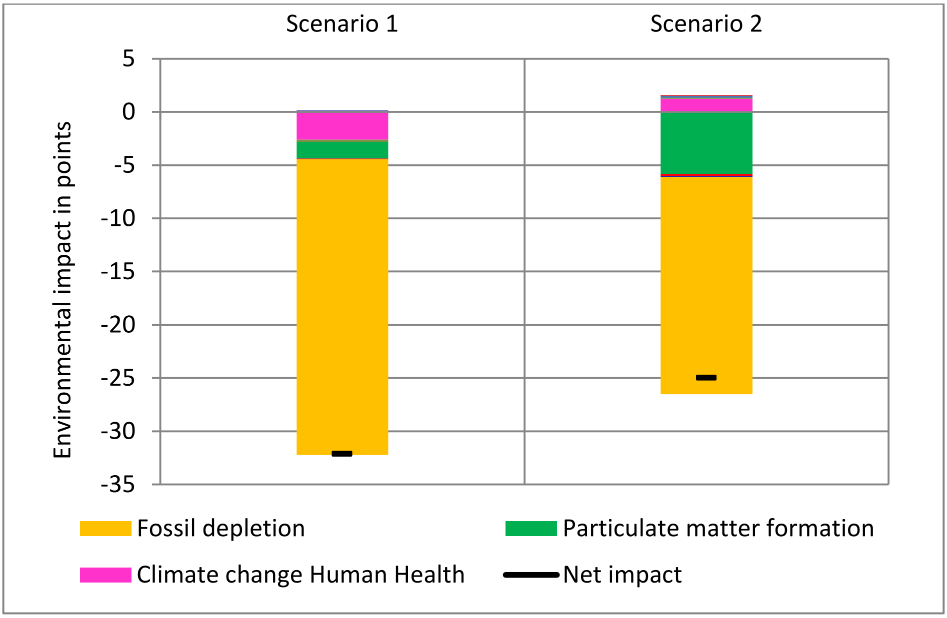


Figure 7.7: Most significant impact categories of scenario 1 and 2 (reference flow- 1 tonne of waste in the dump site)

The evolution of gas and leachate that can be produced by the hypothetical dump site has been analysed in order to identify the current situation. The gas production curve for the considered dump site was obtained from the LandGEM model (version 3.02) and is presented in Figure 7.8. LandGEM is an automated estimation tool with a Microsoft Excel interface that can be used to estimate emission rates for total landfill gas, methane, carbon dioxide, non-methane organic compounds, and individual air pollutants from MSW landfills (USEPA 2005). The gas production curve reveals that this dump site is, in 2015, at the peak of gas production; the gas production will then decrease over time and become considerably low after 100 years. In order to decide whether or not the valorisation of waste present in the dump site is environmentally beneficial against the existing situation (Do nothing scenario), the residual impact of the dump site should be determined. In this study the residual impact starts from year 2015 (Figure 7.8), as the waste valorisation activities are assumed to have started in 2015. The respective residual environmental impact was calculated for 100 years starting from 2015. The leachate emission and composition data present in Sathees et al. (2014) were used to determine the emission to water and soil.

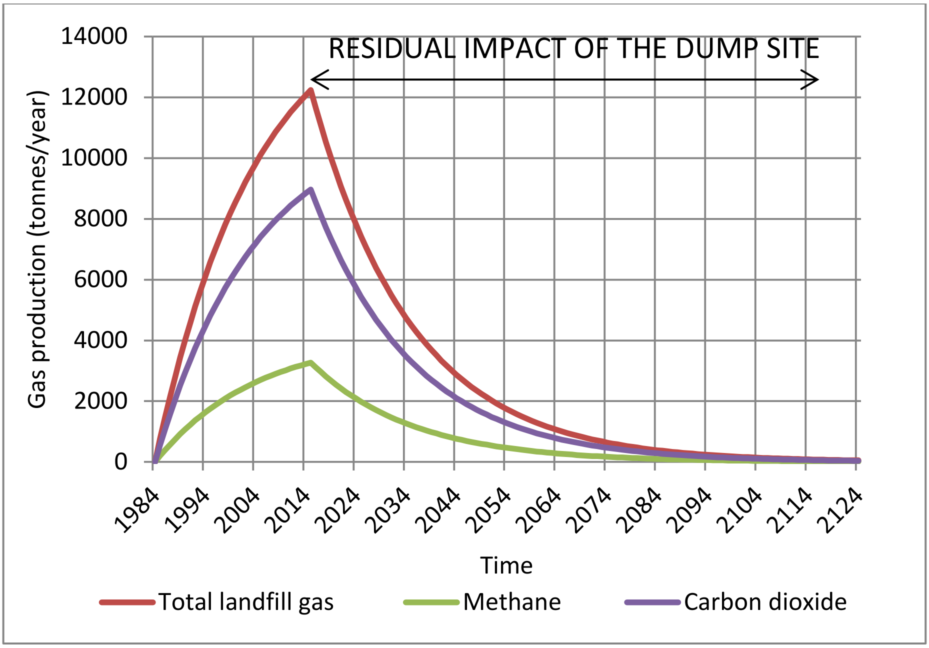


Figure 7.8: Gas production curve of studied dump site as delivered by LandGEM model (version 3.02)

Figure 7.9 shows the environmental impact of the Do nothing scenario for the total amount of waste. In addition, those impacts were compared with Scenario 1 and Scenario 2. The impact of the two scenarios were calculated for the total amount of waste present in the dump site (valorisation of total waste present in the dump site).

The net environmental impact of the Do-nothing scenario turns out to be very detrimental compared to the waste mining/valorisation scenarios. In the Do-nothing scenario the burdens are mainly found in the impacts of climate change on human health and climate change on ecosystems. These burdens are mainly due to the 66,758 tonnes of total methane emission for 100 years, starting from 2015. CO_2 emission in the Do-nothing scenario was considered as CO_2 neutral because of its biogenic origin. This scenario is not responsible for any environmental benefit, as the produced methane is not used in energy production and no materials are recuperated whatsoever.

Another impact assessment was performed by using the method of IPCC 2007 GWP 100a (PRéConsultants 2008); the results are illustrated in Figure 7.10. The figure reveals that the CO_2 equivalent emission of the Do-nothing scenario can completely be eliminated by Scenario 2. Not only elimination, but also a CO_2 equivalent saving is foreseen for Scenario 1. Additionally, Scenario 2 reduces the CO_2 equivalent burden of the Do-nothing scenario up to 98 percent. From figures 7.8–7.10, it can be concluded that a higher fraction of environmental burden taken place due to open waste dumps can be eliminated by applying appropriate mining and valorisation scenarios at the early stages of the waste degradation of a dump site. Over time, a large fraction of methane is freely emitted to the environment and the dump site reaches its maturation/long-term phase (final state of waste stabilisation). Performing waste mining and valorisation during the maturation phase still allows for environmental benefits through materials and energy recuperation, but is not advantageous in mitigating the emission of CO_2 equivalent, as shown by the case study analysed in chapter 4.

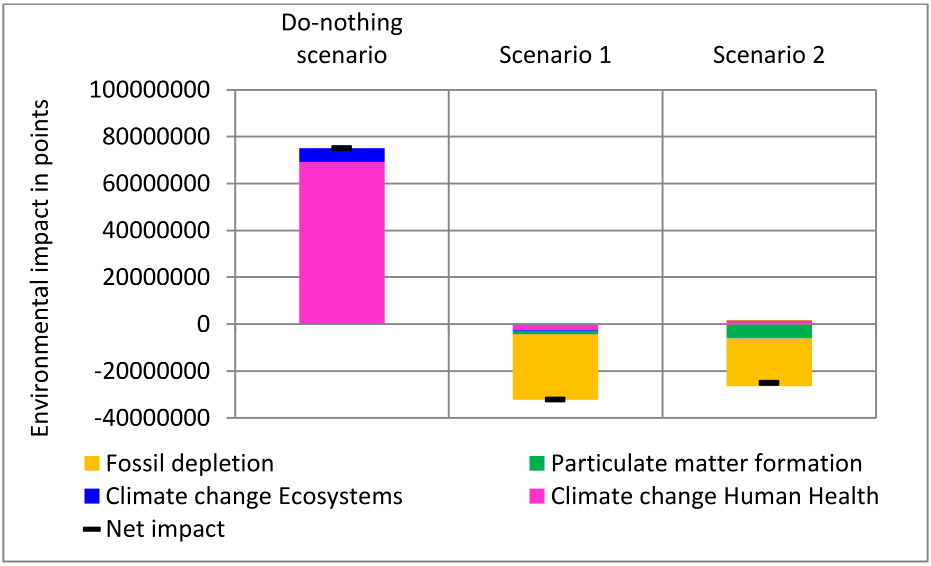


Figure 7.9: Environmental impact of Do-nothing scenario and waste valorisation scenarios

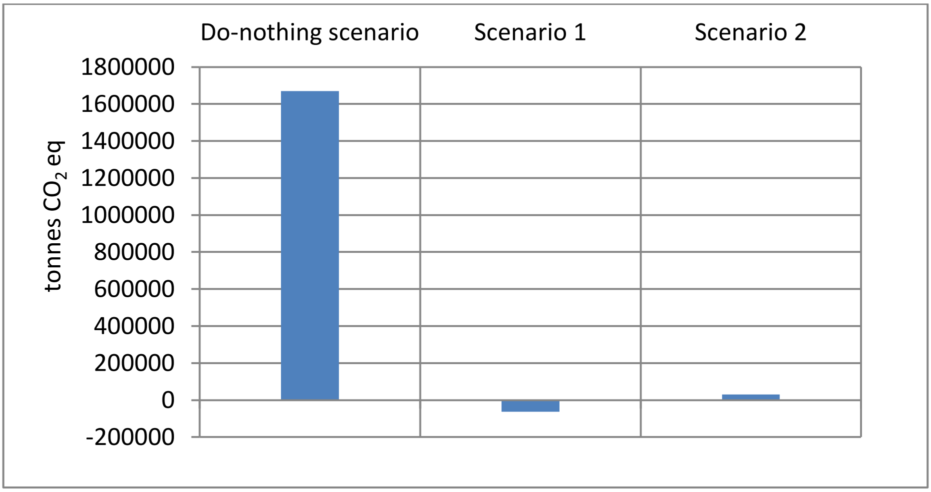


Figure 7.10: CO_2 equivalent emission of Do-nothing scenario and waste valorisation scenarios

7.6.2. Sensitivity analysis in environmental profiles

From the analysis of the above open waste dump mining scenarios, it was identified that the transportation, separation, and thermal treatment are the most influencing processes to the environment. Likewise, waste transportation distance, RDF recovery in the separation process, and electricity production in the thermal treatment process were recognised as the main factors that dominate the environmental profiles. The amount of produced electricity depends on the calorific value of RDF and the net electrical efficiency of the thermal treatment system. In addition, the recovery efficiency of RDF in the separation process is also a factor to decide net electricity production. Hence, the parameters of transport distance, RDF recovery efficiency, calorific value of RDF, and the net electrical efficiency of the thermal treatment process were subjected to a sensitivity analysis. Table 7.5 provides a summary of those parameters on which the sensitivity analyses are performed. Figure 7.11 illustrates the comparative environmental profile of the scenarios with the sensitivity analyses.

Table 7.5: Overview of the sensitivity analyses

Parameter	Basic value	Best case value	Worst case value
Transport distance from dump site to the separation plant (km)	150	50	250
RDF recovery efficiency in separation process (%)	80	90	70
Calorific value of RDF (MJ/kg)	20	25	15
Net electrical efficiency of thermal treatment system (%)	22	30	20

Transportation of excavated waste from the dump site is necessary when there is no sufficient space to construct further processing plants at the dump site, or when it is essential to process the waste in a specific processing plant, away from the dump site. Reducing the transport distance obviously increases the environmental benefit of the two suggested scenarios. However, this increment is not well pronounced, ranging from five to nine percent only (Figure 7.11). RDF plays a significant role in both scenarios. When the RDF recovery is higher, the amount of coal replacement is also higher in Scenario 1; this results in a 13 percent increment of environmental benefit in Scenario 1 when the RDF recovery efficiency increases by 10 percent. Similarly, the higher RDF recovery efficiencies positively affect the electricity production in Scenario 2. As illustrated in Figure 7.11, a 10 percent increment of RDF recovery efficiency leads to a 15 percent increase in the net environmental impact (benefit) of Scenario 2. As explained earlier, this benefit can further be improved with higher calorific values and higher electrical efficiencies. The calorific value of RDF is mainly dependent on the biodegradables and plastic content. When they are not in larger fractions, then lower calorific values are expected; similarly, when the dump site is in its maturation phase the calorific value of the waste displays lower values due to the waste degradation. Considering these facts, in the sensitivity analysis a 15–25 MJ/kg range was used as the calorific value of RDF (Menikpura et al. 2008).

According to Figure 7.11, the net environmental benefit of Scenario 2 increases by 60 percent for a 25 percent enhancement of calorific value of RDF. Although the average electrical efficiency of a typical incinerator is 22 percent, higher efficiencies such as 30 percent are also reported (Bosmans et al. 2013). Hence, an upper margin of 30 percent was applied for the sensitivity analysis of net electrical efficiency of thermal treatment system. It expands the environmental benefit of Scenario 2 by 92 percent. Apart from improving the calorific value and electrical efficiency, the thermal treatment technology can also be altered for obtaining higher benefits. Bosmans et al. (2013) concluded that plasma gasification/vitrification is a viable candidate for combined energy and material valorisation in the framework of ELFM. Moreover, chapter 6 highlights that the environmental performance of plasma gasification is few times better than that of incineration. This finding can also be applied in open waste dump mining in order to improve the current environmental benefits. However, application of such a technology in a developing country is arbitrary due to the financial constraints.

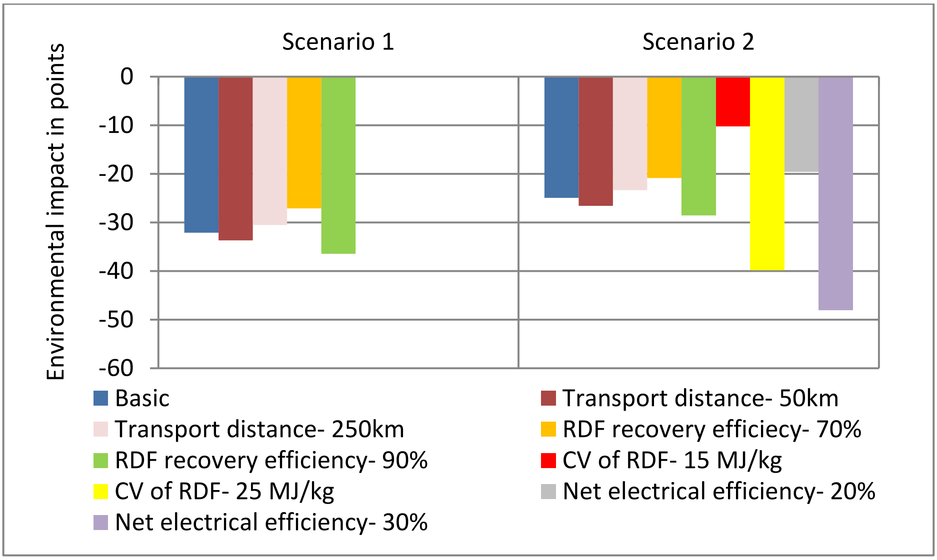


Figure 7.11: Environmental profile of open waste dump mining scenarios with sensitivity analysis- reference flow: 1 tonne of waste in the dump site (basic scenario comprise 150km transport distance, 80% RDF recovery efficiency, 20 MJ/kg CV of RDF and 22% net electrical efficiency of thermal treatment system)

7.6.3. Economic performance of waste valorisation

In Figure 7.12 the economic performances of the two scenarios are plotted against the environmental performances. NPVs and environmental impacts were calculated for the hypothetical case explained in section 7.5.2. As explained in the previous chapters, negative values indicate benefits in environmental performance and burdens in economic performance. Hence, Figure 7.12 shows that none of the scenarios are beneficial in both aspects. Although both scenarios produce environmental benefits, the NPVs are negative within the data used in Table 7.4. Scenario 2 shows better economic results compared to Scenario 1.

The contributions of the most influencing parameters to the NPV are illustrated in Table 7.6. An increase in the NPV with an increase of a parameter is specified by a positive value, and the opposite situation is designated by a negative value.

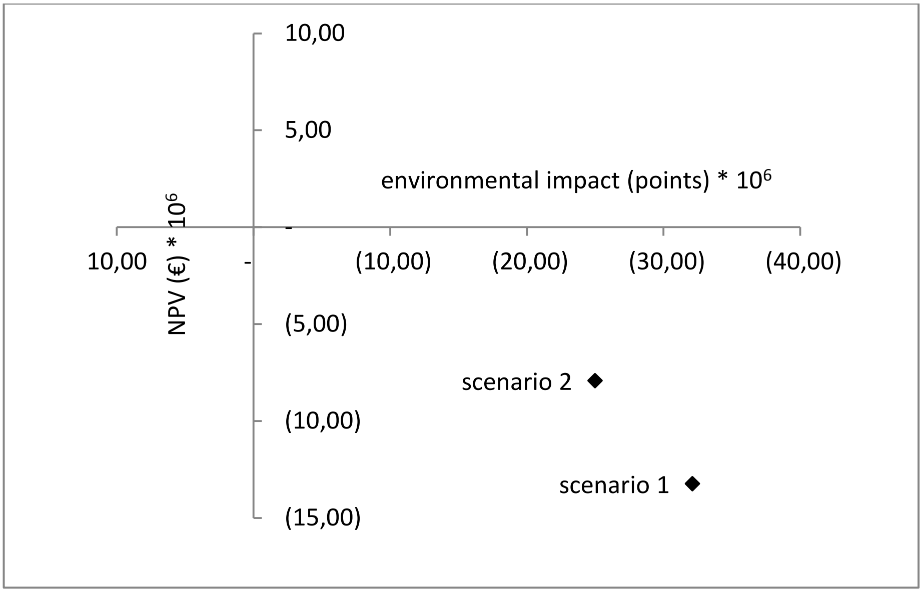


Figure 7.12: Economic performance against the environmental impact of scenarios

Table 7.6: Monte Carlo sensitivity analysis

Parameter	Minimum value	Maximum value	Contribution to variance of NPV (%)
Scenario 1			
Transport cost (€/tkm)	0.09	0.23	54.9 (-)
Transport distance (km)	50	250	31.5 (-)
RDF selling price (€/t)	25	42	12.1 (+)
RDF recovery efficiency (%)	70	90	1.5 (+)
Scenario 2			
Calorific value of RDF (MJ/kg)	15	25	31.3 (+)
Electrical efficiency of thermal treatment system (%)	20	30	23.1 (+)
Electricity price (€/Mwh)	100	150	19.6 (+)
Transport distance (km)	50	250	13.6 (-)
Transport cost (€/tkm)	0.09	0.23	6.2 (-)
RDF recovery efficiency (%)	70	90	2.4 (+)
Investment cost of thermal treatment system (€/t RDF)	55	65	2.1 (-)

In Scenario 1, transport costs contribute 54.9 percent to the NPV. The next highest contribution is given by transport distance. In this study we used 150 km of average transport distance, as the waste has to be transported to a specific ground with enough space, beyond the city limits, for further processing. As the hypothetical dump site is assumed to be in Colombo, the distance from Colombo to a specific ground where the processing plants can be installed is estimated. In the sensitivity analysis 250 km of maximum distance was used, assuming that the northern part of the country can also provide a suitable ground for waste processing due to comparatively less population than the other areas. Reductions in transport costs obviously yield higher NPVs according to Table 7.6. The variation of NPV with the different transport costs for three different transport distances is demonstrated in Figure 7.13. A decrease of transport costs by 10 percent leads to a NPV increase by 12 percent, 11 percent, and 10 percent for the transport distances of 50 km, 150 km, and 250 km, respectively. This figure leads to the conclusion that avoiding waste transportation by implementing all processing plants on the dump site or nearby is a prerequisite to obtaining the economic benefits of open waste dump mining for this scenario.

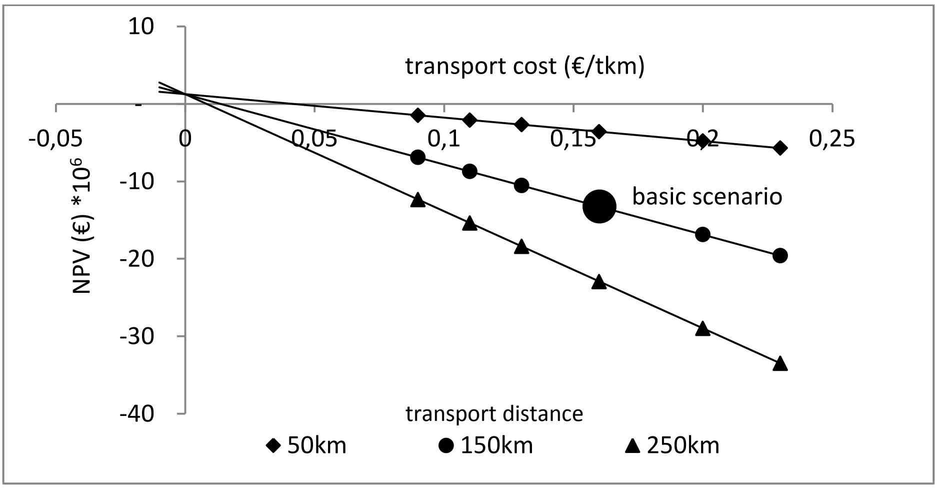


Figure 7.13: The impact of variations in transport cost on NPV in scenario 1

In addition to transport costs and transport distance, the selling price of RDF is another imperative parameter that gives 12.1 percent positive contribution to the NPV. In this study, the selling price of RDF was calculated as 33 €/t by considering the ratio of the calorific values of RDF and coal (20/33) and the average market price of coal (55 €/t). Depending on the composition of the MSW in Sri Lanka, the minimum and maximum values of calorific value of RDF were decided as 15 and 25 MJ/kg. Based on these values, the minimum and maximum values for selling prices of RDF were calculated as 25 and 42 €/t. It is worthwhile to investigate how the selling price of RDF can alter with varying transport costs and distance, as Figure 7.13 confirms that obtaining higher NPVs seems to be less possible by changing only the parameters related to transport. Figure 7.14 shows the variation of NPV with the different transport costs and distances for three different selling prices of RDF.

Figure 7.14 shows that higher selling prices of RDF obviously lead to a gain in higher NPVs for varying transport distances and transport costs. However, a selling price increase fully depends on the calorific value of RDF. Hence, for this study, the selling price cannot exceed the upper margin of 42 €/t. For that selling price, the maximum transport distance and transport costs should be approximately 50 km and 0.05 €/t km in order to make the NPV at least zero instead of having a negative value. Once more, Figure 7.14 further confirms the necessity of avoiding transport in this scenario.

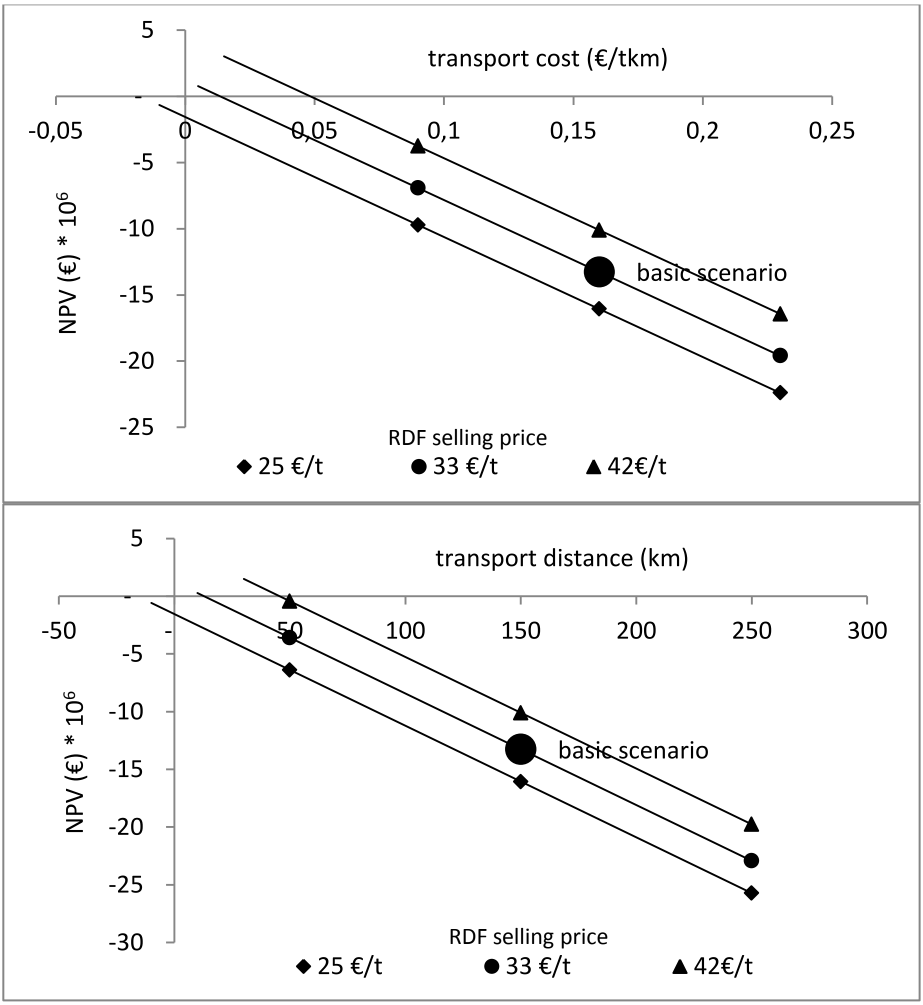
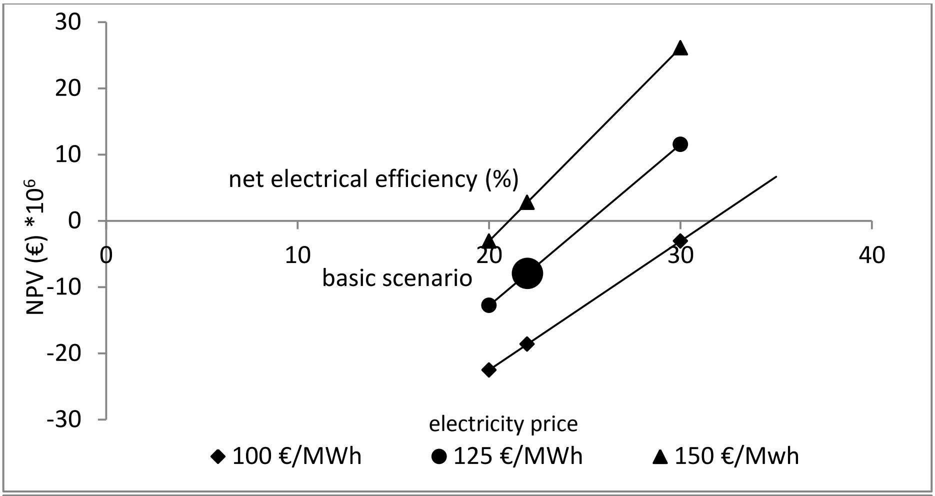


Figure 7.14: The impact of variations in transport distance (top) and transport cost (bottom) on NPV for different selling prices of RDF in scenario 1

For Scenario 2, calorific value of RDF, net electrical efficiency of thermal treatment system, and price of electricity become the highest positively contributed parameters to the NPV (Table 7.6). The range of the price of electricity in the sensitivity analysis was decided as follows: according to the announcement of the Public Utilities Commission of Sri Lanka (PUCSL), the list of rates for electricity purchased by the Ceylon Electricity Board (CEB) from Non-Conventional Renewable Energy (NCRE) sources shows that the rate for electricity from MSW is 26.10 LKR/kWh (1€=165LKR) (PUCSL 2012). As this technology is not yet well developed in Sri Lanka, this price was used in this study as the upper margin (150 €/MWh) of the range of electricity price. For the lowest margin, the price of electricity generated by mini hydro plants that are well developed in Sri Lanka (17.15 LKR/kWh, 100 €/MWh) has been used. Thus, the average electricity price used in this study is 125 €/MWh. Figure 7.15 illustrates the relationship between the net electrical efficiency, calorific value of RDF, and electricity price. The top panel of Figure 7.15 shows that for a fixed calorific value (20 MJ/kg), a 10 percent change in electrical efficiency yields 17 percent and 38 percent increments in NPV for electricity prices of 100 €/MWh and 125 €/MWh, while NPV doubles for the electricity price's upper margin (150 €/MWh). This suggests that Scenario 2 (RDF valorisation via incineration) is economically feasible even with the moderate electrical efficiencies (21–25 percent) if the electricity purchase price by the CEB is high, as suggested above. According to the bottom panel of Figure 7.15, for a fixed electrical efficiency (22 percent), 10 percent, 18 percent, and 36 percent gain in NPV can be foreseen for 100-150 €/MWh of price range when the calorific value increases by 10 percent. The figure reveals that positive NPVs can be obtained even for the calorific values of less than 20 MJ/kg when the electricity price is in its upper margin.

7.7. Conclusions

This chapter discusses the feasibility of open waste dump mining in Sri Lanka. The LCA/LCC model developed for ELFM has been applied to a hypothetical open waste dump. The study comprises two scenarios based on the destination of RDF: Scenario 1 includes the direct selling of RDF as an alternative fuel to replace coal usage in the cement industry, while Scenario 2 consists of processing RDF in an incineration plant in order to produce electricity. The LCA analysis reveals that both scenarios yield higher environmental benefits compared to the 'Do nothing' scenario. The environmental burden due to waste transportation is fully compensated by the avoided burden resulting from the replacement of production and transportation of coal in Scenario 1 and electricity generation in Scenario 2. More than 1.6 million tonnes CO_2 equivalent of GWP that occurred in the Do-nothing scenario can be eliminated by the discussed scenarios. The LCA study concludes that starting the waste valorisation during the early stage of waste degradation of a dump site is beneficial in GWP's viewpoint. The sensitivity analysis concludes that RDF recovery efficiency, calorific value of RDF, and electrical efficiency of thermal treatment system are the most important parameters from an environmental point of view.



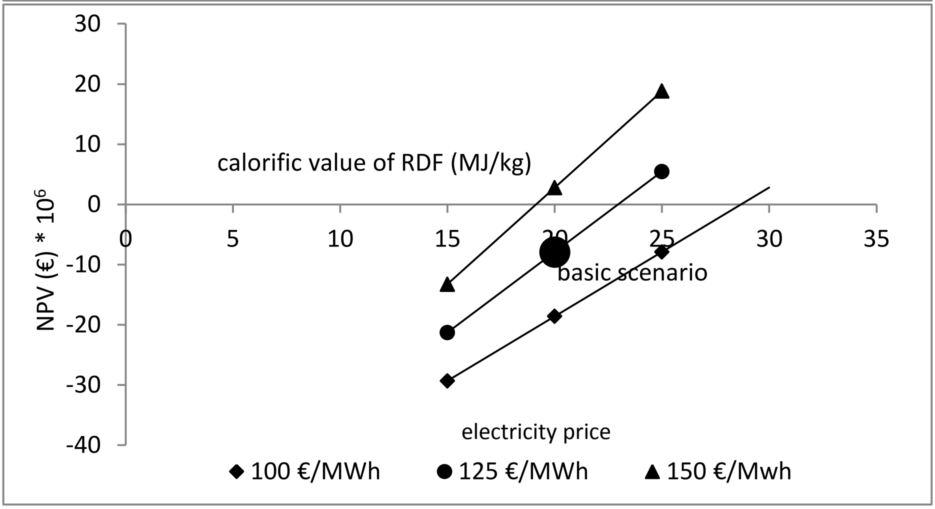


Figure 7.15: The impact of variations in net electrical efficiency (top) and calorific value of RDF (bottom) on NPV for different electricity prices in scenario 2

The LCC analysis shows that none of the scenarios are beneficial economically within the data used for the analysis; nevertheless, Scenario 2 performs better than Scenario 1 in this regard. The analysis further highlights the necessity of avoiding waste transportation in order to obtain economic profits. Overall, the study concludes open waste dump mining is feasible from an environmental point of view. To realize open waste dump mining economically, technological improvements or governmental support will be needed. The environmental benefits can be used to motivate the development of financial support instruments for open waste dump mining. Finally, further research is needed to investigate the possibility of developing the ‘open waste dump mining’ concept as a clean development mechanism (CDM) project.

Limitations of the model structure

The key objective of this research is to perform an environmental and economic evaluation of the novel ELFM concept. For identifying the possible and essential components of ELFM, the focus was mainly on the so-called Closing the Circle (CtC) project of Group Machiels, Belgium: the first complete ELFM project. As a consequence, the development of the LCA/LCC assessment model evolved from very specific to generic characteristics. In fact, landfill mining is different from case to case due to the different geographical conditions, waste composition of the landfill, age of the landfill and also due to the available technologies at the time of mining. In order to assess the environmental impact and economic feasibility of a new ELFM project, one common set of model parameters would be helpful taking into account the above mentioned different characteristics of each single landfill. Owing to these complications, it is extremely difficult to develop a general model that can be used in all landfill mining case studies. The model used in this study basically addresses the geographical conditions, waste compositions, exploitation technologies and material prices found in Europe. To use the model for a landfill which is situated in another part of the world, it is essential to include more exploitation technologies and then to broaden the model parameters. This can be done by analyzing more real cases such as pilot studies. As explained by Frändegärd et al. (2013), the databases related to landfill characteristics, material compositions and the efficiencies of material separation and recovery can be set up and made them accessible within the model. These databases can be used in feasibility assessments of ELFM projects during their planning stages. Currently, the developed environmental model relies on existing data from LCA databases such as Ecoinvent. As ELFM is a recent innovation in waste management, most of the required data however are not included in the available LCA databases. Therefore, use of evolved databases based on more realistic case studies will create more accurate model results. The complex interactions of the ELFM system generated a considerable amount of input parameters as mentioned in Figure 3.3. Although these parameters can be changed to a certain extent according to the nature of the ELFM project to be analysed, the approach should still be general to alter, add or remove different components and parameters dependent on case-specific conditions. Moreover, further critical investigations on landfill characteristics, best available exploitation and recycling technologies as well as the fluctuations of the market prices of recycled materials and energy are absolutely necessary to improve the simplifications and flexibility of the model.

Uncertainty and quality of data

Within each stage of the analysis, uncertainty can result from any number of factors, including insufficient data, an incomplete understanding of the physical or economic process being modeled, model specification, and the inherent uncertainty in the results of any statistical analysis. A thorough treatment of all sources of uncertainty is beyond the scope of this work. Therefore, the remainder of this discussion focuses on the uncertainties associated with the valuation of changes in environmental and economic impacts.

Uncertainty is mainly associated with the input data. We used three types of input data for this study: (i) waste composition of the landfill, (ii) materials and energy flow of processes and (iii) costs and revenues of the processes and products. As our analysis mainly focused on a specific landfill (REMO), all the waste composition data were obtained from the relevant characterisation studies specifically performed for that landfill (Quaghebeur et al. 2013, Spooren et al. 2013). As the data come from one landfill site, the uncertainty level is comparatively low. This justification is in agreement with the recent studies performed for the same landfill site: Winterstetter et al. (2015) worked with very low levels of uncertainties while Van Passel et al. (2013) used waste composition data without uncertainty ranges.

Data gathering for the ELFM processes has been done based on the lab scale and pilot scale experiments and also on the available literature. Recovery efficiencies, materials and electricity consumption of the separation processes were mainly according to the separation tests performed specifically for the REMO waste and to the experts of the separation industry (Busschers, Galoo). This study also includes processes that are not commercially proven yet such as plasma gasification and its residue valorisation techniques. Due to the insufficiencies of data of these processes, emission data, auxiliary material inputs and energy consumption data were obtained from the pilot scale experiments performed by Advanced Plasma Power (APP) for plasma gasification and from lab scale experiments for residue valorisation. This may results in a high level of uncertainties. Moreover, uncertainties can also be originated due to selected costs and market prices. For example, costs for potential treatment of fine fraction are largely depending on its level of contamination and thus on the landfill's specific composition.

We addressed the uncertainties associated with above input data in two different ways. Firstly sensitivity analyses were performed by identifying the most affected areas of the environmental analysis. For example, as the thermal treatment and separation processes became the highly contributing processes to the total environmental impact, parameters related to those processes such as recovery efficiencies, calorific values, electrical efficiencies, etc. were subjected to sensitivity analyses. This approach led to identify how the impacts of the worst-case and best-case deviate from that of the most likely case. The other approach was the scenario analysis which estimates possible range of outcomes for the innovative technologies against the conventional ones. Scenario analysis was mainly applied for thermal treatment and its residue valorisation. In addition, to address the uncertainty of the data in the economic analysis, a Monte Carlo approach which is simulating a distribution of the results by randomly drawing from the probability distributions of input variables and repeating the analysis numerous times has been used.

All these analyses were based on the available data in hand mainly obtained from the pilot and lab scale studies and the relevant experts attached to the CtC case. However, to broaden the objectives of this research beyond the CtC case, thorough analysis and collection of data are necessary. This should include the data related to waste composition of landfills in different geographical areas, materials and energy consumption of processes according to different industrial references and costs and market prices according to continental differences. Meta-analysis is proposed to combine the data or results from a number of different studies to estimate a more general model or to characterize the range or distribution of key input variables. Furthermore, when considering the economic data, time scale is suggested to be taken into account to avoid the time gap, for instance, data from 2005 and 2010. To minimize this uncertainty, a possible inflation rate can be applied. As we used unit prices (€/t) for investment costs economies of scale cannot be taken into account directly. This issue can be solved by introducing scaling factors. However, it does not affect the results as we assumed similar production lines. Learning Effect leads to fall in the cost of production per unit because with the increased involvement in the production process labor and managers become more and more familiar with the production process. Learning effects can be incorporated with operational costs, hence use of unitary investments costs is not problematic in this study.

Effect of chosen methodologies and assumptions

Besides uncertainties resulted due to input data, chosen evaluation methodologies and related assumptions also create uncertainties. In this study, ReCiPe and IPCC GWP methods have been used as impact assessment methods. The main reason to use ReCiPe was that ReCiPe addresses a wide range of both midpoint (problem oriented) and endpoint (damage oriented) impact categories. ReCiPe endpoint method provides aggregated results in one common unit (single score: environmental points) which makes easier to compare them with overall economic performance of ELFM scenarios. However the methodology of providing single scores results in higher uncertainties compared to ReCiPe midpoint method. Among the impact categories of ReCiPe, a category such as "spoiled groundwater resources" has not been considered. This is somehow a drawback of our selection as groundwater pollution can be one of the major impacts of landfills (Manfredi et al. 2009). The impact categories related to acidification, ecotoxicity and eutrophication include the contaminants coming through the ground water but not to a full extent. Nevertheless, the considered REMO landfill in the CtC case has the required protection layers and leachate collection and treatment facilities, which avoid leaching to the ground water table. Hence omitting above impact category would not be problematic. In contrast, the considered dump site in case study in Sri Lanka contains no protection layers or leachate collection and treatment systems. In such a situation, impact to ground water resources is not fully assessed through the ReCiPe method.

The IPCC GWP method has been used where GWP becomes the priority impact category. GWP is calculated over a specific time interval, commonly 20, 100 or 500 years while commonly a time horizon of 100 years is used by regulators. GWP is expressed as a factor of carbon dioxide whose GWP is standardized to 1. The GWP value depends on how the gas concentration decays over time in the atmosphere. A gas which is quickly removed from the atmosphere may initially have a large effect but for longer time periods as it has been removed becomes less important. Thus methane has a potential of 72 over 20 years but 25 over 100 years and 7.6 over 500 years (Forster et al. 2007). Hence, user related choices such as the time horizon can greatly affect the numerical values obtained for carbon dioxide equivalents. For a change in time horizon from 20 to 100 years, the GWP for methane decreases by a factor of approximately 3 (IPCC 2013). In our study, We used GWP method to assess the impacts associated with plasma gasification, plasmastone valorisation and the Do-nothing scenario of case study in Sri Lanka. The effect of time horizon affects the results of the latter case as it includes higher levels of methane emission while the results of the first two cases are not much affected as their emissions mainly contain CO_2 but not methane.

The results considerably vary according to the chosen impact assessment method. For instance, in chapter 6, environmental performance of plasma gasification process is not beneficial from GWP view point but becomes beneficial when considering all impact categories included in the ReCiPe method. These differences between the methods should be taken into account when the results are used in policy analysis.

European normalisation and weighting factors were used throughout the study. Normalisation makes all impact categories comparable with each other. However, applying European normalisation disqualifies a number of impact categories which may not contribute significantly on the EU-level but may be still important worldwide, locally or for a specific group of actors. For instance, in the CtC case for total waste valorisation, the climate change impact on eco systems and the impact on natural land are 5 times higher for EU normalisation than for world normalisation. On the other hand the impacts on human toxicity and particulate matter formation are 1.5 times higher for world normalisation. Due to lack of data for regional normalisation references for other regions such as Asia, same EU normalisation has been used in case study in Sri Lanka as well. This approach may cause severe implications in most impact categories. Although development of normalisation references for different geographic areas as explained by Stranddorf et al. (2005) would solve this issue, it is beyond the scope of this study.

Comparison of Results

In this study, thermal treatment process was identified as the major driving force of ELFM both from an environmental and economic point of view. Moreover, produced electricity as well as metals are recognized as main performance drivers on environmental benefits side and the flue gas emission and the oxygen usage within the plasma gasification process as main drivers on environmental burdens side. This is in line with the recent studies of Winterstetter et al. (2015) and Jain et al. (2014). However, the factor of benefits due electricity to benefits due to metals is 5 in the present study while in the study of Winterstetter et al. it is only 3. In contrast, Jain et al. claimed that benefits coming from metals are 3 times higher than benefits due to thermal treatment of RDF. Nevertheless, it should be noted here that both authors considered only the GWP impact instead of assessing a number of impact categories as performed in the present study.

From economic point of view, the present study confirms the findings of Van Passel et al. (2013): economic performance is mainly dependent on parameters concerning energetic valorisation. These parameters include net electrical efficiency, calorific value of RDF, price of electricity, price of green certificates and investment cost of thermal treatment process. In contrast to Van Passel's study, the present study includes all possible activities that can be conceived within a ELFM project. However, metals do not play an influencing economic role in present study while it became one of the economic drivers in the study of Winterstetter et al. Waste composition of the landfill is the major reason for these different results. The present study and the study of Winterstetter et al. dealt with the same metal composition but the present study used lower overall metal recovery efficiencies and metal prices. In contrast to REMO landfill, the landfill considered in the study of Jain et al. contains higher metal percentage (11% vs 5%) which leads to higher benefits from metal recovery compared to energetic valorisation. The lower metal concentration of REMO landfill is due to the efficient waste separation at waste generation sources in Belgium before disposal. Thus, Higher benefits due to recovery of metals can be expected in the landfill mining projects where the waste separation at the source is not well developed. However, our analysis performed for a dump site in Sri Lanka also implies that metal recovery is not a performance driver both from environmental and economic point of view although Sri Lanka has a very fragmented waste management system. This is due to the scavenger effect that often can be occurred in open waste dumps. Initial metal concentration decreases over the time as collection of metals is one of the major incomes of scavengers.

Apart from thermal treatment and separation processes, reclamation of land also showed environmental benefits in our analysis. However these benefits are limited to only on impact category of natural land transformation. The studies of Van der Zee et al. (2004) and Hull et al. (2005) stress reclamation of land as a possible economic driver for landfill mining, especially in densely populated regions where the value of land can be significant. Nevertheless, in this study, reclamation of land is not economically significant as the land value is comparatively low for lands for nature reserves than for lands for housing, agriculture or industry. The recent study of Frändegård et al. (2015) also highlight that energetic valorisation has a higher importance for the economic outcome while the contribution of land reclamation is comparatively low.

As all the performance drivers of ELFM belongs to the thermal treatment process (plasma gasification), a detailed analysis has been performed in order to benchmark plasma gasification against incineration, a commonly used thermal treatment method in waste processing. Plasma gasification process is more efficient than conventional incineration in converting the energy content of the waste to electricity. Therefore, although both processes give rise to the direct emissions of carbon dioxide from the waste conversion plant, plasma gasification process displaces more conventional electricity generation and is therefore associated with significantly lower lifecycle GWP emissions. Moreover, recovery of metals from the residues is higher than in incineration due to intrinsic advantage of metal recuperation capacity of plasma gasification process (see chapter 6) further reducing the GWP by replacing primary metal production. Incineration shows a higher GWP impact compared with the plasma gasification process, mainly due to the lower net electrical efficiency of the incineration plant. The comparison can alternatively be framed to show that an incineration plant must achieve at least 29% net electrical efficiency to display the same environmental GWP impact (i.e. kg of CO_2eq per kg of RDF treated) as the plasma gasification. Such high efficiencies of more than 30% are also reported with the improvements for energy recuperation (Van Berlo 2010). However it was not reported whether the plant is continuously operating under full load. These results are in agreement with the recent study of UCL (2014) which was used similar electrical efficiencies as in this study. In contrast, the recent analysis of Winterstetter et al. (2015) shows better environmental performance in incineration than in plasma gasification when 30 % and 32% of electrical efficiencies are used for incineration and plasma gasification respectively. The efficiency of the incineration plants mainly depends on the steam turbine size. Hence it is not realistic to compare a certain scale required for a landfill mining project with the best performing scale on the market. Although it was proven in this study that application of plasmastone valorisation options reduces the environmental burdens of plasma gasification further and makes the process more better than the incineration process, it should be noted here that residues of incineration could also potentially be used in high-grade applications other than as aggregates (Bosmans et al. 2013). However, the basic objective of this study was to benchmark plasma gasification (basic scenario- without plasmastone valorisation) with an existing installation (incineration- without bottom ash valorisation) currently relevant and then to investigate how the burdens of plasma gasification can be reduced by applying residue valorisation options.

Policy analysis

This study gives some important insights to make policy decisions for both the CtC case and future ELFM projects. Firstly, the study proves that the type, phase, or average age of the landfill determine whether ELFM is more beneficial than the Do-nothing scenario. The environmental impact of landfills totally depend on the type, phase/age and its pollution control installations. Landfills which contain more MSW produce higher levels of landfill gas including methane. This gas generation gradually increases, becomes stable and then decreases over time. If ELFM is started during the initial phases of the waste degradation, considerable landfill emission can be avoided . Hence higher benefits in GWP aspects can be obtained when the ELFM starts before the maturation phase.

The analysis for the CtC case implies that waste composition of the landfill directly determines the project's profit and loss both from an environmental and economic point of view. As the thermal treatment is the most contributing ELFM process, the amount of RDF present in the landfill is a key factor to decide benefits and burdens of the project. MSW valorisation shows higher burdens and benefits and also higher net impacts due to higher RDF content present in the MSW than in IW. This suggests to valorise IW together with MSW in order obtain higher environmental and economic benefits.

It is a clear conclusion that the thermal treatment process, in this case plasma gasification is the driving force of ELFM. However, the immaturity of plasma gasification process may create higher levels of uncertainties and technical, legislative and institutional barriers for implementation. The situation is the same for the plasmastone valorisation techniques. Despite the immaturity, plasmastone valorisation options decide how and to what extent the plasma gasification process is environmentally and economically beneficial. Referring to Figure 6.7, scenario 8 (plasma gasification with blended cement block production out of plasmastone) is the economically best scenario while scenario 5 (plasma gasification with inorganic polymer block production) is the environmentally best scenario. Implementation of scenario 8 is favorable from company's view point. However, the government may introduce extra taxes to compensate the environmental burden associated with this scenario. On the other hand, if the government stresses to implement scenario 5 instead of scenario 8, higher subsidies for green certificates and higher electricity prices should be offered. The price of green certificates should be increased from 117 €/MWh to minimum 147 €/MWh in order to implement scenario 5 with economic profits. Moreover, the electricity price is needed to be increased by 20% to obtain economic profits from scenario 5. Similarly, in the case study in Sri Lanka, the environmental benefits can be used to motivate the development of financial support instruments.

According to the above discussions following conclusions can be drawn from this PhD research:

- Environmental and economic impact of ELFM depends on the type and composition of the waste, chosen process technologies and quality and the quantity of the output products.
- The type, phase, or average age of the landfill determine whether ELFM is more beneficial than the Do-nothing scenario.
- Thermal treatment (plasma gasification) is the process that has the greatest influence, both from an environmental and an economic point of view.
- The environmental burden of thermal treatment process can be largely reduced by using residues in production of higher value building materials.
- Besides energy production, plasma gasification is capable of producing building materials out of its residues, which contributes to the WtM component of ELFM.
- Application of ELFM concept in open waste dump mining is environmentally feasible, but the governmental support is needed to make the concept economically achievable.

Further research

This work consists of an evaluation of environmental and economic performance of ELFM by using a model based on LCA and LCC. Currently, the developed LCA/LCC model acts as an exploitation tool to identify the appropriate processes to be used in ELFM according to their environmental and economic performances. The LCC model does not address the externalities and only provides a trade-off analysis between the environmental and economic impacts of different ELFM scenarios. Hence, the model should further be developed by including an advanced integration of LCA and LCC to be used as an optimization tool. Multi objective optimization approach can be used to develop this advanced integration (Wang et al. 2009, Arnette and Zobel 2012, De Schepper 2014). This approach is used where optimal decisions need to be taken in the presence of trade-offs between two or more conflicting objectives. Minimizing cost while reducing CO_2 emission and maximizing performance whilst minimizing energy consumption and emission of pollutants of an incineration plant are examples of multi-objective optimization problems involving two and three objectives, respectively.

Technologies, regulations, and markets for the processes and products of ELFM need to be further explored in order to be incorporated within the model. This allows applying the ELFM concept in a wide range of world areas in spite of geographical, economic, and cultural differences.

In this work we proved the feasibility of ELFM, identified the trade-off between environmental and economic performances, and identified the contribution and sensitivity of different ELFM processes and parameters. However, the study can be further elaborated to form a clear set of emission data of the processes related to ELFM. Hence, it can be calculated several environmental indicators such as global warming potential and acidification potential per valorisation of one tonne of landfilled waste. This can be done for different geographical zones and climatic zones. In this way, a robust database will be available for future ELFM projects to screen environmental and economic feasibility.

Finally, a detailed assessment for plasma gasification is necessary for all its syngas valorisation options including hydrogen and methane production. This allows for the identification of the maximum contribution of plasma gasification within the framework of ELFM.

Appendices

Appendix A: Case study input data

General Data			Sources
Total amount of waste (million tonnes)			
MSW	8.2		Case study
IW	6.9		Case study
Time length (years)	20		Case study
Discount rate (%)	15		Van Passel et al. (2013)
Depreciation rate (%)	5		Case study

Waste composition			Sources
	MSW	IW	
Wastes for selective excavation (%)	0.0	26.5	Case study
Metallurgical slags (%)		10.0	Case study
Pyrite ashes (%)		1.5	Case study
Industrial sludge (%)		15.0	Case Study
Wastes for unselective excavation (%)	100.0	73.5	Case Study
Fines (%)	43.0	62.0	Case study, Spooren et al. (2013)
RDF (%)[a]	33.0	19.0	Case study, Spooren et al. (2013)
Metals (%)[b]	2.8	2.4	Case study, Spooren et al. (2013)
Rocks, glass, slag (%)[c]	10.0	8.3	Case study, Spooren et al. (2013)
Undefined (%)	11.2	8.3	Case study, Spooren et al. (2013)

Vegetation and top soil removal		Sources
Vegetation density (t/m^2)	0.01	Case study
Spraying rate of water to minimize dust (t/m^2)	0,01	Case study
Electricity consumption of wood choppers (kWh/t wood)	20	Industrial reference (De Bruin Techniek)
Diesel consumption of excavators (kg/t top soil)	0.281	Ecoinvent database (version 2.2)
Investment cost (€/m^2)	0.48	Industrial reference (Group Machiels)
Operational cost (€/m^2)	0.90	Industrial reference (Group Machiels)

Waste excavation		Sources
Diesel consumption of excavators (kg/t waste)	0.281	Ecoinvent database (version 2.2)
Investment cost (€/t waste)	1.6	Van Passel et al. (2013), Industrial reference (Group Machiels)
Operational cost (€/t waste)	3.0	Van Passel et al. (2013), Industrial reference (Group Machiels)

Separation			Sources
Recovery efficiencies (%)	MSW	IW	
Fines	80	80	Case study
RDF	80	80	Case study
Metals	80	80	Case study
Glass	80	80	Case study
Stones	80	80	Case study

Electricity consumption (kWh/ t waste)	35		Case study, Busschers, Galoo
Water consumption (t/t waste)	0.25		Case study, Busschers, Galoo
Investment cost (€/t waste)	6		Industrial reference (Busschers, Galoo)
Operational cost (€/t waste)	11		Industrial reference (Busschers, Galoo)
Thermal treatment (plasma gasification)			**Sources**
Calorific value of RDF (MJ/kg RDF)	20		Case study, Quaghebeur et al. (2013), Spooren et al. (2013)
Start-up energy (kWh/t RDF)	269		Chapman et al. (2010), Bosmans et al. (2013), Taylor et al. (2013)
Net electrical efficiency (%)	27		Chapman et al. (2010), Bosmans et al. (2013), Taylor et al. (2013)
Plasmastone generation (t/t RDF)	0.17		Industrial reference (APP)
APC residues (t/t RDF)	0.024		Industrial reference (APP)
Metal recuperation (t/t RDF)	0.01		Industrial reference (APP)
Auxiliary materials			Industrial reference (APP)
Oxygen (t/t RDF)	0.55		
$NaHCO_3$ (kg/t RDF)	4		
Activated carbon (kg/t RDF)	0.2		
NaOH (kg/t RDF)	0.8		
H_2O_2 (kg/t RDF)	0.4		
Urea (kg/t RDF)	1.2		
Emission			industrial feedback (APP)
Carbon dioxide (kg/t RDF)			
biogenic[d]	689		
fossil	776		
Carbon monoxide (kg/t RDF)	0.02		
Particulates (kg/t RDF)	0.2		
Nitrogen oxides (kg/t RDF)	0.42		
Sulphur dioxide (kg/t RDF)	0.08		
Hydrogen chloride (kg/t RDF)	0.02		
Green energy factor (%)	47		Van passel et al. (2013), OVAM (2011), Flemish Directive, 5/3/2004
Investment cost (€/t RDF)	50		Industrial reference (Group Machiels, APP)
Operational cost (€/t RDF)	67		Industrial reference (Group Machiels, APP)
Fines treatment			**Sources**
	MSW	IW	
Fines for direct use (%)	10	10	Case study
Fines need a treatment (%)	90	90	Case study
Composition of fines need a treatment			
Ferrous metals (%)	3	24	Spooren et al. (2013)
Non-ferrous metals (%)	0	2	Spooren et al. (2013)
RDF (%)	14	9	Spooren et al. (2013)
Rest (aggregates) (%)	83	65	Spooren et al. (2013)
Recovery efficiencies (%)			
Ferrous metals	10	10	Case study
Non-ferrous metals	10	10	Case study
RDF	90	90	Case study
Rest (aggregates)	90	90	Case study
Electricity consumption (kWh/ t fines)	35	70	Industrial reference (Busschers, Galoo)
Investment cost (€/t fines)	3		Industrial reference (Busschers, Galoo)
Operational cost (€/t fines)	2		Industrial reference (Busschers, Galoo)
Land reclamation			**Sources**

Investment cost (€/m^2)	0.48	Industrial reference (Group Machiels)
Operational cost (€/m^2)	0.9	Industrial reference (Group Machiels)
Solid waste disposal		**Sources**
APC residue disposal (€/t)	96	ETC/SCP (2012), industrial reference (Indaver)
Other waste disposal (€/t)	10	ETC/SCP (2012)
Wastewater treatment plant		**Sources**
Investment cost (€/t wastewater)	5	Industrial reference (Aquafin)
Operational cost (€/t wastewater)	3	Industrial reference (Aquafin)
Temporary storage		**Sources**
Investment cost (€/t waste)	5	Van Vossen and Prent (2011), Van Passel et al. (2013)
Operational cost (€/t waste)	4	Van Vossen and Prent (2011), Van Passel et al. (2013)
Materials and energy prices		**Sources**
Soil for refilling purposes (top soil) (€/t)	5	Gardiner & Theobald (2012), UPEG
Ferrous metals (€/t)	200	Letsrecycle.com (2014), Eurofer Scrap price
Non-ferrous metals (€/t)	1000	Letsrecycle.com (2014), Van Passel et al. (2013)
Glass (€/t)	6	Letsrecycle.com (2014)
Aggregates (€/t)	10	Gardiner & Theobald (2012), UPEG
Electricity (€/MWh)	68	Van Passel et al. (2013), Eurostat (2014)
Green certificates (€/MWh)	117	Van Passel et al. (2013)
Land (€/m^2)	3	Van Passel et al. (2013), Industrial feedback (Group Machiels)

[a] MSW RDF contains 5.7% paper and cardboard, 35% plastic and rubber, 13% textile, 26% wood, 2% mineral, 0.9% glass, 0.07% metals and 18% fines (Spooren et al. 2013). The composition of IW RDF is not reported.

[b] MSW metals contain 2.2% ferrous metals and 0.6% non-ferrous metals. IW metals contain 1.5% ferrous metals and 0.9% non-ferrous metals.

[c] Rocks, glass and slag fraction contains 4% glass and 4% stones in MSW. In IW these values are 3% for both glass and stones.

[d] Calculated by considering 47% of biogenic fraction

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Danthurebandara, M., Van Passel, S., Van Acker, K.. Environmental and economic performance of plasma gasification in Enhanced Landfill Mining. (Revised version under review)

Danthurebandara, M., Van Passel, S., Van Acker, K.. Feasibility of open waste dump mining in Sri Lanka (Revised version under review)

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