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Analysis of the impact associated with mechanical treatment and landfill disposal of municipal solid waste throughout life cycle assessment methodology

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Abstract

The aim of this study is the evaluation of the environmental performances of three systems for the treatment of municipal solid waste (MSW): a mechanical treatment facility (system 1) and two landfills (system 2 and system 3) in 2018 activities.
They are studied by mean of ISO 14040 (2006) standard and ISO 14044 (2017) following the phases of the LCA study.
The defined functional unit is “the total amount of waste treated in 2018” for every system. The system’s burdens also consider transportation of waste and particularly for system 1 they also introduce final waste disposal and for system 2 and system 3 the capping process. The “zero-burden” assumption is applied to all systems. The inventory analysis is performed considering data that refers to waste treated in 2018 while capping and leachate and biogas or emission in air are descried through allocation. The majority of data used to describe the systems and particularly direct emissions one, refer to primary data collected in 2018.
The model used to evaluate the impact is the CML baseline (2002) method that comprehends eleven impact categories. Characterisation phase results show that the main impact for system 1 is transportation of waste to the final treatment and the final disposal (incineration and landfilling), while for system 3 the main contribution it is the energy consumption of diesel and electricity. System 2 differently has several groups that contributes which are capping, utilities consumption and leachate. The impact categories of the three systems are then expressed per tonne of waste and compared. System 1 has the higher impact for all categories respect to system 2 and system 3 while considering only landfills, system 2 is more impactful for abiotic depletion, human toxicity fresh water aquatic ecotoxicity and eutrophication. Sensitivity analysis is performed to investigate transportation, leachate and biogas emissions, hypothesis on capping materials and system burdens. Moreover uncertainty analysis is performed finding that uncertainty for system 2 and 3 is linked to abiotic depletion and ozone layer depletion while for system 1 refers to marine aquatic and terrestrial ecotoxicity.
The study permits to underline the environmental hotspots related to different systems facilities by using primary data and permits to underline the impact related to closure procedure in the evaluation of a landfill performance.

Summary

CHAPTER 4

Introduction

The production of municipal solid waste (MSW) shows an increase linked to a rise in urbanization and in the world population and consequently, municipal solid waste managements and facilities are improved in years (Leme *et al.,* 2014, Yadav *et al.,* 2017 and Behrooznia *et al*., 2018). Landfilling represent the oldest treatment facility (European commission, 2010) and its characterised by the dumbing of waste in a proper dip in soil. The emissions to it associated are leachate and landfill gas whose characteristic composition defines them as pollutants. To ensure their collection a containment structure is built to separate waste from the environment (Daamgard *et al.,* 2011). Considering mechanical treatment facilities, they can be used to pre-treat the waste before disposal by shredding, sifting, metal separation and biostabilization (Di Maria *et al.,* 2013).
The aim of this thesis is the evaluation of the environmental performances of three different systems used for the treatment and disposal of MSW through life cycle assessment (LCA) methodology. These systems refer to a mechanical treatment facility (called system 1) and two landfills (called respectively scenario 2 and scenario 3). Particularly their environmental profile is assessed considering the MSW treated in 2018 inside the different plants.
The use of the primary data collected permits to avoid the use of literature data which can leads to errors in the evaluation of the system behaviour (Henriksen *et al.,* 2018) since waste treatment and disposal system largely depend on site specific characteristic like waste composition, climate conditions and management systems (Laurent *et al.,* 2014, Buratti *et al.,* 2014). The advantage of using primary data permits to compare the three studies by mean of their differences in management and site-specific features, with particular focus to landfills.
In the first section of study the main features of the life cycle assessment (LCA) methodology are proposed with attention to the ISO standards 14040 (2006) and 14044 (2017) which requirements have been followed in this study, reporting also advantages and disadvantages of this methodology.
Literature analysis showed that most of the studies deal with the usage of LCA as a support tool in the implementation of the municipal solid waste management and it also permits to underline the main features of LCA studies applied to waste systems.
According to LCA methodology, firstly goal and scope definition are established together with the functional unit and systems boundaries. The functional unit is defined as the total amount of waste treated in 2018 in every system. Then life cycle inventory (LCI) is performed considering system's input and output with direct and indirect associated process emissions. The so called "zero-burden"

assumption is applied to all systems, meaning that waste is not associate to any environmental impact when it enters the system. Regarding system 1 (mechanical treatment facility) the associated input processes consist of the transportation of entering waste, utility usage, materials for maintenance and their transportation, while outputs comprehends the transportation of waste exiting the plant, their final disposal in landfills or incineration and water linked to plant operation. While for system 2 and system 3 input process are related to waste transportation inside the plant, utilities, materials for plant activities and their transportation and coverage process (capping) with all materials and activities associated. Outputs refer to biogas and leachate emissions. Particularly for system 2 biogas is released in atmosphere without any treatment operation (due to its low methane concentration is gasses) while for system 3 is burned in flare to reduce methane emissions. Moreover, for system 3 the transportation of leachate to the final treatment plant is considered.

Then the fourth phase of the LCA analysis, the life cycle impact assessment, is performed. The model chosen to evaluate impacts is the CML baseline method (Guinèe *et al.*, 2001b) which comprehends eleven impact categories (abiotic depletion, abiotic depletion fossil fuels, global warming potential, ozone layer depletion, human toxicity, fresh-water aquatic ecotoxicity, marine aquatic ecotoxicity, terrestrial ecotoxicity, photochemical oxidation, acidification, eutrophication). Characterisation phase, through a group contribution analysis, shows that the main contribution to the total impact of system 1 is the output waste transportation and the final treatment facility associated with it. While, for system 2, the main contributions are capping, for most of categories, leachate and utilities. Otherwise for system 3 the main contribution can be found in utility group for all categories with the only exception of abiotic depletion (where capping is the main one). Consequently, the performances of the three systems are compared considering the impact per tonne of MSW treated in 2018. Results shows that system 1 is the most impactful one while considering, only landfills, the impact of system 3 is major than the one of system 2 due to the high utility consumption.

Furthermore, two optional analysis for the LCA methodology are performed. Firstly, a sensibility analysis to evaluate the impact of the main assumption on the three systems is performed. Particularly they investigate the transportation EURO 3 characteristic hypothesis for all systems, then assumption on leachate composition are evaluated for the two landfill systems and only for system 2 on biogas composition. Considering again system 2 a final analysis is performed on capping process and for system 3 the selection of boundaries is investigated. Secondly uncertainty analysis is performed through Monte Carlo analysis showing the category that mainly contributes to uncertainty.

In conclusion the main advantages and disadvantages related to this study are reported together with the summarised steps of the study.

Chapter 1

Life cycle assessment methodology

In this chapter the life cycle assessment methodology is described considering the four phases that compose it following ISO 14040 and 14044. The main focus of this chapter is to identify the main features and objective of the methodology following its phases which are: the goal and scope definition phase, the inventory analysis phase, the impact assessment phase, and the interpretation phase.

1.1 Introduction to life cycle assessment methodology

The increasing attention linked to the environment and human actions towards it, has led to the necessity to express their impact on the environment with particular attention to products and management. Consequently, the LCA methodology was proposed to analyse the potential environmental impacts associated with products, processes, and services, following a life cycle approach. LCA is a scientific methodology that provides an objective evaluation. Furthermore, it is the operative tool of the life cycle thinking which can be used to identify all the hotspots along the entire life cycle this also refers to a “from cradle to grave” approach. LCA can assist in several cases (ISO 14040, 2006):

- supply information to develop decisions for organisation, company or administrative entities.
- Evaluating the indicators that describe the environmental assessment.
- Improving a product by mean of the opportunities that may arise from environmental performances.
- Marketing.

The previous assessments are also underlined by the *European platform on life cycle assessment* where an increasing use of the LCA in industry has been reported with the task of reducing the environmental hotspots across the life cycle of goods and services by improving the product design and the decision making procedure. According to this LCA methodology is a tool to improve the competitiveness of a company's products and its communication. The benefit of LCA is that it provides a single tool that is able to defines insights into upstream and downstream trade-offs associated with environmental pressures, human health, and the consumption of resources (European platform of LCA).

This methodology is promoted by several societies and initiatives. Furthermore, LCA is a standardised methodology and guidelines and criteria for the realization of the study are provided mainly by:

- SETAC guidelines (1993).
- US EPA guidelines (2006).
- ISO standard 14040 (2006) and 14044 (2017).

Considering in particular the ISO (International Organization for Standardization) standard, which are the one concerning this study, their content regards:

- UNI EN ISO 14040:2006: "Environmental management – Life cycle assessment – Principles and framework", in which are reported the principles for a correct life cycle assessment evaluation, application and limits. This standard is aimed for potential users and interested parties.
- UNI EN ISO 14044:2017 "Environmental management – Life cycle assessment – Requirements and guidelines", in which guidelines for the impact evaluation, data type and quality evaluation and interpretation of the results are provided. It is used for the development, management and review phase.

Both will be considered in order to perform this study.

1.1.1 History of life cycle assessment methodology

The origin of the LCA methodology are dated back from the late 1960s and early 1970s. The first studies were initially limited to an energy analysis due to development of the petroleum crisis and the demonstration of non-renewability of this energy source but these studies were mainly focused on a comparative analysis (Klöpffer,2014). At that time the use of a life cycle approach was revolutionary because it permits to evaluate not only the single sub-process but to focus on the main system of interest, by highlighting their critical issue.

The first important application of the methodology was in 1969 by some researchers of the Midwest Research Institute in favour of the Coca Cola company and the study was a comparative one concerning different types of container material like glass, aluminium and plastic.

In the following years the interest in the energetic manner diminished, thanks to a decrease in the petroleum crisis but also new issues arose focusing on the industrial, hazardous and waste management system. (Klöpffer,2014).

During the 1990s one common concern was to use the LCA study in the industrial field for marketing manners and the application of the method became more common. This is one of the reasons why environmental organisations wanted a standardisation of the methodology. SETAC (Society of Environmental Toxicology and Chemistry) was the first that underlined the necessity of realizing a standardization of the LCA methodology indeed it was its merit to initiate a standardisation process which culminated in the 'Guidelines for Life-Cycle Assessment: A Code of Practice'(Klöpffer,2014). SETAC was born in order to promote a multidisciplinary approach to solve problems linked to chemical and technological impact to the environment. The first attempt to develop a suitable LCA-structure was achieved during the SETAC workshop 'A Technical Framework for Life Cycle Assessments' in August 1990 in Vermont, USA. The LCA-structure consisted of three components: Inventory, Impact Analysis, Improvement Analysis.

The definition of LCA proposed by SETAC in 1993 refers to an objective process for the evaluation of the environmental loads associated with a product, a process or an activity. It also includes how this evaluation has to include all the life cycle of the product, process or activity, incorporating the extraction and treatment of raw materials, , the production, transportation and distribution, the use, the re-use, the maintenance, the recycle and the final disposal.

After more SETAC defines the main objective of the LCA as (SETAC, 1993):

- To describe in a complete manner the interaction between a product in its life cycle and the environment;
- to contribute to the understanding of the human activities impact on environment;
- to provide decision−makers with information which defines the environmental effects of these activities and identifies opportunities for environmental improvements.

In these years the first scientific journal papers started to appear in the Journal of Cleaner Production, in Resources, Conservation and Recycling, the International Journal of LCA, in Environmental Science and Technology and in others. Moreover different methods still used today were developed in this period, like for example the CML 1992 one. (Guinée *et al.*, 2011)

After the start of work in 1993 it took seven years for the first series of LCA standards to be published (ISO 14040, ISO 14041, ISO 14042, ISO 14043) since finally in 2006 they were condensed into two standards 14040 and 14044 (2006). Based on these classical LCA standards, new approaches have recently been developed which have led to several "Single-issue-LCAs" (Finkbeiner,2014) like carbon footprinting (ISO 14067) or water footprinting (ISO 14046).

In 2002 SETAC and UNEP (United Nations Environmental Program) launched the life cycle initiative with the purpose to improve the Life Cycle Thinking and to provide tools, data, method and indicators. Life cycle thinking continued to grow in importance in the European Policy and in 2005 the European Platform of Life Cycle Assessment was established with the aim to promote the availability and the exchange of information on LCA.

1.2 The phases of the LCA methodology following ISO standard

Life cycle assessment is defined by ISO 14040 (2006) standard as the compilation and the evaluation through all life cycles of the incoming and outcoming fluxes and the potential impacts associated with a product system. The single phases of LCA will be deeply described in the following paragraphs.
ISO 14040:2006 provides the procedure for the LCA study which is composed by:

1. Goal and scope definition;
2. Inventory analysis;
3. Impact assessment;
4. Interpretation.

The scope definition permits to define, together with boundaries and level of detail, the depth of the LCA study. Life cycle inventory phase (LCI) is the second phase of the LCA and contains the input and output data of the studied system together with their collection phase. The life cycle impact assessment phase (LCIA) is the third phase of LCA and its purpose is to evaluate the LCI results and to understand their environmental impact. Finally, the interpretation phase, which is the last phase of the LCA analysis, considers both LCI and LCIA and provides a summary of results and discussion by providing recommendations following the scope of the study. Following the scope and goal definition LCA can be performed by only LCI analysis excluding the LCIA phase. As previously underlined by the phase description the LCA analysis does not address the economic or social aspects of a product since it is an environmental management technique.

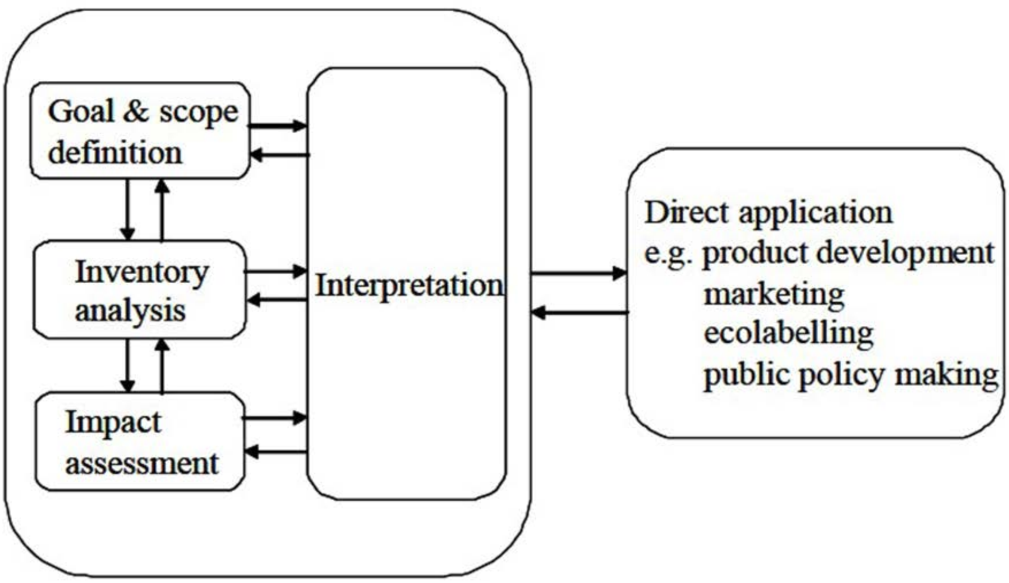


Figure 1.1: Phases of LCA methodology (source: ISO 14040, 2006)

In figure 1.1 are reported the mentioned phase of the study. The use of the two-way arrows permits to underline the iterative of the methodology and so the possibility of rearranging every phase to satisfy a deeper knowledge of the studied process.

In ISO standard, product refers to any good or service with particular reference considering products as hardware, software, services and processed materials, and considering services as an activity performed on a customer-supplied tangible or intangible product, the delivery of an intangible product and the creation of ambience for the customer (ISO 1040,2006). The principle of LCA, that should be used for decisions and planning in conducting the LCA regards the following from a life cycle perspective considering the entire life cycling of a product with particular focus on the environmental aspects of the product, since economic and social aspects are typically outside the scope of LCA. Furthermore, LCA is a relative approach which is structured around a functional unit that defines the object of the study and identifies all the analysis and input and outputs of LCI and the LCIA profile. Principles defines also LCA as an iterative approach where every individual phase is considered while taking into account the other phases of results contributing to the consistency of the study. The LCA study must be transparent and comprehensive of all attributes or aspects of the natural environment, human health and resources and moreover it must give priority to the scientific approach.

1.2.1 Goal and scope definition

The goal definition is the first step in the development of an LCA study. The goal and the application of the study must be unequivocally defined and must contain information about the type of application, the motivations that leads to the study, the type of audience and if the results will be used for comparative assertion or for public divulgation.

Furthermore, the scope must be well defined in order to ensure that the width and the level of detail of the study totally fulfil the goal and are enough to attain it (ISO 14040, 2006)

There are several elements that must be reported in the goal and scope definition phase (ISO 14040,2006):

- The function of the product system, or, in case of comparative studies, of the product systems.
- The functional unit defined as “quantified performance of a product system for use as a reference unit in a life cycle assessment study”.
- The product system to be studied, which is the set of unitary process that leads to the final product and that are connected by elementary mass and energy flows incoming and outcoming the process.
- System boundaries by specifying the unit processes that are part of the product system and the excluded one and considering spatial, functional and temporal limitation.

- The allocation criteria for the distribution of mass and energy fluxes.
- The method for the impact assessment, the impact category and the interpretation used.
- Requirements for the data quality and quantity considering the temporal, geographical and technological coverage, precision, comprehensiveness, representativeness, consistency and reproducibility factor.
- The assumptions.
- The limitations.
- The type of critical re-examination.
- The type of format and report required by the study.

The iteratively of the LCA leads to the possibility of redefining the scope of the study as knowledge of the system increases.

1.2.2 The inventory analysis

The inventory analysis includes the data collection and the calculation procedure which permits to quantify the incoming and outcoming fluxes from the process system. Fluxes may refer to the consumption of natural resources, raw materials, energy or different items linked to the process, but they can also refer to products, co-products, waste, emissions to water, air and soil. Starting from this data is possible to find all the different interpretations concerning the goal and scope of the LCA study. Furthermore, it is the foundation of the impact evaluation.

As all the phases of the LCA study also the inventory analysis is an iterative procedure: a better knowledge of the system leads to the identification of new requirements or limitations concerning data. Consequently, it is necessary to change the procedure for the data collection but also to assess a revision of goal and scope.

Before defining the stages of the life cycle inventory analysis, it is interesting to better define the system product. Indeed, its knowledge may lead to a better comprehension of this stage. Product systems are subdivided into a set of unit processes that are linked one another by flows. The reason why a product system is divided in unit processes is linked to the necessity of facilitating the identification of outputs and inputs to the system inclusive of the elementary flows referring to raw materials and emissions to air, water and land (ISO 14040,2006). The level of detail reached in this modelling phase is defined in order to satisfy the goal and scope definition and refers to system boundaries of the unit process.

The ISO standard defines the mandatory information that must be included in the study: a scheme of the flow diagram of the process, a detailed description of all the units, inputs and outputs, a list of data for all the operative conditions, a list of the unit of measure used, a description of the methodology, calculation procedures and documents of all the possible irregularities of the data collected.

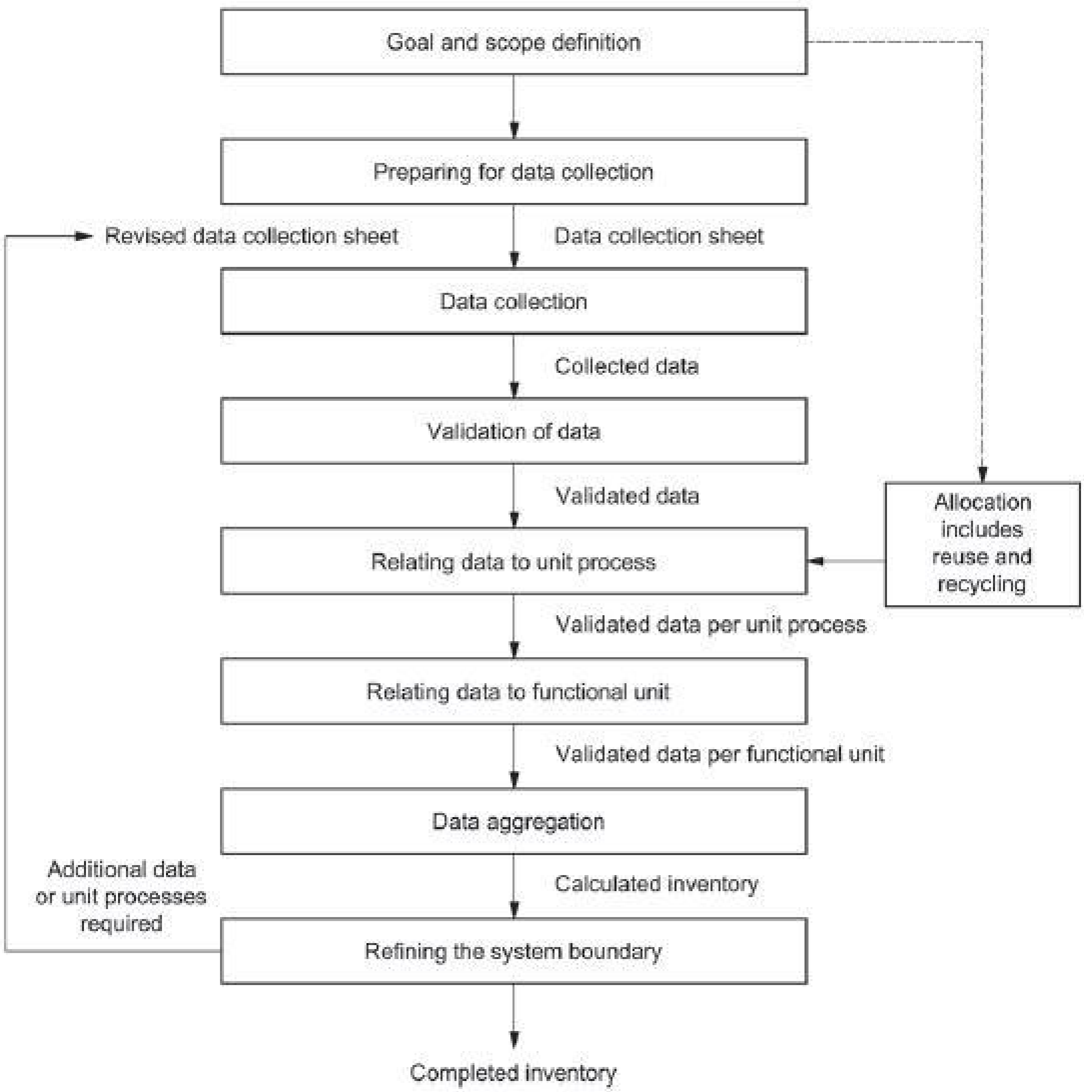


Figure 1.2: Procedure for the inventory analysis: a graphical interpretation (source ISO 14044, 2017)

The inventory analysis can be divided in four phases (ISO 14044,2017):

1. The definition of the scheme of flow diagram which permits a graphical representation the process and its phases.
2. The data collection concerning both quantitatively and qualitatively data that permits to describe all the process units. The goal of this phase is to ensure a complete knowledge of the process units. Data can be defined as:

- Primary data: collected on the process site.
- Secondary data: from the literature or technical manuals (source must be reported).

- Tertiary data: collected by estimate or by technical coefficient (calculation method must be defined); This is usually the most time demanding phase.

3. Review of the system boundaries and definition of the boundary conditions.
4. Data elaboration permits to validate the collected data and to correlate data to the unitary process and to the functional unit. During this phase the allocation procedure can be used to associate to split the incoming or outcoming fluxes of the process or of a product system between the analysed product system and others external.

The previous reported phases can be explained also by looking at figure 1.2.

Regarding allocation the following procedure must be followed (ISO 14044,2017): whenever is possible allocation must be avoided preferring a revision of the boundary system, concerning a division of the process in different sub-units or expanding the system boundaries. In other cases, allocation has to be based on physical criteria and if this is not possible, using an allocation criterion based on economic value.

1.2.3 Impact assessment

The third phase permits to evaluate the potential environmental impacts using the results of the inventory analysis (ISO 14040, 2006). Indeed, the inventory data are associated with specific environmental impacts that are selected thanks to the goal and the scope definition. In particular the categories are related to resource consumption, human health and ecosystem quality.

The impact assessment phase is characterised by mandatory and optional steps. The mandatory ones follow three different steps:

1. Selection of the impact categories, characterisation factors and models;
2. Classification: the results of the inventory analysis are related in different impact categories;
3. Characterization: the potential impacts are calculated by using characterisation factor.

The selection of the impact categories and so of the model shall be consistent with the goal and scope definition. Moreover, considering the characterization models, they reflect the environmental mechanism that relate the LCI results with impact categories results and they are used to derive characterization factors. The previous identification of these elements facilitates the collection, assignment and characterization modelling of appropriate LCI results. Characterisation factors permit the conversion of LCI results to common units and the outcome of the calculation is a numerical indicator result.

In addition to the previous phase there are different optional elements in the LCIA analysis that are reported after:

- Normalization: permits to evaluate how every category contributes to the final impact.
- Weighing: it is used to compare the environmental effects by multiplying for a weight-factor.
- Grouping: sorting and ranking of the impact categories considering process with a certain degree of affinity;
- Data quality analysis.

The last step comprehends additional analysis that can be performed to increase the understanding of the process with particular attention to identifying the negligible LCI results, guiding the LCIA iterative process and underlining important differences. There are three different techniques that can be reported inside this type of analysis: the gravity analysis, the uncertainty analysis and the sensitivity analysis. Gravity analysis or Pareto analysis is a statistical method that permits to evaluate what are the main contributions to a certain category. The uncertainty analysis permits to define how the uncertainty in data and assumption affect the results of the LCIA after propagating through calculations. Finally the sensitivity analysis underlines how the methodological choices affect the impact assessment results. All this further analysis may lead to a reviewing of the inventory results following the iterative method proper of the LCA.

1.2.4 Interpretation

The final phase combines the results of the inventory analysis and of the impact assessment phase, coherently with the goal and scope definition, to draw conclusions and recommendations to improve the environmental performances of the studied system. The objective of this phase is to determine the significant issues, in accordance with the goal and scope definition, proper to the analysed system, by starting from the LCI or LCIA phases results (ISO 14040,2006). The interaction between the LCI and LCIA results permits to draw conclusions on the implications of the methods used and assumptions made, in the previous phases, considering moreover allocation rules, cut-off decisions, selection of impact categories, category indicators and models.

In order to assess the results of the interpretation phase, there are different and meaningful factors that can be identified by an evaluation analysis so composed:

- Completeness check: all the relevant information is considered and if there is a missing one the goal and scope must be redefined;
- Sensibility check: the reliability of the result is evaluated by estimating the data uncertainty and how it declines on conclusions.
- Consistency check: the coherence of assumption, method and data respect to goal and scope.

The conclusion of the LCA analysis refers to the initial problems assessed in the goal and scope definition and recommendations must be the natural consequence of them. Finally the actions, that follow the conclusions, may include technical, economic or social aspects and can go beyond the LCA study, looking to an improvement of the product, system or a new strategy.

1.3 Advantages and disadvantages of the LCA methodology

The results following the LCA methodology can be used as an important tool during the design and decisional phase of a product, process or system. LCA considers a global view of the system following a life cycle approach and analysing all the effects on resources, human health and environment. Indeed, the identification of the critical points or hotspots, thanks to the global and objective approach, permits to set up a less environmental impact system also concerning the possibility of saving resources and producing less waste. LCA methodology can be used not only with this purpose but leads to a competitive advantage and saving money.

On the other hand, limitations are linked to the subjectivity of choices and assumptions that are made during the study. This leads to a less adaptive model concerning the impact assessment and inventory phase. Moreover, this last phase underlines another issuer related to the quality and quantity of data that influence the accuracy of the results. It is important to stress the fact that the inventory phase is the most time and resource demanding one and the absence of a temporal and spatial dimension in the inventory introduces uncertainty linked to a stationary approach. Also, the choice of the system boundaries can set out important unitary processes, introducing significant uncertainties just as the impact category choice. Besides, the LCA analysis does not comprehend all the environmental aspects of a system but only the one related to the goal and scope definition.

Another limitation results from the fact that LCA derives impacts only on global scale neglecting the local effect on the environment. Moreover economic and social aspects related to the system are omitted in the study, but LCA can be used together with other tools like environmental impact assessment, social life cycle and environmental risk assessment or, in the case of a profitability analysis, the life cycle costing.

Chapter 2

LCA in MSW treatment facilities

In this chapter several LCA studies are taken into consideration with reference to their use in the description of MSW treatment facilities in order to better understand the issue of the appliance of life cycle assessment methodology to mechanical- biological treatment and landfills.

2.1 Introduction to application of LCA in MSW treatment facilities studies

Municipal solid waste production has increased over the years and it is presumed to continue growing due to the increasing world population and urbanization. In fact, the actual worldwide production of waste is estimated around 17 billion it is expected to reach 27 billion by 2050 (Karak *et al.*, 2012). Furthermore, the European commission estimate as 16 tonnes the amount of waste produced per year. This condition together with the raising awareness of the importance of environmental protection leads to the necessity of investigating the environmental behaviour of the most common waste treatment facilities. Therefore, to solve this function, LCA has used to evaluate the hotspot correlated to the waste management strategies. Indeed, the use of LCA as a support tool in the waste-treatment decision making has been underlined also by the European commission (2005) with the aim of minimizing the impact through the entire life cycle of products. In his study Laurent (Laurent *et al.*, 2014) underlines how from the first LCA studies, starting from 1995, the numbers of published studies on LCA has increased over years, also driven in Europe by the intensification of the European waste management policy and by the publication of the ISO standard (ISO 14040 and ISO 14044, 2017). Furthermore, with the introduction of the waste hierarchy by the European commission in 2008, in which waste is seen as a secondary raw material and introduce the necessity of distinguish between waste and by-products, calls for the use of life cycle thinking. Even if most of the studies refers to Europe, an increasing number of LCA studies have been investigated in American and Asian territories, in more recent years, respect to the first LCA analysis on waste. In fact, in Asian countries the most used waste management are open dumping of waste and landfilling while incineration is the most used option in developed Asian countries (Yadav *et al*., 2018). Consequently, the development of new MSW management system permits to ensure not only a solution for the increasing waste production but also to ensure that the hazardous emission for human health and environment to be avoided or at least diminished.

The majority of proposed studies regards the comparison between different waste management strategies in order to investigate the best solution in term of environmental performances. These different scenarios can concentrate in a single facility or, often, on different waste facilities that are connected together in succession to provide different treatments for waste and to obtain finally a result, that otherwise cannot be reached in terms of emission. In this prospective and considering landfilling and mechanical-biological treatment of waste which are the main objective to this study, the council directive 1999/31/EC assign the necessity of preventing and reducing the negative effects on the environment in particular on surface water, groundwater, soil, air, and on human health from the landfilling of waste. Indeed the landfill gas contributes to the greenhouse gas emissions in the world (Kormi *et al.*, 2017) and, considering another landfill emission, which is leachate, it is considered as a serious pollutant to natural resources, ground water, human health and hygiene (Naveen *et al.*, 2017). In order to ensure a diminish of pollutant emission, MSW has to be treated before being disposed in landfill and the biodegradable fraction of waste must be reduced before the disposal as underlined also in the studies of Di Maria *et al.*, 2013 and Sauve *et al.*, 2020.

In reference to the above is possible to underline how, inside the European context, increasingly stringent directives on landfilling have been put in places. In this context several studies have been performed to assess how the European policy affects the environmental behaviour of MSW managements, using LCA as evaluation tool. Indeed in the study proposed by Di Maria *et al.,* (2020) the period in between 2007 and 2016 is studied with particular reference to the Italian contest finding an increasing amount of waste processed in mechanical-biological treatment facility (up to 10% in time framework) and a decrease in the amount of waste landfilled (minus 25% about in the time framework) in line with the European policy and its 7th action programme. The results of this study underlines how the targets introduced by the EU policy, considering as previously underlined, the waste hierarchy and the reuse/recycling targets, lead to generally positive consequences both on environment and social aspects dealing with differently waste management strategies (Di Maria *et al.*, 2020). Another study proposed by Wang *et al.,* (2019) evaluate the effects of the European targets on the impact related to the municipal solid waste management in the city of Nottingham, England. The study follows the changes in management action from 2001/2002 to 2016/2017 also considering the future improvements that follows the newest EU directive. Moreover, in this study, an improvement in the environmental performances of the management system is found with particular attention to the global warming potential impact category which diminish both for tonne of MSW and per capita.

In the next paragraphs different meaningful studies selected for their methodological choices are reported. Particularly they relate to the appliance of LCA methodology in MSW management studies in reference to the use of mechanical-biological treatment and landfill.

2.1.1 The organic waste management: an Italian case study

The study proposed by Buratti *et al.*, (2014) will identify the best practice in the waste management strategies of organic material in Italian region of Umbria. It was performed following the four-stage defined in the ISO standard. The goal of the study was to compare the environmental burdens of two common options in Italy: scenario 1: undifferentiated collection, mechanical and biological treatment and disposal of in landfill. Scenario 2: source-separate collection and production of high-quality compost. The software used was SimaPro (7.2) and the method was IMPACT 2002+ considering a 500-year time horizon. The functional unit was defined as one ton OF (organic fraction) treated using different technologies (Buratti *et al.*, (2014)). The studied system comprehended a mechanical treatment facility (MTF) where the waste is firstly processed with metal separation then the dry fraction of MSW is separated from the wet one and disposed in bales in landfill and a wastewater biological treatment plant (Buratti *et al.*, (2014)). The wet fraction is treated in an aerobic biological facility which was located separately from the site (transportation is included in the study). The system's burdens are different for the two scenarios: in the first case are included MTF, WBTP, ABF, landfill, LTP, transportation and energy production while in the second only ABF, compost use, landfill, LTP, electricity production and transportation.

Moreover some assumptions were made: construction, implementation, maintenance or demolition are not included in the study; electricity consumed is drawn from the Italian grid and the one produced by biogas combustion replace the Italian energetic mix one; CO_2 from OF decomposition is considered carbon neutral; organic waste landfilled is not considered as carbon stock. (Buratti *et al.*, (2014)). The data are mainly collected in MTF and ABF plant, referring to 2011 operation and comprehend mass flows, collection quantities and distances, compost recipe, fuel and energy consumption and water emission. (Buratti *et al.*, (2014)). While emission owing to biostabilization, transportation, electricity and fuel consumption came from Econinvent database and literature analysis.

Considering feedstock transportation, in scenario 1 the waste was collected and transport by diesel truck in all the 38 included municipalities, with a calculated specific fuel consumption of 2.46 l/ton. Transport from MTF to ABF happened thanks articulated lorries with a consumption of 0.57 l/ton. In scenario 2 SSOF (source segregated organic fraction) was transported to ABF by diesel full-tracks and considering the distancing between municipalities the consumption was 3.58 l/ton. All the emission factors were considered using EcoInvent database.

Mechanical treatment in scenario 1 only the organic fraction, so 37% of MSW, was accounted for the electricity consumption. Regard biological treatment consisted in four different stages that last 90 days about: mixing the incoming organic waste, aerobic fermentation using a bed system, maturation of compost in outdoor static piles and mechanical refining of compost (Buratti *et al.*, (2014)). For both inventory data energy consumption, emissions, water consumption and chemicals are measured in site. Considering air emission from the aerobic process the main ones are CH_4, H_2S, NH_3, NMVOC and particulate which are directly calculated from OF. Other components were waste productions (refuse and SOF), energy consumption (diesel and electricity), valuable materials (compost) and avoided products (calcium ammonium nitrate, single superphosphate and potassium sulphate).
Landfill was present in both scenarios. The main gases considered in the process were CH_4 and CO_2 while the content of other gases like aromatics, hydrocarbons and H_2S were low. Carbon dioxide emission are considered, as before, biogenic and so they were not included in the inventory. Methane emissions were estimated thanks to the LandGEM (Landfill Gas Emission Model) software using a first order decay equation to define the emission rate due to anaerobic degradation (Buratti *et al.*, (2014)). The methane generation potential L_0 is described as (eq):

$$L_0 = F * DOC * DOC_f * {}^{16}/_{12} * MCF \quad 2.1$$

Where F is the fraction of methane in landfill gas, DOC is the degradable organic carbon, DOC_f fraction of DOC that can be decomposed and MCF the methane correction factor (Buratti *et al.*, (2014)). L_0 result to be 16,5 kg/ton for SOF and 9,1 kg/ton for the refuse. The composition of biogas is analysed thanks to data collected or calculated for the landfill. The biogas is then burned inside a combustion engine in order to recover and energy. The exhausted gasses are discarded to atmosphere.

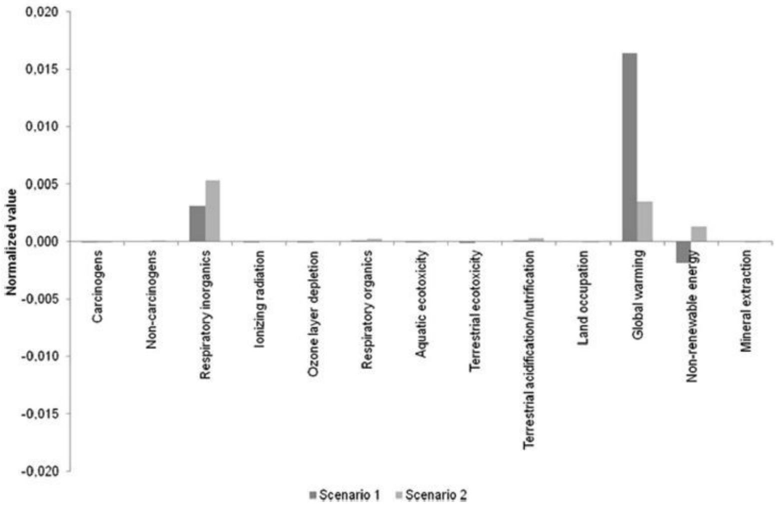


Figure 2.1: Impact assessment results (Buratti et al., 2014)

The method used for impact evaluation is IMPACT 2002+ midpoint and after a normalisation process is performed to define the most relevant impact categories. The method uses the reference value for

Western Europe. The results demonstrate how non-carcinogens, respiratory inorganics, respiratory organic, terrestrial ecotoxicity, terrestrial acidification and nutrification, global warming, non-renewable energy shows the biggest contribution to impact as reported also in Figure (2.1) and Table (2.1). (Buratti *et al.,* (2014)).

Table 2.1: Impact assessment results (Buratti et al., 2014)

Impact category	Units	Scenario 1	Scenario 2
Carcinogens	kg C_2H_3Cl eq	−3.17E-02	−5.05E-02
Non-carcinogens	kg C_2H_3Cl eq	1.43E-01	3.51E-01
Respiratory inorganics	kg $PM_{2.5}$ eq	3.14E-02	5.40E-02
Ionizing radiation	Bq C-14 eq	−5.55E+02	3.91E+01
Ozone layer depletion	kg CFC-11 eq	−9.96E-07	1.47E-06
Respiratory organics	kg C_2H_4 eq	4.12E-01	7.62E-01
Aquatic ecotoxicity	kg TEG water	−6.32E+02	−3.03E+02
Terrestrial ecotoxicity	kg TEG soil	−1.64E+02	5.14E+01
Terrestrial acid/nutri	kg SO_2 eq	1.81E+00	3.37E+00
Land occupation	m^2org.arable	1.12E-02	−5.77E-02
Aquatic acidification	kg SO_2 eq	1.19E+00	3.34E+00
Aquatic eutrophication	kg PO_4 P-lim	−1.88E-03	−5.80E-03
Global warming	kg CO_2 eq	1.62E+02	3.44E+01
Non-renewable energy	MJ primary	−2.87E+02	2.00E+02
Mineral extraction	MJ surplus	2.74E-02	−5.54E-01

Considering non-carcinogens class the impact of scenario 2 is bigger because of the emission of hydrogen sulfide from the biostabilization process while scenario 1 has a smaller impact thanks to the environmental credits due to electricity production (Buratti *et al.*, (2014)). The production of ammonia in the biostabilization phase is the reason why scenario 2 has a bigger impact also for respiratory inorganics while for scenario 1 the contribution is linked to the diesel combustion in transportation. The substances that contributes mainly on respiratory organics for both scenarios are NMVOC (non-methane volatile organic compounds) but also methane plays an important role in this category (Buratti *et al.,* (2014)). Besides for terrestrial ecotoxicity, in both scenarios, the aluminium emission to soil and atmosphere given by diesel and electricity consumption is the most important contribute and so are biostabilization and transportation processes (Buratti *et al.,* (2014)). Also for terrestrial acidification/nutrification scenario 2 has a bigger impact due to the ammonia emission of biostabilization. On the contrary, for global warming the contribution of scenario 1 is bigger than that of scenario 2, owed to methane biogas emission produced by landfilling of organic phase. Finally for non-renewable energy class the electricity recovery from methane gas in scenario 1 permits to reduce its impact. In scenario 2 the main contributions are biostabilization and transportation together with diesel and electricity consumption. (Buratti *et al.,* (2014)).

The damage endpoint categories are then considered for both scenarios for human health, ecosystems, climate change and abiotic resources categories. Considering human health and ecosystem quality damage scenario 2 has the highest contribution to impact (Buratti *et al.*, (2014)). Scenario 1 has the lowest one for resource damage category due to the recovery of electricity from biogas (Buratti *et al.*, (2014)). Human health shows the higher contribution to respiratory inorganics. Furthermore, biostabilization step leads to terrestrial acidification/nutrification as the major contribution, while terrestrial ecotoxicity is the second one because of electricity recover (Buratti *et al.*, (2014)). Finally, resource damage suffers from the use of fossil fuels. The results show that the scenario 2 is less impacting than scenario 1 for greenhouse gasses emission but it shows the worst performances for the other categories (Buratti *et al.*, (2014)).

Table 2.2: Sensitivity analysis (Buratti et al., 2014)

Impact category	Units	Scenario 1				Scenario 2			
		45.3%	55%	65%	75%	45.3%	55%	65%	75%
Carcinogens	kg C_2H_3Cl eq	−3.17E-02	−6.08E-02	−9.04E-02	−1.20E-01	−5.05E-02	−5.24E-02	−5.42E-02	−5.60E-02
Non-carcinogens	kg C_2H_3Cl eq	1.43E-01	1.18E-01	9.22E-02	6.68E-02	3.51E-01	3.50E-01	3.48E-01	3.46E-01
Respiratory inorganics	kg $PM_{2.5}$ eq	3.14E-02	2.76E-02	2.37E-02	1.98E-02	5.40E-02	5.36E-02	5.32E-02	5.28E-02
Ionizing radiation	Bq C-14 eq	−5.55E+02	−7.49E+02	−9.46E+02	−1.14E+03	3.91E+01	2.68E+01	1.47E+01	2.80E+00
Ozone layer depletion	kg CFC-11 eq	−9.96E-07	−1.89E-06	−2.80E-06	−3.71E-06	1.47E-06	1.42E-06	1.36E-06	1.30E-06
Respiratory organics	kg C_2H_4 eq	4.12E-01	3.88E-01	3.63E-01	3.38E-01	7.62E-01	7.60E-01	7.58E-01	7.57E-01
Aquatic ecotoxicity	kg TEG water	−6.32E+02	−1.03E+03	−1.43E+03	−1.83E+03	−3.03E+02	−3.28E+02	−3.53E+02	−3.77E+02
Terrestrial ecotoxicity	kg TEG soil	−1.64E+02	−2.69E+02	−3.76E+02	−4.84E+02	5.14E+01	4.48E+01	3.82E+01	3.17E+01
Terrestrial acid/nutri	kg SO_2 eq	1.81E+00	1.76E+00	1.72E+00	1.68E+00	3.37E+00	3.36E+00	3.35E+00	3.35E+00
Land occupation	m^2org.arable	1.12E-02	−2.16E-03	−1.57E-02	−2.93E-02	−5.77E-02	−5.86E-02	−5.94E-02	−6.02E-02
Aquatic acidification	kg SO_2 eq	1.19E+00	1.16E+00	1.12E+00	1.09E+00	3.34E+00	3.34E+00	3.34E+00	3.33E+00
Aquatic eutrophication	kg PO_4 P-lim	−1.88E-03	−3.04E-03	−4.22E-03	−5.40E-03	−5.80E-03	−5.87E-03	−5.94E-03	−6.01E-03
Global warming	kg CO_2 eq	1.62E+02	1.22E+02	7.97E+01	3.78E+01	3.44E+01	3.18E+01	2.93E+01	2.67E+01
Non-renewable energy	MJ primary	−2.87E+02	−4.48E+02	−6.11E+02	−7.73E+02	2.00E+02	1.90E+02	1.80E+02	1.70E+02
Mineral extraction	MJ surplus	2.74E-02	9.08E-03	−9.47E-03	−2.80E-02	−5.54E-01	−5.55E-01	−5.56E-01	−5.57E-01

The sensitivity analysis, which results are reported in Table(2.2), permits to analyse the assumption made in the study. The first one considered is the methane collection efficiency which is equal to 45,3% and it reflects the current situation of Italian landfills (Buratti *et al.*, (2014)). The studied regards different collection efficiency (55%, 65% and 75%). Increasing the collection fraction leads to an increase in electricity generation and, consequently, environmental impacts decreases particularly for scenario 1 except for respiratory organics, terrestrial acidification/nutrification and aquatic acidification ionizing radiation ionizing radiation.,, For scenario 2 the environmental benefits are less important apart from ionizing radiation and terrestrial ecotoxicity. Another sensitivity analysis was performed in reference of carbon sequestration in landfill and from the use of compost. This contribution refers to a decrease in GHG emissions for both scenarios but it is not always assumed in the previous LCA studied (Buratti et al.). However carbon is partially oxidized over 100 year scale but no information on degradation rate in 500 years are known. The carbon sequestration considering the compost is estimated to be 2%-10%, particularly 6% and this leads to a decrease by 45,2% for scenario 1 and 40,1% for scenario 2.

In conclusion two waste treatment scenarios considering the organic fraction were evaluated: scenario 1 with undifferentiated collection and subsequently biostabilization of OF and its disposal in landfill and source separated collection of the organic fraction and followed by the production of compost, in scenario 2. Results, calculated with the Midpoint 2002+ method, show how scenario 1 has the best performances in ten of fifteen impacts while scenario 2 has the lowest ones considering carcinogens, land occupation, aquatic eutrophication, global warming and mineral extraction categories. This trend is maintained also at endpoint level were scenario 2 has the best performances only for climate change category. Sensitivity analysis made it possible the evaluation of the main hypotheses regarding carbon sequestration and biogas collection in landfill. In conclusion, the study underlines how LCA applied to waste management systems has a great potential in supporting the decisions of planners and companies that manage waste collection, transportation and recovery activities but it also underlines the necessity of the acquisition of primary data to ensure a consistent result in of the LCA study.

2.1.2 Municipal solid waste management for landfilling and composting-landfilling in Iran

The study proposed by Behrooznia *et al., (*2018) has as objecting the identification of the main sustainable MSW management system between two scenarios: the first one consider only landfilling of waste and composting-landfilling for 48% of treated waste, 50,5% landfilling and 1,5% recycled. To accomplish this main objective, the following four specific objectives are identified (Behrooznia *et al.*, (2018)):"1) investigate energy profiles of available MSW management scenarios in the case study region; 2) analyse exergy demand and the main hot spots of MSW management processes via the LCA methodology; 3) analyse the sensitivity of environmental impact categories to the individual independent parameters; and 4) identify potential options for improving the sustainability of MSW management systems in the north of Iran based on the results of energy analysis, LCA and sensitivity analysis". The study is performed considering Rash city, in north Iran, where the management waste system includes composting and landfilling sites distant 10 km and 30 km respectively. The waste production per capita in the region is of 931g every day. The composting process is supported by the organic waste fraction, which is the main component of waste in this region (64%).

Firstly, following the LCA phases, goal and scope definition are settled. Particularly the goal of the study is defined as the evaluation of the energy demand, exergy demand and identification of the main hotspots through LCA analysis (Behrooznia *et al.,* (2018)). The functional unit is defined as 100 ton of MSW. The system boundaries included the production of inputs and MSW generation in the background processes, transportation, separation and composting of the organic fraction and landfilling of the inert fraction. Considering the two scenarios landfilling (L) consist of a landfill where waste is disposed with a daily treatment stream of 350 ton. Furthermore, there is no energy recovery and neither leachate treatment in the considered landfill. The composting-landfilling (CL) system is otherwise described by the transportation of waste to the separation centre where recyclables (1,5%) are separated and transported to market (10 km far). The remaining fraction is divided in inert material (50,5%) that is then send to landfilling (20 km far) and the remaining 48% is separated as organic material and transferred to decomposition hall for composting (Behrooznia *et al.*, (2018)). Composting and landfilling sites of Rasht were monitored during 2017 and primary data regarding inputs and products for this study were gathered by monitoring the and it consider also the waste composition machinery and equipment, as well as amounts of utilities and materials needed for the processing MSW during operations. The data of products included compost production yield and amounts of recyclable materials. Indirect and direct emissions associated with the production of inputs were estimated using LCA databases while direct emissions during the landfilling and composting processes were estimated literature models.

Energy analysis is one of the goals of this study. In order to perform this study each of the scenarios was analysed multiplying the amounts of physical inputs to their associated energy coefficients (Behrooznia *et al.*, (2018)). Energy coefficients are reported in Table (2.3).

Table 2.3: Energy coefficient used (Behroozia et al., 2018)

Items	Unit	Energy equivalent (GJ unit^{-1})	Reference
A. Inputs			
1) Human labor	hr	1.96×10^{-3}	Fathollahi et al., 2018
2) Diesel fuel	L	47.8×10^{-3}	Fathollahi et al., 2018
3) Transportation	t.km	4.5×10^{-3}	Tabatabaeefar et al., 2009
4) Electricity	kWh	11.93×10^{-3}	Mousavi-Avval et al., 2011
5) Lubricant	L	42×10^{-3}	Kitani, 1999
6) Pesticide	kg	101.2×10^{-3}	Mousavi-Avval et al., 2017
7) Machinery	kg	62.7×10^{-3}	Fathollahi et al., 2018
8) Natural gas	m^3	49.5×10^{-3}	Kitani, 1999
B. Output			
1) Organic fertilizer	t	0.3	Mousavi-Avval et al., 2017

The LCI phase is then performed. Transportation is defined as the product between the amount of waste transported and the distance and reported as tonnes plus kilometer (tkm) while data on electricity usage were considered as a combination of two energy sources in Iran: 92% fossil fuels and 8% renewable energy sources. In CL scenario, diesel fuel was mainly consumed by machinery, (loaders, trucks, windrower, bulldozer and shovel) and direct emissions associated to diesel combustion were estimated thanks to Ecoinvent 2.2 database. Considering landfilling process, the estimation of direct emissions to air is performed using LandGem model (version 3.02) while direct emission to water are estimated thanks to literature parameters. Direct emissions to soil, associated to the use of chemicals, are calculated by the LCA database. Similarly, during the composting process the direct emissions were estimated using the emission factors provided from the study conducted by the EPA (2010) (Behrooznia *et al.*, (2018)).

Before looking at the impact assessment results its necessary to compare the two scenarios. Indeed, considering CL scenario the human labour and diesel consumption are higher, caused by the separation phase. Considering transportation, the landfilling system have an higher value due to a major distance from landfilling respect to the composting site while methane emissions associated to landfill are higher than the composting ones. Referring to the energy analysis total energy consumption for CL scenario was 22.54 GJ/100t of MSW, while it was found to be 17.49 GJ/100t of MSW for L scenario (Behrooznia *et al.,* (2018)). Furthermore, results show that transportation, diesel fuels and machinery are in order the main contribution in energy consumption. Consequently a bigger attention to transportation shall be considered in order to reduce energy consumption. (Behrooznia *et al.,* (2018)).

Then the impact assessment of the systems are calculated by using the CML-IA baseline V3.04/World 2000 model in the SimaPro 8.3.0 software. The impact categories of this model include abiotic depletion potential (AD), abiotic depletion of fossil fuels (ADF), acidification potential (AC), eutrophication potential (EP), global warming potential (GWP), terrestrial ecotoxicity potential (TE), ozone layer depletion potential (OLD), human toxicity potential (HT), freshwater aquatic ecotoxicity potential (FE), marine aquatic ecotoxicity potential (ME) and photochemical oxidation potential (PO). Considering the exergy analysis, exergy is defined as maximum useful energy that should be validated within the life cycle framework and should be achieved when the system is compatible with the reference circumference. While cumulative exergy demand (CExD) is a quantitative index which evaluates the quality of energy consumption and it contains also the non-energetic materials and it is the sum of all exergy required to produce a product. Exergy analysis methods are based on life-cycle viewpoint and cradle-to-grave concept and is part of the LCA.

Impact assessment refers to two different contributes: gross impacts, that do not comprehend positive environmental impacts and net impacts that are described by the difference between negative and positive ones. Considering this impact description is possible to underline that L scenario is associated to an amount of positive impacts equal to zero since no outputs are related to this process. On the contrary the compost produced is associated to a positive environmental impact as outcome of this MSW management scenario and the impact associated to CL is subtracted by the compost production (Behrooznia *et al.,* (2018)). Results of the impact assessment are reported in Table (2.4) considering group contribution that refer to direct, indirect and total emissions.

CL scenario has avoided environmental emission in the forms of AD, FE and TE. Avoided emissions account for the substitution of compost synthetic fertilizer with nutrients (nitrogen N, phosphorus P

Table 2.4: Impact assessment result for CL and L systems (Behrooznia et al., 2018)

Impact categories	Unit	CL			L			Difference
		Direct emissions	Indirect emissions	Total net emissions (TCL)	Direct emissions	Indirect emissions	Total net emissions (TL)	(%) $\left[\frac{TL-TCL}{TCL}\times 100\right]$
AD	kg $Sb_{eq.}$	0	-1.48×10^{-2}	-1.48×10^{-2}	0	4.37×10^{-3}	4.37×10^{-3}	−129
ADF	MJ	6.89×10^{3}	1.76×10^{4}	2.45×10^{4}	1.61×10^{3}	1.29×10^{4}	1.45×10^{4}	−41
GWP	kg CO_{2eq}	3.70×10^{4}	6.71×10^{2}	3.76×10^{4}	2.23×10^{5}	7.59×10^{2}	2.24×10^{5}	495
OLD	kg CFC_{11eq}	0	1.86×10^{-4}	1.86×10^{-4}	0	1.52×10^{-4}	1.52×10^{-4}	−18
HT	kg 1,4-DB_{eq}	2.02×10^{2}	3.57×10^{2}	0.56×10^{3}	1.47×10^{3}	4.28×10^{2}	1.90×10^{3}	239
FE	kg 1,4-DB_{eq}	0.36	−30.40	-0.30×10^{2}	0.68	1.55×10^{2}	1.56×10^{2}	−618
ME	kg 1,4-DB_{eq}	3.37×10^{2}	2.90×10^{5}	2.91×10^{5}	1.01×10^{2}	4.34×10^{5}	4.34×10^{5}	49
TE	kg 1,4-DB_{eq}	2.96×10^{-2}	−20.38	−20.35	5.05×10^{-2}	2.12	2.17	−111
PO	kg C_2H_{4eq}	18.10	0.10	18.20	1.41×10^{2}	1.58×10^{-1}	142.00	679
AC	kg SO_{2eq}	3.16×10^{2}	3.28	0.32×10^{3}	2.46×10^{3}	2.85	2.46×10^{3}	669
EP	kg $PO_4^{-3}{}_{eq}$	1.24	0.21	1.45	0.23	0.78	1.01	−31

and potassium K) recovered during the production of compost (organic fertilizer). Results shows how ADF, OLD and EP of CL scenario are bigger than the one of L while, on the contrary, results revealed that GWP, HT, ME, PO, AC of L scenario were higher than those of CL scenario by 495%, 239%, 49%, 679% and 669%, respectively. Particularly the high value of GWP is caused by landfill emissions due to high direct CH_4 emissions in atmosphere without a proper energy recover or treatment. Introducing a gas collection facility will lead to a minor impact and, also, to a monetary benefit (Behrooznia *et al.*, (2018)). Transportation showed significant effects on ME, OLD and FE. The contributions of water, lubricant and pesticide were minimal as they were used in minimum quantities. ADF expresses the amount of fossil resources, consumed during the product life cycle, and reflects the transportation and machinery consumption for both scenarios where the CL contribution is higher than the L one. The difference between L and CL systems is mainly due to higher nitrate emissions during the composting process and refer to a higher emission in EP category. For toxicity impacts the main contribution is the combustion of fossil fuels. The results revealed that transportation and machinery has a great impact on FE, ME and TE in both scenarios. Considering CL scenario the positive contribution related to compost production is significant in the previous impact categories. Finally, L scenario contribution is bigger than CL considering AC category (Behrooznia *et al.*, (2018)). Results shows that AD is mainly affected by transportation followed by machinery and fuel while AD for CL is mainly linked to the yield of compost production. GW also shows the highest sensitivity to transportation in both the scenarios while considering L system, the landfill emission is the second parameter, and for CL system, it is electricity.

After the impact assessment results normalization is performed using the normalization factors available in CML-IA baseline V3.04/World 2000 model showing that the main contribution to the total impact of the CL system are ME, AC and GWP and for scenario L are AC, GWP and PO. Moreover, most of the impact categories in CL scenario were smaller than those of L scenario.

Considering the results of the sensitivity analysis the implementation of the transportation efficiency can significantly reduce the environmental emissions. Considering the exergy analysis results, what can be underline is that for the CL scenario the exergy demand in the forms of non-renewable resources of nuclear and primary energy as well as renewable resources of biomass and water has negative values meaning that the compost production positively contributes to the exergy demand. Moreover, increasing the compost production yield means avoiding environmental emissions and decrease emissions for CL. Finally, a machine management can reduce the emissions associated to diesel and fuel production and use improving the environmental behaviour of this system.

2.1.3 Municipal solid waste management improvement through life cycle assessment: an Indian case study

The goal of this study proposed by Yadav *et al.,* 2018, is to evaluate the best waste management scenarios in the area of Dhanbad City, India thanks to the use of LCA for four scenarios with particular attention to the use of primary data. The area of the study is the amount of municipal solid waste produced by population is 147.06 ton/day of MSW with an average per capita waste generation rate of 0.41 kg/c/day. The waste composition is firstly evaluated using primary data and particularly by sampling the solid waste collected, founding that waste is composed for 37,77% of compostable material, 31,05% of inert waste, 26,5% of recycling material and 4,68% of incinerable (Yadav *et al.,* (2018)). To perform the study a functional unit of 1 ton of MSW with composition founded by sampling results. The first scenario called S1 represents the existing collection and transportation to landfill (10 km far from the city). The second scenario, called S2, corresponds to the actual MSW management where recycling activities in the studied area are included, considering only collection and separation of mixed plastics, metals, and glass products from the waste stream (only 2% of total) (Yadav *et al.,* (2018)). The emission associated to the recycling process are excluded. The uncollected waste (23,5%) is supposed to be open dumped. The remaining 73,5% are collected and sent to unsanitary landfill without energy recovery. S2 is the baseline scenario of this study. The third scenario (called S3) is characterized by the composting of the organic fraction (37,77%) in aerobic way and the rest of waste is transported to landfill without energy recovery due to the small amount of biogas produced and mainly due to the organic fraction of waste (Rajcoomar and Ramjeawon 2017). The fourth scenario consist of recycling plastic, paper, cardboard, metal, glass, waste, and recyclable textile products (28,67%) while organic fraction (37,77%) is sent to composting and remaining waste is landfilled, and since only inserts material are disposed, no energy recovery is assumed.

Then the life cycle inventory phase is performed. Considering the transportation it is assumed the same for S3 and S4 considering that sorting and recycling plant, the composting plant, and the landfill site are at the same place and average distance can be assumed as 10 km. Open burning is assumed to be one of the possible scenarios and it is characterised by the burning of waste and the consequent emission of pollutants in the atmosphere is considered. Indeed, it is recognised as a source of emission of carbon monoxide and particulates along with the dust, dirt, soot, smoke, and liquid droplets (Yadav *et al.,* (2018)). Furthermore, open dumping is another possible scenario which leads to a high possibility of pollution to soil and water (Yadav *et al.*, (2018)). Air emission are calculated by the help of chemical formula of biodegradable MSW and the emissions (CO2,CH4, and NH3) due to anaerobic degradation of biodegradable wastes was calculated and found as 705.1 kg/ ton of CO2, 361.7 kg/ton of CH4, and 7.50 kg/ton of NH3. The emissions of CO, HC, NO_x and $PM_{2,5}$ is calculated from Babu et al. (2014) study. The recycling procedure can decrease the reducing of direct emission of greenhouse gasses, reducing the amount of virgin material being processed and avoiding emissions of CO2 and CH4 and it affects both direct and indirect emissions since waste is not disposed in landfills and there is a decrease in energy and raw materials consumption (Yadav *et al.,* (2018)). The recycling rate for the study is supposed equal to 40% and the indirect emissions associated to electricity and diesel consumption are calculated thanks to Ecoinvent 3 database (SimaPro 2014). Considering compost its composition of N, K and P is taken from literature. Air emissions from composting were estimated using the chemical formula of the bio-waste calculated using the fractional mass composition of the MSW of the study area. Following the Andersen study (2010) it is assumed that CH4 is oxidized by the microorganisms in the upper layer of the compost material. Data leads to the assumption that from 1 ton of wet MSW it can be produced approximately 0.14 ton of residues (Yadav *et al.,* (2018)). Lastly, considering landfilling, in the base case scenario, leachate and biogas are not collected. To evaluate the biogas emission it was considered the amount of organic fraction leading to a significant amount of methane (59.67 kg/ton), biogenic carbon dioxide (25 kg/ton), non-methane volatile organic compounds (0.388 kg/ton) as well as the smaller amounts of nitrous oxide (1.47 g/ton) and carbon monoxide (3 g/ton) (Yadav *et al.*, (2018)). While considering the amount of emission in water they are taken from Samadder et al. (2017) study. S3 is characterised by gas recovery and leachate collection system, while S4 does not comprehend the emissions streams collection since only inter material is landfilled.

The life cycle impact assessment of the four scenarios is performed using CML 2 baseline 2000 method. Results of the characterisation phase are reported in Table (2.5).

Table 2.5: characterization impact assessment results for four scenarios in Yadav study

Impact category	Unit	MSWM scenarios			
		S1	S2	S3	S4
Abiotic depletion	kg Sb eq.	4.20E − 05	4.46E − 07	7.20E − 06	7.15E − 06
Abiotic depletion (fossil fuels)	MJ	2.09E + 02	6.32E + 01	2.71E + 02	2.52E + 02
Global warming (GWP100a)	kg CO_2 eq.	1.24E + 01	9.42E + 03	4.92E + 03	3.43E + 03
Ozone layer depletion (ODP)	kg CFC-11 eq.	2.24E − 06	7.60E − 07	2.73E − 06	2.47E − 06
Human toxicity	kg 1,4-DB eq.	4.76E + 00	2.25E + 01	5.61E + 00	5.50E + 00
Fresh water aquatic ecotoxicity	kg 1,4-DB eq.	2.61E + 00	1.57E + 00	6.54E + 00	5.39E + 00
Marine aquatic ecotoxicity	kg 1,4-DB eq.	1.86E + 04	1.23E + 03	1.86E + 04	1.81E + 04
Terrestrial ecotoxicity	kg 1,4-DB eq.	2.42E − 02	7.87E − 03	3.36E − 02	3.19E − 02
Photochemical oxidation	kg C_2H_4 eq.	4.13E − 03	2.12E + 00	1.20E + 00	8.72E − 01
Acidification	kg SO_2 eq.	6.60E − 02	1.15E + 01	4.63E + 00	4.63E + 00
Eutrophication	kg PO_4^{3-} eq.	1.54E − 02	2.63E + 00	2.08E + 00	1.99E + 00

Results shows that S1 has the higher impact in abiotic depletion due to the use of fossil fuels. The main contribution to global warming potential is landfilling and open dumping to which is associated to high methane emission and consequently S2 is the scenario with the biggest impact. Bromotrifluoro-Halon 1301 is a product of the fossil fuels production and the main cause to ozone depletion. This is the reason why S3, which includes composting and sanitary landfilling, has the higher impact on this category (Yadav *et al.*, (2018)). The open burning scenario is the one to which can be associated the highest contribution in terms of human toxicity category and so scenario 2. Considering the terrestrial and fresh-water aquatic ecotoxicity, the emissions of nickel, arsenic, lead, zinc, mercury, and barium associated to landfill, composting and open burning, leads to a bigger impact of S3 in this category. While marine ecotoxicity bigger contribution is S2. Photochemical oxidation is related to VOCs emission and methane, that together with SO_x due to transportation leads to a bigger impact resulted from scenario 2. Furthermore, S2 correlated emissions of NOx, HCl, SO_2, and NH_3 leads to a higher impact for this category. Finally considering eutrophication S3 scenario is the most impactful one. After the characterisation step normalization is performed underling how global warming potential represent the highest impact among all the 11 environmental impact categories except scenario S1. Marine aquatic ecotoxicity is the second in all environmental categories except to scenario 2. Finally, the results of the impact assessment analysis show that S2 is the less impactful scenario between the one compared in this study and so that increasing the recycling rate would reduce the environmental impact (Yadav *et al.*, (2018)). The highest environmental impact is observed in scenario 2 from landfilling without energy recovery, open dumping, and open burning of mixed waste. While the actual management system is not appropriated considering the environmental impacts to it associated (Yadav *et al.*, (2018)).

2.1.4 LCA applied to the definition of the best MSW management option

The study proposed by Yay (2014) is performed with the aim of analysing three different scenarios and to identify the best waste management strategy for the Sakarya province of Turkey, in terms of environmental impacts, by using LCA. Firstly the composition of waste in this area has been studied for one year to identify the different waste fraction. Results shows that waste is composed mainly by kitchen residues (42,4%) followed by plastic (13.4%), other combustibles (12.1%) and the ash (11.3%) and the average annual moisture content of municipal solid waste is 59.7%. Then the four phases of the LCA methodology are followed. The goal of this study is defined as the comparison between three different management systems, considering as functional unit 1 ton of waste generated in Sakarya area (Yay (2014)). The three scenarios refer to landfilling without gas recovery, which is the baseline case, it represents the actual situation, and it is defined as alternative 1. The second scenario, called alternative 2, is material recovery facility (MRF), where metals, paper/cardboard, glass and plastics are separated and recycled at a 40% rate, while the rest of waste is landfilled 19 km away. The third scenario (alternative 3) refers to MRF with 40% rate and the organic fraction is composted and the rest sent to landfill. Alternative 4 describes the incineration of all waste and the landfilling of the residues while the last one (alternative 5) refers to MRF with 40% rate, the combustible MSW and the rest of plastic and paper/cardboard is transported to the incineration plant, then the residues are landfilled. To perform life cycle inventory analysis SimaPro 8.0.2 database is used to valuate indirect emissions. Transportation of waste refer both to collection process and to transportation to the final facility plant. Considering this 19 km is assumed as distance between every treatment facility and the final landfill disposal and the selected process in SimaPro was 'Transport, municipal waste collection, lorry 21t/CH U' (Yay (2014)). For electricity, a proper mix of different sources was considered in order to describe the actual situation in Turkey. The MRF consumption is assumed as 0.059 kWh/ton (Banar, 2009) and the rate of loss for the considered fraction is equal to 17%, 28% and 5% respectively for paper, plastic and metal. Aerobically composting permits to reduce the moisture content and increase the lower heating value of MWS (Yay (2014)). Compost rate of production is 38% and composition in N, P and K are assumed as at 0.83%, 0.2% and 0.99%, respectively. Air emission of composting are evaluated thanks to the chemical formula proposed by Tchobanoglous et al. (1993) giving to a production of CO_2 and NH_3 1.82 and 0.033 ton/ton bio-waste, respectively while 60% of emissions are assumed to be removed in the biological filter. For the incineration facility energy recovery, incineration and air pollution control (APC 7 is assumed) equipment are considered. Finally considering landfill for scenario 1 no methane recovery is assumed and it is all emitted to air and landfill gas composition is assumed to be composed as 53% CH_4, 38% CO_2, 1.4% O_2, 7.3% N_2, 2 ppm CO, and 425 ppm H2S. In the second scenario, methane is burnt to recover energy and the amount of electricity produced for one ton of waste is estimated as 218 kWh while 30% of the emission. Emissions to air are calculated using LandGEM. the methane generation was set at 100 m3 CH4 per tonne of wet waste, corresponding approximately to 190 m3 landfill gas (LFG) per tonne of wet waste. Emissions to soil are calculated thorough analysis. Leachate produced is assumed to be treated for 80% while 20% leaks to aquatic recipients. Considering the residual of incineration, no gas collection occurs since waste is inert.

Life cycle assessment was performed investigating eleven impact categories included in the CML method and results are reported in Table (2.6).

Table 2.6: impact assessment results for different scenarios (Yay et al., 2014)

Impact	Indicator	A1	A2	A3	A4	A5
Abiotic depletion	kg Sb eq	1.99E-6	1.76E-6	4.74E-6	9.93E-6	6.54E-6
Abiotic depletion (fossil fuels)	MJ	136	121	325	681	448
Global warming (GWP 100a)	kg CO_2 eq	1.84E3	512	−874	346	−1.03E3
Ozone layer depletion (ODP)	kg CFC-11 eq	3.83E-6	3.82E-6	3.71E-6	5.4E-7	2.54E-6
Human toxicity	kg 1,4-DB eq	47.9	42.8	25	20.6	9.79
Freshwater aquatic ecotoxicity	kg 1,4-DB eq	20.8	18.4	20.7	29.8	19.6
Marine aquatic ecotoxicity	kg 1,4-DB eq	7.16E4	6.35E4	6.37E4	8.33E4	5.49E4
Terrestrial ecotoxicity	kg 1,4-DB eq	1.28	1.13	0.568	0.0658	0.0265
Photochemical oxidation	kg C_2H_4 eq	0.405	0.112	−0.0237	0.0143	−0.0748
Acidification	kg SO_2 eq	0.169	0.162	−3.31	0.414	−3.27
Eutrophication	kg PO_4— eq	0.0662	0.057	−1.21	0.181	−1.18

For abiotic depletion, the higher impact can be found in incineration in alternative linked to the consumption of fossil fuels such as hard coal, natural gas and lignite for electricity. Considering GWP the main contribution is linked to the methane emission is scenarios A1 and A2 since methane is not recovered (and in the case of A2 only the 70 % of biogas emissions are collected and burned). Emissions that effects GWP linked to incineration are due to the combustion of fossil carbon and in MSW. For A3 and A5 the prevention of carbon dioxide dinitrogen monoxide releases due to production of compost and fertilizer creates a positive impact for the global warming potential. Ozone layer depletion is linked to emission of the crude oil related process production and the best alternative to reduce this impact is the fourth one. Alternatives 1 and 2 have the highest human toxicity effect due to barium, chromium, lead and nickel produced during landfilling and transportation. For marine aquatic ecotoxicity and fresh water ecotoxicity the emissions of arsenic, PAH (polycyclic aromatic hydrocarbons), cadmium, barium and chromium associated to alternative 4 are the bigger contribution to this category. For terrestrial ecotoxicity Nickel, copper and barium are primary pollutants emitted during landfilling leads to a greater impact of this category particularly for A1 but also for A2. Landfilling causes the most adverse impact on photochemical oxidation due to methane emissions and consequently the higher impact can be found for A1 and A2. Considering A3 and A5 it is possible to notice how this solution, has a minor impact thanks to the production of compost. The major acidifying pollutants are SO2, NOx, HCl and NH3 associated to A1, A2 and A4 scenarios, while in the composting process is associated to a lower impact value for this category. The main causes of eutrophication are nitrogen oxides and phosphate arising from the transport, incineration, and composting procedures in alternatives 3, 4 and 5. Normalization analysis underlines that marine aquatic ecotoxicity and global warming potential are the greater impacts. The final analysis performed is sensitivity that investigates the impact results considering increasing recycling ratios, finding a better performance of alternative 5. A sensitivity analysis is also performed evaluating different impact methods resulting that no chances can be found. Results has underline how alternative 5 is the best scenario option considering environmental benefits. However, this solution may not be performed due to hight investment demand and operation cost in the long term. Furthermore alternative 3 (MRF, composting and landfilling) can also be considered as a favourable option.

2.1.5 A study on the mechanical-biological treatment of MSW

The study proposed by Di Maria *et al.,* 2013 permits to underline the environmental behaviour of a mechanical-biological treatment facility of waste followed by landfilling with biogas treatment, under different work conditions. Mechanical-biological treatment (MBT) can be used to process MSW or residual MSW thanks to mechanical and biological processes to stabilize biological and degradable fraction. This can be done in two ways: by increasing the amount of solid recovered fuels and by increasing the amount of material recovery together with mass reduction and stabilization (Di Maria *et al.,* 2013). In the first case the waste is biologically dried and then mechanically refined. In the second case the waste is firstly shredded, screened and metal materials are sorted in order to separate the organic fraction from the other materials (Figure (2.2)). The mechanically sorted organic fraction is then biologically pre-treated to reduce its reactivity. Several already proposed studies underlines how MBT permits to diminish the long terms-emission (up to 90%) from landfills (Adani et al., 2004; Binner and Zach, 1999; Cossu et al., 2003; De Gioannis et al., 2009; Frike et al., 2005; Komilis et al., 1999; Leikman and Stegmann, 1999; Lornage et al., 2007; Van Praag et al., 2009). In the present study an existing MSW management system was analysed in which the residual MSW is firstly processed in a MBT facility with aerobic treatment of the organic fraction and then landfilling. The studied system is characterised by the first removal of bulk waste and followed by the bags opening and the conveyor to the metal separation. Then waste is screened with drum sieves passing through the sieve holes (100 mm diameter) that permits to collect the organic fraction of MSW stream which is so moved to the aerobic treatment. Aerobic treatment consists of a chamber in which a continuous flux of air, produced by a proper fan, is emitted through the floor. The residence time inside this basin is of about 2 weeks and consequently is posed in static windrows heap a concrete platform for Further Aerobic Treatment (FAT) to reach the proper degree of stability. It is necessary to ensure that anaerobic conditions and/or anoxic conditions are not introduced inside the basin in order to avoid the production of gasses like methane and ammonia and if this conditions are avoided these substances concentration in the exhaust air is usually <1% v/v.

The goal of this study is to evaluate the environmental performances of the system composed by a mechanical-biological treatment followed by the landfill disposal characterised by the a system for the landfill gas that can be burned in flare or in an internal combustion engine for energy recovery. Particularly these performances are evaluated by assuming different residence times in the basin of 0, 4, 8 and 16 weeks of aerobic stabilization and analysed with different amounts of collected gas, 50%, 60% and 70%. (Table (2.7)).

Table 2.7: Different scenarios of the study (Di Maria et al., 2013)

Week	% Of collected gas		
	50	60	70
0	1.1	1.2	1.3
4	2.1	2.2	2.3
8	3.1	3.2	3.3
16	4.1	4.2	4.3

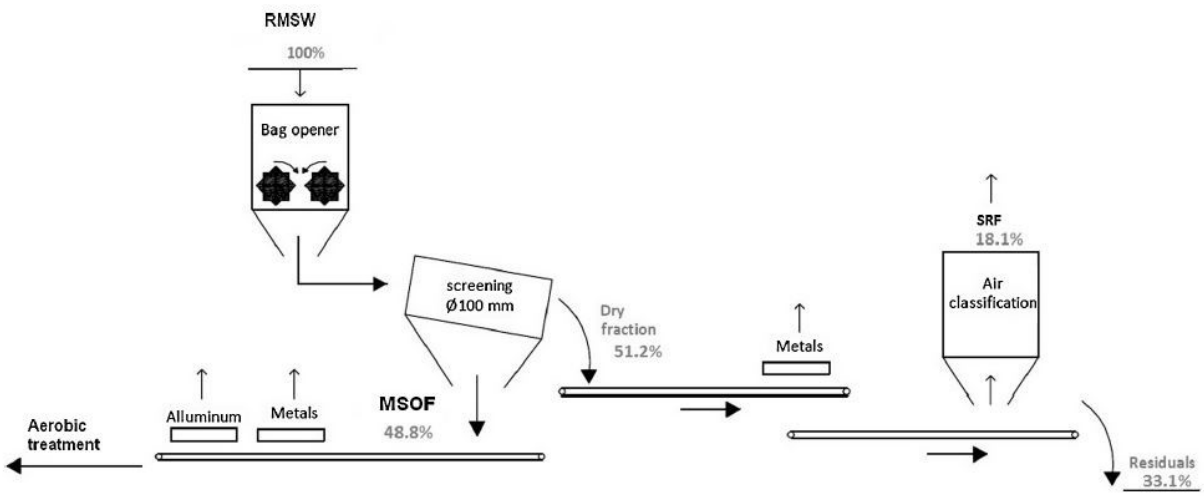


Figure 2.2: Diagram of the mechanical MSW sorting section (Di Maria et al., 2013)

Considering the system inputs electricity used in the MBT operations is considered from the Italian grid, while outputs refer to electricity produced thanks to the landfill gas combustion and emission to air (Di Maria *et al.,* 2013). Since the study is performed with a constant amount of MSW entering the plant the output waste streams (the metal one) are not considered in this study since they are considered as constant. The functional unit considered is the amount of waste entering the system.
The global impact of the scenarios analysed was evaluated using the LCA method CML 2001 (Guinée *et al.* 2001) that consist of the following impact categories: abiotic resource depletion (ADP), climate change expressed as the global warming potential at 100 years (GWP100), eutrophication (EP), acidification (AP), human toxicity (HTP) and photo-oxidant formation (POCP). The results of the impact are reported in Figure (2.3).

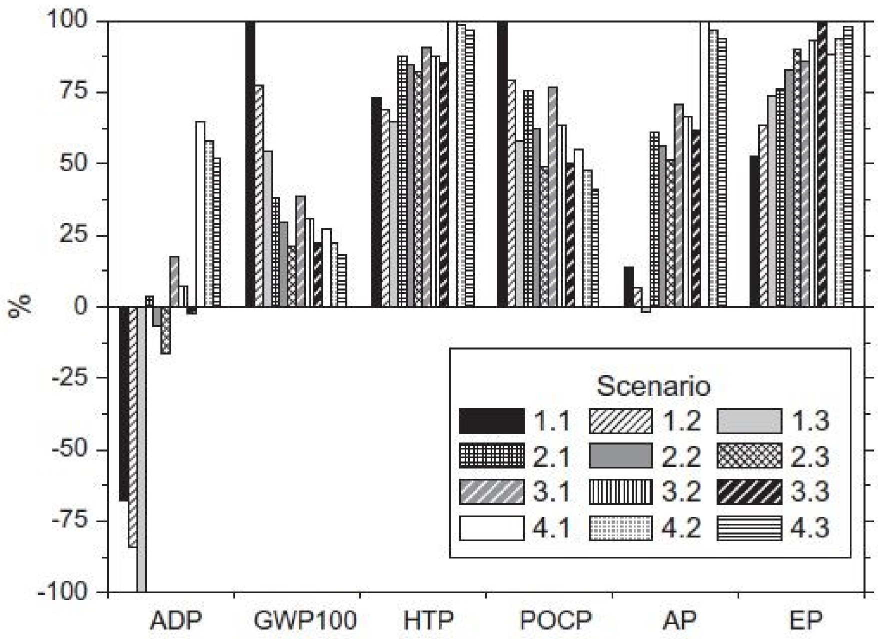


Figure 2.3: Impact assessment results for different investigated scenarios (Di Maria et al., 2013)

Considering firstly global warming potential shows that significant reduction can be found for 4 weeks of pre-treatment (Di Maria *et al.,* 2013). The maximum energy recovery was about 87,000MWh and was achieved for scenario 1.3 and considering scenarios 2.3 and 4.3 the total energy recovered was reduced by 57% while methane and carbon dioxide were reduced respectively of 43% and 15%. Considering the other impact categories ADP, HTP, AP and EP shows lower values for the scenarios from 1.1 to 1.3 because of the positive effects arising from the higher amount of energy production. GWP100, ODP and POCP were greatly influenced by the amount of gases released that is maximum for the lower pre-treatment period (Di Maria *et al.,* 2013). Considering a collected gas percentage of 50%, scenario 4.1 is the one with lower global impact, even if differences with scenarios 2.1 and 3.1 are quite limited. Looking at scenario 2.3 (related to 4 weeks of aerobic pre-treatment and 70% of collected gas) emissions are lower respect to all others. The results permit to underline how impact reduction of strong gaseous emissions was achieved after 4 weeks of waste pre-treatment and longer pre-treatment times showed negligible influence on the gaseous emission impact. Furthermore, it underlines how increasing the amount of energy recoverable from landfill gas positively affected the global impact of the scenarios analysed (Di Maria *et al.,* 2013).

2.1.6 The environmental impacts of municipal solid waste landfills in the European context

The study proposed by Sauve *et al.,* 2020 focus on the evaluation of the impact of MSW sanitary landfills in the European context, focusing also in understanding the environmental implications of landfills and waste directives. The aim of the study is to assess and compare, by a consistent LCA framework, the environmental impacts of landfills varying site-specific conditions (Sauve *et al.,* 2020). Together with this aim the study wants to assess and compare the potential impact on the MSW Europe landfills. Indeed, the possibility of comparing different LCA studies on landfills, thanks to the definition of a proper LCA framework, permits to better underline the differences that arise from site-specific conditions. Whit this in mind results of the study can be used as a support tool in implementing landfill targets (Sauve *et al.,* 2020). The study consists of a contribution analysis conducted on 48 studies regarding landfills to evaluate the methodological choices together with the effects related to landfill gas treatment methods and investigating the difference on climate conditions. To ensure the representativeness of the LCA model to the European context the reference cases of the study are evaluated by analysing the landfill potential emissions (Sauve *et al.,* 2020). A particular attention is placed in the leachate and landfill gas potential emissions that are estimated from landfill climatic conditions and waste compositions. These potential emissions are derived from literature analysis and particularly to estimate the landfill gas emissions the first order decay model is used with reference to the US EPA LandGem model (version 3.02) to evaluate the amount of gas produced by a single tonne of waste. To use the first order decay model, it is necessary to evaluate the methane generation potential (L_0) and the methane generation rate (k). L_0 is the expression of the total amount of methane that can be generated considering the landfilled carbon fraction. Since L_0 can be evaluated for all the different European country they represent a way to compare the potential impacts of landfills in Europe (Sauve *et al.,* 2020). Considering the methane generation rate k, it describes the rate of degradation of the waste and it depends on the climatic conditions and environmental conditions. k values are derived from literature. The possible variation of k and L_0 due to variation in climate condition or waste composition, are not considered in this study. To finally perform the first order decay method, it is necessary to evaluate the waste design capacity and the landfill opening and closure. These values are evaluated from literature and refers to 20 years of filling and an average bulk waste density of 1 t/m^3. Emissions from landfill gas and leachate are evaluated thought experimental model proposed by Manfredi et al. 2010. The reference value for this calculation is L_0=87 [m3 CH4/Mg MSW] and this approach is a simplification that do not consider the other variables. Considering each reference case, 12 additional scenarios are developed to define different variable factors as descried in Figure (2.4).

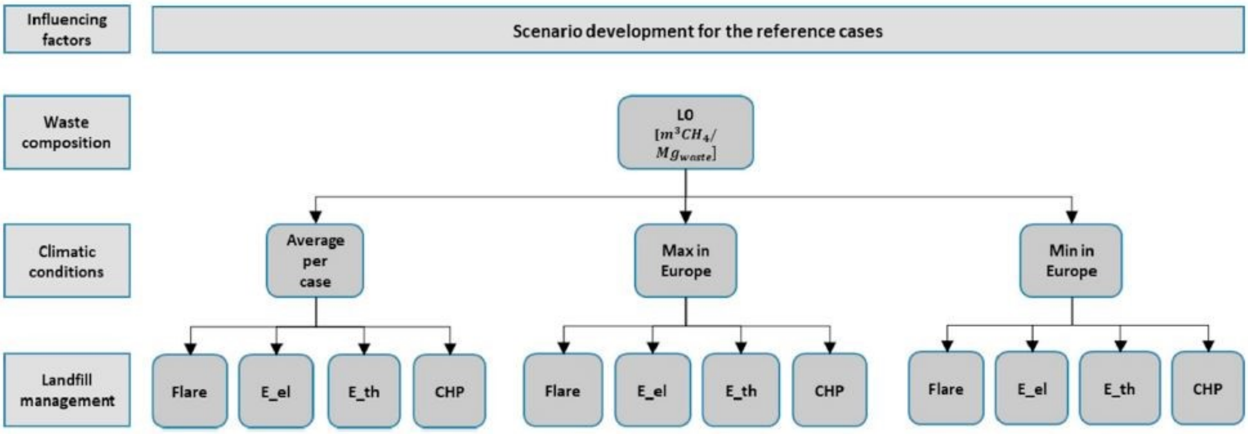


Figure2.4: The 12 scenarios that arises from each reference case, by varying site-specific factors (Sauve et al., 2020)

For each of the proposed case 3 leachate production rates are defined: one for the average, one for the minimum (111 mm/year) and one for the maximum (875 mm/year) values obtained in Europe. Furthermore, different scenarios concerning landfill gas to energy, are considered in the study like heat recovery or electricity production or its conversion to bio-diesel or methanol. Flares are used only when recovery plant are not installed.

The chosen functional unit to perform the study is 1 ton of MSW disposed in a landfill with an average height of 20 m and a waste density of 1 t/m3 (Sauve *et al.*, 2020). The time frame considered for the study is 100 years. The system boundaries comprehend the transportation of waste, the capping procedure and the entering MSW defines the system inputs, while the system outputs refer to on-site operations, together with landfill gas and leachate emissions. Particularly considering landfill gas emissions it comprehends direct emissions, top cover oxidation, flaring, landfill gas to energy and electricity produced. Moreover leachate emissions refer to direct emissions, waste-water treatment plant and landfill residual (Sauve *et al.*, 2020).

The inventory data mainly refers to previous studies and statistical information and, in fact, to evaluate the data for the modelling of landfill was taken from Doka (2009) together with the energy consumption. For the energy mix the average European process were used. According to literature analysis the biogas collection efficiency is assumed to cover a landfill efficiency between 45% and the ideal value 100%, since is affected by the landfill coverage and collection system. On the contrary for leachate a constant collection efficiency of 95% is assumed and the remaining 5% is assumed to be directly released in groundwater (Sauve *et al.*, 2020). After the aftercare period of 30 years, a 0% collection efficiency is assumed for both leachate and landfill gas for the remaining 50 years. For landfill gas flaring an efficiency of 100% is considered while for energy recovery an efficiency of 80% is assumed for heat recovery and 30% for electricity production. Moreover the treatment of LFG for combined heat and power recovery (CHP) is assumed with an electrical efficiency of 30% and a thermal efficiency of 45%. The emissions are calculated thought literature like the methane oxidation potential (Sauve *et al.*, 2020). Considering the emissions after the flare, carbon dioxide is calculated from the methane combustion reaction, obtaining a factor of 2.75 which is then multiplied by the amount of methane emitted [g/m3 LFG]. Considering the waste water treatment plant (WWTP) the model is created using the Ecoinvent database and the documentation by Doka, (2009) in his study. To evaluate the pollutants that are precipitates and ends in the sludge, the percentage that is emitted to water and the one that is emitted to air. The final disposal of sludge is derived from the WWTP is instead considered to be re-landfilled. Considering the life cycle assessment, the ILCD methodology is utilised, defining as impact categories are refers both to toxicity-related and ecosystem-related impacts. Particularly they are climate change (Global Warming Potential), acidification potential Ozone Depletion Potential (ODP), Human Toxicity Potential (HT), Ecotoxicity (ET), Terrestrial and Aquatic (Marine and Freshwater) Eutrophication (EUx). The support software used to perform this study is GaBi which is not widely used in waste management studies and for this reason it is interest its usage in this study. Together with the characterisation (which is mandatory for the studies that follows the ISO 14040 and 14044 standard (2006)), a normalisation phase is performed using the factor recommended in the ILCD guidelines. Considering firstly the results of the contribution analysis the first methodological choice investigated concern boundaries and particularly the introduction of waste transportation inside the system boundaries (Sauve *et al.*, 2020). Indeed, results shows that transportation can describe from 2% up to 85% for climate change category that underlines how a proper municipal solid waste management can be fundamental in defining impacts. For the studied systems it may vary from 0 to 17%. Particularly the categories that are mainly influenced by transportation are in order AP and EUt, EUm and GWP. The infrastructures related to landfill mainly impacts on acidification potential, terrestrial eutrophication, ozone depletion, and the toxicity categories. The influence of infrastructures decreases with the increasing of the L_0 parameter where impacts related to emission in water, air and soil increase too (Sauve *et al.*, 2020). Considering waste composition, it is possible to underline how a variation of it, particularly in the organic fraction, leads to a major landfill impact independently from the landfill management and the climate condition. Scenarios characterised by LFG treatment for energy recovery show higher avoided impacts that are

achieved for higher L_0 values, since more landfill gas is produced. However, this leads to an increase in climate change and ozone depletion impact and the electricity produced is not able to outweigh the impacts, while it able to outweigh impacts on waste water treatment. The influence on the leachate production rate can be seen in EUm, EUf, ET and HT categories. Normalisation is then performed finding that freshwater eutrophication has the hight contribution on the total impact, caused by sludge, and this condition can be linked to modelling choices. Considering other analysis sensitivity is performed on site specific parameter that underlines how flaring efficiency influence particularly GWP category. Results shows also that a minor leachate collection efficiency leads to a smaller impact which is not true and only related to the model choice. Also, the sensitivity analysis underlines that environmental behaviour of landfills refer mostly on waste composition.
Finally, it is possible to assess that climate condition, landfill gas management and waste composition has an impact in the final result (Sauve *et al.,* 2020). Particularly waste composition impacts in all categories, landfill gas management in GWP, AP, EUt, OD, HT, ET and climate condition on HT, ET, EUm, EUf that are in line with literature. The main limitation of the study is the gathering of different LCI, and an uncertainty analysis is not performed in the study to investigate its effects. Furthermore, the study confirms the need to reach the landfill targets described in European directions particularly considering the diminish of the organic fraction that must be disposed.

2.2 Critical literature analysis

In the previous paragraph different studies have been deeply analysed underling the methodological choices. Results that can be used in the evaluation of the following analysis. Literature analysis has permitted to acquire a general knowledge on the most important methodological choices about the description of MSW management systems and facilities. The aim of this paragraph is to identify these methodological choices on several studies that has been selected by mean of their goal and scope, that refer, particularly, on the environmental assessment of MSW systems facility, including landfilling and mechanical treatment facilities. These studies are: Leme *et al.*, 2014, Buratti *et al.*, 2014, Yay *et al.,* 2014, Rajaeifar *et al.,* 2014, Fernandez-Nava *et al.,* 2014, Chi *et al.,* 2014, Ripa *et al.,* 2015, Mali *et al.,* 2016, Hong *et al*., 2016, Coelho *et al.*, 2016, Liu *et al.,* 2017, Nabavi-Pelesaraei *et al.,* 2017, Ayodele *et al.,* 2017, Wang *et al.,* 2019, Di Maria *et al.,* 2019, Lima *et al.,* 2019, Wang *et al.,* 2019, Khandelwal *et al.,* 2019, Sauve *et al.,* 2020. The main features regarding the cited above studies are reported in Table (2.8).
Following the above, the studies are analysed with attention to some methodological choices and particularly to: type of study, mention of the ISO standard (called “ISO” in table) functional unit, impact evaluation method (called “method” in table), other LCIA analysis performed (called “other

analysis” in table), database. Considering firstly the type of study conducted, it is possible to notice that all studies are comparative underlining how the main feature of LCA, perhaps in the last years, is to evaluate the best waste management solution for a certain territory, contrary to single studies, that permits to identify the environmental performances of an existent and static plant or to evaluate its future performances.

Another element investigated refers to the fact that the studies are carried out following the methodology reported in the standards ISO (14040,2006 and 14044, 2017). Only six of the twenty-one studies do not cite the standards demonstrating how this methodology is followed, in most cases, in its main and mandatory points. Moreover, this allows to define a common basis for the comparison of the studies cited above.

Regarding the functional unit defined, most studies uses a unitary functional unit (for 12 studies over 21) expressed as one ton of MSW. Particularly they can refer to single tonne of MSW treated inside the plant or landfilled, or on the contrary they can refer to one tonne of MSW entering the plant. The other functional unit defined are expressed as the total amount of MSW produced in one year in a certain country or town (for 6 studies over 21). The last ones proposed are correlated particularly to study conditions like a specific amount of waste produced (Nabavi-Pelesaraei *et al.*, 2017) or are different from the unitary amount of waste (Behrooznia *et al.,* 2018). Results are also in line with Laurent *et al.* (2014) which underlines how the main proposed functional unit affecting solid waste management’s systems, it is the unitary one followed by generation-based one.

Concerning impact assessment methods there is heterogeneity in its choice, but it is possible to underline that CML method (considering all the different version used) is one of the most used for LCIA phase. However, the other methods discussed, that are Impact 2002þ, ReCiPe, ILCD, are less chosen. Two studies (Liu *et al.,* 2017 and Wang *et al.,* 2019) are related only to the evaluation of climate change impact category referring for impact assessment and characterisation factors to literature studies (Liu *et al.,* 2017) and to the IPCC guidelines (Wang *et al.,* 2019). Particularly only the study proposed by Hong *et al.*, 2016 uses another model (USEtox) to estimate toxicity impact related to reduce the geographic variability of this categories. Only one study uses two different impact assessment method for purpose of sensitivity analysis (Yay *et al.,* 2014).

The majority of the studies proposed other optional analysis after the characterisation one, which is mandatory for those who follows the LCA methodology in ISO standards. Particularly thirteen studies out of twenty-one have introduced a sensitivity analysis while ten out of twenty-one have performed normalization and five have performed both. Only two studies stop the LCIA at characterisation phase (Ayodele *et al*., 2017 and Lima *et al*., 2019). Considering finally the database chosen to evaluate the direct and indirect emissions associated to the system the main used one is the Ecoinvent database.

The majority of the studies refers to the implementation of municipal solid waste management system, comparing different options and finding the best one for a certain territory, which is particularly characterised by mean of climate conditions, waste composition, specific distances between facilities and waste composition. Due to this site-specific condition is not possible to underline a best solution for MSW treatment as also underlines in Laurent *et al.,* (2014) and Buratti *et al.,* (2014). The results founded by Henriksen *et al.,* (2018) stresses the necessity of increase the compatibility between the system context and the LCI analysis and to review goal and scope definition when LCI is developed. Furthermore, considering transportation Ripa *et al.,* (2015) underlines the importance of specific data on transportation (primary data) to ensure that this process does not appear to be over or under rated. Indeed transportation is founded to be an important contribution to the final system impact also in Fernandez-Nava *et al.* (2014), Yadav *et al.,* (2018), Wang *et al., (*2018), Behrooznia *et al*., (2018) which also refers to machinery and fuel consumption. Another aspect that needs to be underlined regards the organic fraction in MSW. In fact, the landfill disposal of undifferentiated MSW leads to an increase in the landfill gas emission and consequently, due to its composition based mainly on methane and carbon-dioxide, on climate change impact category. The implementation of several waste treatment facilities that permits to treat before the landfill disposal, the organic fraction is fundamental to ensure this condition (Di Maria *et al.,* 2013, Fernandez-Nava *et al.,* 2014, Lima *et al.,* 2019, Wang *et al.,* 2019). The mechanical-biological treatment facility follows this concept and its mainly associated as a pre-treatment step in the MSW management system. Considering landfill is possible to notice how the direct emission associated to this system are leachate and landfill gas emission. Indeed, unsanitary landfill and open dumping are the most impactful scenarios due to emission of pollutants in air, water and soil (Leme *et al*., 2014, Fernandez-Nava *et al.,* 2014, Yay *et al.,* 2014, Yadav *et al.,* 2018). Particularly considering landfill gas emission there are two methods used to ensure the treatment of this gasses and particularly the methane fraction: flaring and energy recovery by the use combustion engines. Flaring option is usually avoided from MSW management studies, which usually refer to landfill gas as an energy source, but as underlined by Di Maria *et al*., 2013 its used whenever the fraction of methane in landfill gas is lower than the one necessary to recover enough energy, depending on engine size. Otherwise landfill systems are associated to an energy recovery system to produce both electricity or heat.

Furthermore all the proposed studies in Table 2.8 do not refer on capping procedures (with the only exception of the study proposed by Sauve *et al.,*2020) inside them and the contribution to the final impact assessment results related to the material usage and operations to ensure coverage, insulation and retraining of the landfill area. Indeed, as underlined by Turner *et al.,* (2017) landfill aftercare is an important feature that must be faced in order to reduce the time-scale impact of landfills.

Furthermore Damgaard *et al.,* (2011) underlines how capping can ensure the diminish of leachate formation in landfills and consequently its impact. Capping can also increase the collection ability inside the plant both for leachate and biogas. Damgaard *et al.,* (2011).

Table 2.8: Analysis of several studies related to the management of MSW in order to underline their methodological choices. (A personal rework).

Autors	Type of study	ISO	Functional unit	Method	Other analysis	Database
Di Maria et al., 2013	Comparative study	no	MSW entering the plant	CML 2001	Normalization	*Not menthioned*
Leme et al., 2014	Comparative study	yes	1 ton of MSW entering the system	CML 2000	Sensitivity	*Not menthioned*
Buratti et al., 2014	Comparative study	yes	1 ton of OF treated	IMPACT 2002þ	Sensitivity	EcoInvent
Yay et al., 2014	Comparative study	no	1 ton of MSW generated	CML-IA	Sensitivity	SimaPro 8.0.2
Rajaeifar et al., 2014	Comparative study	yes	1 ton of MSW treated	Impact 2002þ	Normalization	Ecoinvent e v3.0
Fernandez-Nava et al., 2014	Comparative study	yes	MSW produced in one year in Asturias	Impact 2002þ	Normalization	Ecoinvent v2.0
Chi et al., 2014	Comparative study	yes	Annual MSW generation in the city in 2010	EDIP 97	Sensitivity, normalization	*Not menthioned*
Ripa et al., 2015	Comparative study	yes	Annual production in City of Naples in 2012	ReCiPe Midpoint	Sensitivity, normalization	Ecoinvent
Mali et al., 2016	Comparative study	yes	1 ton of MSW	CML	Normalization	SimaPro7
Hong et al., 2016	Comparative study	yes	1 ton of MSW	ILCD and USEtox (for the human health category)	Sensitivity, normalization	*Not menthioned*
Coelho et al., 2016	Comparative study	no	Annual amount of waste generated	CML 2001	Sensitivity, normalization	*Not menthioned*

Liu et al., 2017	Comparative study	yes	1 ton of wet MSW processed	*Literature*	Sensitivity	EaseTech
Nabavi-Pelesaraei et al., 2017	Comparative study	yes	8500 ton of daily produced of MSW in Tehran	CML 2 baseline 2000 V2/world,	Sensitivity	EcoInvent 2.2
Ayodele et al., 2017	Comparative study	yes	ton of MSW generated in between 2016 and 2035	Eco-indicator 99	*Not menthioned*	*Not menthioned*
Behrooznia et al., 2018	Comparative study	no	100 ton of MSW treated	CML-IA baseline V3.04	Sensitivity	Ecoinvent 2.2
Yadav et al., 2018	Comparative study	yes	1 ton of MSW	CML	Normalization	SimaPro 8.0.1.
Wang et al., 2019	Comparative study	yes	1 ton of MSW processed	ReCiPe 2008	Sensitivity	*Not menthioned*
Lima et al., 2019	Comparative study	no	MSW produced in one year from 2017 to 2037	ILCD	*Not menthioned*	Ecoinvent 3
Wang et al., 2019	Comparative study	no	1 ton of MSW treated	IPCC 2013 GWP 100a	Sensitivity	*Not menthioned*
Khandelwal et al., 2019	Comparative study	yes	1 ton of MSW	CML-1A	Sensitivity	Gabi 8.5.0.79
Sauve et al., 2020	comparative study	yes	1 ton of MSW waste disposed in a landfill	ILCD	Sensitivity, normalization	GaBi 8.0

Chapter 3

Goal, scope definition and inventory analysis

In this chapter the first and the second step of the LCA are presented. Following this methodology, the goal and scope definition are primary defined considering all the three cases, namely system 1, system 2 and system 3. Then the inventory analysis was reported considering all the input and output fluxes and the assumption made to fulfil the inventory.

3.1 Goal and scope definition

The first step of the LCA methodology following ISO 14040 is the goal and scope definition. The aim of this study is to evaluate the environmental performances of three systems for the treatment of municipal solid waste (MSW). These are: mechanical treatment facility (system 1) and two sanitary landfills (system 2 and system 3). Particularly the environmental performances of the three system are evaluated considering the operation due to MSW treated in 2018.

The study has been conducted following the ISO 14040 (2006) and ISO 14044 (2017) standards and the functional unit has been defined as "1 ton of MSW treated inside the systems in 2018". The system function has been defined for system 1 as the treating of MSW by mean of mechanical and biological operation and for system 2 and 3 as the disposal in landfill of MSW, including materials and activity performed in the sites. Boundaries of the study include all the input waste, materials and energy and output waste and emissions associated to streams in 2018 and they are defined in Figure(2.1), Figure(2.3), Figure(2.4). Particularly transportation processes of input and output streams to all three systems are considered and in reference to landfills described in system 2 and system 3, boundaries include landfill disposal, closure procedure (capping) and after closure treatment of biogas and leachate. For mechanical and biological treatment plant construction and dismission operation are not considered in the study.

3.2 Inventory analysis

Inventory analysis represents the most demanding task in the LCA procedure. It is an iterative procedure that involves data collection and calculation procedures to quantify the relevant input and outputs of the analysed system (ISO, 20006.). The models for the three systems studied is implemented in the SimaPro software and information on the different inventory analysis are described below.

3.2.1 SimaPro software

The software used to carry out the study is Simapro, developed by the Dutch company PRè Sustainability. The version of the software used is the 8.0.3.14. Simapro software allows to follow ISO 14040 (2006) and ISO 14044 (2017) standards in all the different steps of the methodology and it also allows to put it into practice its iteratively. Indeed, it is possible to describe goal and scope definition by adding also documentation on data sources, to define and describe the aim of the study, that can be easily reminded if necessary. Following LCA methodology then it is feasible to enter data inside the inventory which represent the most demanding task together with data collection. Data can be distinguished in two main groups (Goedkoop et al. 2013):

- Foreground data: refer to data need for modelling the system which particularly describe its features.
- Background data: it is the data to produce generic materials, energy, transport and waste management. This data usually refers to Simapro database or they may derive from literature analysis.

The distinction between data depends on the object of the LCA study and permits to define the level of deepness required in the data collection.

The system is described thought “processes” which are the building blocks that can be filled in with input and output data. These are classified as (Goedkoop et al. 2013):

- Known inputs from nature (resources)
- Known inputs from tecnosphere (material/fuels)
- Known inputs from tecnosphere (electricity/heat)
- Known outputs to tecnosphere. Products and co-products
- Known outputs to tecnosphere. Avoided products
- Emission to air
- Emission to water
- Emission to soil
- Final waste flows
- Nonmaterial emissions
- Social issues
- Economic issues
- Known outputs to tecnosphere. Waste and emission to treatment

As underlined before, data may come from database which are already integrated inside SimaPro and it is necessary to select the more appropriate one for the type of study. Examples of this databases are Ecoinvent, Agri-footprint and ELCD (Goedkoop et al. 2013). Considering impact evaluation, by mean of the previous step, SimaPro can generate life cycle inventory results which contains the elementary flows of emission or extraction to the environment. This step is founding in order to perform the characterization phase, that is a mandatory one for impact assessment, in which elementary flows of the inventory are associated through the proper impact category by mean of their characteristics. To define impact categories, it is necessary to choose a method through which perform the impact assessment. SimaPro proposes the use of several methods which can be distinguish in (Prè, 2019):

- European methods as CML, IMPACT 2002+, EDIP 2003;
- North American as BEES+, TRACI 2.1;
- Single issue as Cumulative Energy Demand, Ecological footprint, USEtox;
- Superseded as CML, Eco-indicator 99, TRACI;
- Global as Recipe 2016;
- Water footprint as Berger et al 2014, Wave (water scarcity).

The choice of the method represents one issue in the impact evaluation phase and it has to be selected considering the type of study to perform since no method can be described as better than the others. For this study, as it will be seen, the CML method has been chosen. The impact assessment result can be described thought proper table or histogram but also by the use of process tree which permits to identify the critical point of the studied system. Information on the assessment can be described also by process or substances definition to simplify the research of hotspots. Together with the mandatory steps following the ISO 14040 (2006) methodology, SimaPro allows to implement normalisation and weighting. Normalization shows if the impact category indicator result is higher or lower compared to a reference. The main advantage of this step is that it permits to compare results that are expressed in different unit of measure. For each impact method it is possible to define different normalization values like Dutch, European and worldwide ones. Weighting is a step of impact assessment that it is not allowed to be used in comparative studies disclosed to public following ISO 14040 (2006). Indeed, by definition, it is not based on natural science and it is very subjective (Goedkoop *et al*., 2013). The final step regards interpretation of the result which can comprehend data uncertainty analysis. Particularly it can be performed using statistical technique like the Monte Carlo one which is based on distribution description expressed as a range of standard deviations. Finally, it is possible to perform sensitivity analysis which describes how the most important assumption influence the results by changing the same assumption and comparing the obtained results. The previous chapter has underline how SimaPro software allows to follow all the steps of the methodology being an important support tool to perform the LCA analysis.

3.2.2 Model definition for inventory analysis

In this study primary, secondary and tertiary data are used. Most of them are primary because they derive from proper analysis, quantifications and information of the three studied systems. Considering secondary data, they may refer to previous studies but also and mainly, on the database present in SimaPro. The dataset considered for this study is Ecoinvent version 3.0, a result of the joint effort of different Swiss institutions of the integration of several life cycle inventory database (Goedkoop *et al*., 2013). Considered the dataset versions there are six of them implemented in SimaPro:

- Allocation default, unit process;
- Allocation default, system process;
- Allocation recycled content, unit process;
- Allocation recycled content, system process;
- Consequential, unit process;
- Consequential, system process.

Choosing between unit and system process does not influence the result of the study. The main difference between them is that unit process describes the process unit by a series of upstream unit which can be opened individually. While in system processes emissions are already inside the chosen process unit and there is no possibility of separate the supply chain. The dataset chosen for this study is the "allocation, cut-off by classification-unit" which refer to end-of-life allocation.

In the Ecoinvent database the process selected in the inventory can be described thorough mean of "market process" or "transformation process": transformation processes contain all the inputs for making a product or service, except for transport, and all the associated emissions and resource extractions while market processes include inputs from production in several countries as well as inputs of transport processes. (Goedkoop *et al.*, 2013). Usually market processes have been selected for this study except for transformation processes, for which transportation is assumed to be considered in raw materials processes.

All the study follows the so called "zero-burden" assumption, so it considers waste from the moment it becomes such, until through processes and treatments the material ceases to be waste. The life cycle assessment does not consider the stages that generated the waste, which are supposed to be common to all waste systems. This hypothesis is valid as the goal of the study is not to assess how waste prevention can best be achieved but to evaluate the assessment of three management waste system (Ekvall *et al.,* 2007).

3.2.3 CER code

In the following inventory analysis waste stream can be characterised by CER code. CER code (in Italian "*Codice Europeo del Rifiuto*") is the European code of waste that permits to identify waste stream by mean of its characteristic. Particularly they refer on the European directive 75/442/CEE where waste is described as "every objective of which the holder discards or has decided or has the obligation to discard" and the waste management as "the collection, transport, recovery and disposal of waste, including the control of these operations as well as control landfills after their closure". The commission decision 2000/532/CE describes all the codes references to the type of waste dividing them in hazardous and non-hazardous waste. To easily describe waste categories they are defined through from a series of six identification numbers where the first two refer to the source that generates the waste and that refers to proper chapters. In order to favor the description of waste streams in the following inventory analysis in Table (3.1). all the type of waste considered are reported, by describing their characterization in the CER code.

Table 3.1: CER code description

CER code	Description
13 02 05	Mineral oil waste for engines, non-chlorinated gears and lubrication
15 01 02	Plastic packaging
15 01 06	Mixed material packaging
17 02 03	Plastic
17 09 04	Mixed construction waste and demolition, other than those referred to in items 17 09 01, 17 09 02 and 17 09 03
19 05 03	Unspecified compost
19 12 02	Iron materials
19 12 12	Other wastes (including mixed materials) produced by the mechanical treatment of waste, other than those mentioned in item 19 12 11
20 03 01	Municipal solid waste undifferentiated
20 03 07	Bulky waste

3.2.4 Mechanical treatment facility for MSW

The first studied system is a mechanical treatment facility for MSW and it will be called system 1. It represent the second line of a plant in which the municipal solid waste of 42 municipality is processed in order to separate the dry fraction, which is sent later on to the final facility that are landfill and incineration, and the wet fraction that is stabilised and intertied.

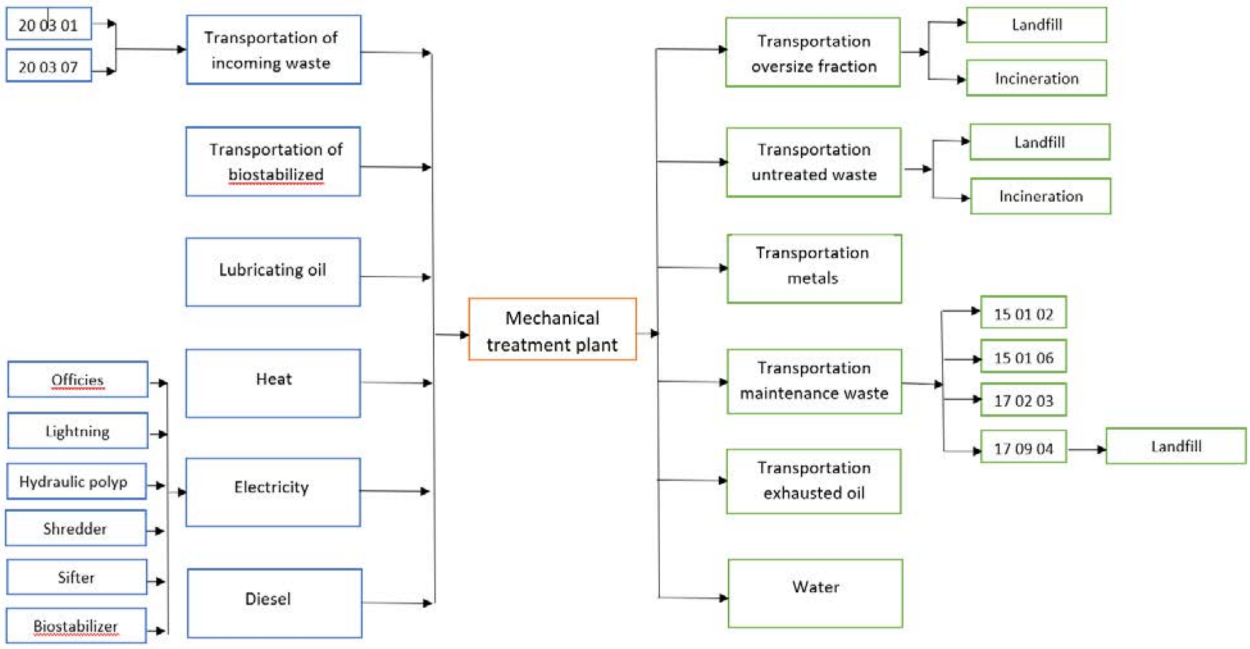


Figura 3.1: Burdens of system 1 (A personal rework)

Mechanical treatment facility is also called mechanical "pre-treatment" and it is the first step in the waste processing after the waste collection. The aim of the mechanical treatment plant is to treat the entering waste by separating it in different fractions. The separation usually occurs in a completely mechanical way or the sorting can be done manually at hand picking stations. In this specific case only, operating machines are used which comprehends a shredder, conveyor belt, sifter, iron removal machine, biostabilizer and hydraulic polyp. All this component will be better considered in the following description of the plant. The separation of the dry and the wet fraction permits to treat the last one inside a biostabilizer. The biostabilizer is composed of biocells in which the wet fraction is stabilized through oxidative processes which occurs in aerobic conditions. Particularly in biocells the air is forced inside the waste mass to favour the oxidation process and to ensure uniform diffusion of air in the biomass, thus avoiding the creation of "preferential ways" of outflow with areas that are too dry and others that are too humid. Also the blowing of air allows to maintain an optimal temperature for the microbial fauna in an interval in which the activity of these organisms is maximum. So biocells permits the acceleration of the degradation process of organic matter and to sanitise it. The stabilised waste is then transferred to landfill or it can be used, if it is within the necessary specifications, as compost.

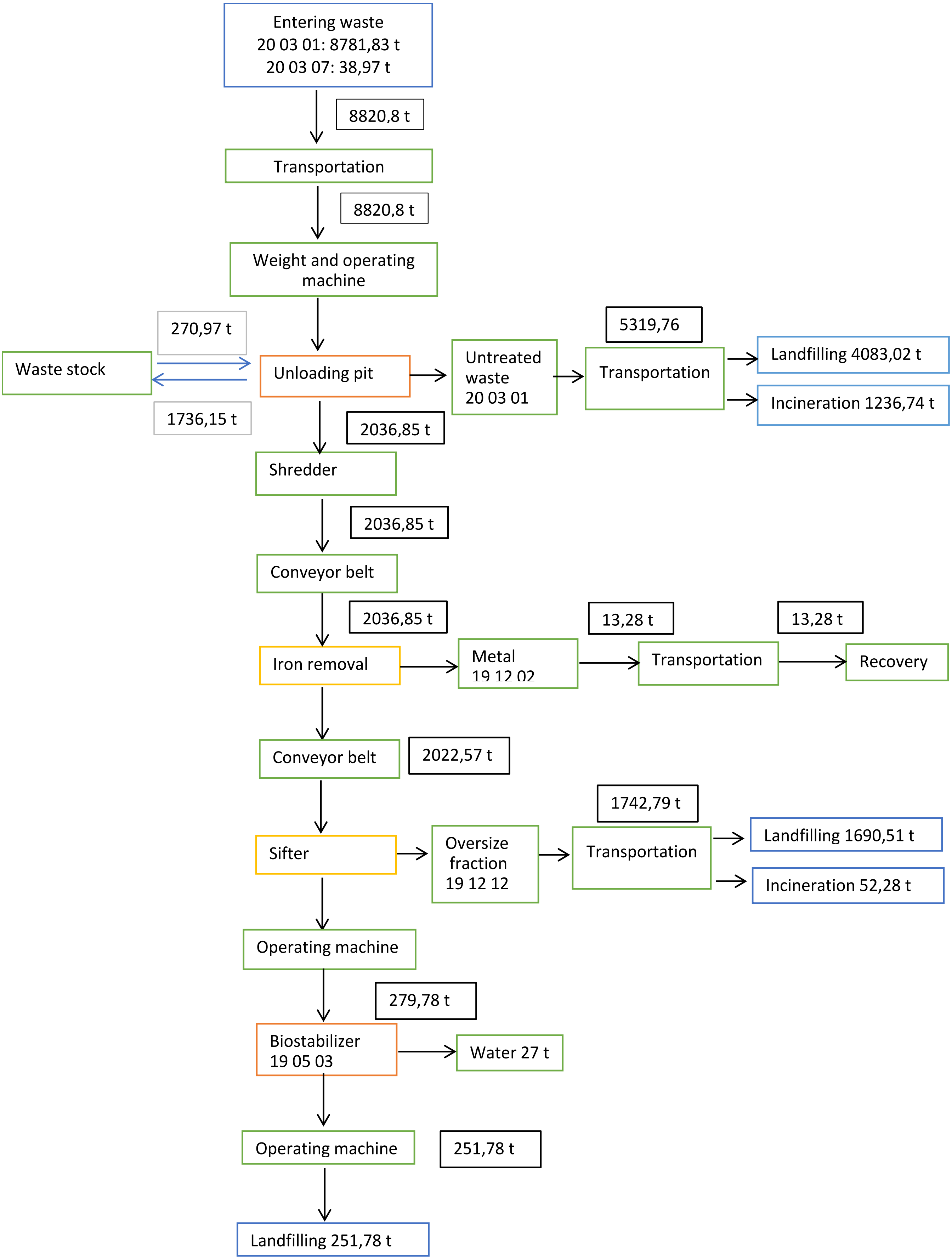


Figure 3.2: Material balance considering waste in system 2 (A personal rework)

In order to describe the plant it is necessary to consider fluxes of energy and mass entering and exiting the plant and the equipment that interacts with this fluxes. Considering system 1 the boundaries of the study are settled in order to consider MSW treated in mechanical treatment facility plant during all 2018 and all streams will refer to this time span. System boundaries are described in Figure (3.1). This includes waste, which is treated, entering and leaving the plant, the energy used for operating machines and other utilities in the form of electricity, diesel fuel or natural gas, general materials, emissions of water (correlated to the biostabilizer) and transportation of waste from and to the plant. The final disposal scenarios for treated waste are also considered, which are landfill and incineration, with the proper percentage of disposal characterizing every stream.

The mass balance regarding waste streams is reported in Figure (3.2) and all streams refer to total amounts in 2018.

The total amount of waste entering the plant is 8820,8 ton and it can be catalogued, considering the CER classification, in:

- 8781,83 ton of 200301.
- 38,97 ton of 200307.

The entering waste is transported to the scales where it is weighted and then sent, thanks to an operating machine which in this case is a hydraulic polyp, to the unloading pit. At this node of the plant it is important to underline a mass balance between the input to the system and the output. Indeed at 31/12/2017 are already present, inside the plant, 270,97 ton of waste which represent together with the entering waste the two input of our system. The outputs to the unloading pit are the waste treated in 2018 (2036,85 ton), the waste stocked at 31/12/2018 (1736,15 ton) and the untreated waste (5319,76 ton). Particularly, considering this last stream, the waste, once separated, is transported to the final disposal that it is represented for the 77% of waste (4083,02 ton) to sanitary landfill and for the remain 33% (1236,74 ton) to incineration plant.

Considering the main stream as the one referred to the treated waste inside the plant, and being also one ton of treated MSW the chosen functional unit, the waste is then transferred to the shredder where it is grinded to the necessary dimension and then moved with a conveyor belt to the iron removal. Here the waste is cleaned of ferrous materials giving rise to an output stream equal to 13,28 ton (19 12 02) which is then moved thanks to an operating machine to the storage area and finally transported to the recovery facility. The deferred waste (2022,57 ton) is moved with a conveyor belt to the sifter where it is separated in two different streams by mean of their size and composition. The first stream is the oversize fraction (1742,79 ton) is characterised by the dry fraction of waste which is then transported to the final facility that are sanitary landfill and incineration. Particularly only 3% of the waste (52,28 ton) goes to incineration and the remain 97% (1690,51 ton) is treated in landfill. The undersize fraction (279,78 ton), characterised by the wet part of waste, is moved with an operating machine to the biostabilization unit to make it inert by a single bio-oxidation phase. During this operation 28 ton of water are vaporised and exit from the plant. The remain biostabilzed fraction (251,78 ton) is unloaded with an operating machine and then transported to a landfill to be used as covering.

3.2.5 Data collection and quantification: system 1

The system inputs are: MSW entering the plant, their transportation, electricity consumption, heat generated by natural gas and lubricating oil used in machineries. The impact associated to the construction of the operating machine used inside the plant is not considered in the inventory also by the reason of the system boundary.

The entering waste is marked out by two different streams considering their property and dimension: the first one refers to undifferentiated MSW (CER code 200301) and the second one by bulky MSW (CER code 200307). As underlined the waste is modelled by mean of the zero burden assumption as empty considering the associated impact. The weight of the streams, regarding overall year 2018 are reported in Table(3.2).

Table 3.2: Input stream: MSW entering the plant and their characteristic (system 1)

CER	Amount (ton)	Percentage
20 03 01	8781,33	99,56 %
20 03 07	38,97	0,44 %
Totale	8820,3	100 %

Together with the entering waste it is necessary to consider their transportation to the plant. The distance is calculated thought averages consideration on the collection of the two waste stream. Distance information between waste source and the plant are reported in Table(3.3).

Table 3.3: Avarage distance regarding transportation of MSW entering the plant (system 1)

Average distance	Amount (unit of measure)
20 03 01	22,49 km
20 03 07	47,36 km

The transportation is assumed to be carried out with lorry characterised by a capacity in between 16 and 32 metric ton and with emission standard EURO 3. This assumption is reported inside the inventory analysis in SimaPro by mean of Ecoinvent inventory as: “transport, freight, lorry 16-32 metric ton, EURO 3 {RER}| Cut-off, U”. This type of lorry is assumed to be used in all the transportation processes of waste in system 1. Waste transportation is described, in terms of unit of measure, as tonne per kilometre (tkm) and so it is represented by the product of the quantity of waste and their characterising distance. The result of the product calculation is given in Table(3.4).

Table 3.4: Transportation assessment regarding the transportation of the entering MSW (system 1)

Transportation assessment	Amount (unit of measure)
20 03 01	197’487,55 tkm
20 03 07	1’845,67 tkm

Considering again waste transportation it is necessary to consider also the transportation of biostabilized waste to the landfill disposal. The biostabilized fraction is described as compost out of specification (CER code 190503) and its transport assessment is evaluated as the product of the quantity of waste and the average distance, as underlined before. The information are reported in Table(3.5).

Table 3.5: Transportation assessment of biostabilized waste fraction to its landfill (system 1)

Biostabilized	Amount (unit of measure)
19 05 03	251, 78 ton
Distance	125 km
Transport assessment	31’472,5 tkm

Electricity consumption represent an input for the system, and it is used in order to move the operating machines inside the plant and for general use in offices and internal and external illumination. Electricity is assumed to be taken from the Italian grid and it measured in kilowatt hour. It is modelled thanks to the Ecoinvent database as: “electricity, medium voltage {IT}| market for| Cut-off U”. Information on the consumption of the different unit operation in 2018, reported in Figure(3.1),are written in Table(3.6). Power and usage are reported in appendix (Table(A1)).

Table3.6: Electricity consumption of different unit operation in system 1

Unit operation	Amount (unit of measure)
Hydraulic polyp	3830 kWh
Shredder	18935 kWh
Sifter	6123 kWh
Biostabilizer	5722 kWh

As said previously electricity consumption also comprehends its use in other sections of the plant, different from the treatment one. Indeed they consist in interior and exterior lighting and offices usage consumption. They are all defined in Table(3.7).

Table 3.7: Electricity consumption in other section of the plant in system 1

Unit	Amount (unit of measure)
Interior lightning	3601,88 kWh
Exterior lightning	7017,89 kWh
Offices	2109,79 kWh

Another input is related to the energy consumed in order to heat up the plant and environments. Heating information are in Table(3.8).

Table 3.8: Energy consumed for heating system 1

Heating energy	Amount (unit of measure)
Heat, central or small-scale, natural gas {RoW}| heat and power co-generation, natural gas, 160 kW electrical, lambda=1| Cut-off U	26’372,38 kW

Diesel usage is another input to the plant and the amount of diesel used in 2018 inside the system is expressed in mass unit and it is accounted in Table(3.9).

Table 3.9: Diesel consumption in system 1

Diesel	Amount (unit of measure)
Diesel {RER}| Market group for | Cut-off, U	8,97 ton

The last element of input streams is the lubricating oil used for maintenance of the different unit operation. It is expressed in mass unit and its quantity is reported in Table(3.10).

The output streams are characterised by the waste that exit the plant, their transportation and the final facility destination.

Table 3.10: Lubricating oil consumption in system 1

Lubricating oil	Amount (unit of measure)
Lubricating oil {RER}| market for lubricating oil| Cut-off U	0,45 ton

Considering firstly the untreated waste stream (CER code 200301), it represents the part of the waste exiting from the unloading pit and not treated inside the plant. The waste is then transported to two final treatment facilities, as plant data reported, whereby 77% of the stream goes to sanitary landfill and 23% to incineration. Data on the quantity of waste, distance, transportation assessment and final treatment facility are reported in Table(3.11). Final facilities are expressed as tonne of waste treated in their plant.

Table 3.11: Final treatment facility for untreated waste in system 1

Final treatment facility for untreated waste (20 03 01)	Amount (ton)	Percentage
Municipal solid waste {RoW}| treatment of, sanitary landfill | Cut-off U	4083,02	77%
Municipal solid waste (waste scenario) {IT}|treatment of municipal solid waste, incineration | Cut-off U	1236,74	23%
Total	5319,76	100%

Transportation of waste to the two final destinations is described in Table(3.12)

Table3.11: Untreated waste exiting the plant: information of quantity of waste and transportation (system 1)

Untreated waste (20 03 01)	Amount (unit of measure)
Distance for landfill	190 km
Transport assessment for landfill	775’773 tkm
Distance for incineration	98,4 km
Transport assessment for incineration	121’695 tkm

One of the output of the iron removal refers to metals collected from the mainstream and then sent to the recovery plant. As they are still considered as waste, they follow the zero burden assumption and therefore the only impacts attributable to them concern the transport outside the plant as reported in Table(3.12).

Table 3.12: Metals exiting the plant: information of quantity of waste and transportation (system 1)

Metals	Amount (unit of measure)
15 01 06	0,79 ton
Distance	98,4 km
Transport assessment	77,736 tkm

As underlined before there are two streams exiting the sifter: one for the undersize and the second one for the oversize. Particularly the oversize one (CER code 191212) is then transported to the final treatment facility which is, for the 97% of the stream, landfill disposal and for the 3% incineration. (Table 3.13).

Table 3.13: Final treatment facility for oversize fraction in system 1

Final treatment facility for oversize fraction (19 12 12)	Amount (ton)	Percentage
Municipal solid waste {RoW}| treatment of, sanitary landfill | Cut-off U	1690,51	97%
Municipal solid waste (waste scenario) {IT}|treatment of municipal solid waste, incineration | Cut-off U	52,28	3%
Total	1742,79	100%

Table 3.13: Final treatment facility for oversize fraction in system 1

Transportation to the final destinations of the oversize fraction are in Table(3.14).

Table 3.14: Oversize fraction exiting the plant: information of quantity of waste and transportation (system 1)

Oversize fraction (19 12 12)	Amount (unit of measure)
Distance for landfill	190 km
Transport assessment for landfill	321’196 tkm
Distance for incineration	98,4 km
Transport assessment for incineration	5’144 tkm

Another output that must be taken into consideration is related to the biostabilization phase: in this phase, in fact, due to the temperatures maintained inside the biostabilizer and due to the action of aerobic microbial agents, there is water evaporation which exits the plant and so it is considered as an output. The amount of evaporated water is written in Table(3.15).

Table 3.15: Water exiting from the biostabilizer in system 1

Evaporated water	Amount (unit of measure)
Water	28 ton

During 2018, maintenance interventions were carried out inside the plant which included the use of lubricating oil as already pointed out above. Together with this element different groups of waste were generated. each of these elements, being a waste, is modelled through the “zero burden” assumption and characterized, as an impact, solely by their transport outside the system. The first one considered refer to the lubricating oil used for machine maintenance. The amount of exhausted oil refers to the lubricating oil that has been used for maintenance and that now is considered as waste. (Table(3.16))

Table 3.16: Lubricating oil disposal: : information of quantity of waste and transportation (system 1)

Lubricating oil disposal (13 02 05)	Amount (unit of measure)
Lubricating used oil	0,45 ton
Distance	59 km
Transport assessment	26,55 tkm

The other waste generated by maintenance works concern different categories and all of them have a different destination and are characterized by different transport assessment. Waste refer to plastic packaging (Table(3.17)), mixed material packaging (Table(3.18)), plastic waste (Table(3.19)), mixed waste from construction and demolition (Table(3.20)) and all information on quantity of waste and transportation are in the following tables.

Table 3.17: Plastic packaging waste of maintenance works in system 1

Plastic packaging	Amount (unit of measure)
15 01 02	0,79 ton
Distance	121 km
Transport assessment	95,59 tkm

Table 3.18: Mixed material packaging of maintenance works in system 1

Mixed material packaging	Amount (unit of measure)
15 01 06	0,46 ton
Distance	134 km
Transport assessment	61,64 tkm

Table 3.19: Plastic waste of maintenance works in system 1

Plastic waste	Amount (unit of measure)
17 02 03	0,21 ton
Distance	121 km
Transport assessment	121 tkm

Table 3.20: Mixed waste from construction and demolition of maintenance works in system 1

Mixed waste from construction and demolition	Amount (unit of measure)
17 09 04	0,16 ton
Transport assessment	10,5 tkm
Municipal solid waste {RoW}| treatment of, sanitary landfill | Cut-off U	0,16 ton

In the following chapter the inventory analysis for landfilling system are described.

3.2.6 Sanitary landfill for MSW

The last two studied system are sanitary landfill. Landfill is the oldest way to treat waste and the most common. Usually the disposal waste are both MSW and/or all type of waste, considering also the wet ones, that it was not possible to recycle or treat in previous stages. Indeed the directive of the European Union (99/31/EC) has established that only materials with low organic carbon content and non-recyclable materials must end up in landfills so, giving priority to material recovery, the directive provides composting and recycling as primary strategies for waste disposal. This is due to the fact that the residues of waste, particularly organic MSW, undergoes an anaerobic degradation process in which biogas and leachate are produced (Andreottola *et al.*, 1992). This degradation occurs over long periods of time, which go beyond the closure times of the landfill and therefore require particular attention also in the disposal phase. Furthermore, decomposition can be described through three phases (Andreottola *et al.*, 1992):

- The first, of rather short duration compared to the others, regards aerobic reactions, possible thanks to high concentration of oxygen with respect to other phases. From this oxidation the organic material is decomposed to CO_2 and energy and may comprehend the formation of aldehydes, ketones and alcohols.
- The second is an acid anaerobic phase in which the main products are hydrogen, ammonia, carbon dioxide and partially degraded organic compounds. The leachate that is produced in this phase has hight values of BOD and COD which is acid and aggressive. The components in leachate are mainly volatile carboxylic acids, esters and thioesters.
- Third phase: it is methanogenic anaerobic where anaerobic bacterial biodegradation decomposes organic acids and other products developed in the previous phase. The transformation of the acids causes an increase of the PH up to values close to neutrality or higher. The main products of the third stage of decomposition are methane and carbon dioxide.

The different degradation phases underline the necessity of collecting and treating both biogas and leachate and to avoid their dispersion in the environment. Usually landfill is constructed following a layer structure that permits to isolate waste and its residue from the environment. Starting from the bottom layer there is the foundation of the landfill followed by an unpermeablised barrier made of geomembranes to avoid the loosing of leachate and to permits its collection, thanks also to a drainage system and pumps. Then the pre-treated waste is placed: indeed the waste is first weighed, then through a conveyor belt, it is sent to the pressing phase where they are compacted and subsequently placed inside the landfill by means of excavators. The subsequent layers concern the coverage of the landfill. The covering can be done daily by disposing of layers of inert material (such as soil, silt or clay) or it can be definitive to allow the requalification of the soil. Considering the final coverage its procedure is called capping. In this phase it is necessary to take into account the possibility of erosion, due to the degradation of the waste, and consequently it is required to consolidate the soil and the upper layer of the landfill as best as possible. The landfill also needs to be from the environment to avoid rainwater leading to the formation of leachate or erosion phenomena and to avoid the release of biogas. to achieve this result, a layered structure is introduced due to the presence of geomembranes, geotextiles, draining material or gravel. This layer permits to capture biogas and send it to combustion to recover energy or simply to a flare to reduce the methane which is one of the main components of biogas. Finally, it is possible to cover with soil the impermeable layer in order to proceed with grassing of the landfill area.

3.2.7 Data collection and quantification: system 2

System 2 is a sanitary landfill and its burdens are described in Figure(2.3). Burdens refer to stream of waste entering in landfill and their transportation, the usage of electricity for the scales, press, leachate pump and other activities necessary inside the plant and diesel for excavators. Also they refer on the use of natural resources, like water and land occupation and materials, like iron wire and silt (considering also their transformation). Capping, which will occur in 2020, is also included in burdens, considering all material and operations made in order to ensure landfill closure. Emissions considered are leachate and biogas and together with capping they need to be allocated considering the percentage attributable at 2018 treated MSW waste.

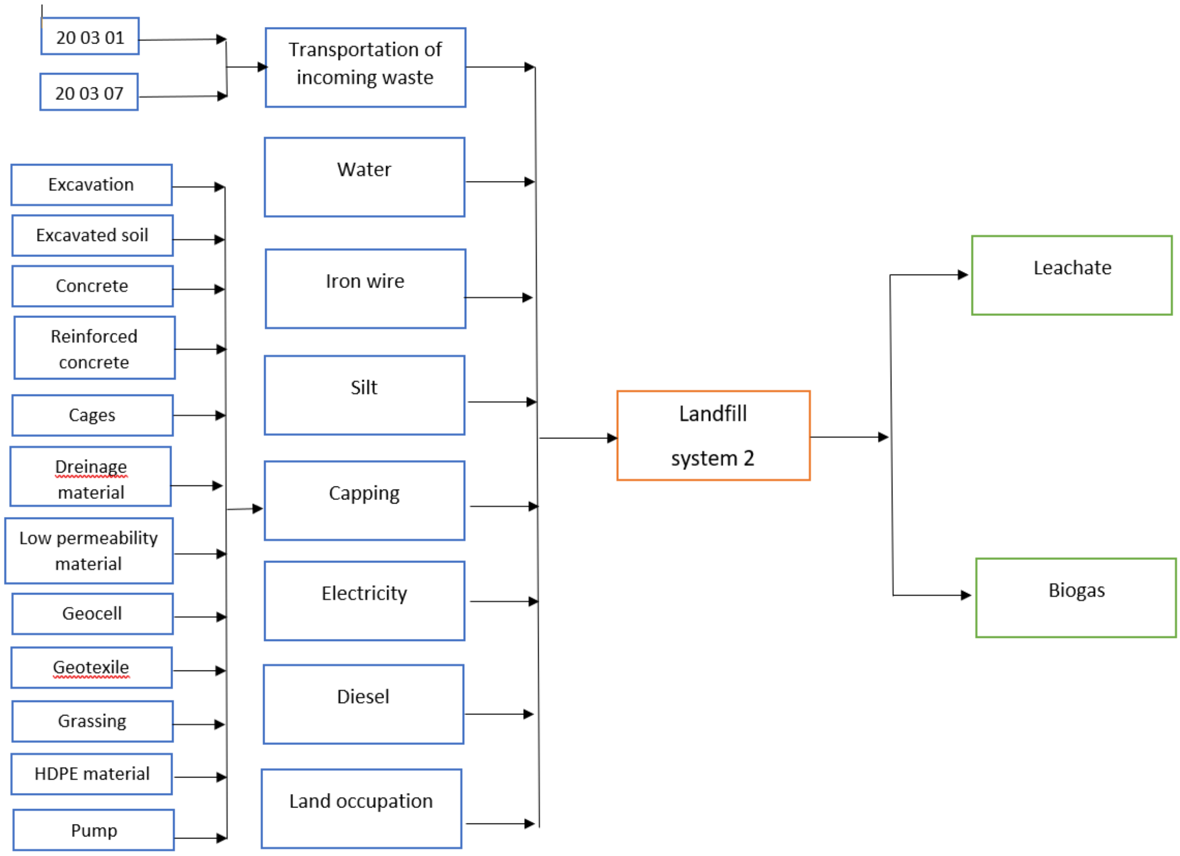


Figure 2.3: Burdens of system 2

The amount of MSW treated in 2018 inside system 2, which is the functional unit for this system, is reported in Table(3.21).

Table 3.21: Treated waste in 2018 in system 2

Treated waste in 2018 in system 2	Amount (unit of measure)
Waste	2378,42 ton

As the zero burden assumption defines, the waste is considered “empty” regarding the associated impact when it enters inside the plant. The only associated impact consists of the transportation of waste to the plant. There are two main types of MSW entering the plant, following the CER code: 200301 and 200307. Transportation lorry used for waste are the same of system 1 and the referred parameter in the Ecoinvent database is “transport, freight, lorry 16-32 metric ton, EURO 3 {RER}| Cut-off, U”. The waste comes from different destination and information on distance and quantity of waste for each of them are written in Table(3.22) and Table(3.23).

Table 3.22: MSW entering system 2: amount, distance and transportation assessment

20 03 01	Amount (ton)	Distance (km)	Transportation assessment (tkm)
Path 1	810,4 ton	23 km	18’639,2tkm
Path 2	1152,34 ton	41 km	47’254,94 tkm

Table 3.23: MSW entering system 2: amount, distance and transportation assessment

20 03 07	Amount (ton)	Distance (km)	Transportation assessment (tkm)
Path 3	107,22 ton	5 km	536,1 tkm
Path 4	64,96 ton	25 km	1’617,25 tkm
Path 5	144,58 ton	41 km	5’927,78 tkm
Path 6	98.92 ton	25 km	2’473 tkm

Considering the system 2 input it is possible to identify two main categories of input together with the entering waste: the first one refers to the use of natural input while the second one derives from technosphere. Starting from the nature one it comprehends water used in plant and the transformation of land. The information on this two parameter are reported in Table(3.24).

Table 3.24: Natural input to system 2: quantity and definition in SimaPro

Natural input	Amount (unit of measure)
Water, process, unspecified natural origin	127 m^3
Occupation, dump site, temperate grassland and savannah	1000 m^2

Input from technosphere are related to materials and their transportation, energy consumption and capping procedure. The first material considered is silt which is used for daily coverage of the landfill. Since silt is not present inside the Ecoinvent database the clay parameter is used since it differs from silt only for texture (particularly silt is 10^{-2} m thinner than clay). Table(3.25) contains information on the amount of silt used and on the transportation assessment.

Table 3.25: Silt usage and transportation in system 2

Silt	Amount (unit of measure)
Clay {RoW}| clay pit operation | Cut-off	104 ton
Distance	3 km
Tranportation assessment	312 tkm

Another input, considering materials, consist of iron wire used in the pressing procedure in the pressing procedure. However iron wire is not a parameter of the Ecoinvent database, so in order to assess for this material production two elements are considered: the first one refers to iron pellet, which is the raw material, the second one refers to a transformation process which permits to describe the metal processing. Material, transformation process and transport are written in Table(3.26).

Table 3.26: Iron wire usage and transportation in system 2

Iron wire	Amount (unit of measure)
Iron pellet {RoW}| production| Cut-off U	3,3 ton
Metal working, average for metal production	3,3 ton
Distance	385 km
Trasportation assessment	1270,5 tkm

Considering energy consumption there are two main sources, electricity and diesel, used both for the required operations inside the plant. Electricity is used for the scales, press, leachate pump, pressure washer, tire washer, whether station, illumination, offices and storage. Their usage, expressed in hours of working in year, and their power consumption is reported in Table(A.1).

Furthermore electricity is taken from the Italian grid and information on total consumption are in Table(3.27).

Table 3.27: Electricity consumption from the Italian grid in system 2

Electricity	Amount (unit of measure)
Electricity medium voltage {IT}| market for| Cut-off U	32050 kWh

Diesel is the other source of energy used inside the plant for the movement of excavator. The primary data refers to litres of diesel but since it is necessary to express it in ton inside the life cycle inventory, a density of the diesel The amount used in 2018 is written in Table(3.28).

Table 3.28: Diesel consumption in system 2

Diesel	Amount (unit of measure)
Diesel {RER}| Market group for | Cut-off, U	5,168 ton

Capping procedure in the final input in the inventory analysis. Capping is needed in order to cover the landfill once the closing time has arrived. It permits to avoid the rainfall water to give more leachate but also it permits to collect the formed biogas and to avoid erosion. In order to ensure the correct functioning of capping several material and operation are used. Since the capping procedure refers to all the waste landfilled and not only to MSW treated in 2018, which refer to the chosen functional unit, it is necessary to perform an allocation procedure. Therefore, the percentage of capping attributable to the MSW treated in 2018 is given by the ratio between the treated waste in 2018 and all the waste inside the landfill up to the capping procedure. Particularly capping occurred in 2020 when the total amount of waste inside the landfill is equal to 13958 ton giving to an allocation percentage equal to 17 %.

In Table(3.29) all the elements which are attributable to capping are reported while in Table(3.30) the allocation percentage is considered in the description of capping.

Table 3.29: Capping procedure elements for system 2

Capping	Amount (unit of measure)
Excavation	752 m^3
Excavated soil	20273,5 ton
Concrete	15,84 ton
Reinforced concrete	8,67 m^3
Cages	154 ton
Drainage material	19417 ton
Low permeability material	9917 ton
Geocell	9,6 ton
Geotextile	12,29 ton
Grassing process	1 piece
HDPE materials	1685,8 kg
Pump	1 piece

Table 3.30: Capping procedure and its allocation in system 2

Capping	Amount (unit of measure)
Capping	0,17 piece

The first element that is necessary to consider is the excavation in order to build up the necessary area. Excavation comprehends also the levelling of the bottom and wall profiling while the only specific regarding the deep is to avoid the excavation of the rocky mantle. For the excavation a hydraulic digger is used and its work is expressed as cubic meter of soil excavated (Table(3.31)).

Table 3.31: Excavation work for capping in system 2

Excavation	Amount (unit of measure)
Exacavation, hydraulic digger {GLO}| market for | Cut-off, U	752 m^3

Together with the previous excavated soil other soil was used. This soil comes from other excavation and it is used for the surface covering layer. Soil and its excavation with hydraulic digger are defined in Table(3.32). Since soil must be expressed in tonnes and excavation in cubic meter it was assumed an average density of soil of 1700 kg/m^3.

Table 3.32: Excavated soil for capping in system 2

Exacavated soil	Amount (unit of measure)
Exacavation, hydraulic digger {GLO}| market for | Cut-off, U	34464 m^3
Soil (natural input)	20273,5 ton

Concrete is another material used in capping procedure. Particularly it is used for six inspection pit extension each of which weight 840 kg, with square section of side 100 cm. It is also used for two flat roof slabs of the inspection pit extension with square section of side 100 cm each with weights 720kg each. The weight of all the pieces in concrete material is reported in Table(3.33).

Table 3.33: Material in concreate used for capping in system 2

Concreate	Amount (unit of measure)
Inspection pit extension	15,12 ton
Flat roof slab	0,72 ton
Total: concrete block {GLO}| market for| Cut-off U	15,84 ton

Reinforced concrete is used for channels and their wells. Channels run through 620 m and they are used for the rainfall water collection. Their upper edges are made in steel and they are build following the din 19580 standard for type of load which comprehends the basement foundation. From literature analysis the weight of 1 m is estimated as 24 kg giving to a total weight of 14,88 ton. The number of well used is equal to five and each of them weights 1,2 ton to a total weight of 6,1 ton. Since reinforced concrete needs to be expressed in cubic meters an average density of 2420 kg/ m^3 is assumed. Information wells and channels, in cubic meter, are in Table(3.34).

Table 3.34: Material in reinforced concrete used for capping in system 2

Reinforced concreate	Amount (unit of measure)
Channels	6,15 m^3
Wells	2,52 m^3
Total: concrete, high exacting requirements {RoW}| market for| Cut-off U	8,67 m^3

In order to reinforce the soil steel wire mesh cages full of rubble are used. The steel wire used for the cages follows UNI EN 10223-3 with diameter equal to 3 mm with strong galvanization and double twist. The mesh has a dimension equal to 8x10 cm and the steel structure is reinforced with steel tie-rods, of dimension 2,2 mm, and with side borders in galvanized steel of 3,9 mm in order to avoid deformation of the cages and to increase their mechanical strength. Cages are fulfilled with rocks of minimum dimension 15x15x15 cm. The total amount of cages used is of 70 m^3 which are divided between rubble and wire. Assuming that for every cubic meter of rubble are necessary 1,26 m^3 of wire, for 70 m^3 of rubble 1,26 m^3 need to be used. In the inventory information on rubble, wire and its production are reported (Table(3.35)), considering that in order to obtain rocks of proper dimension a crushing process is need and that steel wire need to be galvanized. Galvanization needs to be expressed in square meters by assuming a thickness of 3 mm of the wire. Rocks, rock crushing and wire steel must be defined in mass unit by requiring an average density of rocks of 2400 kg/ m^3 and a density of wire of 2000 kg/ m^3.

Table 3.35: Cages in capping procedure in system 2

Cages	Amount (unit of measure)
Stone (natural)	84 ton
Rock crushing {RoW}| processing| Cut-off U	84 ton
Wire drawing steel {GLO}| market for | Cut-off U	70 ton
Zinc coat, coils {RoW}| zinc-coating, coils| Cut-off U	11667 m^2

The coverage must ensure stability of the soil and it is necessary to use different materials by mean of different properties, building a layered structure. Drainage material refer to one of this layers and in case of system 2, it is made by recycled materials of structural demolition for a total amount of 16181 m^3. In the inventory it needs to be expressed in mass unit by considering a density, given from literature analysis, of 1200 kg/m^3. Since drainage material is a waste of demolition process and following the zero burden hypothesis, it is assumed as "empty" for the associated impacts. However together with the material, it is necessary to consider all the operation needed to prepare the drainage material for its purpose. Those operations are excavation with hydraulic digger, mechanical shovel, separation, screening, crushing, iron removing. Information on the electricity consumption for tonne of waste treated are taken from previous studies and reported in Table(3.36). As before electricity is assumed to be taken from the Italian grid (Electricity medium voltage {IT}| market for| Cut-off U).

Table 3.36: Drainage material in system 2 for capping: quantity and production process

Drainage material	Amount (unit of measure)
Recycled materials of structural demolition	19417 ton
Exacavation, hydraulic digger {GLO}| market for | Cut-off, U	16181 m^3
Mechanical shovel	32038,05 kWh
Separation	213,587 kWh
Screening	7825,051 kWh
Crushing	11766,702 kWh
Iron removing	1206,261 kWh

Gravel forms the subsequent stage and it is used for its drainage property to ensure the passage of biogas. The gravel dimension is in between 3 and 6 mm with non-calcareous characteristic (Table(3.37)).

Table 3.37: Gravel for capping in system 2

Gravel	Amount (unit of measure)
Gravel, crushed {RoW}	150 ton

Another material used for the same purpose is a low permeability soil and consist of a mix of different soil texture, from sand to clay. The average density of the low permeability soil 1560 kg/m^3 and together with the amount of soil used it is considered also its excavation with hydraulic digger. Information on low permeability material are in Table(3.38).

Table 3.38: Low permeability material for capping in system 2

Low permeability material	Amount (unit of measure)
Sand and clay (natural)	9917 kg
Exacavation, hydraulic digger {GLO}| market for | Cut-off, U	20273,5 m^3

The use of geocells in polyethylene permits to avoid erosion in escarpments and heights. Their structure is characterised by honeycomb shape with a maximum traction resistance of 1,1 kN, resistance of junctions of 0,7 kN together with laying and anchoring. Following the production process low density polyethylene granulate is used as raw material and subjected to an extrusion film characterised by an efficiency of 0,969 so that 1 kg of polyethylene gives to 0,969 kg of extruded material. Geocells amount and processes are reported in Table(3.39).

Table 3.39: Geocell for capping in system 2: amount and production

Geocell	Amount (unit of measure)
Polyethylene, low density, granulate {GLO}| market for| Cut-off U	9,6 ton
Extrusion plastic film {RoW}| extrusion, plastic film| Cut-off, U	9,9 ton

Together with geocells also a geotextiles are used to increase the soil stability and also to ensure separation function and filter. It is made of polypropylene without the use of recycling fibres, stabilised at UV-ray, in continuous filaments, bonded with mechanical needling without the use of adhesives or chemical compounds or even thermal treatments. The density per square meter is of 0,2 kg/m^2 and the number of square meters used is 61450 m^2. To build geotextiles, together with granulated polypropylene as raw material, it is necessary to consider extrusion into plastic fibre and its weaving to create the fabric. As for the extrusion process in the geocell construction, the efficiency for this process is of 0.969. Inventory information on geotextiles are in Table(3.40).

Table 3.40: Geotextile for capping in system 2: amount and production

Geotextile	Amount (unit of measure)
Polypropylene, granulate {GLO}| market for| Cut-off, U	12,29 ton
Extrusion plastic film {RoW}| extrusion, plastic film| Cut-off, U	12,68 ton
Weaving, bast fibre {RoW}| processing| Cut-off, U	12,29 ton

The grassing process represent the final layer for the coverage of waste and it allows the consolidation of the soil. To ensure an effective grassing of soil a hydroseeding process is used and the total expansion of the grassing area is 20273 m^2. Together with seed natural and chemical fertilizer are used to favour the process as reported in Table(3.41). Particularly chemical fertiliser is assumed to be constituted of nitrogen, potassium chloride and phosphate in equal part.

Table 3.41: Grassing process for capping in system 2

Grassing process	Amount (unit of measure)
Grassed seed, organic, for sowing {GLO}| market for| Cut-off, U	12,67 kg
Nitrogen fertiliser, as N {GLO}| market for| Cut-off, U	0,4147 kg
Phosphate fertiliser, as P2O5 {GLO}| market for| Cut-off, U	0,4147 kg
Potassium chloride fertiliser, as K2O {GLO}| market for| Cut-off, U	0,4147 kg
Solid manure loading and spreading, by hydraulic loader and spreader {GLO}| market for| Cut-off, U	3 kg

Another important feature in capping process is the collection of leachates. Leachate is mainly due to precipitation and percolating through waste disposed in landfill and once in contact with it, it become contaminated. To ensure that leachate does not disperse into the soil and subsoil it is necessary to collect it by a system of pipes and pump. Particularly the collection system is made in HDPE and consist of pipe, wells and wells extension. Pipes and their connection joints follow the requirements of UNI ISO 4437 and two types are used: HDPE gas De90 S8 for 750 m and HDPE gas De200 for 85 m. Density per meter of HDPE pipe is 1,48 kg/m as reported in literature for similar process. Wells and their extension have a diameter of 1200 mm and they are produced through an injection moulding process that permits to create the needed shape for the purpose. Injection moulding, similarly to extrusion, has a proper efficiency considering the amount of material used to build an object of 0.994. Every well with extension has an average weight of 150 kg and in total and three wells are used in the system. All the amount and process used for HDPE materials in capping are reported in Table(3.42).

Table 3.42: HDPE materials for capping in system 2

HDPE materials	Amount (unit of measure)
Polyethylene pipe, DN 200, SDR 41{RoW}| production| Cut-off U	85 m
Polyethylene pipe, DN 200, SDR 41{RoW}| production| Cut-off U	750 m
Polyethylene, high density, granulate {GLO}| market for| Cut-off U	450 kg
Potassium chloride fertiliser, as K2O {GLO}| market for| Cut-off, U	452,7 kg

A pump is also needed in order to remove the leachate. The pump is a pneumatic one characterises by a double membrane. Double membrane pumps are volumetric pumps which exploit flexible diaphragms and move alternately, creating a temporary chamber, which attracts and expels fluid through the pump. The diaphragms function is to create a separate surface between the air and the liquid. Since in the Ecoinvent inventory there is no reference to a double membrane pump a generic pump with a power of 40 W is selected, as reported in Table(3.43).

Table 3.43: Pump for leachate for capping in system 2

Pump	Amount (unit of measure)
Pump, 40W {GLO}| market for| Cut-off U	1 piece

Considering the system outputs, they are two: biogas and leachate. The biogas, which is formed by the degradation process of the organic component of the waste, is intercepted by several extraction wells, homogeneously distributed throughout the landfill area, and sent to flare combustion. Since methane is the main component of emissions linked to biogas, its combustion allows to reduce emissions and consumed energy recovery, thanks to internal combustion gas engines. However the amount of biogas produced was too low to permit a continuous functioning of the energy production system and in 2018 either the flare was never used because of low methane concentration. Biogas composition refers to sample analysis carried out in July 2018 and the composition measured was assumed as the average one during all reference year. Analysis report composition both as mg/m^3 and as volume ratio and a density of the air of 1,184 kg/m^3 is considered (reference value for 25°C and 1 atm for dry air). Information regarding the standard followed for the measurements of the different elements in biogas analysis are in Table(A2). The results of the analyses showed that the concentration of some elements was below a certain threshold of attention for the element itself. Therefore, since the known value was not reported in the analysis table, the limit value was inlayed in the inventory, thus allowing to describe the impact related to system 2 with an approach as conservative as possible. Considering one cubic meter of air, the different components amount are reported in Table(3.44).

Since the biogas produced refers not only to the amount of waste landfilled in 2018 but on all the waste landfilled since 2018 it is necessary to consider an allocation factor. Particularly allocation factor refers to the ratio between the waste treated in 2018 (2378 ton) and the total waste landfilled at the end of 2018 (11477 ton) giving a percentage of 20,72%. The amount of waste imputable to 2018 is reported in Table(3.45). CH_4 CO_2

*Table 3.44: Biogas composition considering 1 m^3 of air analysed in system 2. * indicates components that are expressed as the upper limit value following the composition hypothesis*

Biogas composition	Amount (unit of measure)
Oxygen	0,22 kg
Carbon dioxide*	0,001184 kg
Carbon monoxide*	0,0001184 kg
Methane*	0,001184 kg
Hydrogen*	0,001184 kg
Sulfuric acid*	0,01 mg
Mercaptans*	0,05 mg
Volatile organic compounds (VOC)*	0,05 mg
Dust	0,37 mg

Table 3.45: Biogas produced in 2018 and biogas imputable to 2018 treated waste in system 2

Biogas	Amount (unit of measure)
Total biogas produced in 2018	276 m^3
Biogas imputable to 2018 treated waste	57,18 m^3

Leachate is the other system output due to in the infiltration of water into the mass of waste or from its decomposition. Leachate production will occur also after the landfill closure giving to an estimate production of 1500 ton in the first five years, 750 ton in the next 5 years, 250 t for the next 5 years and 750 t for the remaining 15 years, for a total of 13,250 t. However, it has been found in recent years that good surface waterproofing combined with a small amount of biodegradable component present in the waste reduces the production of leachate by at least 20% compared to what was expected (Based on collected data). As for biogas its composition is given by proper analyses which are carried out in two distinct points of the plant to ensure homogeneity of the results. Information regarding the standard followed for the measurements of the different elements in leachate analysis are in Table(A3). As in biogas inventory process, the value reported in the analysis for some elements, refer to an attention threshold value which is chosen as the composition value to ensure a conservative condition for the study. Analysis refers mainly on metals and organic compounds together with some general parameters that simply reflect the pollutants condition in leachate as COD and BOD5 which permits to describe indirectly the content of organic substances present in a water. COD or chemical oxygen demand is expressed as mgO_2/L and it represent the amount of oxygen needed for the complete chemical oxidation of the organic and inorganic compounds in the water solution. BOD5 or biochemical oxygen demand is the amount of oxygen needed by aerobic microorganism to decompose, by oxidation, the organic substances in an ambient at 20°C and without the sunlight in five days. It is expressed in mgO_2/L as the chemical oxygen demand. A major value of COD and BOD5 leads to a reduction in dissolved oxygen levels that can lead to anaerobic conditions, which is deleterious to aquatic life forms.

Analysis reports that leachate has a specific density of 1 kg/dm^3 with pH value of 7,2. Some parameters are not defined in the Ecoinvent database and they are so characterised as “unspecified organic compounds”. Particularly an entire class of components, which are the perfluoro alkyl substances (PSAS) are not present inside the database(Table A8) and leading to a degree of uncertainty in the assessment of the impact of the leachate, linked to the characteristics of environmental persistence. All the parameters introduced in the inventory are reported in Table(3.46).

Table 3.46: Leachate composition considering 1 kg of leachate in system 2 indicates components that are expressed as the upper limit value following the composition hypothesis*

Leachate composition	Amount (unit of measure)
<u>*Metals*</u>	
Arsenic	0,02 mg
Calcium	111,5 mg
Iron	7,45 mg
Magnesium	36 mg
Manganese	0,46 mg
Nickel	0,12 mg
Lead	0,02 mg
Potassium	273 mg
Copper	0,19 mg
Sodium	404 mg

Zinc	0,15 mg
Cadmium *	0,01 mg
Chromium VI*	0,1 mg
Mercury*	0,005 mg
Acid-base substance	
Surfactants	2,9 mg
Ammonia, as N	84,75 mg
TOC	923 mg
Phosphorus, total	10,3 mg
COD, Chemical Oxygen Demand	964 mg
Kjeldahl-N	163,7 mg
BOD5, Biological Oxigen Demand	206,5 mg
Chlorides, unspecified	341,5 mg
Sulfate	39 mg
Nitrate*	5 mg
Nitrite*	0,5 mg
Cyanide compounds *	0,05 mg
Chlorinated solvents	
Ethane, 1,1-dichloro*	0,1 mg
Ethane, 1,1,2-trichloro-*	0,2 mg
Ethane, 1,2-dichloro-*	0,1 mg
Benzene, 1,2-dichloro-*	0,1 mg
Propane, 1,2-dichloro-*	0,1 mg
Propane, 1,2,3-trichloro-*	0,1 mg
Benzene, 1,2,4-trichloro-*	0,1 mg
Ethane, 1,2-dibromo-*	0,1 mg
Benzene, 1,3-dichloro-*	0,1 mg
Benzene, 1,4-dichloro-*	0,1 mg
Methane, bromodichloro-*	0,1 mg
Ethane, 1,1,2,2-tetrachloro-*	0,1 mg
Trichlorobenzenes*	0,1 mg
Chloroform*	0,1 mg
Methane, chloro-, HCC-40*	0,1 mg

Methane, dibromochloro-	0,1 mg
Butadiene, hexachloro-	0,1 mg
Benzene, chloro-	0,1 mg
Methane, tetrachloro-, CFC-10	0,1 mg
Dichlorophenol	0,1 mg
2,3,5- Trichlorophenol	0,1 mg
3-Chlorophenol	0,1 mg
Phenolic compounds	
Phenol*	0,1 mg
Phenol, pentachloro-*	0,1 mg
P-ethylphenol *	0,1 mg
2,5-Dimethylphenol*	0,1 mg
Organic compounds (unspecified) [Includes all items, considering PFAS class, that were not found in the inventory]	0,0074 mg
Organic compounds (unspecified) [Includes all items that were not found in the inventory: 1,1-dichloroethylene, 1,2-dichloroethylene, vinyl chloride, hexachlor-butadiene, tetrachlorethylene, tribromo methane, trichlorethylene]	0,7 mg
Organic compounds (unspecified) [Includes all items, considering phenol class, that were not found in the inventory: 3-methyl-phenol, 4-methyl-phenol, 4-chloro-3-methyl phenol]	0,3 mg

Since the leachate produced refers not only to the amount of waste landfilled in 2018 but on all the waste landfilled since 2018 it is necessary to consider an allocation factor. Particularly allocation factor refers to the ratio between the waste treated in 2018 (2378 ton) and the total waste landfilled at the end of 2018 (11477 ton) giving a percentage of 20,72%. The amount of waste imputable to 2018 is reported in Table(3.21). Together with the amount of leachate produced in 2018 caused by waste landfilled it is necessary consider the quantity of leachate that will be produced in the following years and evaluate that attributable to the 2018 waste. Similarly, to capping this allocation percentage is given by the ratio between waste treated in 2018 and all the amount of waste that has been treated since the landfill closure in 2020. The allocation percentage value is 17 %. Finally leachate attributable to MSW treated in 2018 is given by the sum of this two parameters as reported in the expression in Table(3.47).

Table 3.47: Leachate produced in 2018 and leachate imputable to 2018 treated waste in system 2

Leachate	Amount (unit of measure)
Total leachate produced in 2018	2124 ton
Leachate produced after 2018	13250 ton
Leachate imputable to 2018 treated waste	2690,7 ton

The composition of the leachate is assumed to be constant and equal to that of the analyses referred to 2018, also for the leachate formed after 2018.

3.2.8 Data collection and quantification: system 3

As system 2, system 3 refers to a sanitary landfill and its burdens are reported in Figure(3.4).

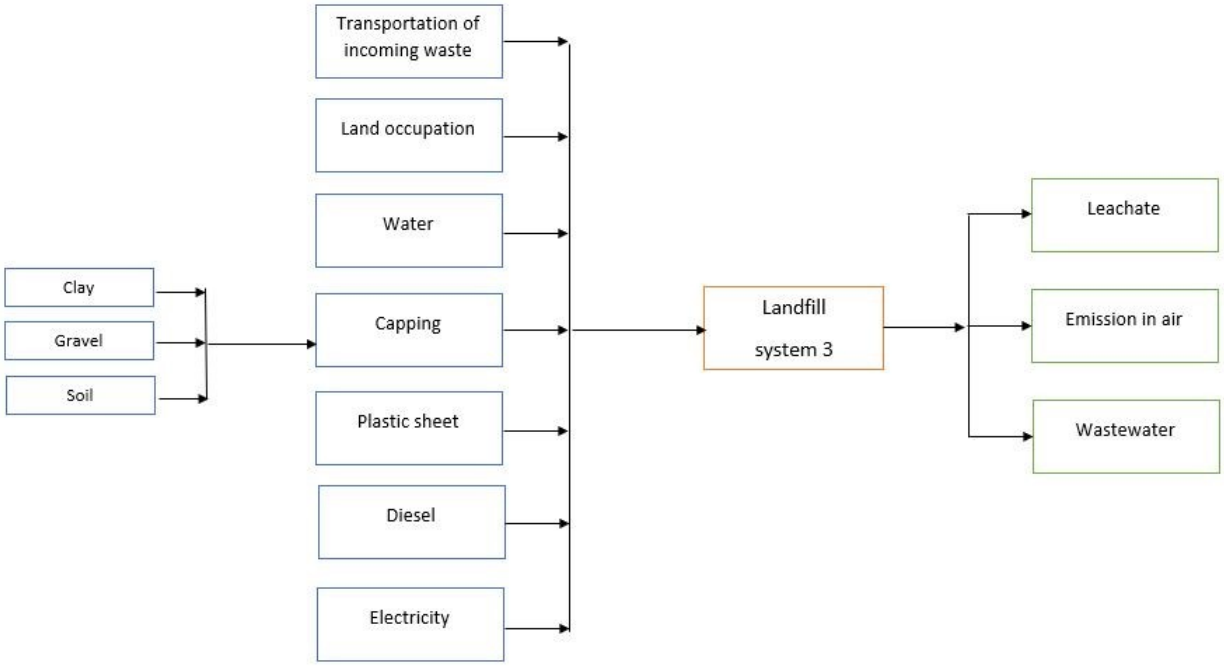


Figure3.4: Burdens of system 3

The input to the system are land occupation referred to 2018 waste, the transportation of the incoming waste, the utility usage which are diesel and electricity, the water used, the use of plastic sheet for the daily coverage of waste and capping which will occur in 2026, referred to the quantity of waste treated in 2018. System outputs are wastewater, leachate and emissions to air. As for the previous systems the temporal burdens refer to all the streams entering and exiting the system in 2018 or that are referrable to waste treated in 2018.

The total amount of waste treated in 2018 is 4398,95 ton while the amount of previously landfilled waste at 31/12/2017 is 174'787 ton and consequently at the 31/12/2018 the amount of waste landfilled in system 3 is 179'185,95 ton.

Municipal solid waste is transported through the plant from different location by using lorries with EURO 3 characteristic and the referred parameter in the Ecoinvent database is "transport, freight, lorry 16-32 metric ton, EURO 3 {RER}| Cut-off, U". Transportation assessment is described as the product between the amount of MSW, expressed as tonnes, and the distance, expressed in kilometres, of the path (Table(3.48)).

Table 3.48: Transportation assessment of the incoming waste in system 3

Incoming waste	Amount (ton)	Distance (km)	Transportation assessment (tkm)
Path 1	2241,74 ton	4 km	8'966,96 tkm
Path 2	369,77 ton	12,1 km	4'474,21 tkm
Path 3	147,96 ton	79,52 km	11'765,78 tkm
Path 4	79,52 ton	33,2 km	2'640 tkm
Path 5	47,94 ton	17,9 km	858,12 tkm
Path 6	182,42 ton	26,7 km	4'870,61 tkm
Path 7	1329,5 ton	57,6 km	76'579,2 tkm

Also considering system 3 is possible to identify two main categories of inputs which are natural ones and technosphere ones.

Considering firstly the natural ones it is possible to identify water and land occupation to this category, similarly to what happened in system 2. Particularly land occupation is calculated by considering the volume of the excavation, which shape can be described as a cone, and the base area of the cone. The area defendant to the amount of waste treated in 2018 in system 3 is calculated by multiplying the total area to an allocation factor. The allocation factor is defined as the ratio between the amount of waste treated in 2018 (4'398,95 ton) and the total amount of waste at the end of 2018 (179'185,95 ton) which is so equal to 0.02455. Results are reported in Table(3.49).

Table 3.49: Natural input considering system 3 and their definition in the SimaPro model

Natural input	Amount (unit of measure)
Water, process, unspecified natural origin	2'000 l
Occupation, dump site, temperate grassland and savannah	14'952,86 m^2

Before being landfilled waste is pre-treated inside the plant: firstly it is shredded and moved thanks to a wheel loader, then using compactors, excavators and backhoes, it is placed in landfill. All this operating machine are powered by diesel and their proper consumptions are reported in Table(3.50). considering diesel from market process. Since diesel consumption must be expressed is mass unit inside the model, a density of 0,835 kg/l is assumed for diesel. (Table(3.51))

Table 3.50: Machine operator diesel consumption in system 3

Diesel	Amount (unit of measure)
Compactor	9118 l
Backhoe	627 l
Excavator	9560 l
Shredder	8940 l
Wheel loader	4244 l

Table 3.51: Diesel consumption in system 3 and its definition in the SimaPro model

Diesel	Amount (unit of measure)
Diesel {RER}| Market group for | Cut-off, U	27’128,315 kg

Electricity is the other source of power used in system 3. It is assumed to be taken from the Italian grid, considering a medium voltage and the data of consumption are reported in Table(3.52).

Table 3.52: Electric energy consumption in system 3 and its definition in the SimaPro model

Electricity	Amount (unit of measure)
Electricity medium voltage {IT}| market for| Cut-off U	216’974 kWh

In order to ensure the waste coverage, avoiding it to be in contact with the environment and trying to prevent the formation of a larger amount of leachate, due to rainfall water percolating through waste, and HDPE plastic sheet is used for the daily coverage. Plastic sheet is characterised by a thickness of 1 mm and the total area covered is of 11’000 m^2. Since results need to be reported in mass units, a density of 940 kg/m^3 is assumed for the HDPE. The process considered refer to an extrusion into plastic film, which is characterised by an efficiency of 0,976 kg/kg as previously underlined. Together with the plastic sheet production also its transportation to landfill is considered. Results are written in Table(3.53).

Table 3.53: Plastic sheet consumption for daily coverage and its definition in the SimaPro model

Plastic sheet	Amount (unit of measure)
Polyethylene, high density, granulate {GLO}| market for| Cut-off U	10340 kg
Extrusion plastic film {RoW}| extrusion, plastic film| Cut-off, U	10594,262 kg
Distance	50 km
Transportation assessment	517’000 kgkm

Also for system 3 capping procedure is considered inside the inventory analysis. Capping allows the landfill to be isolated from the environment in the years following its closure, also allowing the collection and subsequent treatment of the leachate and biogas. To ensure a proper insulation, multiple layers of materials are used, with first waterproofing and then draining properties, starting from the outside towards the inside. This will allow the correct action of the cover both for insulation and emission recover. Capping will occur on march 2026 and it in order to refer it to the quantity of waste treated in 2018 an allocation parameter is defined as the ratio between the waste treated in 2018 and the total amount of waste landfilled at the closure. To calculate this value the amount of waste treated per year since 2026 is estimated as 4’000 ton/year. The value of the allocation factor is 0.02067 and reported in the model as descripted in Table(3.54).

Table 3.54: Capping procedure for system 3 and its definition in the SimaPro model

Capping	Amount (unit of measure)
Capping	0,02067 piece

Capping comprehends the use of clay for the impermeabilization layer, drainage material (which is gravel) and soil. The information on quantity are reported in Table(3.55).

Table 3.55: Process that contributes to capping in system 3

Capping	Amount (unit of measure)
Clay	18900 ton
Gravel	13500 ton
Soil	30600 ton

Clay is used for the impermeabilization for a total amount of 9000 m^3 and with a density of 2.1 ton/m^3. Information are reported in Table(3.56).

Table 3.56: Clay used for capping in system 3

Clay	Amount (unit of measure)
Clay {RoW}| clay pit operation | Cut-off	18900 ton

After the clay layer gravel is used to ensure the drainage of gas and liquid. The total amount of gravel used is 9000 m^3 with a density of 1,5 ton/m^3. Data are reported in Table(3.57).

Table 3.57: Gravel used for capping in system 3

Gravel	Amount (unit of measure)
Gravel, crushed {RoW}	13500 ton

Finally to complete capping procedure, soil is used with a total amount of 18000 m^3 and a density of 1,7 ton/m^3 as reported in Table(3.58) where excavation of soil through hydraulic digger is also considered.

Table 3.58: Soil used for capping in system 3

Soil	Amount (unit of measure)
Excavation, hydraulic digger {GLO}| market for | Cut-off, U	18000 m^3
Soil (natural input)	30600 ton

Considering the system output there are three of them: waste water, air emission and leachate.
Waste water regards all the water used in 2018 for normal plant operation mostly linked the cleaning of operating machines and them collected inside a tank. The cleaning tank operation in 2018 lead to the removal of 2000 l of waste water as underlined in Table(3.59) where the process considered in SimaPro is also reported.

Table 3.59: Waste water outing the plant in 2018 in system 3

Waste water	Amount (unit of measure)
Wastewater, average {Europe and Switzerland}| market for, average| Cut-off, U	2000 l

Considering air emission it is firstly necessary underline the fact that it is different from biogas characteristic of system 2. Indeed, in system 3 the biogas that forms is still collected but, due to the high concentrations of methane that characterize it, it needs to be disposed of. In this regard, the biogas is then burned to reduce the plant's methane emissions together with all those potentially dangerous components. In the previous years of operation of the plant, the high concentrations of methane made it possible to recover energy using a cogenerator. However, methane production in 2018 did not yield to an economically feasible recover of energy and this plant of the plant has been shutter down. The need to reduce emissions however remains and for this reason the gases are sent to a flare where they are burned at temperatures higher than 800 ° C in order to favour the combustion of all the compounds present. To evaluate the concentration of the air emission firstly it was considered the analysis of gas exiting the flare which comprehend information on dust, nitrogen oxides, sulfur oxides, fluorides and chlorides concentration together with CO and total organic carbon (TOC) that permits to describe if total combustion has occurred. As for the other systems analysis concentration analysis are assumed to be constant and representative of all 218. However analysis do not report information on the amount of CO_2 produced thought combustion and the one proper to the biogas emission, which are remarkable in the following impact evaluation of system 3. To evaluate this concentration monthly analysis on biogas composition are accounted, which report information on CH_4 and CO_2, by mediating the values of concentration. Those values are reported in Table(3.60) as volume percentage and permits to underline how the methane concentration for system 3 is bigger than the one founded in biogas in system 2. (In Table(A6) monthly concentration values are reported).

Table 3.60: Biogas concentration of methane and carbon dioxide in system 3

Biogas concentration of CH4 and CO2	Amount (unit of measure)
Methane	45,382 %
Carbon dioxide	34,32 %

To evaluate the carbon dioxide derived from methane combustion is assumed to have complete combustion for methane and considering 1 Nm^3 of biogas. Since the model need to report CO_2 data in mass unit a density of 0,717 kg/m^3 has been assumed for methane at normal conditions (T=0°C and P = 1 atm). The result of the combustion leads to a production of CO_2 of 892,75 g. The amount of CO_2 already present in biogas has been calculated considering a density at normal condition of 1,976 kg/m^3 leading to 678,16 g of CO_2. Finally, it is necessary to consider the organic compounds that has been reported in analysis and also their contribution to the CO_2 emission. Table containing all components founded in biogas is reported in Table(A7) where they are expressed in mass units. Similarly to CH_4 the amount of CO_2 is calculated considering complete combustion of the every organic compound while the emission of sulfur oxides, nitrogen oxides, chlorides (as HCl) and fluorides (as HF) are assumed to be equal to the one founded in gas after combustion analysis. The amount of CO_2 produced from this combustion is equal to 93,9 mg. Finally results are reported in Table(3.61) and they describe the characteristic components of emission in air process considering 1 Nm^3 of gas. Information on the amount of CO and TOC founded in the analysis are symptom of a lack of complete of combustion and so they describe how carbon present in the analysed components it is not completely oxidised. Particularly TOC is described as the amount of carbon founded in an organic compound and it can be used also to describe the quality of water and air. To describe this situation the amount Of CO and TOC is subtracted from the previous founded value of CO_2, as underlined in Table(3.61). As for system 2 analysis refers to concentration of some components as lower than a certain attention threshold value which is chosen as the composition value of the component to ensure a conservative condition for the study. This components has been defined in Table(3.61) whit an asterisk. (Calculation method are in Table(A4)).

*Table 3.61: Emission in air composition of 1 Nm^3 in system 3. * indicates components that are expressed as the upper limit value following the composition hypothesis*

Emission in air composition of 1 Nm^3	Amount (unit of measure)
Particulates	0,96 mg
Carbon monoxide	101 mg
Nitrogen oxides	137 mg
Total Organic Carbon	43,7 mg
Sulfur oxides	23,7 mg
Fluoride compounds *	1,64 mg
Hydrogen chloride	4,47 mg
Carbon dioxide = $678{,}16 + 892{,}75 + 0.0939 - 0{,}101 - 0{,}0437$	1570,8592 g

The total amount of emission in air produced by the combustion in 2018 are linked to the total amount of waste landfilled since 2018 and it is equal to 71732,9 Nm^3. It is necessary so to refer to an allocation parameter, which is the same considered in the land occupation calculation in system 3, equal to the ratio between the amount of waste treated in 2018 and the amount of waste landfilled since 2018. This allocation value is equal to 0.02455. Considering the system boundaries, it is necessary also to take into account the amount of emission in air associated to waste landfilled in 2018 after the plant closure. The amount of biogas that is necessary to treat after the landfill closure is 20’000 Nm^3 which have assumed to be characterised by the same composition of the one in 2018. The allocation parameter necessary to describe this contribution is equal to the one previously defined for capping and it is equal to the ratio between waste treated in 2018 and the total amount of waste landfilled at the landfill closure, equal to 0,02067. Table(3.62) report the "emission in air" inside the model which is described by the sum of the two previous defined contribution.

Table 3.62: Emission in air process in system 3

Emission in air	Amount (unit of measure)
Emission in air	2174,485 Nm^3

Finally the last output of system 3 is leachate emission. As previously underline leachate formation is caused by water which percolates through waste and to decomposition processes. The total amount of leachate produced in 2018 its equal to 5469,8 ton characterised by a density of 1 kg/l. First of all it is necessary to consider the composition of 1 kg of leachate that can be found in 2018 analysis. As for leachate in system 2 also in this case some component concentration is expressed as lower than a certain threshold value, which is defined as the concentration value in order to perform a conservative study. Analysis are reported in Table(3.63) and components that suffers the previous hypothesis, are underlined with an asterisk. Furthermore some values are not defined inside the database and they are expressed in term of "generical organic compounds" inside the model. (In Table(A5) evaluation method are reported).

Table 3.63: Leachate composition for system 3 indicates components that are expressed as the upper limit value following the composition hypothesis*

Leachate composition	Amount (unit of measure)
Oxygen	0,0325 mg
Fluoride	35,33 mg
Chlorides	372,275 µg
Nitrite	7,833 µg
Nitrate	7,25 µg
Sulfate	41,46 mg
Ammonia, as N	621,75 mg
Iron	2163 µg
Manganese	194,11 µg
COD, Chemical Oxigen Demand	3800 mg
BOD5, Biological Oxigen Demand	120 mg
TOC, Total Organic Carbon	990 mg
Cyanide compounds *	5 µg

Arsenic*	0,06 mg
Cadmium *	0,01 mg
Chromium	0,45 mg
Chromium VI*	100 µg
Copper	0,09 mg
Mercury *	0,06 mg
Nickel	0,14 mg
Lead	0,08 mg
Zinc	0,64 mg
Calcium	133 mg
Magnesium	44,5 mg
Potassium	535 mg
Sodium	624 mg
Benzene *	0,0 µg
Benzene, -ethyl- *	0,1 µg
Styrene *	0,1 µg
Toluene *	0,5 µg
Xylene *	0,1 µg
Benzene, 1,2,4-trimetyl-*	0,2 µg
Benzene, -butyl- *	0,2 µg
Benzene, 1-propyl- *	0,1 µg
Hydrocarbon aromatic *	0,1 µg
Methane, chloro-, HCC-40 *	0,1 µg
Organic compound * (vinyl chloride, PFAS)	8,21 µg
Chloroform *	0,005 µg
Ethane, 1,2-dichloro- *	0,1 µg
Ethane, 1,1-dischloro- *	0,005 µg
Ethane, trichloro- *	0,1 µg
Ethane, tetrachloro- *	0,1 µg
Butadiene, hexachloro- *	0,005 µg
Ethane, dichloro-(cis) *	0,05 µg
Ethane, dichloro-(trans) *	0,05 µg
Propane, 1,2-dichloro- *	0,005 µg

Ethane, 1,1,2-trichloro- *	0,005 µg
Propane, 1,2,3-trichloro-	0,0001 µg
Ethane, 1,1,2,2-tetrachloro-	0,0001 µg
Bromoform *	0,005 µg
Ethane, 1,2-dibromo-*	0,001 µg
Methane, bromodichloro-*	0,005 µg
Organic compounds *(dibromodichloromethane)	0,005 µg
Acrylonitrile *	8 µg
Benzene, 1-methyl-2-nitro *	8 µg
Benzene, 1-methyl-3-nitro *	8 µg
4-nitrotoluene*	8µg
Etridiazole *	0,004 µg
Lindane, alpha-*	0,004 µg
Lindane, beta- *	0,004 µg
Atrazine *	0,004 µg
Delta-hexachlorocyclohexane *	0,004 µg
Lindane *	0,004 µg
Alachlor *	0,004 µg
Heptachlor *	0,004 µg
Aldrin *	0,004 µg
Terephthalate, dimethyl 2,3,5,6-tetrahloro *	0,004 µg
Heptachlor, epoxide *	0,004 µg
Chlordane, cis *	0,004 µg
Chlordane, trans *	0,004 µg
Trans-nonachlor *	0,004 µg
Endosulfan sulfate *	0,004 µg
Cis-permethrin *	0,004 µg
Trans-permethrin *	0,004 µg
Ametryn	0,115 µg
Bromacil *	0,004 µg
Chloropropham *	0,004 µg
Cycloate *	0,004 µg
Cyanazine *	0,004 µg

Diphenamid *	0,004 µg
Dipropylthiocarbamic acid S-ethyl ester *	0,004 µg
Fenarimol *	0,004 µg
Fluridone *	0,004 µg
Hexazimone *	0,004 µg
Metolachlor *	0,004 µg
Molinate *	0,004 µg
Napropamide*	0,004 µg
Norflurazon *	0,004 µg
Pebulate *	0,004 µg
Prometon *	0,004 µg
Prometryn *	0,036 µg
Pronamide *	0,004 µg
Propachlor *	0,004 µg
Propazine *	0,004 µg
Tebuthiuron *	0,107 µg
Terbacil *	0,004 µg
Terbutryn *	0,004 µg
Triadimefon *	0,004 µg
Tricyclazole *	0,004 µg
Diazinon *	0,004 µg
Dichlorvos *	0,004 µg
Disulfoton *	0,004 µg
Fenamiphos *	0,004 µg
Methyl paroxan *	0,004 µg
Mevinfos *	0,004 µg
Terbufos *	0,004 µg
Tetrachlorvinphos *	0,004 µg
Trifluralin *	0,004 µg
Naphthalene *	0,11 µg
Fluorene *	0,11 µg
Phenanthrene *	0,11 µg
Anthracene *	0,11 µg

Fluoranthene *	0,11 µg
Pyrene	0,24 µg
Benzo(a)anthracene *	0,11 µg
Chrysene *	0,11 µg
Benzo(b,j,k)fluoranthene *	0,13 µg
Benzo(a)pyrene *	0,01 µg
Diben(a,h)anthracene *	0,01 µg
Benzo(g,h,i)perylene *	0,01µg
Indeno(1,2,3-cd)pyrene *	0,01 µg
Phenol	0,6 µg
Phenol, 2-chloro *	0,02 µg
o-cresol	0,16 µg
m-cresol	2 µg
3-nitrophenol	0,08 µg
Phenol, 2,4-dimethyl- *	0,02 µg
Phenol, 2,4-dichloro- *	0,02 µg
Metacresol, parachloro-*	0,02 µg
2,6-dichlorophenol *	0,02 µg
Phenol, 2,4,6-trichloro-	0,02 µg
Phenol, 2,4,5-trichloro-*	0,02 µg
Phenol, pentachloro- *	0,02 µg
Phenol, 2,3,4,6 -tetrachloro- *	0,02 µg

Furthermore the total amount of leachate produced in 2018 is linked to all waste landfilled production and it is necessary to refer only to the part imputable to 2018 waste. In order to this an allocation parameter is used, as for emission in air process. This parameter is defined as the ratio between the amount of waste treated in 2018 and the amount of waste landfilled since 2018, consequently equal to 0.02455. It is also needed to take into account the production of leachate after the landfill closure which is estimated as 1’200 ton of leachate produced per year. Since the period of post-management considered is 30 years the total amount of leachate produced is 36’000 ton. The allocation parameter needed to refer this amount to waste landfilled in 2018 is equal to the ratio between this quantity and the total amount of waste landfilled at the closure, equal to 0.02455. Leachate is then transported

Table 3.64: Leachate and its transportationin system 3

Leachate and its transportation	Amount (unit of measure)
Leachate	878,515 ton
Distance	141 km
Tranportation assessment	123’870 tkm

After the life cycle inventory phase, proposed in this chapter, following the LCA methodology proposed by ISO 14040 (2006) and ISO 14044 (2018), the life cycle assessment phase will be performed in chapter 4.

Chapter 4

Impact assessment and interpretation

Chapter 4 contains the third and the fourth phase of the LCA methodology (ISO 2006a) regarding impact evaluation and result interpretation.

Firstly, the selected method, CML baseline, is described together with its impact categories and then it is applied to all the system considered thanks to the SimaPro software. The characterisation results are interpreted as a whole and considering all the single impact categories, considering the system grouping, and underlining the most impactful processes and materials. Then a comparison between the three systems in terms of environmental performances is considered by analysing the results are impact over tonne of MSW treated. Together with the mandatory characterisation analysis, results are studied by mean of uncertainty analysis and sensitivity analysis underling the peculiarity of the three systems. Finally the result interpretation is reported.

4.1 Impact assessment

The impact assessment represents the third phase of the LCA methodology and permits to evaluate the potential impacts of the systems. In this step the collected component of the inventory analysis is translated in terms of environmental impacts considering different impact categories, proper to the method, thanks to characterisation factor. Particularly in this study impact evaluation is considered only by mean of characterisation.

4.1.1 Description of the CML baseline method

CML method was firstly proposed in 2001 by the Centre of Environmental Science of Leiden University within the publication “operational guide to the ISO standards” (Guinèe et al. (2002)) where information about impact categories and characterisation method are reported. Differently from others method of representation, as Eco-indicator 99 and EPS, which are based on a “damage approach”, CML method is the set of impact categories defined for the “problem oriented approach”. The selected method is the CML baseline method which is composed of ten impact categories, differently from the extended version made of sixteen impact categories, and they are all midpoint impact categories. The categories are reported in Table (4.1).

Table 4.1: CML baseline method addressed at midpoint level (Guinèe et al. (2002))

Impact categories	**Unit**	**Notes**
Abiotic depletion	kg Sb eq.	It evaluates to the depletion of non-biological resources such as metals, minerals and other raw materials.
Abiotic depletion (fossil fuels)	MJ	It evaluates the depletion of fossil fuels and their use for the energy production.
Global warming potential	kg CO_2 eq.	It evaluates the impact on global warming caused by the emission of greenhouse gases.
Ozone layer depletion	kg CFC-11 eq.	It evaluates the depletion of the ozone layer due to different gases.
Human Toxicity	kg 1,4-DB eq.	It evaluates the effect of toxic substances on humans.
Fresh water aquatic ecotoxicity	kg 1,4-DB eq.	It evaluates the impact of toxic substances on air, water and soil to the fresh-water ecosystem.
Marine aquatic toxicity	kg 1,4-DB eq.	It evaluates the impact of toxic substances on air, water and soil to the marine ecosystem.
Terrestrial ecotoxicity	kg 1,4-DB eq.	It evaluates the impact of toxic substances on air, water and soil to the terrestrial ecosystem.
Photochemical oxidation	kg C_2H_4 eq.	It evaluates the impact of smog caused by photochemical reactions.
Acidification	kg SO_2 eq.	It evaluates the impact connected to the formation of acid rain.
Eutrophication	kg PO_4 eq.	It evaluates the impact due to the potential eutrophication of water bodies.

4.1.2 Impact assessment results of system 1

The process was grouped as follows:

- Energy consumption: contains the use of electricity for the mechanical treatment of MSW inside the plant (weights, shredding, conveyor belt, iron removal, screening procedure and general plant utilities) and diesel consumption.
- Input: it includes all the plant input's comprehending the MSW entering the plant and their transportation and the use of lubricant oil.
- Output: it includes all the plant output's comprehending the waste exiting the plant and their transportation and their final disposal (incineration or landfilling), transportation and disposal of lubricant oil, transportation of maintenance waste and disposal.

Input, output and energy consumption are compared through a group analysis. The absolute values of the impact assessment results are obtained with the CML method and reported in Table(4.2)

Table 4.2: Characterisation phase results of system 1

Impact **category**	**Unit of Measure**	**Total**	**Energy consumption**	**Input**	**Output**
Abiotic depletion	kg Sb eq	8,94E-01	1,73E-02	9,84E-02	7,79E-01
Abiotic depletion (fossil fuels)	MJ	6,21E+06	7,86E+05	5E+05	4,93E+06
Global warming (GWP100a)	kg CO2 eq	4,53E+06	2,82E+04	3,28E+04	4,47E+06
Ozone layer depletion (ODP)	kg CFC-11 eq	7,62E-02	9,02E-03	6,1E-03	6,11E-02
Human toxicity	kg 1,4-DB eq	9,59E+06	6,44E+03	1,09E+04	9,58E+06
Fresh water aquatic ecotox.	kg 1,4-DB eq	6,46E+07	5,94E+03	3,16E+03	6,46E+07
Marine aquatic ecotoxicity	kg 1,4-DB eq	3,13E+11	2E+07	8,84E+06	3,13E+11
Terrestrial ecotoxicity	kg 1,4-DB eq	9,36E+03	5,7E+01	4,65E+01	9,26E+03
Photochemical oxidation	kg C2H4 eq	8,3E+02	7,64 E+00	6,06E+00	8,17E+02
Acidification	kg SO2 eq	2,34E+03	2,07E+02	1,67E+02	1,96E+03
Eutrophication	kg PO4--- eq	1,54E+04	4,64E+01	4,03E+01	15326,21

The results are graphically represented in the bar chart in Figure(4.1) and are reported in a percentage scale because of the different unit of measure. The colours permits to underline the different groups: green refers to energy consumption, light green to input and orange to output.

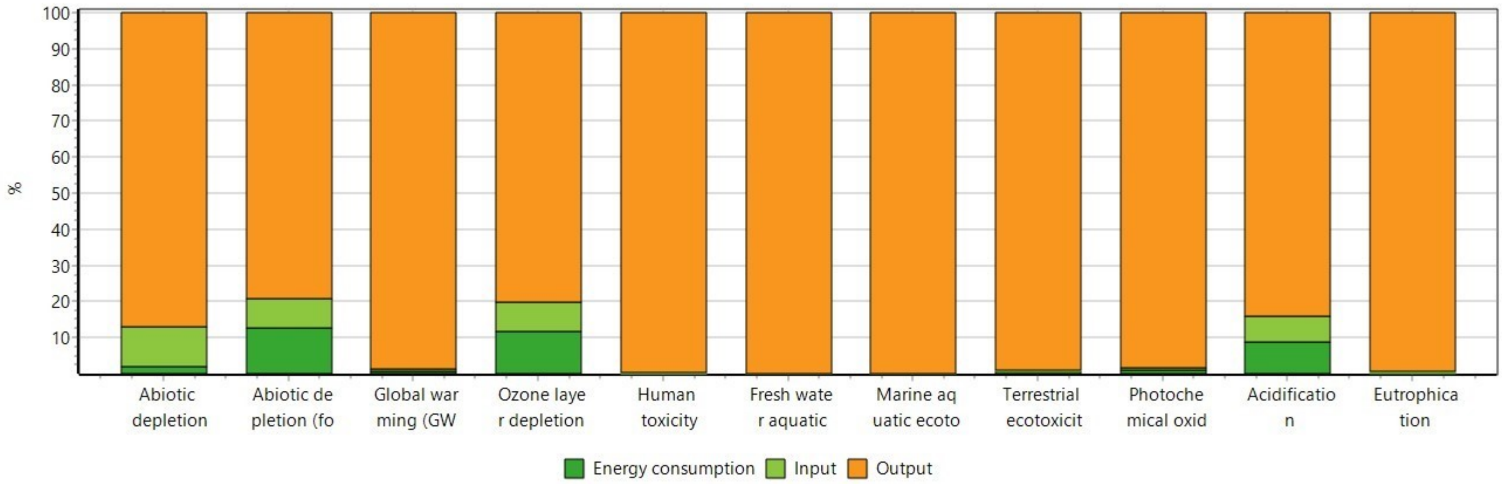


Figure 4.1: Characterisation results described through a histogram representation (personal rework of SimaPro data)

Looking at the previous graph Figure(3.1) is possible to underline how the main contribution for every impact category is due to the output while energy consumption and input contributes less than this group. Moreover the contributions of the last two groups is almost the same with energy consumption slightly bigger than input.

Finally considering the single impact categories and considering Table(4.2) it is possible to fully evaluate environmental performances of the system.

- Abiotic depletion: 0.894 kg Sb eq.

The major contribution to this category is the output (87% with 0,779 kg Sb eq.) while input (11% with 0,098 kg Sb eq.) and energy consumption (2% with 0,017 kg Sb eq.). Abiotic depletion is mainly due to transportation of waste: particularly the transportation of waste entering the process contributes for the 0.098 kg Sb eq. (11%) while the transportation from the process to the subsequent processes, incineration and sanitary landfill, contributes for 0.604 kg Sb eq. (67.57%). This is mainly due to the presence of materials like cadmium, lead, silver chromium, copper and gold found in the lorry transportation process.

- Abiotic depletion (fossil fuels): 6214636,4 MJ

The main contribution to abiotic depletion is the transportation of waste in entrance to the plant for the 500752 MJ (8%) but mostly for the output transportation with 3077766 MJ (49,5%). Another important contribution is due to the use of diesel fuel for the activities inside the plant with 754268

MJ (12%). Reinforcement of what has been said, crude oil contributes for the 81,9 %. The two subsequent process, incineration and sanitary landfill respectively contributes for the final result with 430457 MJ (7%) and 1339931 MJ (21,5%) considering that the last one is mainly caused by the use of pitch in the construction.

- Global warming potential (GWP): 4528726,3 kg CO_2 eq.

Global warming potential is mainly characterised by the contribution of the two final destination of waste: landfill with 3538059 kg CO_2 eq. (79,3%) and incineration with 66944 kg CO_2 eq. (14,8%). Transportation of waste entering and exiting the plant contributes only to the 5,3% about (4.45 % for the exiting transportation and 0,8% for the entering one). The main substance which describes the impact is the biogenic methane with 3332550 kg CO_2 eq. (73%) associated mainly to landfilling.

- Ozone layer depletion: 0.076 kg CFC-11 eq.

There are different processes that contributes to the ozone layer depletion: the main one is the transportation of waste outside the plant with 0,037 kg CFC-11 eq. (49,2%) while transportation of waste inside the plant is characterised by 0,006 kg CFC-11 eq. (8 %) and the use of diesel fuel for the plant activity by 0,009 kg CFC-11 eq. (12,8%). Other two significant contribution are sanitary landfill with 0,017 kg CFC-11 eq. (23%) and incineration with 0,005 kg CFC-11 eq. (6,6 %). All of them can be strictly referred to the petroleum industry production and the use of halon 1301 which contributes for the 88% to ozone layer depletion.

- Human toxicity: 9596927,4 kg 1,4-DB eq.

The main contribution to human toxicity impact category is related to the two final scenarios incineration and sanitary landfill. Indeed, the first one represents with 8388904 kg 1,4-DB eq. the 87,4% of the total and sanitary landfill, with 1082048 kg 1,4-DB eq., the 11,3%. The most impactful substance is indeed beryllium as a result of the incineration residues.

- Fresh water aquatic ecotoxicity: 64621613 1,4-DB eq.

Fresh water aquatic ecotoxicity category is mainly due to the incineration treatment with 53366501 1,4-DB eq. (82,6%) and to sanitary landfill with 11200103 1,4-DB eq. (17,3%). The substance that contributes to this category are beryllium (80,3%), copper (14,3%) and nickel (3,5%) mainly linked the incineration residues. Contribution is due only to the output group (99,99%) while input and energy consumption are so small that they can be considered as null.

- Marine ecotoxicity: 3,125 E11 1,4-DB eq.

Incineration 3,0703821E11(98,2%) and sanitary landfill 5,3995944E9 (1,72%) are the main contribution to this category and they are associated to the output group as the end of life of MSW.

Output so represents 99.99% of the total impact while the contribution of the other two groups is so small that can be almost neglected.

- Terrestrial ecotoxicity: 9363,385 1,4-DB eq.

There are two main processes that contributes to the terrestrial ecotoxicity: sanitary landfill with 7934,694 1,4-DB eq. (84,7 %) and incineration with 747,114 1,4-DB eq. (8%). The distribution network for electricity has a lower contribution of 115, 082 1,4-DB eq. (1,2%) to this impact category. The impact for this category is mostly due to mercury emission in water (83,4%) and in air (11,7%).

- Photochemical oxidation: 830,974 kg C_2H_4 eq.

Photochemical oxidation is mainly characterised by the sanitary landfill contribution with 756,03817 kg C_2H_4 eq. (91%) and by the incoming and outcoming waste transportation 12,943155 kg C_2H_4 eq. (1,6%). As a matter of fact, the main contributing substance are methane (biogenic 86% and fossil 4,2%) and sulphur dioxide (4,41%).

- Acidification: 2341,575 kg SO_2 eq.

There are several processes that contributes to acidification: the first one is transportation for both transportation of waste outside the plant 1027,7684 kg SO_2 eq. (43,9%) and transportation of waste inside the plant 167,218 kg SO_2 eq.(7,1%), then sanitary landfill with 619,907 kg SO_2 eq. (26,5%) and incineration 292,538 kg SO_2 eq. (12,5%) and finally the electricity used for the operations inside the plant with 151,589 kg SO_2 eq. (6,5%).

- Eutrophication: 15412,961 kg PO_4 eq.

The major contribute to eutrophication is given by the sanitary landfill with 14571,274 kg PO_4 eq. (94,5 %). The remains processes are incineration with 357,839 kg PO_4 eq. (2,3%) and transportation of waste inside and outside the plant with 180,523 kg PO_4 eq. (1,2%).

4.1.3 Impact assessment results of system 2

The process was grouped as follows:

- Materials: contains contributions due to the use of iron wire and silt for the operations inside the landfill plant.
- Utility: contains the contribution of electric energy consumption and diesel consumption used as utility for plant operations and machines.
- Capping: contains all the contributes to the capping procedure.
- Incoming waste: contains the contribution due to the input waste and their transportation to the plant.

- Leachate: contribution due to the leachate formation which is an output of the system.
- Biogas: contribution due to biogas formation which is an output of the system.

The absolutes values of the impact assessment results are represented in Table(4.3).

Table 4.3: Characterisation results of system 2

Impact category	**Unit of measure**	**Total**	**Materials**	**Utility**	**Capping**	**Incoming waste**	**Leachate**	**Biogas**
Abiotic depletion	kg Sb eq	3,9E-01	2,32E-02	7,65E-03	3,46E-01	1,28E-02	0E-00	0E-00
Abiotic depletion (fossil fuels)	MJ	1,02E+06	1,59E+04	4,38E+05	4,58E+05	1,09E+05	0	0
Global warming (GWP100a)	kg CO2 eq	4,8E+04	1,27E+03	1,63E+04	2,34E+04	6,9E+03	0	1,96E+00
Ozone layer depletion (ODP)	kg CFC-11 eq	8,25E-03	1,39E-04	5,13E-03	1,64E-03	1,34E-03	0	0
Human toxicity	kg 1,4-DB eq	5,65E+04	1,70E+03	3,85E+03	5,15E+03	2,77E+03	4,31E+04	0
Fresh water aquatic ecotox.	kg 1,4-DB eq	3,54E+04	7,47E+02	3,77E+03	3,88E+03	6,25E+02	2,64E+04	0
Marine aquatic ecotoxicity	kg 1,4-DB eq	3,35E+07	4,57E+06	1,28E+07	1,33E+07	,87E+06	9,07E+05	0
Terrestrial ecotoxicity	kg 1,4-DB eq	9,9E+01	6,62E+00	3,66E+01	3,07E+01	1,03E+01	1,46E+01	0
Photochemical oxidation	kg C2H4 eq	1,11E+01	4,37E-01	4,5E+00	4,93E+00	1,25E+00	0	5,8E-04
Acidification	kg SO2 eq	3,1E+02	9,5E+00	1,31E+02	1,33E+02	3,59E+01	0	4,46E-06
Eutrophication	kg PO4--- eq	2,19E+02	3,48E+00	2,98E+01	3,4E+01	8,6E+00	1,43E+02	0

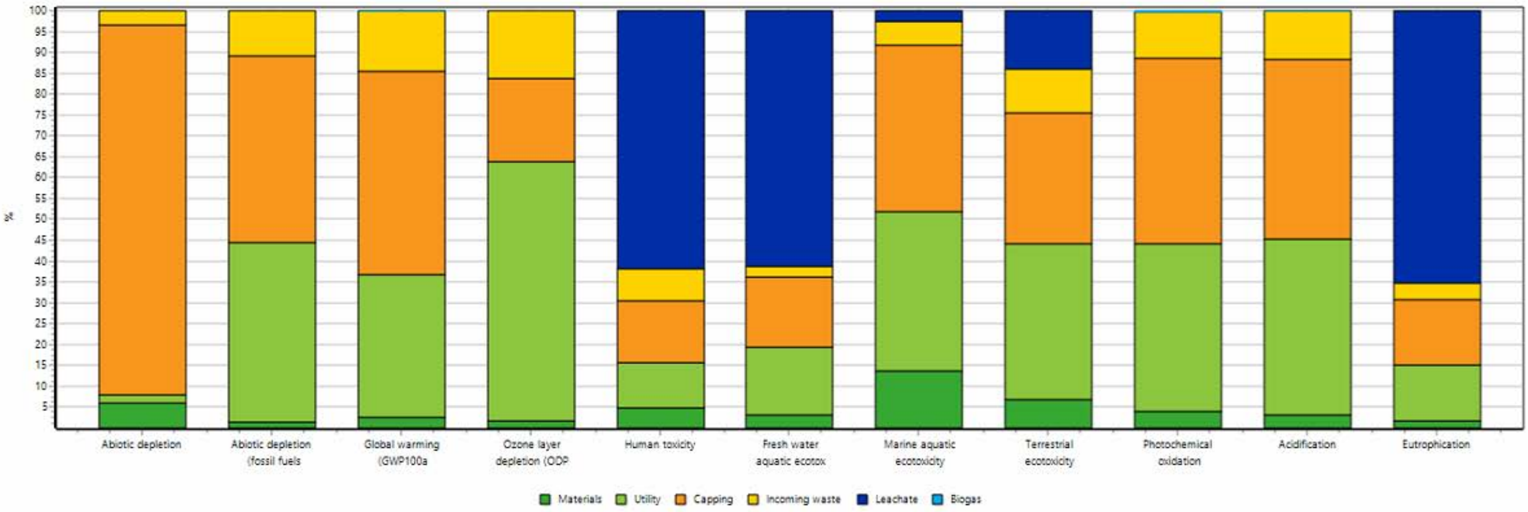


Figure 4.2: Characterisation results described through a histogram representation (personal rework of SimaPro data)

Differently from the previous case, there is no group whose contribution prevails over the others for each impact category. Certainly, it is possible to underline that capping represent the one with higher weight for all categories followed by the use of electricity as utility. The leachate produced is predominant describing human toxicity, fresh-water aquatic ecotoxicity and eutrophication while is relevant in describing marine aquatic ecotoxicity and terrestrial ecotoxicity. For all the other categories it is null. Biogas is considerable in the description of global warming potential, photochemical oxidation and acidification but his contribution is in every of this case really small, due also to a really low concentration of biogenic methane in it. Contribution of silt and iron wire are not so wire in all the considered impact categories.

Lastly all the categories are described thanks to values reported in Table(4.3) and Table(A.10).

- Abiotic depletion: 0,39 kg Sb eq.

The contribution to this category is mainly linked to capping (88,8% with 0,346 kg Sb eq.) while silt contributes for 5,5% (0,021), transportation of waste inside the plant for 3,3% (0,012) and electricity as utility for 1,5% (0,006). Diesel and iron wire contribute is almost negligible. Indeed, for the capping procedure, zinc steel is used and its production comprehends the galvanization process which becomes the most relevant one (84,6%) in terms of contribution.

- Abiotic depletion (fossil fuels): 1182995,502MJ

The main contribution to abiotic depletion for fossil fuel is given by capping (44,8 % with 458075,252448045 MJ) followed by diesel used as a plant utility (26,9% with 275030,313 MJ) and electricity utilities (16% with 163233,686 MJ). Small contributes are given by the waste transportation (10,7 % 109492,546 MJ). As a matter of fact the petroleum production process is the bigger help for abiotic depletion but it is also relevant the production of plastic components used in the capping like polyethylene and polypropylene.

- Global warming potential (GWP): 48018,31 kg CO_2 eq.

Global warming potential is mainly characterised by the contribution of capping (48,8% with 23464,901 kg CO_2 eq.) followed by the use of electricity as utilities (28,5% with 13681,369 kg CO_2 eq.) and the transportation of MSW to the landfill (14,4% with 6907,633 kg CO_2 eq.). The use of diesel inside the plant contributes only for the 5% (with 2691,468 kg CO_2 eq.). The two substances that are linked to global warming potential are carbon dioxide (90%) and methane (8,2%) both from fossil sources underlining that the contribution lined to the exiting biogas is minimal, as also the data regarding biogenic methane underlines. Indeed biogas helps only with 1,9635 kg CO_2 eq. equal to the 0,004% of the total.

- Ozone layer depletion: 0,00824319742940645 kg CFC-11 eq.

To the ozone depletion category contributes mainly the two utilities: diesel (42,9% with 0,003 kg CFC-11 eq.) and electricity (19,3% with 0,0016 kg CFC-11 eq.) followed by capping (19,8% with 0,0016 kg CFC-11 eq.) and the transportation of MSW inside the plant (16,2% with 0,0013 kg CFC-11 eq.). As for abiotic depletion and abiotic depletion fossil fuels leachate and biogas do not contribute to this category of impact.

- Human toxicity: 35182,468 kg 1,4-DB eq.

The main contribution to is given by leachate (61,7% with 21701,679 kg 1,4-DB eq.) followed by capping (14,6% with 5149,946 kg 1,4-DB eq.), electricity utility (8,6% with 3038,58 kg 1,4-DB eq.) and transportation of MSW inside the plant (7,9% with 2772,241 kg 1,4-DB eq.). The use of silt, iron wire and diesel as utility comprehend the remain 15,8% of the human toxicity category. The biogas exiting the system does not contribute to the impact. Indeed, the presence of hexachlorobutadiene in leachate contributes alone for the 60,8% of the impact and considering substances is the most influent one.

- Fresh water aquatic ecotoxicity: 23236,463 1,4-DB eq.

Fresh water aquatic ecotoxicity category biggest contribute is leachate (61,2% with 14231,327 1,4-DB eq.). Capping represent the 16,6% (3861,839 1,4-DB eq.) of the impact and electricity utility the 14,6% (3391,740 1,4-DB eq.). The remain 7,6% is linked to diesel use, silt and iron wire while biogas contribution is null. Like for the human toxicity impact category the substance that main contributes to the impact is hexachlorobutadiene (60,8%) included in the leachate production.

- Marine ecotoxicity: 33515476 1,4-DB eq.

Capping and electricity used as utility are the main contribution to marine ecotoxicity category corresponding respectively to 40 % (13398692 1,4-DB eq.) and 34,4% (11526827 1,4-DB eq.) followed by iron wire (9,4% with 3143096 1,4-DB eq.). The transportation of MSW inside the plant is linked to 11526827 1,4-DB eq. (5,6%), silt to 1428884 1,4-DB eq. (4,2%), diesel to 1262335 1,4-DB eq. (3,7%) and leachate to 887438 1,4-DB eq. (2,7%).

- Terrestrial ecotoxicity: 97,919 1,4-DB eq.

There are two main contributions to terrestrial ecotoxicity: the first one is the use of electricity inside the plant (32,5% with 31,811 1,4-DB eq.) and the second one is capping (31,3% with 30,7 1,4-DB eq.). They are followed by leachate (13,8% with 13,596 1,4-DB eq.), transportation of MSW inside the plant (10,5% with 31,811 1,4-DB eq.), silt (6% with 5,929 1,4-DB eq.) and diesel used as utility (4,9% with 4,862). Iron wire contribution is almost negligible (1% with 0,698 1,4-DB eq.) while the one of biogas is null.

- Photochemical oxidation: 11,119 kg C_2H_4 eq.

Photochemical oxidation impact category consists mainly of capping contribution for 44,3% (4,935 kg C_2H_4 eq.) and electricity used as utility for 24,9% (2,775 kg C_2H_4 eq.). Then diesel consumption (15,4% 1,722 kg C_2H_4 eq.) and the transportation of MWS inside the plant (11,2% 1,249 kg C_2H_4 eq.) are two relevant ones followed by silt, iron wire and biogas for a total of 4,2%. The biogas contribution is for this category almost negligible with 0,00589 kg C_2H_4 eq. (0,005%).

- Acidification: 310,4156 kg SO_2 eq.

The main contribution of acidification impact category is capping (43,1 % with 133,785 kg SO_2 eq.). The other relevant contributions are electricity as utility (33% with 102,642 kg SO_2 eq.), transportation of waste inside the plant (11,6% with 35,931 kg SO_2 eq.) and diesel consumption (9,2% with 28,529 kg SO_2 eq.). Silt and iron wire contribute for a total of 3% while the contribution of biogas is negligible. Finally leachate does not contribute to acidification.

- Eutrophication: 219,347 kg PO_4 eq.

Leachate is the main contribution with 65,3% (143,349 kg PO_4 eq.) followed by capping with 15,5% (34,065 kg PO_4 eq.) and electricity utility with 11,9% (26,205 kg PO_4 eq.). Transportation of MSW inside the plant represent the 3,9% (8,589 kg PO_4 eq.) while the sum of diesel, silt and iron wire contribution is equal to 3,4%. The contribution of biogas to eutrophication is null.

4.1.4 Impact assessment results of system 3

The process was grouped as follows:

- Incoming waste: contains the contribution linked to the entering waste transportation.

- Capping: contains the contribution of capping procedure.
- Plastic sheet: contains the contribution of the usage of HDPE plastic sheet for the daily coverage and its transportation.
- Utility: contains the contribution of electricity used in plant and diesel used for compactor, backhoe, excavators, shredder and wheel loader
- Waste water: contains the contribution of waste water for 2018.
- Emission in air: contains the contribution of the emission in air after the flare combustion.
- Leachate: contains the contribution of leachate emission and its transportation to the final treatment facility.

The absolute values of the impact results are reported in Table(4.4).

Table 4.4: Impact assessment results for system 3

Impact category	**Unit of measure**	**Totale**	**Utility**	**Incoming waste**	**Capping**	**Plastic sheet**	**Waste water**	**Leachate**	**Emission in air**
Abiotic depletion	kg Sb eq	2,39E-01	4,93E-02	4,97E-02	1,09E-01	1,02E-02	4,32E-06	2,07E-02	0E+00
Abiotic depletion (fossil fuels)	MJ	3,81E+06	2,54E+06	2,53E+05	8,38E+04	7,484E+05	8,96E+00	1,76E+05	0E+00
Global warming (GWP100a)	kg CO2 eq	1,7E+05	1,06E+05	1,65E+04	6,95E+03	2,01E+04	9,8E-01	1,11E+04	3,41E+03
Ozone layer depletion (ODP)	kg CFC-11 eq	3,58E-02	2,93E-02	3,08E-03	7,2E-04	5,93E-04	6,93E-08	2,15E-03	0E+00
Human toxicity	kg 1,4-DB eq	4,57E+04	2,84E+04	5,55E+03	6,8E+03	3,83E+03	1,72E+00	4,68E+03	3,62E-01
Fresh water aquatic ecotox.	kg 1,4-DB eq	3,52E+04	2,49E+04	1,59E+03	3,62E+03	3,38E+03	8,83E-01	1,69E+03	0
Marine aquatic ecotoxicity	kg 1,4-DB eq	1,12E+08	8,46E+07	4,46E+07	9,66E+06	9,81E+06	1,35E+03	3,33E+06	0
Terrestrial ecotoxicity	kg 1,4-DB eq	3,78E+02	2,4E+02	2,35E+01	3,15E+01	1,69E+01	2,1E-02	6,56E+01	0
Photochemical oxidation	kg C2H4 eq	4,26E+01	2,78E+01	3,06E+00	2,2E+00	7,56E+00	0,000329	2,01E+00	5,93E-03
Acidification	kg SO2 eq	1,12E+03	8,44E+02	8,44E+01	4,28E+01	9,53E+01	8,47E-03	5,79E+01	1,49E-01
Eutrophication	kg PO4--- eq	3,41E+02	1,96E+02	2,03E+01	1,64E+01	2,06E+01	2,69E-02	8,72E+01	3,87E-02

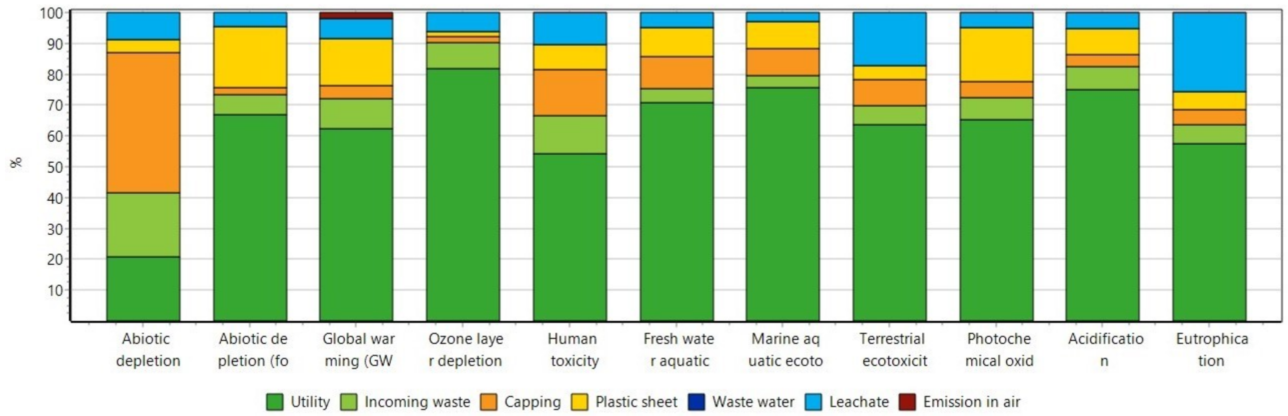


Figure 4.3: Impact assessment results for system 3 (a personal rework of SimaPro data)

Results for system 3 show that utility group is the main contribution to the environmental impact for all the categories with the exception of abiotic depletion for which the main contribution is capping. Leachate group contribution is low compared to the others remaining groups except for eutrophication and terrestrial ecotoxicity where its relevance increase. Plastic sheet contribution is considerable in the description of abiotic depletion fossil fuels, global warming potential and photochemical oxidation while capping is important for abiotic depletion, as previously underlined, and human toxicity. Incoming waste group is considerable in all impact categories while emission in air contribution is relevant only in the description of global warming potential event if it is present for human toxicity, photochemical oxidation, acidification and eutrophication. The contribution of waste water group is negligible for all impact categories.

All the categories are singularly described thanks to values reported in Table(4.3) and Table(A.11).

- Abiotic depletion: 0,239206 kg Sb eq.

The main contribution to abiotic depletion category is capping procedure with 0,109 kg Sb eq. (45%) linked to clay usage in impermeabilization and to its production process. Furthermore, incoming waste (with 0,0497 kg Sb eq.) and utility (with 0,0493 kg Sb eq.) impact respectively for 20,77% and 20,63%. Leachate contribution (8,66% with 0,0207 kg Sb eq.) is due only to its transportation to the final facility. Plastic sheet contributes only for 4,29% (with 0,01 kg Sb eq.) while waste waster impact is negligible.

- Abiotic depletion (fossil fuels): 3810064 MJ

Utility represents the main contribution to abiotic depletion (fossil fuels) with 66,88% (with 2548367 MJ) due to diesel usage inside the plant and fossil fuels used for electricity production. Plastic sheet group impacts for 19,6% (with 748369 MJ) caused by its HDPE plastic material characteristic.

Transportation influence is 6,64% of the total (with 253011 MJ) considering the incoming waste and 4,63% (with 176459 MJ) considering leachate transportation. Capping shows a percentage of 2,2% (with 83847 MJ) while the contribution of waste water is negligible.

- Global warming potential (GWP): 170934,4 kg CO_2 eq.

The main contribution is given by utility group with 62,44% (6958 kg CO_2 eq.) followed by the use of plastic sheet with 15% (with 11132 kg CO_2 eq.). Incoming waste contribution is the 9,7% of the total (with 16573 kg CO_2 eq.), the one of leachate is of 6,5% (with 11132 kg CO_2 eq.) linked only to its transportation and the one of capping is 4% (with 6958 kg CO_2 eq.). Emission in air contribution is relevant for this category and equal to 2% (with 3415 kg CO_2 eq.).

- Ozone layer depletion: 0,03589 kg CFC-11 eq.

The main contribution to ozone layer depletion category is utility (81,7% with 0,0293 kg CFC-11 eq.) linked to diesel production which is then used inside the plant or in the electricity production. Incoming waste (0,00308 kg CFC-11 eq.) and leachate (0,002158 kg CFC-11 eq.) show a contribution of 8,6% and 6% associated to the transportation impact. Capping and plastic sheet contribute only for 0,00072 kg CFC-11 eq. and 0,00059 kg CFC-11 eq. while waste water contribution is negligible.

- Human toxicity: 45720,5 kg 1,4-DB eq.

Human toxicity main contribution is utility (54% with 24843 kg 1,4-DB eq.) and in particular electricity production. Incoming waste (with 5550 kg 1,4-DB eq.), capping (with 6802 kg 1,4-DB eq.) and leachate (with 4686 kg 1,4-DB eq.) contribution stays between 10% and 15%. Plastic sheet impacts for 8,4% (with 3835 kg 1,4-DB eq.). Emission in air and waste water contribution are negligible. The substances that main contributes to this category are selenium (17%) and chromium VI (17%).

- Fresh water aquatic ecotoxicity: 35245 1,4-DB eq.

The main impact to this category is associated to utility group (70,8% with 24948 1,4-DB eq.) associated to the hard coal process that interests the electricity production. Capping (3619 1,4-DB eq.) and plastic sheet (3384 1,4-DB eq.) contribution is smaller and respectively equal to 10% and 9% while the one of leachate (1696 1,4-DB eq.) and incoming waste (1596 1,4-DB eq.) is similarly around 4%.

- Marine ecotoxicity: 111941776 1,4-DB eq.

Utility is the main contribution to marine ecotoxicity with 75,6% (84659463 1,4-DB eq.) while all the other categories show an impact percentage less than 10%. Particularly the percentage is of 8,77% for plastic sheet (9817470 1,4-DB eq.), 8,63% for capping (9664100 1,4-DB eq.), 3,98% for incoming waste (4465785 1,4-DB eq.) and leachate with 2,97% (3333601 1,4-DB eq.).

- Terrestrial ecotoxicity: 378,5 1,4-DB eq.

Utility is the main contribution to terrestrial ecotoxicity category with 63,6% (with 240,8 1,4-DB eq.) followed by leachate with 17,3% (with 65,6 1,4-DB eq.). Particularly the contribution of leachate to this category is linked to the emission of mercury in water (13% with 49,21 1,4-DB eq.). Considering the other categories the contribution is much lower respect to the utility one and equal to 31,5 1,4-DB eq. for capping (8,3%), to 23,5 1,4-DB eq. for incoming waste (6,2%) and to 16,9 1,4-DB eq. for plastic sheet (4,5%).

- Photochemical oxidation: 42,68 kg C_2H_4 eq.

The main contribution to photochemical oxidation is associated to the use and production of diesel and electricity (for a total of 65,2% with 42,7 kg C_2H_4 eq.) followed by the use of plastic HDPE sheet (for 17,7% with 7,57 kg C_2H_4 eq.). Furthermore incoming waste contributes for 7,2% (with 27,8 kg C_2H_4 eq.), capping for 5,1% (with 2,2 kg C_2H_4 eq.) and leachate for 4,7% (with 2 kg C_2H_4 eq.). the contribution of air emission is of 0,014% while the contribution of waste water is negligible.

- Acidification: 1125,268 kg SO_2 eq.

Acidification category biggest contribution is utility with 844,6 kg SO_2 eq. due to the sulfur dioxide and ammonia emissions linked to electricity and diesel production and consumption. The others contribution to acidification are plastic sheet (8,5% with 95,3 kg SO_2 eq.), incoming waste (7,5% with 84,5 kg SO_2 eq.), leachate (5,14% with 57,9 kg SO_2 eq.) and capping (3,8% with 42,8 kg SO_2 eq.). Emission in air contribution is small and equal to 0,149 kg SO_2 eq. together with the waste water one (0.00848 kg SO_2 eq.).

- Eutrophication: 219,347 kg PO_4 eq.

The main contribution to eutrophication is given by utility group which represents the 57,6% of the impact with 341,4 kg PO_4 eq. followed by leachate with 25,5% (with 87,3 kg PO_4 eq.). The substances correlated to this impact are phosphate which is mainly linked to electricity utility and chemical oxygen demand associated to leachate emission. The other categories contributions are 20,66 kg PO_4 eq. for plastic sheet, 20,3 kg PO_4 eq. for incoming waste and 16,4 kg PO_4 eq. for capping. Emission in air contribution is equal to 0,0387 kg PO_4 eq. .

4.1.5 Comparison between system 1, system 2 and system 3

Looking at the previous results and considering the impact characterisation of the three systems, is possible to make a comparison between them and analyse their environmental performances.

In order to do this, it is necessary to refer results on functional unit, which is equal to tonnes of municipal solid waste treated inside every plant. So impact results for every system are divided by tonnes of MSW treated inside them in 2018 and then finally compared which are respectively 2036,85 tonnes for system 1, 2378,42 tonnes for system 2 and 4398,95 tonnes for system 3. Results are reported in Table(4.5).

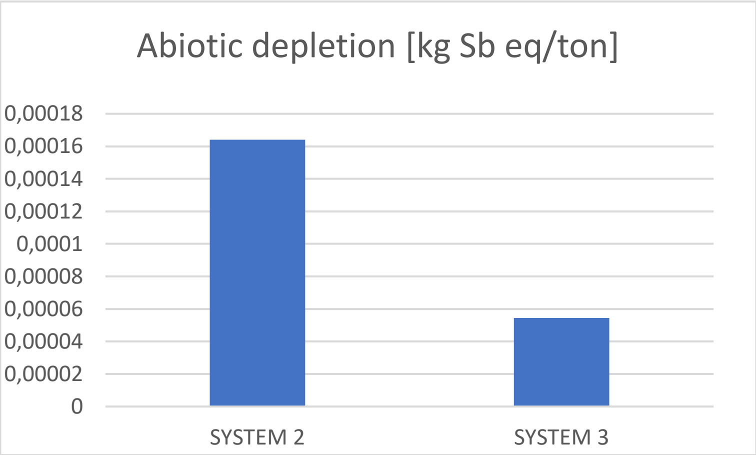
Table 4.5: Comparison between system 1, system2 and system 3 results considering impact per treated tonne inside the plants in 2018

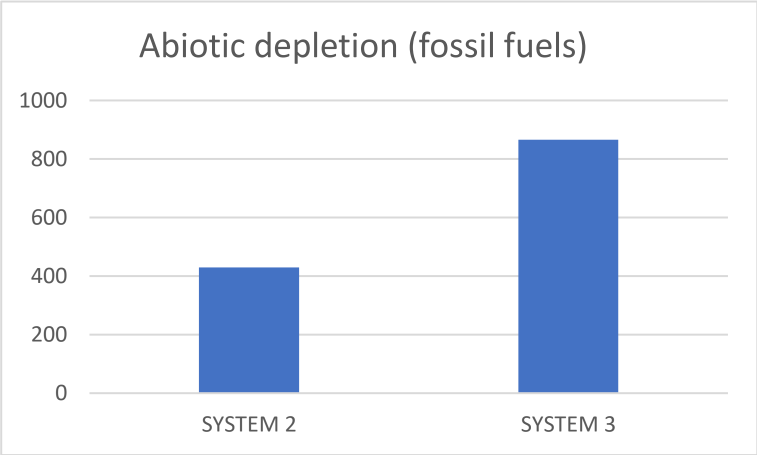
Impact category	**Unit of measure**	**SYSTEM 1**	**SYSTEM 2**	**SYSTEM 3**
Abiotic depletion	kg Sb eq/ton	4,39E-04	1,64E-04	5,44E-05
Abiotic depletion (fossil fuels)	MJ/ton	3,05E+03	4,29E+02	8,66E+02
Global warming (GWP100a)	kg CO2 eq/ton	2,22E+03	2,01E+01	3,88E+01
Ozone layer depletion (ODP)	kg CFC-11 eq/ton	3,74E-05	3,47E-06	8,16E-06
Human toxicity	kg 1,4-DB eq/ton	4,71E+03	2,37E+01	1,04E+01
Fresh water aquatic ecotox.	kg 1,4-DB eq/ton	3,17E+04	1,49E+01	8,01E+00
Marine aquatic ecotoxicity	kg 1,4-DB eq/ton	1,53E+08	1,4E+04	2,54E+04
Terrestrial ecotoxicity	kg 1,4-DB eq/ton	4,59E+00	4,16E-02	8,6E-02
Photochemical oxidation	kg C2H4 eq/ton	4,07E-01	4,67E-03	9,7E-03
Acidification	kg SO2 eq/ton	1,14E+00	1,3E-01	2,55E-01
Eutrophication	kg PO4--- eq/ton	7,56E+00	9,22E-02	7,76E-02

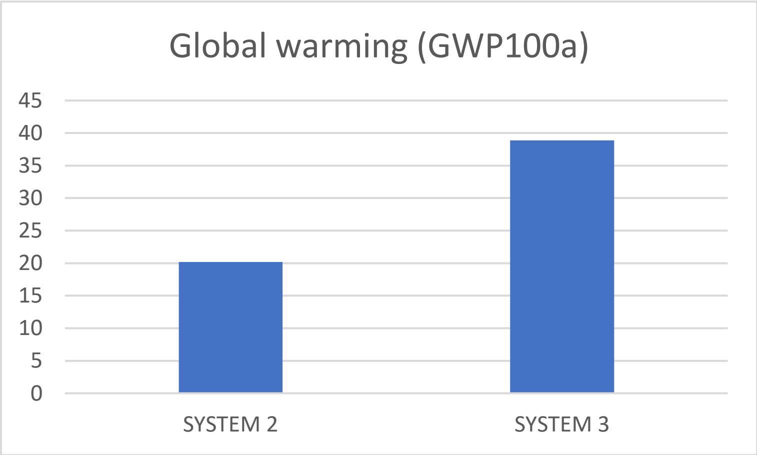
This comparison underlines that system 1 is the most impactful one considering all categories and showing results greater than several orders of magnitude for all of them. The reason of this behaviour is linked to two main contributions present in system 1: waste transportation and final disposal in landfill or incineration. Indeed, as previously underlined for all system, emission associated to transportation are remarkable in the impact description for all systems and linked both to the quantity of waste treated and distance of the path. For system 1 particularly it is considered both transportation for waste which is entering the plant and for waste which is outing the plant, differently from system

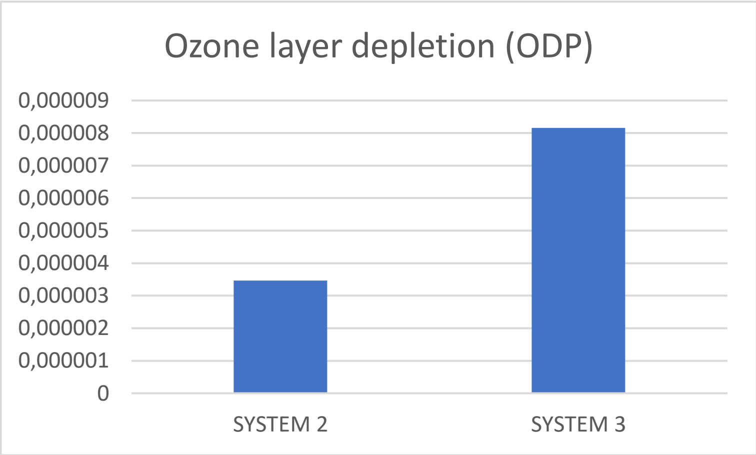
2 and 3 where only entering transportation is considered due to the fact that they refer to landfill. As previously underlined the impact category that are main characterised by this contribution are abiotic depletion, abiotic depletion fossil fuels, ozone layer depletion and acidification. For the other categories the main contribution is linked to the final treatment facility of waste and particularly they are associated to incineration for human toxicity, fresh water aquatic ecotoxicity and marine aquatic ecotoxicity. While global warming potential, terrestrial ecotoxicity and photochemical oxidation are mainly related to landfill emissions and consequently they seem to be more impactful considering the categories mentioned above.

It is interesting now to consider only the results for system 2 and system 3 which refer to landfill facilities to evaluate the most impactful one. Results are reported considering every impact category in the following figure.









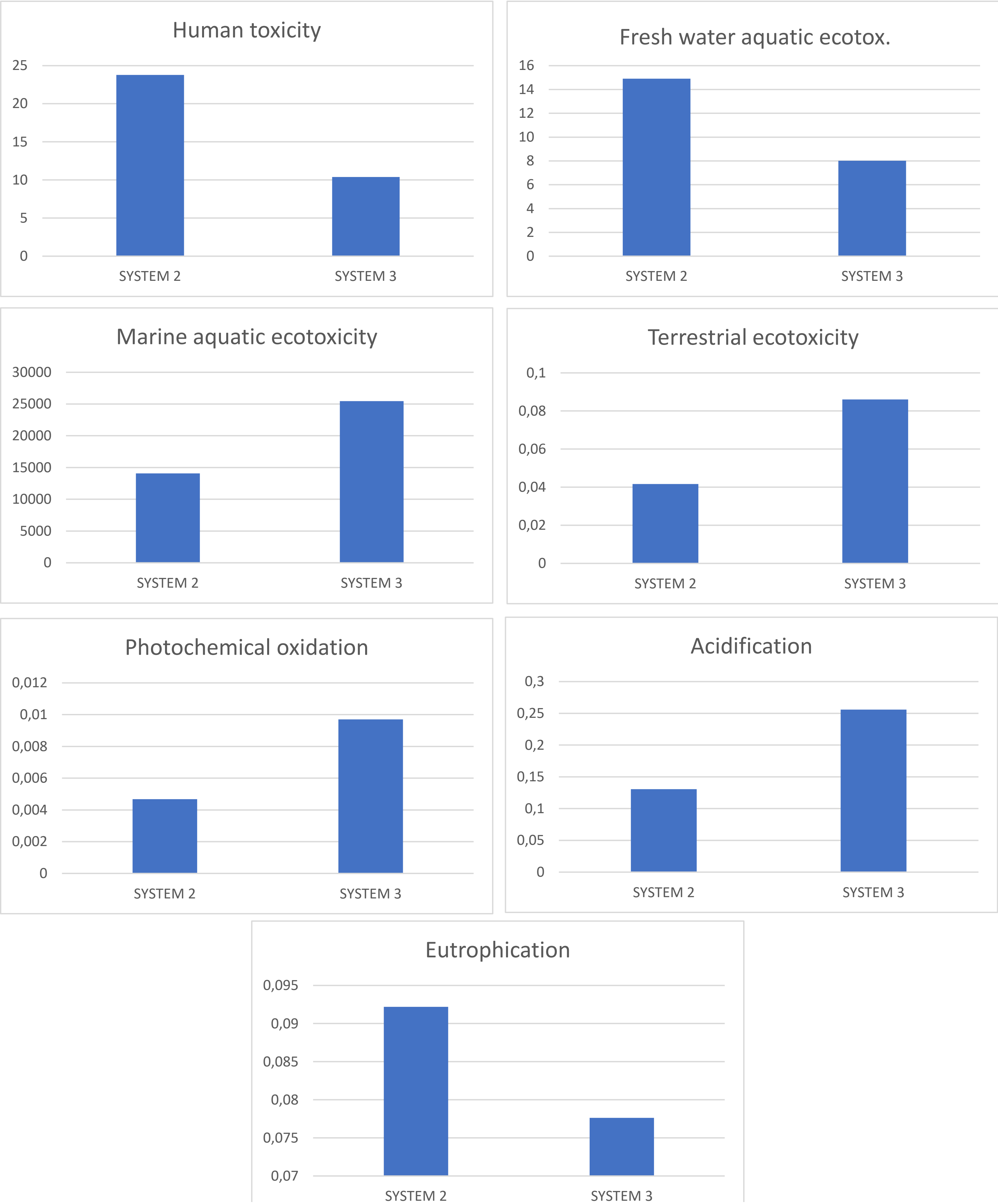


Figure 4.4: Comparison between system 2 and system 3. Impact is expressed for tonnes of waste treated. (A personal rework).

Abiotic depletion system 2 impact is greater than system 3 one (Figure(4.4)) and the reason of that can be found in the capping procedure for this system which is characterised by the use of zinc coated steel while for system 3 it is linked only to clay production with a smaller impact. Considering, firstly, utility group, which is the main contribution in the impact assessment of system 3, it is interesting to evaluate the electricity and diesel consumption per tonne of the two systems in order to better understand the major impact that system 3 shows for all the other categories. The electric energy used for tonne of treated waste in system 3 is equal to 49,325 kWh/ton which is more than three times bigger than the one of system 2 (13,48 kWh/ton). A similar result can be found for the diesel consumption which is equal to 2,6 l/ton for system 2 and 7,4 l/ton for system 3 which is again bigger than the previous one. The utilities consumption is the reason why the utility group is predominant in the description of impact assessment for system 3 respect to system 2 and also why the impact of abiotic depletion fossil fuels, global warming potential and ozone layer depletion. The same consideration can be done also for photochemical oxidation, acidification and marine aquatic ecotoxicity mainly associated to emission linked to fossil fuels usage in all different process. For human toxicity, fresh water aquatic toxicity and eutrophication the impact associated to system 2 is bigger than the one of system 3. The reason of that can be found in the impact associated to leachate which is a relevant contribution to those categories. Particularly leachate emission for system 2 are more impactful than the one of system 3. It is interesting to underline that biogas, even if it is not treated by flare combustion, it is not relevant in the determination of the most impactful category of system 2. On the contrary, as previously underlined, emission in air are relevant only in the description of global warming potential underlining the fact than CO_2 produced by the waste degradation and the one produced by methane combustion are bigger than the one of system 2 per tonne of treated MSW.

4.2 Sensitivity analysis

Sensitivity analysis is a tool in the result interpretation and in the knowledge of the studied system. Sensitivity is mentioned also in ISO 14040 (2006) and it permits to evaluate how a parameter influence the results. Particularly it is a significant tool to investigate the robustness of the study and the assumptions, together with the identification of the most important set of parameters. The study is performed considering the change of one studied parameter at a time underlining its impact on the characterisation results.

Sensitivity analysis has been performed in order to evaluate the impact of assumption and parameters as the choice of lorry, used for waste transportation, with EURO 3 or EURO 4 characteristic in system 1, system 2 and system 3. The change in lorries is considered only for waste transportation since it is

the only one for which the different systems can intervene. Furthermore, considering system 2 a sensitivity study on leachate and biogas composition assumption are performed, followed by a study on zinc coat and welding for the production of cages used in capping. Finally considering system 3, leachate composition assumption is studied and the impact of the introduction of a group category in the model.
Results will be reported with the same precision given by SimaPro to appreciate the changes in the results.

4.2.1 Hypothesis on waste transportation in system 1

In all the three considered systems the transportation of waste is assumed to be performed with lorries, with a capacity in between 16 and 32 metric tonnes and characterised by EURO 3 standard emission efficiency. As previously defined, transportation of waste is an important contribution in the evaluation of the environmental performances of the three considered systems and therefore it is interesting to investigate how the use of lorries with and higher efficiency in emissions, as the EURO 4, with the same capacity, will effect on the study results. The impact assessments result for the sensibility analysis performed on system 1 are reported in Table(4.6).

Table 4.6: Impact results for system 1 implemented following the standard EURO 3 hypothesis and the new implemented EURO 4 hypothesis

Impact category	**Unit of measure**	**EURO 3**	**EURO 4**
Abiotic depletion	kg Sb eq	0,894897111	0,894709816
Abiotic depletion (fossil fuels)	MJ	6215611,103	6184070,164
Global warming (GWP100a)	kg CO2 eq	4528726,341	4526903,23
Ozone layer depletion (ODP)	kg CFC-11 eq	0,07621094	0,075805501
Human toxicity	kg 1,4-DB eq	9596927,442	9595986,069
Fresh water aquatic ecotox.	kg 1,4-DB eq	64621613	64621049,21
Marine aquatic ecotoxicity	kg 1,4-DB eq	3,12573E+11	3,12573E+11
Terrestrial ecotoxicity	kg 1,4-DB eq	9363,385157	9343,410763
Photochemical oxidation	kg C2H4 eq	830,9740425	825,7852757
Acidification	kg SO2 eq	2341,575249	2048,251163
Eutrophication	kg PO4--- eq	15412,96155	15337,71995

As showed previously in the results of system 1, transportation of the incoming and outcoming waste is one of the main contributions to the environmental impact in system 1. Therefore, changing the hypothesis on the efficiency on emission of lorries will affect the impact results. Particularly transportation impacts on all categories, with a different weight depending on the category characteristic. Relative difference in impacts are showed in Figure(4.5). Terrestrial ecotoxicity, marine aquatic ecotoxicity, fresh water aquatic ecotoxicity, human toxicity, global warming potential and abiotic depletion reports a really small change (in the other of 10^{-4}) while for abiotic depletion fossil fuels, ozone layer depletion, terrestrial ecotoxicity, photochemical oxidation and eutrophication the improvement is more appreciable (in the order of 10^{-3}). However, the most important gain in impact assessment can be found in acidification category. Acidification describes the contribution connected to the formation of acid rain and so a significant reduction in SOx and NO_x leads to a percentage reduction of impact for this category of 12,5%, as also reported in Figure(4.5).

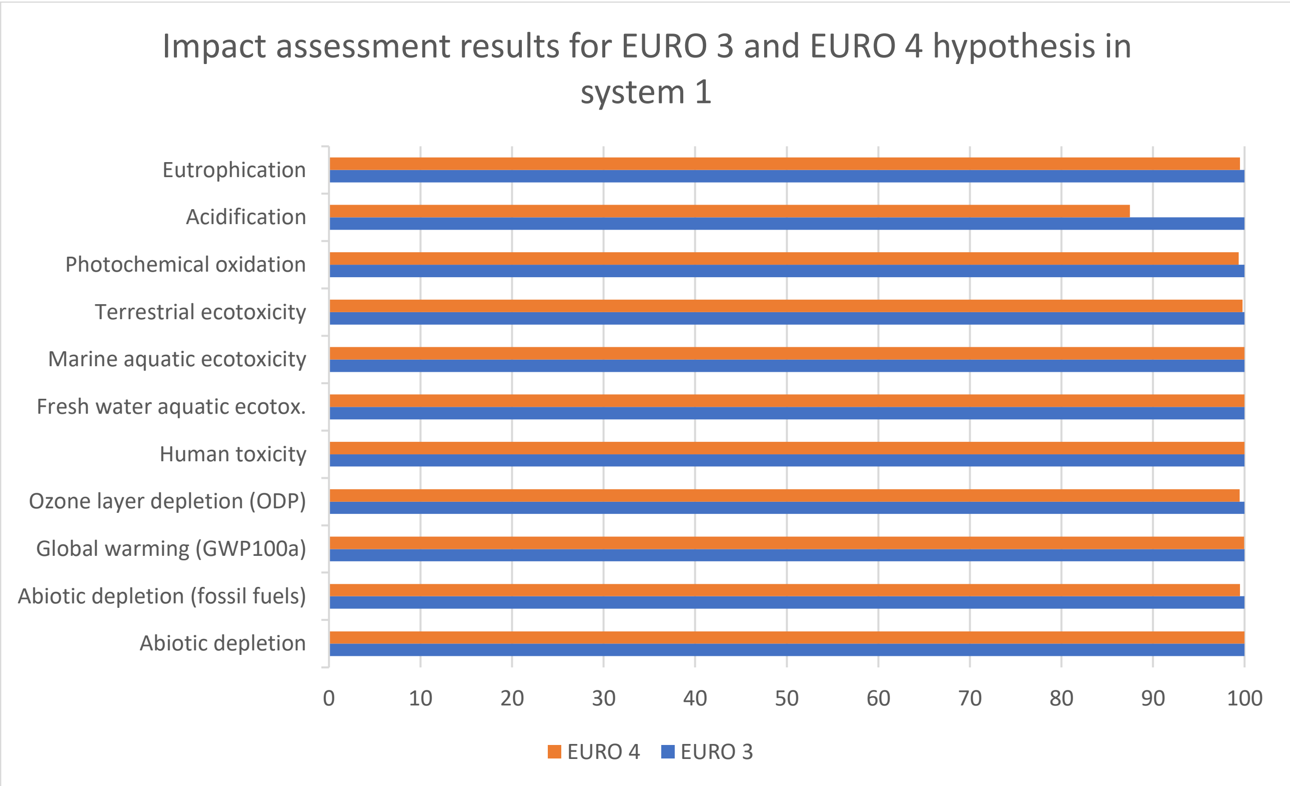


Figure 4.5: Impact assessment results for EURO 3 and EURO 4 hypothesis in system 1 reported as percentage respect to the base case. (A personal rework).

4.2.2 Hypothesis on waste transportation in system 2

The same hypothesis considered in system 1 is also be included in system 2. Differently from system 1 transportation affects only the transportation of the incoming MSW, so from the waste collection centre to landfill. Consequently, in this case only the "incoming waste" group suffers the hypothesis for the usage of EURO 3 lorries. As noted above using EURO 4 lorries for the transportation of incoming waste will have a positive effect on all impact categories and results under this hypothesis are reported in Table(4.7).

Table 4.7: Impact results for system 2 implemented following the standard EURO 3 hypothesis and the EURO 4 hypothesis

Impact category	**Unit of measure**	**EURO 3**	**EURO 4**
Abiotic depletion	kg Sb eq	0,390334	0,390323
Abiotic depletion (fossil fuels)	MJ	1021920	1020113
Global warming (GWP100a)	kg CO2 eq	48018,31	47909,23
Ozone layer depletion (ODP)	kg CFC-11 eq	0,008251	0,008227
Human toxicity	kg 1,4-DB eq	56589,69	56558,77
Fresh water aquatic ecotox.	kg 1,4-DB eq	35474,34	35459,18
Marine aquatic ecotoxicity	kg 1,4-DB eq	33521178	33512564
Terrestrial ecotoxicity	kg 1,4-DB eq	98,99468	98,48784
Photochemical oxidation	kg C2H4 eq	11,11924	10,98111
Acidification	kg SO2 eq	310,4156	301,167
Eutrophication	kg PO4--- eq	219,2961	216,9326

In Figure(4.6) results are expressed as percentage variation respect to the base casa allowing a better understanding on the influence of the hypothesis. Results shows that the impact on categories like abiotic depletion, human toxicity, fresh water aquatic ecotoxicity and marine aquatic ecotoxicity is low (in the other of 10^{-4}) while the one for abiotic depletion fossil fuels, global warming potential, ozone layer depletion and terrestrial ecotoxicity the variation is more appreciable in the order of 10^{-3}. Finally considering photochemical oxidation, acidification and eutrophication they show a tangible variation considering impact.

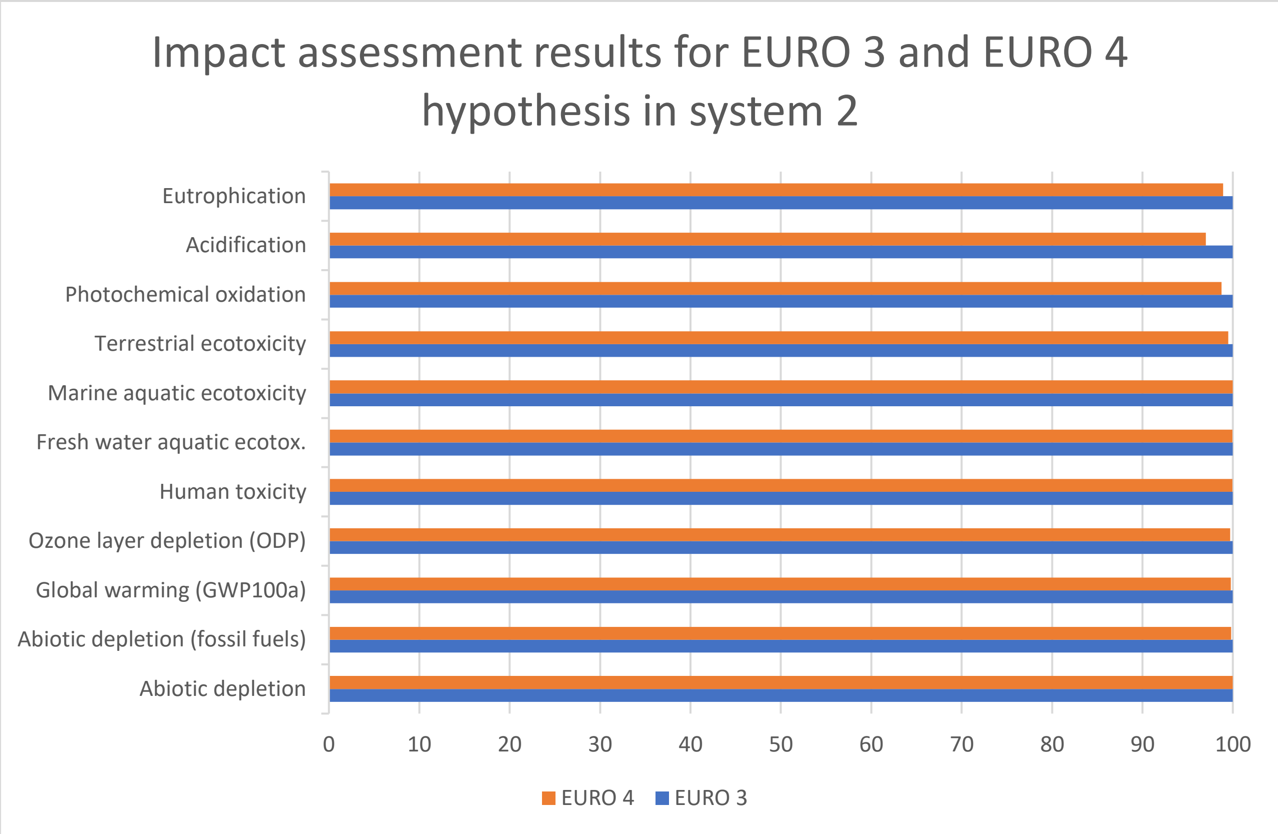


Figure 4.6: Impact assessment results for EURO 3 and EURO 4 hypothesis in system2 reported as percentage respect to the base case. (A personal rework).

Since "incoming waste" is the only group affected by EURO 3 hypothesis the impact associated to this group is studied in Table(4.8) in order to better represent changes between EURO 3 and EURO 4 results. This also permits to evaluate precisely how more efficient lorries in terms of emission leads to an improvement in the results.

Table 4.8: Impact results for "incoming waste" group in system 2 implemented following the standard EURO 3 hypothesis and the new implemented EURO 4 hypothesis

Impact category	Unit of measure	EURO 3	EURO 4	Improvement (%)
Abiotic depletion	kg Sb eq	0,012857	0,012846	0,08
Abiotic depletion (fossil fuels)	MJ	109492,5	107685,7	1,65
Global warming (GWP100a)	kg CO2 eq	6907,634	6798,547	1,58
Ozone layer depletion (ODP)	kg CFC-11 eq	0,001339	0,001316	1,73
Human toxicity	kg 1,4-DB eq	2772,241	2741,328	1,11
Fresh water aquatic ecotox.	kg 1,4-DB eq	625,9255	610,7666	2,42
Marine aquatic ecotoxicity	kg 1,4-DB eq	1868201	1859587	0,46
Terrestrial ecotoxicity	kg 1,4-DB eq	10,31981	9,812976	4,91
Photochemical oxidation	kg C2H4 eq	1,249791	1,11166	11,05
Acidification	kg SO2 eq	35,9316	26,68302	25,74
Eutrophication	kg PO4--- eq	8,589232	6,225749	27,51

The major difference can be found in eutrophication category which shows an improvement in emission and consequently on impact of 27,5% respect to EURO 3 case. Indeed, eutrophication describes is mainly linked to the nitrogen pollution due to emission of NO_x from combustion of fossil fuels. Together with eutrophication, acidification and photochemical oxidation, establish the highest value of betterment with 25,7 % and 11% respectively using EURO 4 lorries. Abiotic depletion and marine aquatic ecotoxicity are the two categories for which the impact is smaller.

4.2.3 Hypothesis on waste transportation in system 3

Following the previous considerations, also for system 3 a sensitivity analysis has been done in order to understand the impact of EURO 3 lorries characteristic for transportation respect to EURO 4 ones. Like in system 2 the only group that suffers from this hypothesis is incoming waste which is a relevant contribution for all categories keeping consequently to an improve in the impact of system 3. Results can be found in Table(4.12).

Table 4.12: Impact results for system 3 implemented following the standard EURO 3 hypothesis and the EURO 4 hypothesis

Impact category	**Unit of measure**	**EURO 3**	**EURO 4**
Abiotic depletion	kg Sb eq	0,239206	0,239193
Abiotic depletion (fossil fuels)	MJ	3810065	3807884
Global warming (GWP100a)	kg CO2 eq	170934,4	170808,4
Ozone layer depletion (ODP)	kg CFC-11 eq	0,035891	0,035863
Human toxicity	kg 1,4-DB eq	45720,53	45655,44
Fresh water aquatic ecotox.	kg 1,4-DB eq	35245,69	35206,71
Marine aquatic ecotoxicity	kg 1,4-DB eq	1,12E+08	1,12E+08
Terrestrial ecotoxicity	kg 1,4-DB eq	378,4894	377,1081
Photochemical oxidation	kg C2H4 eq	42,67911	42,3203
Acidification	kg SO2 eq	1125,269	1104,985
Eutrophication	kg PO4--- eq	341,4355	336,2326

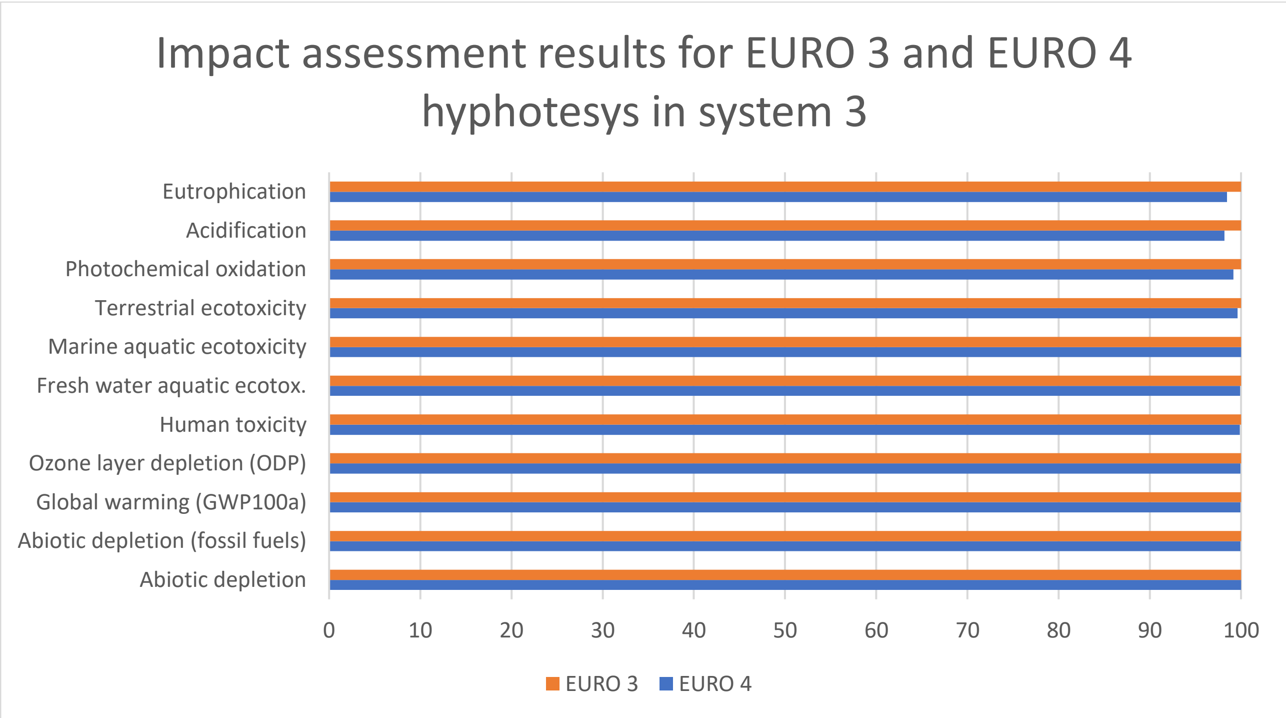


Figure 4.9: Impact assessment results for EURO 3 and EURO 4 hypothesis in system 3 reported as percentage respect to the base case. (A personal rework).

Results shows that the most impactful hypothesis categories are photochemical oxidation, eutrophication and acidification. Looking at Figure(4.9) is possible to underline that for this categories the improvement is around 2% while for all the others it is less than 1%. The results are in line with the previous sensitivity study and they are linked to a smaller characteristic emission of EURO 4 in NO_x and SO_x. The reason why this hypothesis does not effect deeply the results can be found in incoming waste group which is not predominant in the determination of the impact assessment for system 3.

4.2.4 Leachate composition hypothesis for system 2

In the inventory analysis of system 2 leachate is considered as an output of the process and its composition is described by proper analysis executed in year 2018. Howevfor some components the composition is reported as lower of a certain value of confidence. In order to carry out the study following a conservative logic, the upper value has been considered as the concentration value of the component. To investigate how this hypothesis influence the results a sensitive analysis was performed considering that the error made is 50% i.e. that the concentration value of a given component is half respect to that of the base case and then, considering a bigger error, a value of the component equal to 10% than the base case one, so an almost null concentration of the component. Composition is so calculated multiplying the base case value for a corrective parameter ERR equal to 0,5 and 0,1 to describe to two previous assumptions, while ERR equal to 1 describes the base case. As seen in chapter 4.2 leachate influence only human toxicity, fresh water aquatic ecotoxicity, marine aquatic ecotoxicity, terrestrial ecotoxicity and eutrophication categories as reported in Table(4.9).

Table 4.9: Impact results for the categories influenced by leachate group considering values of ERR equal to 0,1 0,5 and 1 to the describe the assumptions on leachate composition in system 2

Impact category	**Unit of measure**	**ERR = 0,1**	**ERR = 0,5**	**ERR = 1**
Human toxicity	kg 1,4-DB eq	17940,43	35117,88	56589,69
Fresh water aquatic ecotox.	kg 1,4-DB eq	13190,83	23094,61	35474,34
Marine aquatic ecotoxicity	kg 1,4-DB eq	33476291	33496241	33521178
Terrestrial ecotoxicity	kg 1,4-DB eq	85,78447	91,65567	98,99468
Eutrophication	kg PO4--- eq	217,9642	218,5561	219,2961

To better understand the changes in the environmental impact, results are described through a percentage expression in Figure(4.7).

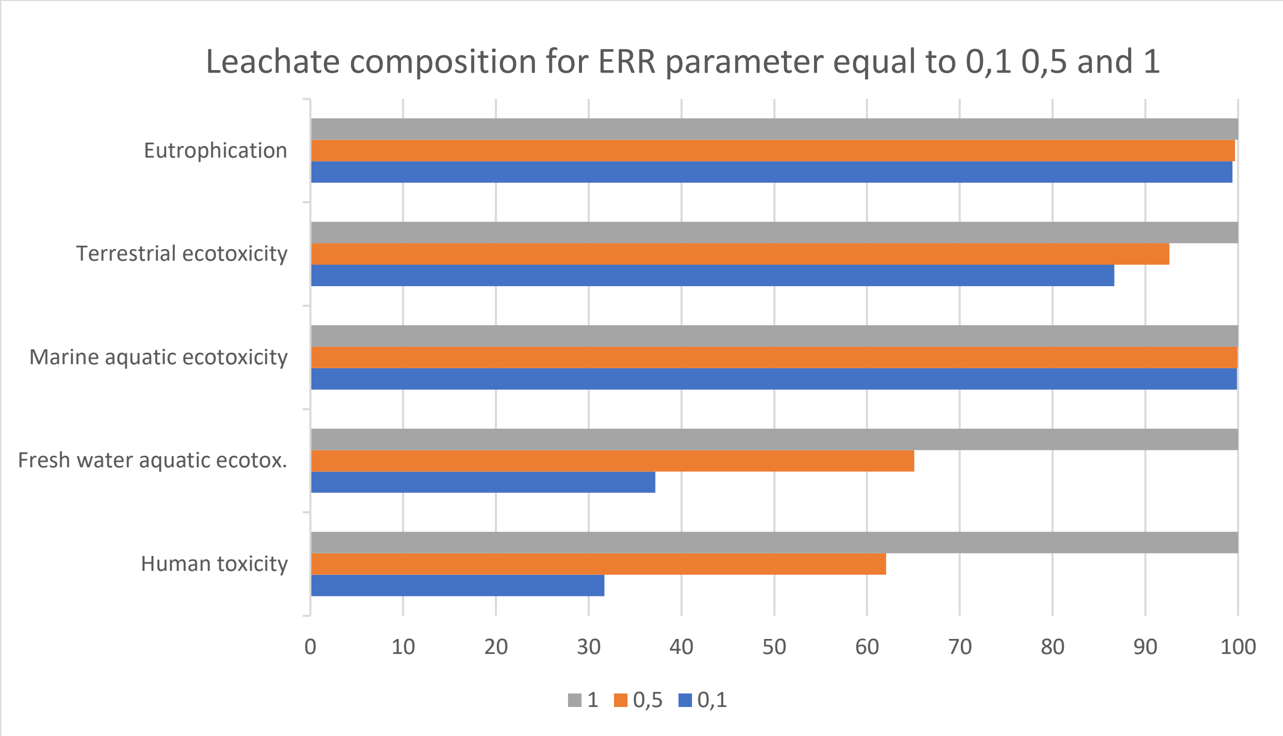


Figure 4.7: Leachate composition for multiplying factor ERR equal to 0,1 0,5 and 1 expressed as percentage respect to the base case (ERR=1). (A personal rework)

Human toxicity is the category for which the assumption on composition shows the highest impact. Indeed, human toxicity major contribution is given to the presence of hexachlorobutadiene which is a component that shows characteristics of toxicity and carcinogenicity. The improvement is of 62% for ERR = 0,5 and of 31,7% for ERR = 0,1 respect to the base case. Also fresh water aquatic ecotoxicity depends on the amount the concentration of hexachlorobutadiene in leachate but also on copper and nickel, two components that do not suffer the studied hypothesis and so the betterment for this category are less than human toxicity one. However the results are for ERR = 0,1 equal to 37,1% and for ERR = 0,5 equal to 65,1% respect to the base case. The components that better describe terrestrial ecotoxicity are hexachlorobutadiene and mercury and concentration of both depend on the previous hypothesis. The diminish in concentration leads to an improvement of the results equal to 86,6% for ERR = 0,1 and 92,5% for ERR = 0,5 respect to the base case. Marine aquatic ecotoxicity and eutrophication show the smaller differences respect to the base hypothesis. This condition is due to the fact that the components that mainly contributes to this category are elements that do not suffer the previous hypothesis and consequently the base case can be considered as a good estimation of the impact results for this category.

4.2.5 Leachate composition hypothesis for system 3

As for system 2 also in system 3 leachate composition has been calculated thought proper analysis where some concentration are referred as lower than a certain value of uncertainty. Even for system

3 these values are considered as the concentration one in order to improve the study in a conservative way. Consequently it is interesting to underline how those values will affect only human toxicity, fresh water aquatic ecotoxicity, marine ecotoxicity, terrestrial ecotoxicity and acidification since other impact associated to leachate is caused by its associated transportation process. Results are reported in Table(4.13) only for the previous mentioned categories.

Table 4.13: Impact results for the categories influenced by leachate group considering values of ERR equal to 0,1 0,5 and 1 to the describe the assumptions on leachate composition in system 3

Impact category	**Unit of measure**	**ERR = 0,5**	**ERR = 1**
Human toxicity	kg 1,4-DB eq	45632,37	45720,53
Fresh water aquatic ecotox.	kg 1,4-DB eq	35173,14	35245,69
Marine aquatic ecotoxicity	kg 1,4-DB eq	1,12E+08	1,12E+08
Terrestrial ecotoxicity	kg 1,4-DB eq	353,9788	378,4894
Eutrophication	kg PO4--- eq	341,4355	341,4355

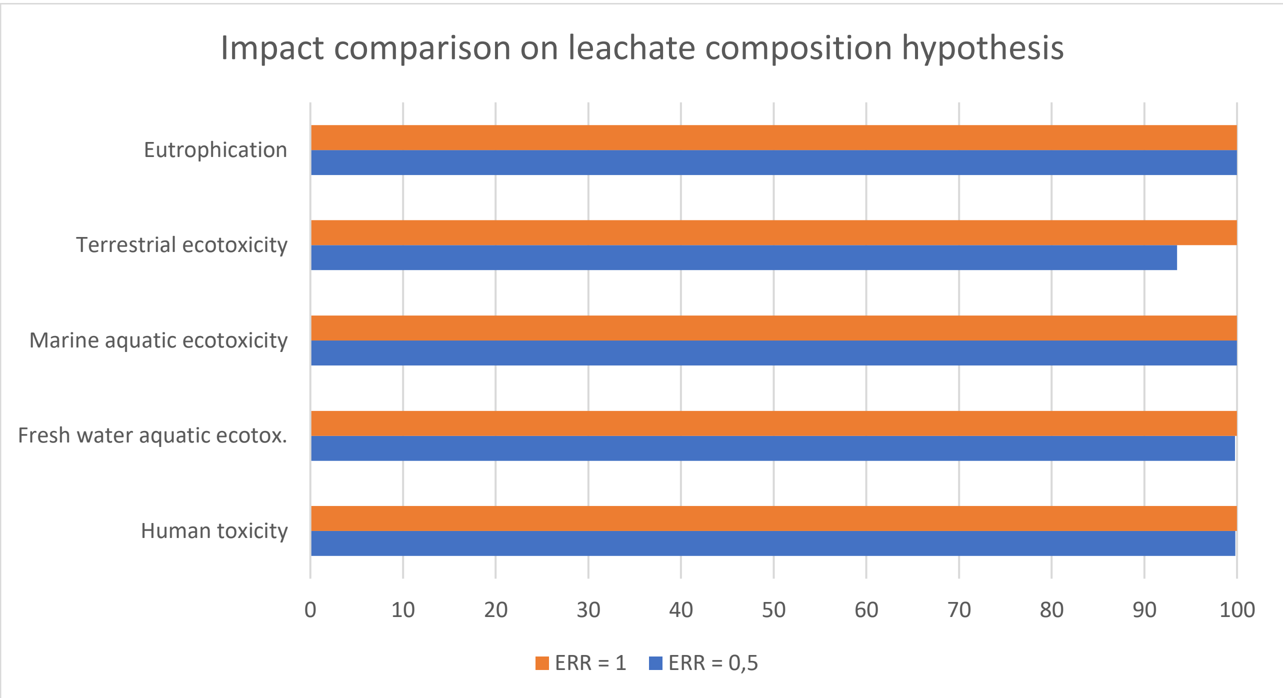


Figure 4.10: Impact assessment for category affected by leachate composition hypothesis in system 3 referred as percentage respect to the base case. (A personal rework).

Results shows that the most affected category by the hypothesis on leachate composition is the terrestrial ecotoxicity one which is 93% of the base case Figure(4.10). This in caused by the concentration of mercury considering water emission which suffer the leachate hypothesis. The impacts of fresh water aquatic ecotoxicity, marine aquatic ecotoxicity and human toxicity are barely noticeable. Eutrophication on the contrary does not show any difference since COD concentration, which is the main contribution to this category linked to leachate, is precisely defined in the inventory.

4.2.6 Biogas composition hypothesis for system 2

As for leachate description, biogas composition is expressed for some components as the limiting value of the lower interval given in the analysis. This leads to the necessity of investigate how this hypothesis reflect the environmental impact results for system 2. As seen in the previously impact assessment description biogas affect only global warming potential, photochemical oxidation and acidification and its contribution is not so determining considering impacts of system 2. This leads to the choice of investigating only an error in the composition description of 50% therefore defining a multiplying parameter of composition value equal to ERR = 0,5. Results for system 2 under this two hypothesis are reported in Table(4.10) considering only the involved categories.

Table 4.10: Biogas under different hypothesis of composition for multiplying parameter ERR equal to 0,5 and the base case for ERR=1

Impact category	**Unit of measure**	**ERR=0,5**	**ERR=1**
Global warming (GWP100a)	kg CO2 eq	48017,33319	48018,31497
Photochemical oxidation	kg C2H4 eq	11,11924124	11,11924124
Acidification	kg SO2 eq	310,415628	310,415628

As results show difference in the impact values are minimal due to the low contribute of biogas in the final environmental assessment of system 2 so underling how an error in the concentration estimation does not influence the final result.

4.2.7 Cages construction hypothesis for system 2

For system 2 cages are used in order to ensure soil stability in the capping procedure. Cages are made of properly crushed rocks surrounded by steel wire grid. However the extension of the grid is not known and it is dimension has been found through geometrical approximations. Particularly they refer to tonnes of wire drawing used and to square meters of material processed through zinc coating. In order to examinate the correctness of this hypothesis and how much it affects the final result it is assumed to have overestimated the two process by 50% and to have underestimated it by 50%. To describe properly this conditions a multiplying factor has been used which is equal to 0,5 to describe the overestimation and to 1,5 to describe the underestimation both of them respect to base case for which the multiplying parameter is equal to 1. Results are reported in Table(4.11) for the different parameters and in Figure(4.8) as percentage respect to the base case.

Table 4.11: Impact results for system 2 studied under different multiplying parameter in the description of zinc-coiling and welding for cages used in capping

Impact category	Unit of measure	P = 0,5	P = 1	P = 1,5
Abiotic depletion	kg Sb eq	0,225835	0,390334	0,554833
Abiotic depletion (fossil fuels)	MJ	1019862	1021920	1023977
Global warming (GWP100a)	kg CO2 eq	47840,92	48018,31	48195,71
Ozone layer depletion (ODP)	kg CFC-11 eq	0,008237	0,008251	0,008264
Human toxicity	kg 1,4-DB eq	56397,05	56589,69	56782,32
Fresh water aquatic ecotox.	kg 1,4-DB eq	35344,79	35474,34	35603,89
Marine aquatic ecotoxicity	kg 1,4-DB eq	33104148	33521178	33938207
Terrestrial ecotoxicity	kg 1,4-DB eq	95,72789	98,99468	102,2615
Photochemical oxidation	kg C2H4 eq	11,06566	11,11924	11,17282
Acidification	kg SO2 eq	302,8427	310,4156	317,9886
Eutrophication	kg PO4--- eq	217,3811	219,2961	221,2111

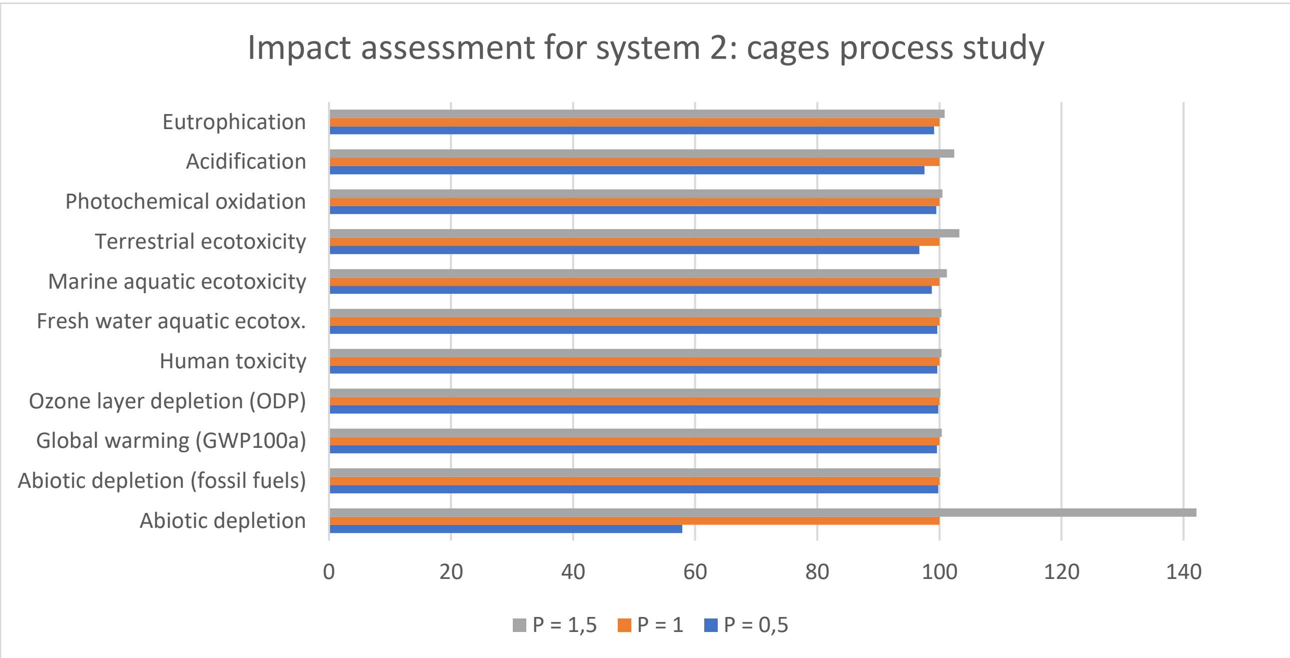


Figure 4.8: Impact assessment for system 2 under different multiplying parameters in the description of cages used in capping in system 2: a percentage study taking as reference the base case P=1. (A personal rework).

Results shows that abiotic depletion is the category that highly suffers from differences in the cages process description. Indeed as previously underlined in the impact results of system 2, the use of zinc for the zinc coating process represent the main contribution in abiotic depletion category. Consequently an error in its description of 50% leads to a diminishing in the impact of this category of 50%. This behaviour can be found both considering overestimation and underestimation of zinc-coating. The other categories differences in impact are smaller than the previous one and only terrestrial ecotoxicity and acidification shows a percentage difference of more than one percentage point.

4.2.8 Boundaries hypothesis in system 3: plant works

At the end of 2018 inside the landfill in system 3 construction works were carried out that it was decided not to consider in the impact analysis by defining them outside the boundaries of the study. These works refer to the excavation and waterproofing of the bottom of it and its walls, in order to isolate the waste from the environment and to subsequently be able to collect the leachate and the biogas produced. The contributions to the aforementioned works are shown in Table(4.14).

Table 4.14: Process contained in works group in system 3

Works	Amount (unit of measure)
Clay	2287,1492 ton
Bentonite	236,77196 ton
Bentonite geomembrane	22160 kg
Gravel	275,31623 kg
Excavation	3426,1569 m^3
Geotexile	146,83532 kg
HDPE pipes	82,88 kg

First of all the excavation is described by considering the volume associated to the quantity of waste treated in 2018 as reported in Table(4.15).

Table 4.15: Soil excavation in works in system 3

Excavation	Amount (unit of measure)
Exacavation, hydraulic digger {GLO}| market for | Cut-off, U	752 m^3

Clay is used to perform impermeabilization of the excavation, both at the bottom and for the side walls. The excavation area was associated with that relating to the waste treated in 2018 and the thickness of the clay waterproofing of the bottom is 1,3 m. Clay is also used for side wall covering a total height of 115 m with a total thickness of 0,8 m. Results are reported in Table(4.16) where clay is expressed in tonnes with a density of 2,1 ton/m^3.

Table 4.16: Clay used in works in system 3

Clay	Amount (unit of measure)
Clay {RoW}|market for clay| Cut-off, U	2287,1492 ton

Bentonite material is in the bottom waterproofing inserting it inside two layers of polypropylene geotextile. As for the previous case it refers to the area associate to waste treated in 2018 with a thickness of 0,3 m and a density of 2,15 ton/m^3 (Table(4.17)).

Table 4.17: Bentonite used in works in system 3

Bentonite	Amount (unit of measure)
Bentonite {RoW}|market for clay| Cut-off, U	236,77196 ton

The two geotextile that covers the bentonite are described in Table(4.18) which comprehends the production processes, similarly to what is described in 3, considering that the extrusion process has an efficiency of 0,96 kg/kg.

Table 4.18: Geotextile used in works in system 3

Geotextile	Amount (unit of measure)
Polypropylene, granulate {GLO}| market for| Cut-off, U	220,25 kg
Extrusion plastic film {RoW}| extrusion, plastic film| Cut-off, U	227,3 kg
Weaving, bast fibre {RoW}| processing| Cut-off, U	220,25 kg

For the impermeabilization of the side walls bentonite geomembrane are used covering a total height of 115 m. The geomembrane can be described by the bentonite and geotextile processes previously reported since it is constituted of a bentonite layer with thickness of 1 mm and interposed to two geotextile layer of thickness 1 mm each. (Table(4.19)).

Table 4.19: Bentonite geomembrane used in works in system 3

Bentonite geomembrane	Amount (unit of measure)
Bentonite geomembrane	22160 kg

Over the waterproofing layer gravel is used in order to ensure the drainage of gas and water to permit their collection through proper pipes. Gravel has a density of 1,5 ton/m^3 and with a thickness of 0,5 m. information are reported in Table(4.20).

Table 4.20: Gravel used in works in system 3

Gravel	Amount (unit of measure)
Gravel, crushed {RoW}	150 ton

Finally HDPE pipes are used to collect leachate and biogas. To evaluate the weight of HDPE used a density of 1,48 kg/m is considered for 56 m of pipe. Results are in Table(4.21).

Table 4.21: HDPE materials in works in system 3

HDPE materials	Amount (unit of measure)
Polyethylene pipe, DN 200, SDR 41{RoW}| production| Cut-off U	56 m

Adding this process as a group, it is possible to describe the impact results of system 3 under a different study burden condition. Results are reported in Table(4.22), in Figure(4.11) and Figure(4.12).

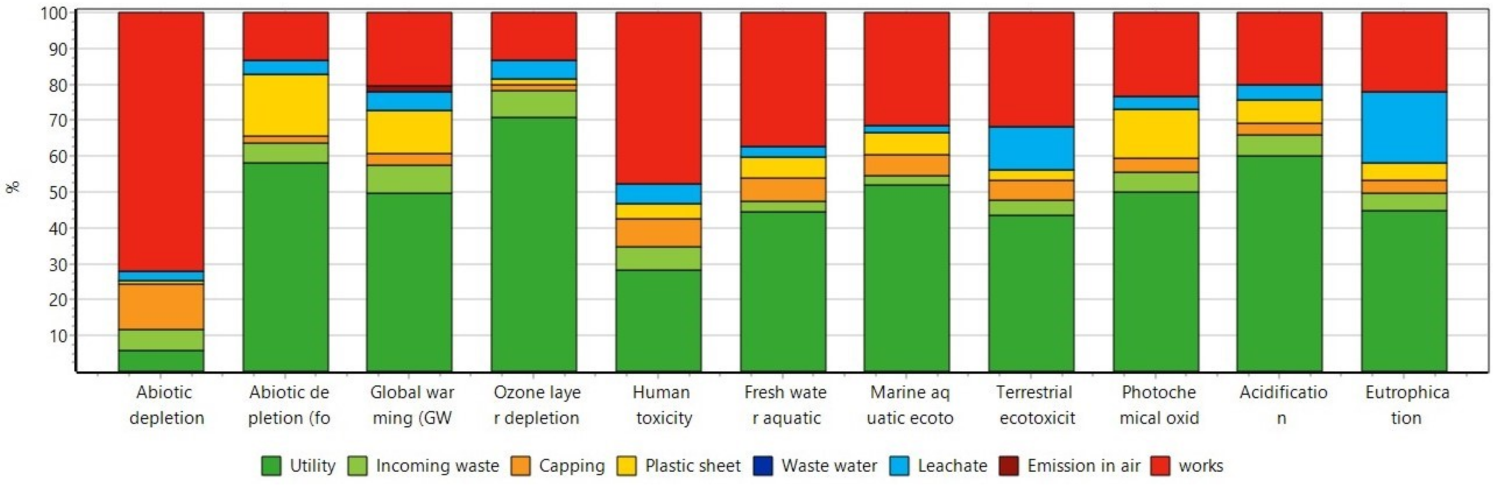


Figure 4.11: Impact assessment for system 3 considering works group. (A personal rework of SimaPro data)

Table 4.22: Impact assessment for system 3 and system 3 with works group

Impact category	**Unit of measure**	**System 3 considering works group**	**System 3**
Abiotic depletion	kg Sb eq	0,858225	0,239206
Abiotic depletion (fossil fuels)	MJ	4394838	3810065
Global warming (GWP100a)	kg CO2 eq	214697,9	170934,4
Ozone layer depletion (ODP)	kg CFC-11 eq	0,041458	0,035891
Human toxicity	kg 1,4-DB eq	87466,12	45720,53
Fresh water aquatic ecotox.	kg 1,4-DB eq	56131,99	35245,69
Marine aquatic ecotoxicity	kg 1,4-DB eq	1,63E+08	1,12E+08
Terrestrial ecotoxicity	kg 1,4-DB eq	555,6962	378,4894
Photochemical oxidation	kg C2H4 eq	55,70172	42,67911
Acidification	kg SO2 eq	1408,323	1125,269
Eutrophication	kg PO4--- eq	437,3404	341,4355

Considering the impact results, it is possible to underline that works group has a great impact in the environmental assessment of system 3. This impact is mainly related to abiotic depletion category for which an increase of 358% more than the base case. The reason of that can be found in the clay usage which production has a noticeable impact in the description of abiotic depletion of system 3 related to capping. Figure(4.12) show results expressed in percentage terms considering the base case as the unitary one (100%).

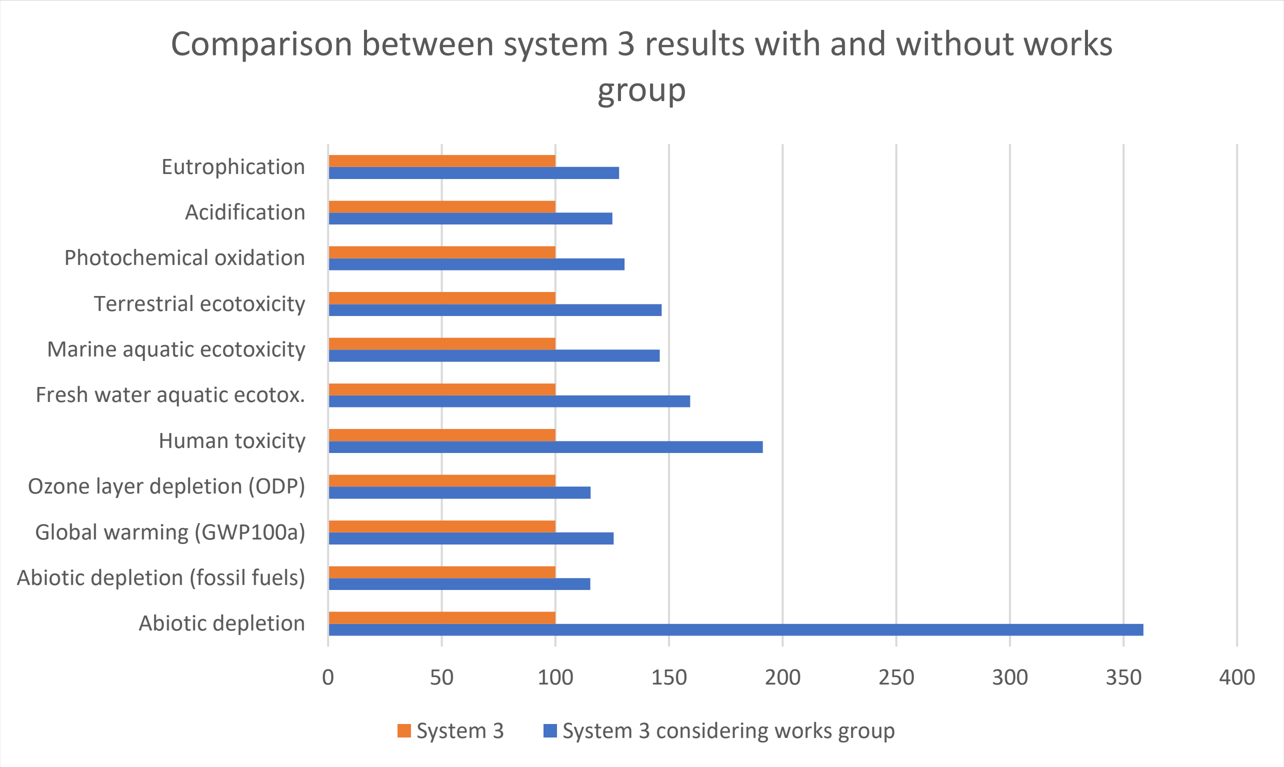


Figure 4.12: Impact results for system 3 and system 3 with work group contribution considering. (A personal rework).

Considering the other impact categories human toxicity shows a higher impact of almost 191% followed by fresh water aquatic ecotoxicity and terrestrial and marine ecotoxicity. All the other impact categories show an increase respect to the base case of lower than 130% particularly considering abiotic depletion fossil fuels, ozone layer depletion and acidification. Indeed, this category are mainly linked to the usage of fossil fuels which affects this category only in secondary way. The introduction of work group leads to a significant increase in the impact respect to system 3.

4.3 Uncertainty analysis

Uncertainty analysis permits to analyse the uncertain aspect of the model. In building the LCA process uncertainty may arise from (Kohler *et al.*, 2002):

- Data quality (incomplete, inaccurate, not appropriate, obsolete etc.)
- Building description (incomplete, inaccurate)
- Building life span and building components of lifecycle (assumption on life span)
- Building operation (user influence, long term evolution of cost etc.)

- Upstream process analysis since inventories of material and energy flows are highly depended on geographical localisation and technology

A single data can have a great uncertainty even if its contribution to the uncertainty result could be smaller (Kohler et al., 2002). To deal with this problem a Monte Carlo analysis can be performed. The analysis is made by repeating many times the calculation where each time a random value is chosen for each flow. The resulting range of all calculation results form a distribution from which uncertainty information can be derived with statistical methods. The value chosen in the Monte Carlo are within a specified distribution that can be selected in SimaPro. Otherwise the Ecoinvent database selected in this study supplies uncertainty data with the inventory data. Analysis results will report information on (Pré, 2019):

- Mean: this is the average score of all results (the sum of all results divided by the number of results). This value can be heavily influenced by outliers.
- Median value: middle value of all results calculation. It is useful whenever the mean is influenced by outliers.
- Standard error of mean: can be also described as the amount by which the last calculation influenced the mean.
- Variability coefficient: it is the ratio between the standard deviation and the mean and a useful parameter to describe the relative magnitude of uncertainty.

The Monte Carlo analysis is performed for all three systems by defining an interval of 95%.

4.3.1 Uncertainty analysis for system 1

The result of the Monte Carlo analysis for system 1 are reported in Figure(4.13).

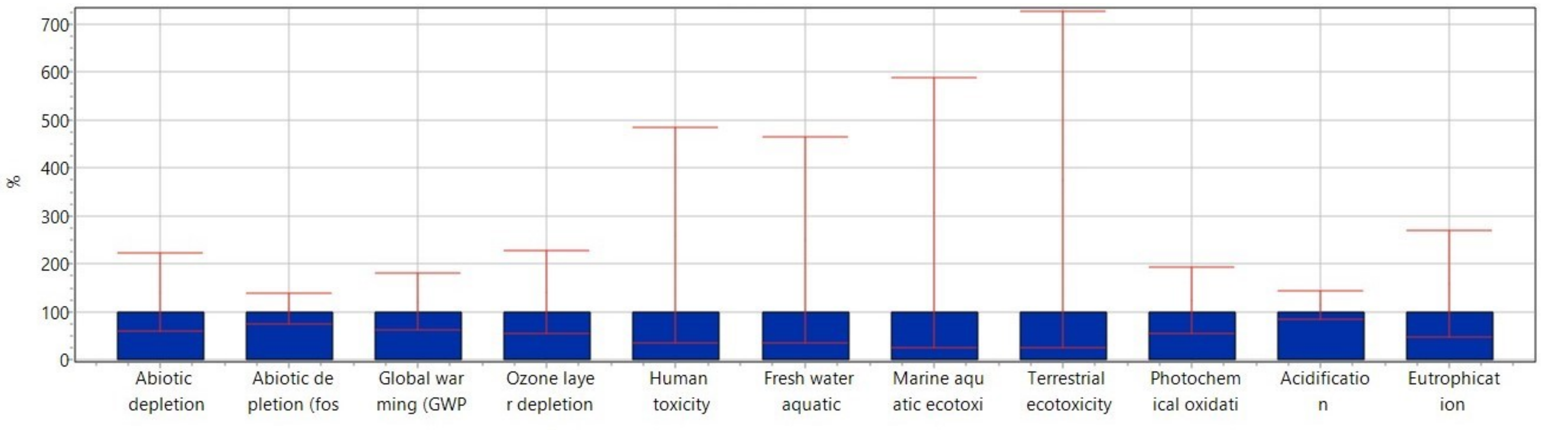


Figure 4.13: Uncertainty analysis for system 1, a graphical interpretation. (A personal rework of SimaPro data)

The variability coefficient permits to describe the relative magnitude of uncertainty referred to impact categories. Considering it possible to underline how terrestrial ecotoxicity is the category main affected with the highest variability coefficient with a value of 725% followed by marine aquatic ecotoxicity with 590 % and human toxicity with 490 % while fresh water aquatic ecotoxicity assess on 470%. The value of the other impact categories is much lower and never exceed 300%. Particularly the categories that show the minimum magnitude of uncertainty are acidification and abiotic depletion fossil fuels which are equal to 140%. Results on the other values are reported in Table(4.23).

Table 4.23: Uncertainty analysis result for system1

Impact category	**Unit of measure**	**Mean**	**Median**	**SD**	**CV**	**2,5%**	**97,5%**	**SEM**
Abiotic depletion	kg Sb eq	8,74 E-01	8,11E-01	3,11E-01	3,56E+01	4,76E-01	1,80E+00	1,39E-02
Abiotic depletion (fossil fuels)	MJ	6,19E+06	6,12E+06	9,73E+05	1,57E+01	4,47E+06	8,44E+06	4,35E+04
Acidification	kg SO2 eq	2,31E+03	2,24E+03	3,46E+02	1,49E+01	1,86E+03	3,2E+03	1,54E+01
Eutrophication	kg PO4--- eq	1,52E+04	1,31E+04	9,74E+03	6,38E+01	6,09E+03	3,54E+04	4,35E+02
Fresh water aquatic ecotox.	kg 1,4-DB eq	6,59E+07	4,81E+07	5,45E+07	8,26E+01	1,71E+07	2,24E+08	2,43E+06
Global warming (GWP100a)	kg CO2 eq	4,45E+06	4,23E+06	1,31E+06	2,59E+01	2,58E+06	7,6E+07	5,88E+04
Human toxicity	kg 1,4-DB eq	9,78E+06	7,1E+06	8,3E+06	8,48E+01	2,45E+06	3,42E+07	3,71E+05
Marine aquatic ecotoxicity	kg 1,4-DB eq	3,19E+11	2,12E+11	3,19E+11	1E+02	5,34E+10	1,25E+12	1,43E+10
Ozone layer depletion (ODP)	kg CFC-11 eq	7,43E-03	6,68E-02	3,04E-02	4,09E+01	3,78E-02	1,52E-01	1,36E-03
Photochemical oxidation	kg C2H4 eq	8,15E+02	7,7E+02	2,8E+02	3,43E+01	4,25E+02	1,49E+03	1,25E+01
Terrestrial ecotoxicity	kg 1,4-DB eq	9,64E+03	5,36E+03	1,24E+04	1,29E+02	1,34E+3	3,89E+04	5,56E+02

4.3.2 Uncertainty analysis for system 2

The result of the Monte Carlo analysis for system 2 are reported in Figure(4.14).

Uncertainty biggest contribution is linked to the abiotic depletion category with 190% immediately followed by ozone layer depletion for which the value is 185%. Marine aquatic ecotoxicity and terrestrial ecotoxicity shows the same value of 145% while photochemical oxidation and acidification values are respectively 130% and 122%. All the other categories show a value smaller than 120% while the lowest variability coefficient is correlated to global warming potential with 110%. Results on the other uncertainty analysis values are reported in Table(4.24).

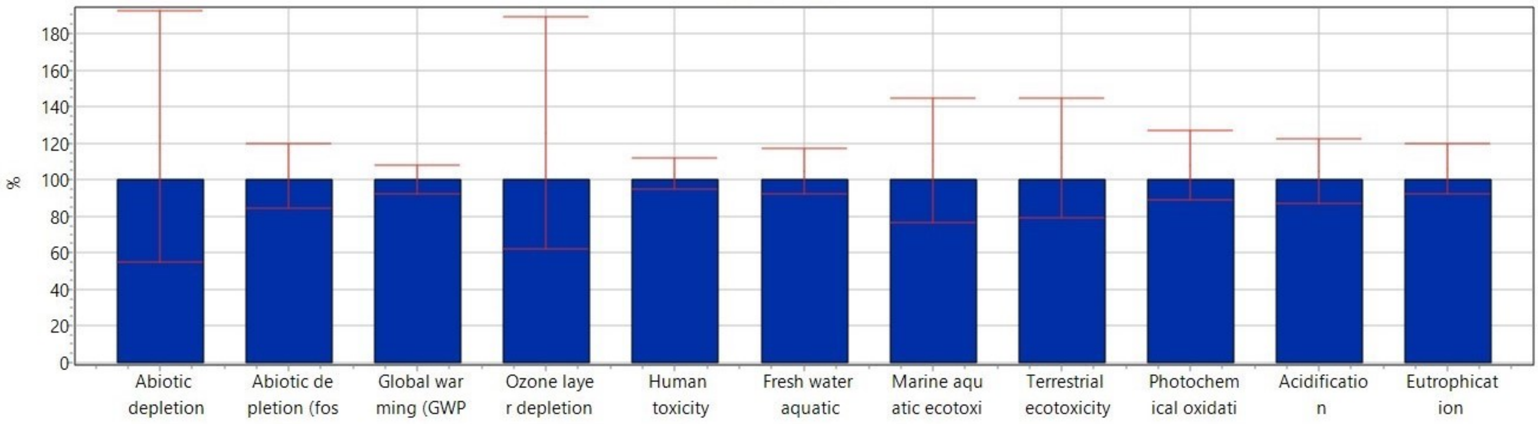


Figure 4.14: Uncertainty analysis for system 2, a graphical interpretation. (A personal rework of SimaPro data)

Table 4.24: Uncertainty analysis result for system 2

Impact category	**Unit of measure**	**Mean**	**Median**	**SD**	**CV**	**2,5%**	**97,5%**	**SEM**
Abiotic depletion	kg Sb eq	3,82E-01	3,66E-01	1,16E-01	3,03E+01	2,04E-01	6,53E-01	5,2E-03
Abiotic depletion (fossil fuels)	MJ	1,02E+06	1,01E+06	9,21E+04	9,03E+00	8,65E+05	1,24E+06	4,12E+03
Acidification	kg SO2 eq	3,09E+02	3,07E+02	2,46E+01	7,94E+00	2,66E+02	3,64E+02	1,1E+00
Eutrophication	kg PO4--- eq	2,21E+02	2,17E+02	1,82E+01	8,25E+00	1,98E+02	2,67E+02	8,16E-01
Fresh water aquatic ecotox.	kg 1,4-DB eq	3,55E+04	3,5E+03	2,37E+03	6,69E+00	3,24E+04	4,13E+04	1,06E+02
Global warming (GWP100a)	kg CO2 eq	4,8E+04	4,78E+04	2,1E+03	4,37E+00	4,4E+04	5,25E+04	9,39E+01
Human toxicity	kg 1,4-DB eq	5,66E+04	5,61E+04	2,54E+03	4,48E+00	5,31E+04	6,25E+04	1,13E+02
Marine aquatic ecotoxicity	kg 1,4-DB eq	3,39E+07	3,31E+07	6,66E+07	1,96E+01	2,48E+07	4,85E+07	2,97E+05
Ozone layer depletion (ODP)	kg CFC-11 eq	8,19E-03	7,56E-03	3,13E-03	3,82E+01	4,46E-03	1,58E-02	1,4E-04
Photochemical oxidation	kg C2H4 eq	1,1E+01	1,09E+01	9,58E-01	8,65E+00	9,66E+00	1,31E+01	4,28E-02
Terrestrial ecotoxicity	kg 1,4-DB eq	9,96E+01	9,81E+01	1,45E+01	1,46E+01	7,59E+01	1,34E+02	6,5E-01

4.3.3 Uncertainty analysis for system 3

The result of the Monte Carlo analysis for system 3 are reported in Figure(4.15).

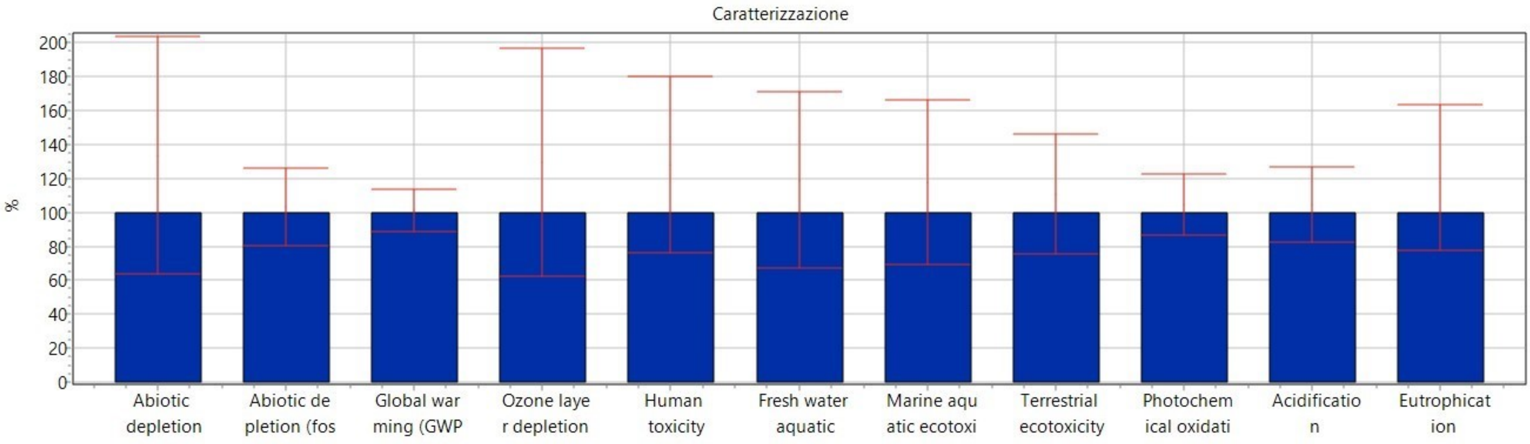


Figure4.15: Uncertainty analysis for system 3, a graphical interpretation

Uncertainty results can be described, as for the other systems, in term of variability coefficient. Considering that abiotic depletion is the category with the highest magnitude of uncertainty showing a value for the variability coefficient of 205 % followed by ozone layer depletion with 195%. Human toxicity, fresh water aquatic ecotoxicity, marine aquatic ecotoxicity and eutrophication variability coefficient are in between 180% and 160%. All the other variability coefficients stand under 130% and global warming potential shows the lowest value with 115%. Results on the other uncertainty analysis values are reported in Table(4.25).

Table 4.25: Uncertainty analysis result for system 3

Impact category	**Unit of measure**	**Mean**	**Median**	**SD**	**CV**	**2,5%**	**97,5%**	**SEM**
Abiotic depletion	kg Sb eq	2,38E-01	2,18E-01	7,74E-02	3,25E+01	1,39E-01	4,44E-01	2,45E-03
Abiotic depletion (fossil fuels)	MJ	3,81E+06	3,78E+06	4,40E+05	1,15E+01	3,05E+06	4,76E+06	1,39E+04
Acidification	kg SO2 eq	1,13E+03	1,11E+03	1,34E+02	1,19E+01	9,21E+02	1,40E+03	4,26E+00
Eutrophication	kg PO4--- eq	3,41E+02	3,24E+02	7,57E+01	2,21E+01	2,52E+02	5,32E+02	2,39E+00
Fresh water aquatic ecotox.	kg 1,4-DB eq	3,59E+04	3,34E+04	3,33E+04	9,24E+01	2,23E+04	5,71E+04	1,05E+03
Global warming (GWP100a)	kg CO2 eq	1,71E+05	1,7E+05	1,08E+04	6,35E+00	1,51E+05	1,94E+05	3,44E+02
Human toxicity	kg 1,4-DB eq	4,64E+04	4,37E+04	1,59E+04	3,43E+01	3,31E+04	7,88E+04	5,05E+02
Marine aquatic ecotoxicity	kg 1,4-DB eq	1,13E+08	1,08E+08	4,2E+07	3,72E+01	7,5E+07	1,79E+08	1,33E+06
Ozone layer depletion (ODP)	kg CFC-11 eq	3,56E-02	3,30E-02	1,14E-02	3,20E+01	2,06E-02	6,52E-02	3,62E-04
Photochemical oxidation	kg C2H4 eq	4,26E+01	4,22E+01	4,45E+00	1,04E+01	3,52E+01	5,17E+01	1,40E+01
Terrestrial ecotoxicity	kg 1,4-DB eq	3,75E+02	3,65E+02	6,84E+01	1,82E+01	2,74E+02	5,33E+02	2,16E+00

After the impact assessment analysis, the last phase of the LCA methodology is performed, which is the result interpretation phase. This phase permits to draw the results of the analysis by considering together the LCI and LCIA phases.

4.4 Results discussion

In the light of the previous results concerning the characterization of the three systems and the sensitivity and uncertainty analysis it is possible to draw some consideration about their environmental performances. The great part of the data introduced in the model are primary ones and refer to proper system's analysis and information on streams and operations linked to functional unit and following the burdens assumption. Indeed, as underlined by Henriksen *et al.*, (2018) the use of primary data is fundamental for a site-specific assessment. Regarding the hypothesis introduced, they were studied through sensitivity analysis.

Considering firstly system 1 results have underlined that the two main contribution are waste transportation and the final disposal of waste. Indeed, transportation of waste is the main contribution in the description of abiotic depletion, abiotic depletion fossil fuels, ozone layer depletion and acidification. Particularly the main impact on transportation is linked to the exiting waste that represents the 84,2% of the total transportation assessment for system 1 and consequently input one consist only in 15,86%. The reason why output transportation assessment and, consequently, its associated impact are bigger than input one can be found in the longest distances associated to this group. In order to improve the transportation performances in terms of associated impact it is possible to use lorries with EURO 4 characteristic instead of EURO 3 ones. This condition studied in the sensitivity analysis underlines an improvement in most categories with a particular contribution, linked a smaller impact, for acidification. Considering now the final waste disposal both landfill and incineration have an hight contribution to the final results for system 1. Particularly incineration process is predominant for human toxicity, fresh water ecotoxicity and marine aquatic ecotoxicity, due incineration emissions, while landfill is the main contribution for global warming potential, considering methane and carbon dioxide in biogas, terrestrial ecotoxicity, photochemical oxidation and eutrophication. The energy consumption of system has a contribution in the description of abiotic depletion, abiotic depletion fossil fuels, ozone layer depletion and acidification, while for the other categories contribution is almost negligible. The impact on this category is linked to the usage of fossil fuel as energy source or in order to produce electricity. Considering finally lubricating oil used for maintenance work its contribution is null in all categories.

For system 2 the main group contribution is capping but also categories like utilities are predominant in the impact description for all categories. Particularly considering abiotic depletion fossil fuels, global warming potential, marine aquatic ecotoxicity, terrestrial ecotoxicity, photochemical oxidation and acidification. These two groups impact is mainly related to the use of fossil sources for the energy production, particularly for electricity, as energy source for operating machines and as raw material considering plastic production and its particular usage in capping for covering and for leachate and gas collection. Transportation of incoming waste is important in the description of all categories together with impact related to material used in plant, but this last one contribution is small compared to all other groups. Abiotic depletion is the category for which capping shows the main contribution linked not only to the usage of clay for covering, which also reflects in a percentage increase with respect to other categories of material group, but particularly on the usage of zinc-coated wire associated to cage process. Since the amount of wire has been calculated with a geometrical correlation and since it has such a big contribution in the abiotic depletion category the results of the sensitivity analysis are moreover important. Indeed it has been found that decreasing the amount of wire used leads to an almost linear decrease in abiotic depletion impact, associated to the a smaller usage of zinc, and the same behaviour can be found also increasing the wire used. Accounting for the incoming waste it is representative in the description of all impact categories and since this category is related only to waste transportation a sensitivity analysis has been done to evaluate the benefits in using a less impactful, regarding emission, type of lorries. However, this change leads only to a few percentage units of improvement in acidification, photochemical oxidation and eutrophication as previously underline also for system 1. Considering leachate group, it is the main contribution in the description of fresh water aquatic ecotoxicity, photochemical oxidation and acidification. The component that has the highest impact is hexachlorobutadiene, associated to water emission, that is present with a concentration of 0,1 mg. Particularly his impact is associated to human toxicity and fresh water aquatic ecotoxicity since it is a suspected carcinogens for humans but certainly associated to an hight toxicity for humans (exposure limit of 0.02 ppm over an eight-hour workday (ILO)) and it can cause long-term effects in aquatic environments and bioaccumulation (ILO). The sensitivity study underlines that a reduction in the hexachlorobutadiene concentration can lead to an important improvement considering these two categories. The other emission associated to system 2 is biogas. Biogas, differently from system 3, it is not characterised by hight concentration in biogas or carbon dioxide. This leads to the fact that it is not necessary to treat by combustion the biogas emission in 2018 and also to a really small contribution in the impact assessment of system 2.

Finally looking at system 3 it is possible to underline how the main contribution is linked to utility usage, comprehended both diesel and electricity. In both cases their usage is bigger than the one of system 2 also if quantity are reported to 1 tonne of waste treated in the plant. Since the main sources for energy production are fossil fuels their associated impact is reflected particularly on ozone layer depletion and acidification categories even if the impact is elevated for all the other ones. Considering this also the usage of plastic sheet it has an appreciable weight in the description of abiotic depletion fossil fuels, global warming potential and photochemical oxidation and is linked to the usage of fossil fuels as raw material and their production processes. Only for abiotic depletion fossil fuels is possible to underline a bigger contribution due to capping, and not to utility, linked to an hight usage of clay which weights also in the description of materials and capping of system 2. With this in mind, the effect of burdens hypothesis on the cut-off of the works group from system 3 was studied. The large amount of material associated to relevant impact as clay and plastic derived materials follow to an hight relative impact of this group that however is still not introduced in system 3. Incoming waste is characterised by transportation of waste emission and its improvement, in terms of emission quality, has been studied by introducing the hypothesis of EURO 4 lorries usage that leads only to a small decrease in impact due to the low impact that this group has compared to utility one. For leachate, its contribution is given both to transportation of leachate outside system 3 and the emission in water characterised by concentration of components. Eutrophication is the category that main suffer from leachate emission in water, particularly due to high concentration in COD followed by terrestrial ecotoxicity which is associated to mercury emission. Mercury particularly suffers from composition hypothesis and diminishing its concentration leads to a 5% smaller impact for terrestrial ecotoxicity.

Differently to system 2, where methane concentration is so low that the treatment of biogas is not required, for system 3 its production is still relevant and average equal to 45% in volume of biogas produced. The emission of carbon dioxide associated to flare combustion leads to a small contribute in global warming potential category thanks to the treatment of biogas before its dismission that, otherwise, will lead to an higher impact since its potential factor is 24 bigger than the CO_2 one. Uncertainty analysis performed show how the main contribution to uncertainty for system 1 are marine aquatic ecotoxicity and terrestrial ecotoxicity while for system 2 and system 3 can be found in abiotic depletion category and ozone layer depletion.

Looking at all results is possible to affirm that the treatment of landfill emission is necessary, both for biogas and for leachate, to diminish the systems impact. However, the main impact assessment contribution for all systems is related to the usage of fossil fuels as base energy source and to the usage of materials linked to fossil sources. Transportation of waste is an important contribution for the impact assessment in all systems and this underlines how primary data ensure a proper description of the systems (Fernandez-Nava et al. (2014), Yadav et al., (2018), Wang et al., (2018), Behrooznia et al., (2018)). Furthermore, capping shows an important contribution for the two landfill systems underling the necessity of insert it as a process in order to fully describe this type of systems. The use of primary data is fundamental to identify the hotspot related to these processes, as demonstrate for system 2 for abiotic depletion category.

Conclusion

The objective of this study is the evaluation of the environmental performances, by applying the LCA methodology, of three different systems: a mechanical treatment plant and two landfills. Mechanical treatment facility consists of several treatment processes that are shredding, iron removal, screening and biostabilization. Moreover, landfilling facility comprehends the disposal process and the collection of leachate and biogas and their treatment together with the capping procedure to ensure the insulation of waste from the environment.

In the first part of this study the LCA methodology is described with particular attention to the ISO standards 14040(2006) and 14044(2017) that respectively defines "principles and framework" and "requirements and guidelines". The four phases of the LCA methodology are goal and scope definition, inventory analysis (or life cycle inventory, LCI), impact assessment (or life cycle impact assessment, LCIA) and interpretation. All these phases are subsequently followed in the study.

Indeed, first goal and scope definition are described as the environmental assessment of the performances of the three systems referred to 2018. The functional unit is defined, for all systems, as the amount of waste treated in 2018. The"zero-burden" assumption is applied where waste as empty concerning previous associated impacts and it is usually applied in the waste LCA's studies (Laurent et al., 2014). The boundaries for system 1 are defined as the energy, waste and materials entering the plant, considering for the last two also transportation, and the waste exiting the plant, its transportation and final disposal in landfill or incineration. System's 2 boundaries comprehend the transportation of waste entering the plant, diesel and electricity usage, materials for plant maintenance (iron wire and silt), natural resources depletion (water and land usage), capping (which comprehends all the different activities and materials for coverage), leachate and biogas emitted (both characterised by proper analysis). Finally considering the boundaries of system 3 they refer to the transportation of the incoming waste, the diesel and electricity usage, natural resources depletion (land occupation and water), materials for daily maintenance (plastic sheet), capping, while considering outputs they comprehend leachate (together with its transportation to the final disposal), emissions in air (since biogas is combusted in flare before being dismissed) and waste water. Particularly for system 2 and system 3 it is necessary to perform allocation for the streams that refer to all waste, like leachate and biogas or emissions in air, to the 2018 waste fraction respect to the waste previously landfilled. While considering capping and leachate and biogas or emissions in air formed after the landfill closure, the allocation is performed referring to the quote associated to the amount of waste treated in 2018 respect to the total amount of waste landfilled at closure.

Then the third phase of the LCA is performed considering as calculation method the CML method 2002. Firstly, characterisation is carried out for all the systems. Results show that the main contribution to the final impact for system 1, considering the groups, is characterised by the output one that represents the transportation of the outcoming waste and the final disposal. Particularly transportation is the most important contribution for abiotic depletion, abiotic depletion fossil fuels, ozone layer depletion and acidification, while for the other impact categories the main contribution is the final disposal. For system 2 leachate mainly contributes to human toxicity, fresh water aquatic ecotoxicity and eutrophication while utility represents mainly only ozone layer depletion. All the other categories are described by capping group while the contribution of biogas is negligible for all impact categories, since its composition presents low values for methane and carbon dioxide. System 3 is described by the utility group for all impact categories except for abiotic depletion. Considering emissions in air, the combustion of biogas permits the abatement of the methane fraction leading to a lower contribution of this group. Consequently, the three systems are compared by evaluating the impact per tonne of municipal solid waste treated. Results show that system 1 is the most impactful one for all categories. Considering the two landfills is possible to underline how system 3 is more impactful with respect to system 2, for all categories with exception of abiotic depletion, human toxicity, fresh-water aquatic ecotoxicity and eutrophication.

Furthermore, the main hypothesis made are studied by sensitivity analysis. Particularly since transportation is founded to be an important contribution for all systems, the use of EURO 4 lorries that shows only small contribution to the total impact. Hypothesis on leachate and biogas composition are investigated for all systems to evaluate how the assumed conservative conditions impact. Moreover, hypothesis on burdens for system 3 are studied founding that avoiding the contributions of works leads to a greater impact for system 3 than the base case one.

Uncertainty analysis show that the main contribution to uncertainty for system 2 and system 3 can be found in abiotic depletion and ozone layer depletion categories while for system 1 they refer to terrestrial ecotoxicity and marine aquatic ecotoxicity.

The advantages provided by this study regard firstly the use of primary data in the life cycle inventory that permits to underline the environmental hotspot of every systems linked to its management or its particular conditions (leachate and landfill biogas concentration). Moreover, the introduction of capping procedures leads to a more complete overview of the systems, representing the management systems choices. Further studies may contribute to a deeper understanding of impact related to the final waste disposal in system 1, introducing specific data in the description of landfills and incineration facilities.

Appendix

TableA1: Operating machine power and functioning in system 1

Operating machine	Power (kW)	Functioning (h/year)
Press	88	300
Leachate pump	2,5	200
Lightning	1,3	4000
Weight	0,08	-
Weather station	0,025	8760
Officies and storages	10	2000
Pressure washer	8,5	150

Table A2: Method for the composition calculation of biogas in system 2

Biogas composition	Method
Oxygen	EPA CTM 034 1999
Carbon dioxide	EPA CTM 034 1999
Carbon monoxide	NIOSH 6604 1996
Methane	UNI EN ISO 25140:2010
Hydrogen	NIOSH 6013 1994
Sulfuric acid	NIOSH 6013 1994
Mercaptans	NIOSH 2542 1994
Volatile organic compounds (VOC)	UNI CEN/TS 13649:2015
Dust	UNI EN 13284-1:2003

Table A3: Method for the composition calculation for leachate in system 2

Leachate composition (System 2)	**Method**
Metals	
Arsenic	EPA 3050B 1996 + EPA 6010D 2014
Calcium	EPA 3050B 1996 + EPA 6010D 2014
Iron	EPA 3050B 1996 + EPA 6010D 2014
Magnesium	EPA 3050B 1996 + EPA 6010D 2014
Manganese	EPA 3050B 1996 + EPA 6010D 2014
Nickel	EPA 3050B 1996 + EPA 6010D 2014
Lead	EPA 3050B 1996 + EPA 6010D 2014
Potassium	EPA 3050B 1996 + EPA 6010D 2014
Copper	EPA 3050B 1996 + EPA 6010D 2014
Sodium	EPA 3050B 1996 + EPA 6010D 2014
Zinc	EPA 3050B 1996 + EPA 6010D 2014
Cadmium *	EPA 3050B 1996 + EPA 6010D 2014
Chromium VI*	EPA 3050B 1996 + EPA 6010D 2014
Mercury*	EPA 3050B 1996 + EPA 6010D 2014
Acid-base substance	
Surfactants	UNI 10511-1:1996/A1:2000 and APAT CNR IRSA 5170 Man 29 2003
Ammonia, as N	APAT CNR IRSA4030 A2/C Man 29 2003
Organic carbon	APAT CNR IRSA5040 A2/C Man 29 2003
Phosphorus, total	APAT CNR IRSA 4110 A2 Man 29 2003
COD, Chemical Oxygen Demand	ISO 15705:2002
Kjeldahl-N	UNI EN 25663:1995
BOD5, Biological Oxigen Demand	APAT CNR IRSA 5120 B2 Man 29 2003
Chlorides, unspecified	APAT CNR IRSA 4020 A2 Man 29 2003
Sulfate	APAT CNR IRSA 4020 A2 Man 29 2003
Nitrate*	APAT CNR IRSA 4020 A2 Man 29 2003
Nitrite*	APAT CNR IRSA 4020 A2 Man 29 2003
Cyanide compounds *	APAT CNR IRSA 4070 A2 Man 29 2003

Chlorinated solvents	
Ethane, 1,1-dichloro*	EPA 5021A 2014+EPA 8260C 2006
Ethane, 1,1,2-trichloro-*	EPA 5021A 2014+EPA 8260C 2006
Ethane, 1,2-dichloro-*	EPA 5021A 2014+EPA 8260C 2006
Benzene, 1,2-dichloro-*	EPA 5021A 2014+EPA 8260C 2006
Propane, 1,2-dichloro-*	EPA 5021A 2014+EPA 8260C 2006
Propane, 1,2,3-trichloro-*	EPA 5021A 2014+EPA 8260C 2006
Benzene, 1,2,4-trichloro-*	EPA 5021A 2014+EPA 8260C 2006
Ethane, 1,2-dibromo-*	EPA 5021A 2014+EPA 8260C 2006
Benzene, 1,3-dichloro-*	EPA 5021A 2014+EPA 8260C 2006
Benzene, 1,4-dichloro-*	EPA 5021A 2014+EPA 8260C 2006
Methane, bromodichloro-*	EPA 5021A 2014+EPA 8260C 2006
Ethane, 1,1,2,2-tetrachloro-*	EPA 5021A 2014+EPA 8260C 2006
Trichlorobenzenes*	EPA 5021A 2014+EPA 8260C 2006
Chloroform*	EPA 5021A 2014+EPA 8260C 2006
Methane, chloro-, HCC-40*	EPA 5021A 2014+EPA 8260C 2006
Methane, dibromochloro-	EPA 5021A 2014+EPA 8260C 2006
Butadiene, hexachloro-	EPA 5021A 2014+EPA 8260C 2006
Benzene, chloro-	EPA 5021A 2014+EPA 8260C 2006
Methane, tetrachloro-, CFC-10	EPA 5021A 2014+EPA 8260C 2006
Dichlorophenol	EPA 5021A 2014+EPA 8260C 2006
2,3,5- Trichlorophenol	EPA 5021A 2014+EPA 8260C 2006
3-Chlorophenol	EPA 5021A 2014+EPA 8260C 2006
Phenolic compounds	
Phenol*	EPA 1653 1996
Phenol, pentachloro-*	EPA 1653 1996
P-ethylphenol *	EPA 1653 1996
2,5-Dimethylphenol*	EPA 1653 1996
PFAS	ISO 25101:2009

Table A4: Method for the composition calculation of emission in air in system 3

Emission in air composition of 1 Nm^3	**Method**
Particulates	UNI EN 13284-1:2017
Carbon monoxide	UNI EN 15058:2017
Nitrogen oxides	UNI EN 14792:2017
Total Organic Carbon	UNI EN 12619:2013
Sulfur oxides	UNI EN 14791:2017
Fluoride compounds *	DM 25 agosto 2000
Hydrogen chloride	DM 25 agosto 2000

Table A5: Method for the composition calculation of leachate in system 3

Leachate composition	**Method**
Oxygen	APHA standard method for examination of water and waste water 2012
Fluoride	UNI EN ISO 10204-1:2009
Chlorides	UNI EN ISO 10204-1:2009
Nitrite	UNI EN ISO 10204-1:2009
Nitrate	UNI EN ISO 10204-1:2009
Sulfate	UNI EN ISO 10204-1:2009
Ammonia, as N	MU 2353:2009
Iron	EPA 3015A: 2007+EPA6010D 2014
Manganese	EPA 3015A: 2007+EPA6010D 2014
COD, Chemical Oxigen Demand	ISO 15705:2002
BOD5, Biological Oxigen Demand	APHA standard method for examination of water and waste water 22th 2013
TOC, Total Organic Carbon	APAT CNR IRSA 5040 Man 2B 2003
Cyanide compounds *	APAT CNR IRSA 4070 Man 2B 2003
Arsenic*	EPA 3015A: 2007+EPA6010D 2014
Cadmium *	EPA 3015A: 2007+EPA6010D 2014
Chromium	EPA 3015A: 2007+EPA6010D 2014
Chromium VI*	Standard methods 3500-Cr B
Copper	EPA 3015A: 2007+EPA6010D 2014
Mercury *	EPA 3015A: 2007+EPA6010D 2014

Nickel	EPA 3015A: 2007+EPA6010D 2014
Lead	EPA 3015A: 2007+EPA6010D 2014
Zinc	EPA 3015A: 2007+EPA6010D 2014
Calcium	EPA 3015A: 2007+EPA6010D 2014
Magnesium	EPA 3015A: 2007+EPA6010D 2014
Potassium	EPA 3015A: 2007+EPA6010D 2014
Sodium	EPA 3015A: 2007+EPA6010D 2014
Benzene *	EPA 5030 C 2003+EPA 8260 C 2006
Benzene, -ethyl- *	EPA 5030 C 2003+EPA 8260 C 2006
Styrene *	EPA 5030 C 2003+EPA 8260 C 2006
Toluene *	EPA 5030 C 2003+EPA 8260 C 2006
Xylene *	EPA 5030 C 2003+EPA 8260 C 2006
Benzene, 1,2,4-trimetyl-*	EPA 5030 C 2003+EPA 8260 C 2006
Benzene, -butyl- *	EPA 5030 C 2003+EPA 8260 C 2006
Benzene, 1-propyl- *	EPA 5030 C 2003+EPA 8260 C 2006
Hydrocarbon aromatic *	EPA 5030 C 2003+EPA 8260 C 2006
Methane, chloro-, HCC-40 *	EPA 5030 C 2003+EPA 8260 C 2006
Chloroform *	EPA 5030 C 2003+EPA 8260 C 2006
Ethane, 1,2-dichloro- *	EPA 5030 C 2003+EPA 8260 C 2006
Ethane, 1,1-dichloro- *	EPA 5030 C 2003+EPA 8260 C 2006
Ethane, trichloro- *	EPA 5030 C 2003+EPA 8260 C 2006
Ethane, tetrachloro- *	EPA 5030 C 2003+EPA 8260 C 2006
Butadiene, hexachloro- *	EPA 5030 C 2003+EPA 8260 C 2006
Ethane, dichloro-(cis) *	EPA 5030 C 2003+EPA 8260 C 2006
Ethane, dichloro-(trans) *	EPA 5030 C 2003+EPA 8260 C 2006
Propane, 1,2-dichloro- *	EPA 5030 C 2003+EPA 8260 C 2006
Ethane, 1,1,2-trichloro- *	EPA 5030 C 2003+EPA 8260 C 2006
Propane, 1,2,3-trichloro-	EPA 5030 C 2003+EPA 8260 C 2006
Ethane, 1,1,2,2-tetrachloro-	EPA 5030 C 2003+EPA 8260 C 2006
Bromoform *	EPA 5030 C 2003+EPA 8260 C 2006
Ethane, 1,2-dibromo-*	EPA 5030 C 2003+EPA 8260 C 2006
Methane, bromodichloro-*	EPA 5030 C 2003+EPA 8260 C 2006
Organic compounds *(dibromodichloromethane)	EPA 5030 C 2003+EPA 8260 C 2006

Acrylonitrile *	EPA 5030 C 2003+EPA 8260 C 2006
Benzene, 1-methyl-2-nitro *	EPA 5030 C 2003+EPA 8260 C 2006
Benzene, 1-methyl-3-nitro *	EPA 5030 C 2003+EPA 8260 C 2006
4-nitrotoluene*	EPA 5030 C 2003+EPA 8260 C 2006
Etridiazole *	EPA 3510C 1996+EPA 8270 E 2007
Lindane, alpha-*	EPA 3510C 1996+EPA 8270 E 2007
Lindane, beta- *	EPA 3510C 1996+EPA 8270 E 2007
Atrazine *	EPA 3510C 1996+EPA 8270 E 2007
Delta-hexachlorocyclohexane *	EPA 3510C 1996+EPA 8270 E 2007
Lindane *	EPA 3510C 1996+EPA 8270 E 2007
Alachlor *	EPA 3510C 1996+EPA 8270 E 2007
Heptachlor *	EPA 3510C 1996+EPA 8270 E 2007
Aldrin *	EPA 3510C 1996+EPA 8270 E 2007
Terephthalate, dimethyl 2,3,5,6-tetrahloro *	EPA 3510C 1996+EPA 8270 E 2007
Heptachlor, epoxide *	EPA 3510C 1996+EPA 8270 E 2007
Chlordane, cis *	EPA 3510C 1996+EPA 8270 E 2007
Chlordane, trans *	EPA 3510C 1996+EPA 8270 E 2007
Trans-nonachlor *	EPA 3510C 1996+EPA 8270 E 2007
Endosulfan sulfate *	EPA 3510C 1996+EPA 8270 E 2007
Cis-permethrin *	EPA 3510C 1996+EPA 8270 E 2007
Trans-permethrin *	EPA 3510C 1996+EPA 8270 E 2007
Ametryn	EPA 3510C 1996+EPA 8270 E 2007
Bromacil *	EPA 3510C 1996+EPA 8270 E 2007
Chloropropham *	EPA 3510C 1996+EPA 8270 E 2007
Cycloate *	EPA 3510C 1996+EPA 8270 E 2007
Cyanazine *	EPA 3510C 1996+EPA 8270 E 2007
Diphenamid *	EPA 3510C 1996+EPA 8270 E 2007
Dipropylthiocarbamic acid S-ethyl ester *	EPA 3510C 1996+EPA 8270 E 2007
Fenarimol *	EPA 3510C 1996+EPA 8270 E 2007
Fluridone *	EPA 3510C 1996+EPA 8270 E 2007
Hexazimone *	EPA 3510C 1996+EPA 8270 E 2007
Metolachlor *	EPA 3510C 1996+EPA 8270 E 2007
Molinate *	EPA 3510C 1996+EPA 8270 E 2007

Napropamide*	EPA 3510C 1996+EPA 8270 E 2007
Norflurazon *	EPA 3510C 1996+EPA 8270 E 2007
Pebulate *	EPA 3510C 1996+EPA 8270 E 2007
Prometon *	EPA 3510C 1996+EPA 8270 E 2007
Prometryn *	EPA 3510C 1996+EPA 8270 E 2007
Pronamide *	EPA 3510C 1996+EPA 8270 E 2007
Propachlor *	EPA 3510C 1996+EPA 8270 E 2007
Propazine *	EPA 3510C 1996+EPA 8270 E 2007
Tebuthiuron *	EPA 3510C 1996+EPA 8270 E 2007
Terbacil *	EPA 3510C 1996+EPA 8270 E 2007
Terbutryn *	EPA 3510C 1996+EPA 8270 E 2007
Triadimefon *	EPA 3510C 1996+EPA 8270 E 2007
Tricyclazole *	EPA 3510C 1996+EPA 8270 E 2007
Diazinon *	EPA 3510C 1996+EPA 8270 E 2007
Dichlorvos *	EPA 3510C 1996+EPA 8270 E 2007
Disulfoton *	EPA 3510C 1996+EPA 8270 E 2007
Fenamiphos *	EPA 3510C 1996+EPA 8270 E 2007
Methyl paroxan *	EPA 3510C 1996+EPA 8270 E 2007
Mevinfos *	EPA 3510C 1996+EPA 8270 E 2007
Terbufos *	EPA 3510C 1996+EPA 8270 E 2007
Tetrachlorvinphos *	EPA 3510C 1996+EPA 8270 E 2007
Trifluralin *	EPA 3510C 1996+EPA 8270 E 2007
Naphthalene *	APAT CNR IRSA 5080 Man 28 2003
Fluorene *	APAT CNR IRSA 5080 Man 28 2003
Phenanthrene *	APAT CNR IRSA 5080 Man 28 2003
Anthracene *	APAT CNR IRSA 5080 Man 28 2003
Fluoranthene *	APAT CNR IRSA 5080 Man 28 2003
Pyrene	APAT CNR IRSA 5080 Man 28 2003
Benzo(a)anthracene *	APAT CNR IRSA 5080 Man 28 2003
Chrysene *	APAT CNR IRSA 5080 Man 28 2003
Benzo(b,j,k)fluoranthene *	APAT CNR IRSA 5080 Man 28 2003
Benzo(a)pyrene *	APAT CNR IRSA 5080 Man 28 2003
Diben(a,h)anthracene *	APAT CNR IRSA 5080 Man 28 2003

Benzo(g,h,i)perylene *	APAT CNR IRSA 5080 Man 28 2003
Indeno(1,2,3-cd)pyrene *	EPA 3510 C 1996+EPA 8270 E 2017
Phenol	EPA 3510 C 1996+EPA 8270 E 2017
Phenol, 2-chloro *	EPA 3510 C 1996+EPA 8270 E 2017
o-cresol	EPA 3510 C 1996+EPA 8270 E 2017
m-cresol	EPA 3510 C 1996+EPA 8270 E 2017
3-nitrophenol	EPA 3510 C 1996+EPA 8270 E 2017
Phenol, 2,4-dimethyl- *	EPA 3510 C 1996+EPA 8270 E 2017
Phenol, 2,4-dichloro- *	EPA 3510 C 1996+EPA 8270 E 2017
Metacresol, parachloro-*	EPA 3510 C 1996+EPA 8270 E 2017
2,6-dichlorophenol *	EPA 3510 C 1996+EPA 8270 E 2017
Phenol, 2,4,6-trichloro-	EPA 3510 C 1996+EPA 8270 E 2017
Phenol, 2,4,5-trichloro-*	EPA 3510 C 1996+EPA 8270 E 2017
Phenol, pentachloro- *	EPA 3510 C 1996+EPA 8270 E 2017
Phenol, 2,3,4,6 -tetrachloro- *	EPA 3510 C 1996+EPA 8270 E 2017

Table A6: Monthly composition of biogas before combustion in system 3

Month	Methane	Oxigen	Carbon dioxide
January	50,7	1,2	40,3
February	55,4	0,7	43,1
March	52,8	1,9	33,1
April	53,1	1,9	40,3
May	51,9	3	40,2
June	47,1	2,8	35,5
July	37	6,7	29,3
August	39,7	3,4	36,1
September	36,3	6,6	34,4
October	39,2	5,9	29,5
November	-	-	-
December	36,0	3,5	15,7

Table A7: Composition, molecular weight and evaluation method for biogas (before combustion) in system 3

Component	Amount (mg)	Molecular weight	Method
Dichloroethane	0,09	98,97	EPA TO 15 1999
Trimethilbenzene	0,646	120,19	EPA TO 15 1999
Trichlorobenzene	0,02	181,45	EPA TO 15 1999
Dichloropropane	0,015	112,98	EPA TO 15 1999
Dichlorobenzene	0,026	147	EPA TO 15 1999
Trimethilpentane	0,036	114,26	EPA TO 15 1999
2-hexanone	0,141	100,16	EPA TO 15 1999
Ethiltoluene	0,057	10,195	EPA TO 15 1999
Ispropyltoluene	22,6	134,21	EPA TO 15 1999
Vinylcyclohexene	0,21	108,18	EPA TO 15 1999
Acetone	0,46	58,08	EPA TO 15 1999
Alpha-metylstirene	0,028	118,18	EPA TO 15 1999
Allyl chloride	0,031	76,53	EPA TO 15 1999
Benzene	0,157	78,11	EPA TO 15 1999
Cyclohexene	0,24	84,16	EPA TO 15 1999
Vinylchloride	0,042	62,5	EPA TO 15 1999
Cumene	0,07	120,19	EPA TO 15 1999
Dichlorofluoromethane	0,007	102,19	EPA TO 15 1999
Diethyletere	0,006	74,12	EPA TO 15 1999
Ethylbenzene	0,204	106,17	EPA TO 15 1999
Isobutylacetate	0,05	116,16	EPA TO 15 1999
Isoprene	0,03	68,12	EPA TO 15 1999
Methyl ketone	0,012	72,11	EPA TO 15 1999
Metyl isobutyl ketone	0,5	100,16	EPA TO 15 1999
Xilene	0,402	106,16	EPA TO 15 1999
Naftalene	0,111	128,16	EPA TO 15 1999
Butylacetate	0,3	116,16	EPA TO 15 1999
Heptane	0,114	100,21	EPA TO 15 1999
Exane	0,084	86,18	EPA TO 15 1999
Propylbenzene	0,067	120,19	EPA TO 15 1999
Butylbenzene	0,017	134,22	EPA TO 15 1999

Terzbutyl benzene	0,005	134,22	EPA TO 15 1999
Tetrachloroethylene	0,01	165,83	EPA TO 15 1999
Tetrahydrofuran	2,28	72,11	EPA TO 15 1999
Thiophene	0,32	84,14	EPA TO 15 1999
Toluene	0,47	92,14	EPA TO 15 1999
Trichloroethylene	0,012	131,79	EPA TO 15 1999

Table A8: PFAS composition in leachate in system 2

PFAS (system 2)	Concentration (mg/l)	Method
PFBA	0,0011	ISO 25101:2009
PFPeA	0,0005	ISO 25101:2009
PFHxA	0,0018	ISO 25101:2009
PFHpA	0,0005	ISO 25101:2009
PFOA	0,0016	ISO 25101:2009
PFNA	0,0005	ISO 25101:2009
PFDoA	0,0005	ISO 25101:2009
PFTriA	0,0005	ISO 25101:2009
PFTeA	0,0005	ISO 25101:2009
PFBS	0,0019	ISO 25101:2009
PFHxS	0,0005	ISO 25101:2009
PFHpS	0,0005	ISO 25101:2009
PFOS	0,0005	ISO 25101:2009
PFDeS	0,0005	ISO 25101:2009
PFAS Total	0,0074	

Table A9: PFAS composition in leachate in system 2

PFAS (system 3)	**Concentration (ng/l)**	**Method**
PFBA	2900	ISO 25101:2009
PFBeA	660	ISO 25101:2009
PFBS	940	ISO 25101:2009
PFHpA	400	ISO 25101:2009
PFOS	1360	ISO 25101:2009
PFAS Total	8160	ISO 25101:2009

Table A.10: Impact assessment results in system 2. Percentage contribution.

Impact category	**Unit**	**Total**	**Materials**	**Utility**	**Capping**	**Incoming waste**	**Leachate**	**Biogas**
Abiotic depletion	%	100	5,95	1,96	88,78	3,29	0	0
Abiotic depletion (fossil fuels)	%	100	1,55	42,88	44,84	10,71	0	0
Global warming (GWP100a)	%	100	2,64	34,09	48,87	14,38	0	0,004089
Ozone layer depletion (ODP)	%	100	1,68	62,16	19,92	16,22	0	0
Human toxicity	%	100	3,01	6,80	9,11	4,89	76,16	0
Fresh water aquatic ecotox.	%	100	2,10	10,62	10,94	1,76	74,55	0
Marine aquatic ecotoxicity	%	100	13,63	38,15	39,92	5,57	2,70	0
Terrestrial ecotoxicity	%	100	6,69	37,04	31,00	10,42	14,82	0
Photochemical oxidation	%	100	3,93	40,44	44,37	11,23	0	0,005298
Acidification	%	100	3,05	42,25	43,11	11,57	0	1,44E-06
Eutrophication	%	100	1,58	13,61	15,51	3,91	65,36	0

Table A.11: Impact assessment results in system 2. Percentage contribution.

Impact category	**Unit**	**Total**	**Utility**	**Incoming waste**	**Capping**	**Plastic sheet**	**Waste water**	**Leachate**	**Emission in air**
Abiotic depletion	%	100	20,63	20,77	45,62	4,29	0,001805	8,66	0
Abiotic depletion (fossil fuels)	%	100	66,88	6,64	2,20	19,64	0,000235	4,63	0
Global warming (GWP100a)	%	100	62,44	9,69	4,07	15,27	0,0005	6,51	1,99
Ozone layer depletion (ODP)	%	100	81,73	8,59	2,00	1,65	0,0001	6,01	0
Human toxicity	%	100	54,33	12,13	14,87	8,39	0,003	10,24	7,93E-04
Fresh water aquatic ecotox.	%	100	70,78	4,52	10,26	9,60	0,0025	4,81	0
Marine aquatic ecotoxicity	%	100	75,62	3,98	8,633	8,77	0,001	2,97	0
Terrestrial ecotoxicity	%	100	63,64	6,20	8,31	4,47	0,0055	17,34	0
Photochemical oxidation	%	100	65,19	7,17	5,16	17,73	0,0007	4,71	0,01
Acidification	%	100	75,05	7,50	3,80	8,47	0,0007	5,14	0,01
Eutrophication	%	100	57,57	5,96	4,81	6,05	0,007	25,56	0,01

References

Andreottola G., Carras P., (1992). Chemical and Biological Characteristic of Landfill Leachate. *Landfilling of waste leachate edited by Christensen H. T., Cossu R., Stegmann R.*.

Ayodele T. R., Ogunjuyigbe A. S. O., Alao M. A., (2017). Life cycle assessment of waste-to-energy (WtE) technologies for electricity generation using municipal solid waste in Nigeria. *Applied Energy*. Elsevier.

Behrooznia L., Sharif M., Alimardani R., Mousavi-Avval S. H., (2018). Sustainability analysis of landfilling and composting-landfilling for municipal solid waste management in the north of Iran. *Journal of Cleaner Production*. Elsevier.

Buratti C., Barbanera M., Testarmata F., Fantozzi F., (2014). Life Cycle Assessment of organic waste management strategies: an Italian case study. *Journal of Cleaner Production.* Elsevier.

Chi Y., Dong J., Tang Y., Huang Q., Ni M., (2015). Life cycle assessment of municipal solid waste source-separated collection and integrated waste management systems in Hangzhou, China. *Journal of Material Cycles and Waste Management.* Elsevier.

Coelho L. M. G., Lange L. C., (2016). Applying life cycle assessment to support environmentally sustainable waste management strategies in Brazil. *Resources, Conservation and Recycling.* Elsevier.

Council of the European Union, (1999). Council Directive 1999/31/EC of 26 April 1999 on the landfill of waste.

Damgaard D. A., Manfredi S., Merrild H., Stensøe S., (2011). LCA and economic evaluation of landfill leachate and gas technologies. *Waste Management.* Elsevier.

Decisione della commissione del 3 maggio 2000 (2000/532/CE). Sostituisce la decisione 94/3/CE che istituisce un elenco di rifiuti conformemente all'articolo 1, lettera a), della direttiva 75/442/CEE del Consiglio relativa ai rifiuti e la decisione 94/904/CE del Consiglio che istituisce un elenco di rifiuti pericolosi ai sensi dell'articolo 1, paragrafo 4, della direttiva 91/689/CEE del Consiglio relativa ai rifiuti pericolosi.

Di Maria F., Sordi A., Micale C., (2013). Experimental and life cycle assessment analysis of gas emission from mechanically–biologically pretreated waste in a landfill with energy recovery. *Waste Management.* Elsevier.

Di Maria F., Sisani F., Contini S., Ghosh S. K., Mersky R. L., (2020). Is the policy of the European Union in waste management sustainable? An assessment of the Italian context. *Waste Management.* Elsevier.

Direttiva del consiglio del 15 luglio 1975 relativa ai rifiuti (75/442/CEE).

Ekvall T., Wondimagegnehu G. A., Björklund A., Finnveden G., (2007). Limitations and amendments in life-cycle assessment on waste management. *11th International Waste Management and Landfill Symposium.*

EPA, EPA/600/R-06/060 (2006), Life Cycle Assessment: principles and practice, Environmental Protection Agency, Ohio.

Fernandez-Nava Y., del Riò J., Rodríguez-Iglesias J., Castrillón L., Marañón, (2014). Life cycle assessment of different municipal solid waste management options: a case study of Asturias (Spain). *Journal of Cleaner Production.* Elsevier.

Finkbeiner M., (2014). The International Standards as the Constitution of Life Cycle Assessment: The ISO 14040 Series and its Offspring. *Background and Future Prospects in Life Cycle Assessment.*

Guinée J.B., Heijungs R., Huppes G, Zamagni A., Masoni P., Buonamici R., Ekvall T. and Rydberg T., (2011). Life Cycle Assessment: Past, Present, and Future. *Environmental Science Technology*. Research Gate.

Henriksen T., Astrup T. F., Damgaard A., (2018). Linking Data Choices and Context Specificity in Life Cycle Assessment of Waste Treatment Technologies: A Landfill Case Study: Representative Technology Choices in LCA. *Journal of Industrial Ecology.* Research Gate.

Hong J., Chen Y., Wang M., Ye L., Qi C., Yuan H., Zheng T., Li X., (2017). Intensification of municipal solid waste disposal in China. *Renewable and Sustainable Energy Reviews.* Elsevier.

ISO, ISO 14040:2006, Environmental management – Life Cycle Assessment – Principles and framework, International Organization for Standardization (ISO), Geneva.

ISO, ISO 14044:2017 Environmental management – Life Cycle Assessment – Requirements and guidelines, International Organization for Standardization (ISO), Geneva.

Karak T., Bhagat R. M., Bhattacharyya P., (2012). Municipal Solid Waste Generation, Composition, and Management: The World Scenario. *Environmental Science and Technology. Research Gate.*

Khandelwal H., Thalla A. K., Kumar S., Kumar R., (2019). Life cycle assessment of municipal solid waste management options for India. *Bioresource Technology.* Elsevier.

Klöpffer W., (2014). Introducing Life Cycle Assessment and its Presentation in 'LCA Compendium'. *Background and Future Prospects in Life Cycle Assessment.* Research Gate.

Kohler N., Lűtzkendorf T. H., (2002). Integrated Life Cycle Analysis. *Building Research & Information.*

Kormi T., Ali N. B. H., Abichou T., Green R., (2017). Estimation of landfill methane emissions using stochastic search methods. *Atmospheric Pollution Research. Research Gate.*

Laurent A., Bakas I., Clavreul J., Bernstad A., Niero M., Gentil E., Hauschild M. Z., Christensen T. H., (2014). Review of LCA studies of solid waste management systems – Part I: Lessons learned and perspectives. *Waste Management.* Elsevier.

Leme M. M. V., Rocha M. H., Lora E. E. S., Venturini O. J., Lopes B. M., Ferreira C. H., (2014). Techno-economic analysis and environmental impact assessment ofenergy recovery from Municipal Solid Waste (MSW) in Brazil. *Resources, Conservation and Recycling. Research Gate.*

Lima P. D. M., Olivo F., Paulo P. L., Schalch V., Cimpan C., (2018). Life Cycle Assessment of prospective MSW management based on integrated management planning in Campo Grande, Brazil. *Waste Management.* Elsevier.

Liu Y., Ni Z., Kong X., Liu J., (2017). Greenhouse gas emissions from municipal solid waste with a high organic fraction under different management scenarios. *Journal of Cleaner Production.* Elsevier.

Mali S. T., Patil S.S., (2016). Life cycle assessment of municipal solid waste management. *Waste and Resource Management.* Research Gate.

Nabavi-Pelesaraei A., Bayat R., Hosseinzadeh-Bandbafha H., Afrasyabi H., Chau K., (2017). Modeling of energy consumption and environmental life cycle assessment for incineration and landfill systems of municipal solid waste management - A case study in Tehran Metropolis of Iran. *Journal of Cleaner Production.* Elsevier.

Naveen B.P., Mahapatra D. M., Sitharam T.G., Sivapullaiah P.V., Ramachandrabc T.V., (2017). Physico-chemical and biological characterization of urban municipal landfill leachate. *Environmental Pollution.* Elsevier.

Goedkoop M., Oele M., Leijting J., Ponsioen T., Meijer E., (2013). Introduction to LCA with SimaPro. *Pré*. https://pre-sustainability.com/

Guinée, J.B.; Gorrée, M.; Heijungs, R.; Huppes, G.; Kleijn, R.; Koning, A. de; Oers, L. van; Wegener Sleeswijk, A.; Suh, S.; Udo de Haes, H.A.; Bruijn, H. de; Duin, R. van; Huijbregts, M.A.J. (2002). Handbook on life cycle assessment. Operational guide to the ISO standards. Part III: Scientific background. *Kluwer Academic Publishers,* ISBN 1-4020-0228-9, Dordrecht, 692 pp.

Rajaeifar M. A., Tabatabaei M., Ghanavati H., Khoshnevisah B., Shahin R., (2014). Comparative life cycle assessment of different municipal solid waste management scenarios in Iran. *Renewable and Sustainable Energy Reviews.* Elsevier.

Ripa M., Fiorentino G., Vacca V., Ulgiati S., (2015). The relevance of site-specific data in Life Cycle Assessment (LCA). The case of the municipal solid waste management in the metropolitan city of Naples (Italy). *Journal of Cleaner Production.* Elsevier.

PRé, various authors, (2019). SimaPro Database Manual: Methods Library. *simapro.com.*

Sauve G., Van Acker K., (2020). The environmental impacts of municipal solid waste landfills in Europe: A life cycle assessment of proper reference cases to support decision making. *Journal of Environmental Management.* Elsevier.

SETAC (1993). Guidelines for Life Cycle Assessment: a code for Practice, Society of Environmental Toxicology and Chemistry, Bruxelles.

Turner D. A., Beaven R. P., Woodman N. D., (2017). Evaluating landfill aftercare strategies: A life cycle assessment approach. *Waste Management*. Elsevier.

Yadav P., Samadder S. R., (2017). Environmental impact assessment of municipal solid waste management options using life cycle assessment: a case study. *Environmental Science Pollution Resource.* https://doi.org/10.1007/s11356-017-0439-7.

Yay A. S. E., (2014). Application of life cycle assessment (LCA) for municipal solid waste management: a case study of Sakarya. *Journal of Cleaner Production*. Elsevier.

Wang Z., Lv J., Gu F., Yang J., Guo J., (2019). Environmental and economic performance of an integrated municipal solid waste treatment: A Chinese case study. *Science of the Total Environment.* Elsevier.

Wang D., He J., Tang Y-T., Higgitt D., Robinson D., (2019). Life cycle assessment of municipal solid waste management in Nottingham, England: Past and future perspectives. *Journal of Cleaner Production.* Elsevier.

Other references

European commission, Environmental action programme to 2020 (7^{th}). (Last access 23/09/2020).
https://ec.europa.eu/environment/action-programme/

European commission and life cycle thinking. (Last access 23/09/2020)
https://ec.europa.eu/jrc/en/publication/life-cycle-thinking-and-use-lca-policies-around-world

European commission, guide on landfilling (Last access 23/09/2020)
https://ec.europa.eu/environment/waste/landfill/pdf/guidance%20on%20landfill%20gas.pdf

European commission, waste. (Last access 23/09/2020)
https://ec.europa.eu/environment/waste/index.htm

European commission, the European waste management approach (Last access 23/09/2020)
https://ec.europa.eu/environment/waste/index.htm

Hexachlorobutadiene characteristic (Last access 20/09/2020)
https://www.ilo.org/dyn/icsc/showcard.display?p_lang=en&p_card_id=0896&p_version=2

Pré sustainability (Last access 19/09/2020)
https://pre-sustainability.com/articles/publication-whats-new-in-simapro-8/

SimaPro database (Last access 19/09/2020)
https://simapro.com/databases/

Bentonite materials (Last access 19/09/2020)
http://www.bentonite.it/proprieta-bentonite.php

Geotexile material (Last access 19/09/2020)
https://www.infobuild.it/approfondimenti/geosintetici-e-geotessili-tessuti-anche-per-ledilizia/

Flare for landfill gas (Last access 19/09/2020)
https://www.prmfiltration.com/flare-systems

Ecoinvent database (Last access 16/09/2020) https://www.ecoinvent.org/database/ecoinvent-37/ecoinvent-37.html

Soil density (Last access 22/07/2020)
https://www.studiopetrillo.com/pesi-specifici-materiali.html

Concrete materials (Last access 22/07/2020)
https://www.concrete.org/topicsinconcrete/topicdetail/materials%20in%20concrete?search=materials%20in%20concrete

Concrete materials (Last access 22/07/2020)
https://www.cmcmanufatticemento.com/Prodotto.aspx?id=2b4b6def-a516-4910-b252-6f63b2cfca07

Polyethylene geocells (Last access 22/07/2020)
http://it.huataogroup.com/geocells/

Polyethylene tube (Last access 22/07/2020)
https://www.oppo.it/materiali/tubi_raccordi/pe_100_tubi.html

Polyethylene materials (Last access 22/07/2020)
https://www.comid.it/it/46-serbatoi-in-polietilene

Zinc coating (Last access 22/07/2020)
https://galvanizeit.org/corrosion/corrosion-protection/zinc-coatings

Gravel properties (Last access 22/07/2020)
https://www.aqua-calc.com/page/density-table/substance/gravel-coma-and-blank-regular

Fertilizer (Last access 22/07/2020)
https://www.politicheagricole.it/flex/files/e/7/8/D.be24b973538d92ad5645/Fertilizzanti.pdf

Diafram pumps (Last access 22/07/2020)
https://ph.parker.com/it/it/diaphragm-pumps

European commission, European platform of life cycle assessment (Last access 10/06/2020)
https://ec.europa.eu/environment/ipp/lca.htm

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