Comparison of the environmental impact of alternative flue gas cleaning systems

Master's thesis in Innovative and Sustainable Chemical Engineering

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Abstract

When municipal solid waste is incinerated, there are many pollutants that needs to be removed from the flue gas due to environmental or health-related considerations. To do this, several different technologies can be used. The aim of this study is to investigate these options from an environmental standpoint and compare them to see which has the lowest impacts. An LCA was performed on five different system configurations; dry + wet, dry, wet, wet with H_2SO_4 recycling, and wet with $CaCl_2$ recycling. The inventory data used to calculate the material requirements and emissions was based on information provided by a Swedish waste incineration facility. The impacts of these five scenarios were then evaluated based on the categories climate change, acidification, respiratory inorganics, ozone depletion and human health effects. The handling of long-term emissions in LCA is also discussed. The results indicate that the direct emissions of the waste incineration facility play a major role in all emissions except ozone depletion. If the $CaCl_2$ recycling can be performed, then that option will have the best environmental impact in all categories. Otherwise, the technology with least environmental impact is a combined system with both dry and wet technologies.

Keywords: Municipal solid waste incineration, air pollution control, acid gas treatment, life cycle assessment, life cycle impact assessment, long-term emissions.

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Jacob Nilsson, Gothenburg, June 2020

Contents

List of Figures

List of Tables

1
Introduction

This chapter covers the context and aim of the study as well as the research questions that the study hopes to answer.

This study is done in cooperation with Babcock & Wilcox Vølund AB. Babcock & Wilcox Vølund AB is a company that designs and delivers solutions for flue gas cleaning and condensation. When selecting which process to use for a specific case, these solutions are evaluated based on their stack emissions and from an economic perspective, but the overall environmental impact is not evaluated. Therefore, this work is meant to provide such a comparison by performing a life cycle assessment (LCA) of the different available technologies to see the extent of the environmental impacts. Since the environmental impacts of waste incineration systems largely depend on the flue gas cleaning technology and the heat and energy produced [1], this comparison could show whether one system is better from an environmental perspective than the other.

The aim of this study is thus to provide information about the environmental impacts of different flue gas cleaning technologies to the environmental engineering sector. The intended outcome of this is for operators of flue gas cleaning systems to be able to make more informed decisions when selecting a technology to implement, based on environmental performance.

This type of analysis of flue gas cleaning systems might become more important as stricter emission limits are implemented. Many studies have been done regarding the environmental impact of municipal waste incineration [2, 3, 4] and landfills [5, 6, 7], but fewer compare the different technology options for cleaning the flue gas relative to each other [1, 8]. The ones found are described in section 2.7.

Since operation of a waste incineration plant, in addition to the direct emissions to air and water, also results in slag and residue that have to be landfilled, this study will also discuss the current LCA approach to long-term emissions and its shortcomings.

The research questions this study will attempt to answer are:

- What are the environmental impacts of the different technologies? Which technology has the least impact?

- Which aspect of the technologies has the greatest effect? Is it the upstream processes, the direct emissions to air and water or the landfill emissions?

- What are the current scientific norms and the future methodological outlook regarding the assessment of elemental and persistent contaminants in an LCA framework?

2
Background

This chapter will provide the background information necessary to understand the study, as well as a summary of the previous research done in this field.

When household waste is thrown away, there are several different ways it can be treated. It can be sent to material recycling, biological treatment, energy recovery or a landfill, in order of descending hierarchy. In Sweden about 50 % of household waste goes to a waste-to-energy (WtE) plant, which in 2018 corresponded to over 2.3 million tonnes of waste [9]. In these plants the waste is incinerated to generate both district heating and electricity, and the waste volume is reduced as well. However, in this process many different pollutants are released into the flue gas. Pollutants of historical concern are NOx, SOx, acid gases and dioxins. Due to these pollutants, waste treatment facilities are equipped with air pollution control (APC) systems, which remove the pollutants from the flue gas through various methods and technologies in order to meet emission requirements.

2.1 Waste-to-energy

Typically, in a WtE-plant the waste is deposited by the garbage collection trucks into a large storage bunker. From there it is then lifted by a crane and brought into the furnace. Here the waste is mixed and transported through the incineration chamber by a moving grate. This, along with a supply of excess air, helps achieve a more complete combustion. The fumes from the incinerated waste rise into the boiler, where the heat generated by the combustion is absorbed by water flowing through pipes in the chamber wall, which vaporizes the water. This steam can then be used to either heat water for district heating or to power a steam turbine in order to generate electricity, or both, depending on the local demand and the season. The residues of the incinerated waste, called bottom ash or slag, are collected and transported to a landfill. After this energy recovery the flue gas arrives at the APC section, before it is released through the stack. The APC system consists of several different steps, since there is no single technology that can remove every pollutant. The different stages and technology options will be described in the following sections.

2.2 Dust separation

Typically, the first step of flue gas cleaning is to remove the fly ash. The different technology options are cyclones, electrostatic precipitators (ESPs), baghouse filters

or venturi scrubbers, all of which have different removal efficiencies for different particle sizes. Since the fly ash has high concentrations of heavy metals and dioxins, this efficiency is a key aspect in choice of technology [10]. In WtE plants, ESPs and baghouse filters are the most common technologies due to their very high efficiencies [11]. These two dedusting techniques are described in greater detail below.

ESPs are a common dust removal method due to their simplicity and efficiency. As the name suggests, the separating mechanism is an electrostatic field generated by negatively charged wires. The flue gas is passed by the wires at low velocity, which ionises the particles. They then pass through sets of parallel metal plates which they are drawn to due to their charge. The particles are thus collected on the plates and the gas is cleaned. The plates are periodically rapped to collect the particles and remove build-up [12]. The collected fly ash is then stored and delivered to a landfill for deposition. An ESP can achieve removal efficiencies of $\geq$ 99 % for particles that are 2.5-10 µm and 95 % for particles below this size [10].

Baghouse filters, or fabric filters as they are sometimes called, contain a large number of bags suspended from a metal housing. The flue gas passes through these bags, which acts as a filter and leaves the particles stuck to the outside of the bag. Once a certain amount has accumulated, a burst of compressed air is blown down into the bag which dislodges the particles and allows them to be collected for landfilling. Baghouse filters reach even higher efficiencies than ESPs, at $\geq$ 99 % for all particle sizes [10].

2.3 Acid gas treatment

The neutralization of acid gases is a very important part of the APC system. The incineration of waste results in several species of acid gases forming, such as HCl, HBr, HF and SO_2 [10]. These gases can be treated with systems that use either solid or liquid reagents, which have their respective strengths and weaknesses. These systems can also be combined in several ways, such as a dry system followed by a wet, or two dry stages combined [13].

2.3.1 Dry systems

Dry technologies usually consist of only a single stage, although two-stage systems show great promise [14]. In these systems, a dry reagent is injected either directly into the gas stream or administered with a spray dryer in order to react with the gas. Most commonly this injected reagent is either a calcium-based compound or sodium bicarbonate. While dry scrubbing methods have the advantages of a simple design and are easy to operate, the poor kinetics of a solid-gas system means that the chosen reagent has to be present in excess [10]. This obviously results in higher amounts of reagents having to be bought and transported, even if the scrubbing residues are recycled to improve efficiency.

The most common of the calcium-based reagents is slaked lime, $Ca(OH)_2$. The

reactions occurring in a dry system using $Ca(OH)_2$ are shown in Equation 2.1 and 2.2.

$$2\,HCl + Ca(OH)_2 \longrightarrow CaCl_2 + 2\,H_2O + \frac{1}{2}\,O_2 \tag{2.1}$$

$$SO_2 + Ca(OH)_2 \longrightarrow CaSO_3 + H_2O \tag{2.2}$$

Alternatives to $Ca(OH)_2$ are limestone ($CaCO_3$) or burnt lime (CaO) [10].
If a calcium-based scrubbing system is used, the preceding fly ash separation can often be emitted since the reaction mostly takes place on the surface of the baghouse filter [10], and the products from the reaction can then be collected along with the fly ash.

Dry systems can also be operated as semi-dry or semi-wet: in a semi-dry system, the flue gas or reactant has been humidified, and in a semi-wet system a slurry is injected instead of a purely dry reactant.

2.3.2 Wet systems

A wet system usually requires two stages, one for the treatment of halogens, and one for the SO_2, due to the differing chemical properties of the acid gases. The first scrubber stage is usually operated at a very low pH value (pH<1), and the second at a higher pH (pH≈7). The neutralizing agent in the first stage is simply water, utilizing the fact that HCl has a very high solubility in water [10].

$$HCl\,(g) \rightleftharpoons HCl\,(aq) \tag{2.3}$$

This aqueous solution is then further treated with $CaCO_3$ and $Ca(OH)_2$ in order to clean the water, as shown in Equation 2.4 and 2.5.

$$2\,HCl\,(aq) + CaCO_3 \longrightarrow CaCl_2 + CO_2 + H_2O \tag{2.4}$$

$$2\,HCl\,(aq) + Ca(OH)_2 \longrightarrow CaCl_2 + 2\,H_2O \tag{2.5}$$

The calcium chloride that forms can then be reacted with Na_2SO_4 to produce gypsum. This is done if there are no separate streams for emitting $CaCl_2$ and water from the sulfate treatment, since this would result in clogging the pipes through precipitation.

In the second stage SO_2 is treated with either sodium hydroxide (NaOH), hydrogen peroxide (H_2O_2) or sodium carbonate (Na_2CO_3) [10]. These reactions are shown in Equations 2.6, 2.7 and 2.8.

$$SO_2 + 2\,NaOH + \frac{1}{2}\,O_2 \longrightarrow Na_2SO_4 + H_2O \tag{2.6}$$

$$SO_2 + H_2O_2 \longrightarrow H_2SO_4 \tag{2.7}$$

$$SO_2 + Na_2CO_3 + \frac{1}{2}O_2 \longrightarrow Na_2SO_4 + CO_2 \tag{2.8}$$

Compared to dry systems, wet systems have much better kinetics, provided a large contact area between the gas and the liquid is achieved [10].
The effluent water from the wet scrubbers are typically cleaned through ultrafiltration, reverse osmosis, neutralization, heavy metal precipitation, flocculation or sedimentation, and then released into the ocean.

2.4 NOx treatment

The two main ways of reducing NOx-emissions due to combustion are selective catalytic reduction (SCR) and selective non-catalytic reduction (SNCR). In both, the reactant used is commonly NH_3, which is injected into the flue gas stream to react with the NOx and form N_2 and water. The catalyst used in the SCR is typically V_2O_5, TiO_2 or WO_3 [15]. Of the two alternatives, SCR has a higher efficiency, up to 90 %, while the efficiency of a SNCR is usually around 50-70 % [16]. Due to this, the SCR is often the preferred choice [10], although some studies show that while the direct environmental impacts are lower using a SCR, the indirect impacts are higher [15]. This is partly due to the pollution related to production of the catalyst, but mainly due to the need to reheat the flue gas before it enters the reactor. This leads to energy losses that outweigh the higher NOx removal efficiency [15]. This energy loss might however be counteracted by placing the SCR directly downstream of the boiler, as a so called high-dust catalyst. This removes the need to reheat the gas, but instead shortens the lifespan of the catalyst [10]. This seems to indicate that the comparison between SCR and SNCR requires further research regarding which one is more environmentally friendly, which is beyond the scope of this study.

2.5 Dioxin treatment

Due to the highly toxic nature of dioxins, efficient removal of this pollutant is very important in WtE processes. Minimisation of dioxin formation can be achieved by proper combustion control, and making sure that fly ash precipitation occurs at temperatures below 200 °C. In dry systems there is also the option of injecting powdered activated carbon (PAC) into the stream, which will form a filter cake together with the other reactants on the surface of the baghouse filter, and adsorb both dioxins and mercury. In wet systems SO_2 can be recirculated from the scrubber into the flue gas stream, since dioxin formation is strongly dependent on the ratio between chlorine and sulphur [10].

2.6 Case study

The plant which the data for this study was based upon is a Swedish waste incineration plant which started operating in 2012. It uses a combined dry and wet system with inputs of slaked lime, PAC, NaOH and HCl. The slaked lime input is purchased as CaO and then hydrated before injection into the flue gas stream, turning it into $Ca(OH)_2$. The HCl is added to the first scrubber stage in the wet system, to improve the separation of ammonia by shifting the equilibrium towards ammonium ions. NOx abatement is done through a SNCR. The condensate water is cleaned using both ultrafiltration and reverse osmosis before being released into the nearby strait. The plant generates both district heat and electricity. A flow sheet for the process is shown in Figure 2.1.

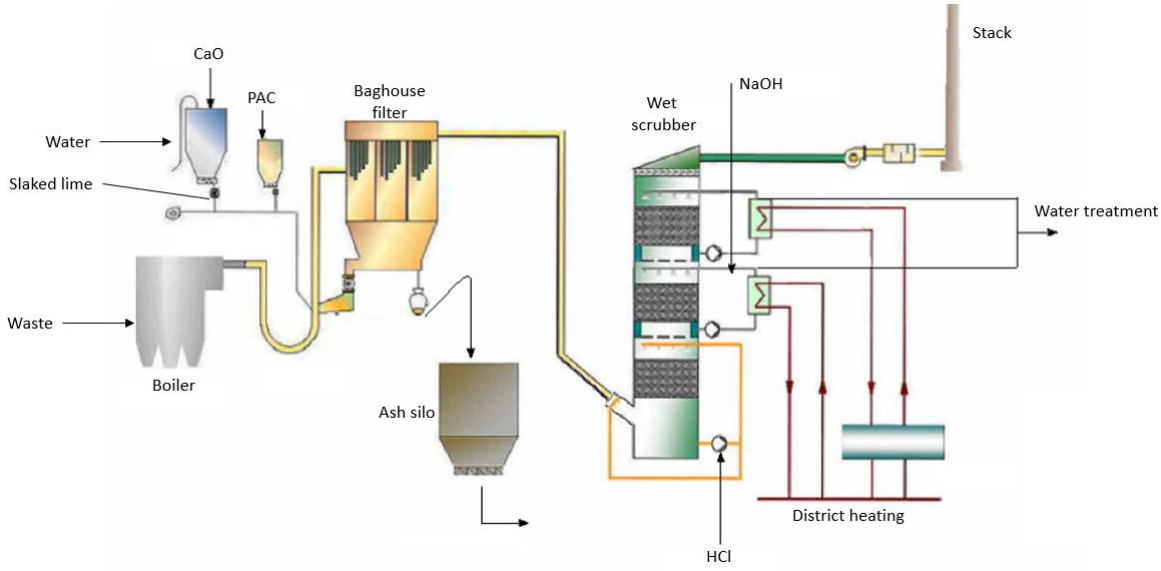

Figure 2.1: The process scheme for the Swedish waste incineration plant.

2.7 Previous research

There are a number of studies which compare two similar technologies: Dal Pozzo et al. (2016) [17], which compares a two-stage dry system with a single stage system, Dal Pozzo et al. (2017) [18] which compares the environmental effects of two dry sorbent injection systems, and Scipioni et al. [19] which compares three different design alternatives for a new Italian incineration line. There are also some that compare new experimental systems with traditional systems as well: Chevalier et al. [20] compares the performance of a new transported droplets column with a wet system, Stasiulaitiene et al. [21] compares two plasma-based technologies with wet flue gas desulphurization, and Biganzo et al. [22] investigates the impact of a pre-abatement step for acid gases using dolomitic sorbent.

The studies of the former type are relevant but difficult to use, since they all compare

only two or three scenarios and all have different impact indicators, while the latter are less relevant as this study is focused on the performance of established technologies. There is also a useful study by Boesch et al. [23] which evaluates the climate change impact of a dry/semidry system and a wet system, as well as the reduction in CO_2 that can be achieved through energy recovery. Another semi-relevant study, by Turconi et al. [4], compares two waste incineration plants in Denmark and Italy, although that study focuses mostly the differences caused by the geographical locations and how that affects the overall impact results.

Studies where several scenarios are compared are more uncommon, but very relevant examples can be found in the works of Dal Pozzo et al. (2018) [13] and Dong et al. (2020) [24].

In the study by Dal Pozzo et al. (2018), five different process schemes are considered and evaluated. The considered process schemes are: a single stage dry system, a two-stage dry system, a semi-dry system, a wet system and a dry + wet system. In the dry systems the residual sodium chemicals are recycled. The study in this report will only compare three of those process schemes, but will investigate additional recycling possibilities, such as sulfuric acid and calcium chloride. There are also some technical differences, such as Dal Pozzo considering a single-stage dry system with $NaHCO_3$ instead of $Ca(OH)_2$. Dal Pozzo's study is also based on the Ecoinvent database, albeit an older version. The study does also not use actual data from a plant for the dry + wet process, but is instead theoretically modeled to obtain consumption rates of the input chemicals.

In the study by Dong et al., 12 scenarios are investigated through combinations of eight different flue gas treatment technologies. The acid gas abatement technologies compared are dry systems, semi-dry systems and wet systems in different configurations. For the dry technologies, both $NaHCO_3$- and $Ca(OH)_2$-based systems are investigated. The wet system is a two-stage process, with HCl and HF removed using a water scrubber and SO_2 removed through absorption using $Ca(OH)_2$. The data used in the study was gathered from 90 French incineration plants during 2012-2015. System modeling is done using GaBi v8.5 and the LCIA is done using the ReCiPe midpoint indicators.

Both of these two studies have significant differences in both methods and assumptions compared to the work presented here, and this thesis should therefore provide an interesting complement to this area of study.

3
Method

In this chapter the theoretical framework of an LCA is described, as well as how this framework has been implemented in this study. Underlying theoretical considerations are also described.

3.1 LCA

In general, an LCA is a technique that can be used to map, sort or compare the environmental impacts of a process or system. This mapping is done by considering the impact of the process itself as well as the impact of upstream or downstream processes. It can therefore be very useful for identifying which parts of a supply chain that have a large impact on the overall emissions, which makes it easier to improve the environmental impact of the whole process. It can also be very useful as an tool for informing decision-makers, such as government officials or operators in industry. Since this technique has many aspects that are open to interpretation, standards have been developed in an effort to ensure that all LCAs follow the same framework. The most important among these standards are the ISO1404x series, which contain guidelines, principles and requirements that help LCA practitioners correctly navigate the intricacies that an LCA entails. However it should be noted that there is no single correct method for performing an LCA, and that it only indicates potential environmental impacts, which are not necessarily absolute.

In ISO 14040:2006 - Environmental management – Life cycle assessment – Principles and framework [25], the four phases of an LCA study are listed as:

1. The goal and scope definition phase

2. The inventory analysis phase

3. The impact assessment phase

4. The interpretation phase

In the first phase, the aim of the study is specified and the system boundaries are defined. The functional unit that is to be used in the study is also decided. This phase defines the depth and breadth of the assessment, both in time and geography, and is instrumental in how the obtained results are interpreted [8]. The second

phase consists of constructing the inventory of the system being studied. This can be done in several ways, such as conducting interviews with persons connected to the chosen system or gathering publicly available data through literature or through the internet. In the impact assessment phase the inventory results are analysed to provide further information, and to understand how the different elements in the inventory affect the environment. The fourth and final phase is the interpretation phase, where the results are interpreted and summarized, as well as discussed in the manner determined in the goal and scope of the first phase.
The relationship between these four phases is illustrated in Figure 3.1.

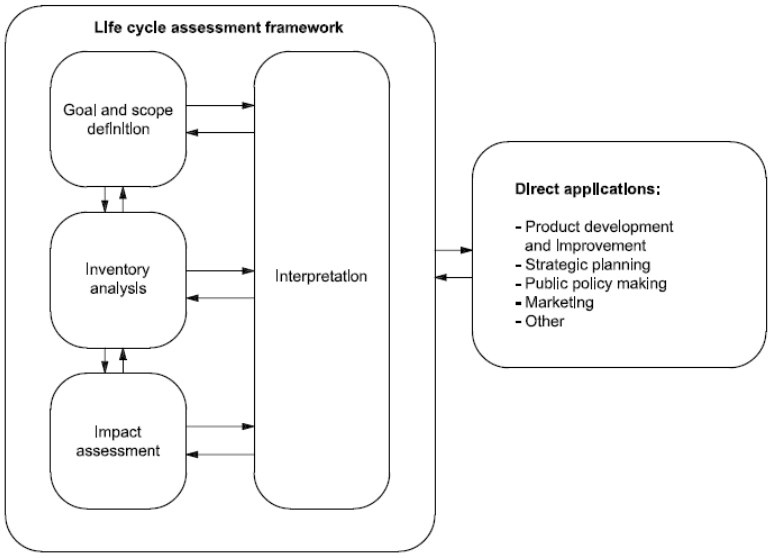

Figure 3.1: The four LCA phases and their relation to one another [25].

As can be seen, this is an iterative process, where the initial goal and scope can change based on the findings in the impact assessment phase and reassessed again.

In this study, the aforementioned ISO 14040 has been used to structure the project work, and the study by Astrup et al. has also been used as a guideline regarding content of goal and scope definitions and descriptions. Astrup et al. has also been used to identify which technical parameters are important and appropriate to include in the goal and scope for a study within this field [8].

3.2 Goal and scope

The goal of the study is to investigate different flue gas cleaning technologies using LCA, to determine the environmental impact of each technology. The different technology scenarios that will be investigated are:

- A combined dry + wet system.

- A dry only system.
- A wet only system.
- A wet only system with recycling of sulfuric acid.
- A wet only system with recycling of calcium chloride.

The intended audience of this are academics and operators of flue gas cleaning systems who are interested in the environmental impact of different technologies. The results of this study will be made public by Chalmers University of Technology and critically reviewed.

3.2.1 Functional unit

The chosen functional unit in this study is treatment of 1 tonne of waste in a WtE facility, since this is more easily described than a complex mixture of gases and particles. This is also a commonly chosen unit in other studies of waste management systems, allowing for easy comparison. The reference flow is therefore 1 tonne of municipal solid waste.

3.2.2 Boundaries

This attributional LCA is essentially a gate-to-grave analysis. This study will not take into account different compositions of waste. While the composition can indeed change the resulting emissions [26], there are too many scenarios to consider for the scope of this study to cover them all. In addition, the software used does not allow for modelling of the waste composition, unlike softwares such as EASEWASTE which is tailored more towards LCA of waste management [2]. While the process used in this study is modeled after Swiss household waste compositions, and the inventory data obtained based on Swedish compositions, it is assumed that these are similar enough that the results should be at least indicative of the correct result. The composition of the Swiss waste used in the model is not specified. The waste composition relevant for the plant this study is based on is 43.4 % residual waste, 26.4 % food waste, 27.2 % newspapers and packaging, 1.5 % garden waste and 1.5 % hazardous waste and electronics.

The emissions resulting from waste collection and transport to the plant will not be considered since all flue gas cleaning methods have this in common. The transport of the input chemicals and the transport of the residues to the landfill will however be considered, since these will differ between different technologies.

The comparison of different NOx abatement technologies, such as SNCR vs SCR, will not be investigated in this study. This is because the performance of these technologies can depends on many factors, and are too complex to cover in this study, as explained in section 2.4.

3.2.2.1 Geographical boundary

The geographical scope of the study is Sweden, since the study is based on data from Swedish plants. However, for the processes and flows where Swedish data was not available, the data will be based on either Swiss or European average values as provided by Ecoinvent.

3.2.2.2 Time horizon

This study aims to study the long-term impacts and burdens of the different flue gas cleaning technologies. As such, the time horizon of the study will be as long as the model used allows, which in this case is 60 000 years. The difficulties of estimating long-term emissions are detailed in section 3.5.

3.2.2.3 Temporal scope

This study is primarily based on current technology, as the inventory of input chemicals and emissions are based on the average data and chemical use of a Swedish WtE-facility for the year 2018. It should however be noted that the base process used in openLCA is based on data from 2010, and many other background processes in Ecoinvent are also older. They should however be applicable to modern cases as well, since this is one of the leading databases available for use in LCA modelling.

3.2.2.4 Allocation

Allocation is built into the Ecoinvent databases, and three different allocation methods can be selected. The allocation cut-off by classification model, or cut-off model in short, allocates primary production of materials to the primary user of that material. Primary producers do not gain any credit for creating recyclable materials, meaning recyclable materials are available without burden to recycling processes. The allocation at the point of substitution model, or APOS, connects processes together into a linked system model, where activities are linked to their respective markets. In this system, all processes or activities resulting in the same product are grouped together, and burdens are allocated proportionally. The third and final system model is called "substitution, consequential, long-term", and can be used for studying the consequences of a market change or demand change in a system.

Because it is easiest to use and offers the user more control, the cut-off model was chosen for this study. In the investigated scenarios where a process resulted in a recyclable product, the cut-off aspect of the model was circumvented by adding the usable output as a negative input. This allows the process to receive the credit for producing a recyclable material.

3.3 Selection of inventory database and software

The LCA software selected to conduct the study is openLCA, due to the fact that it is free and open source, and that it offers several tutorials to teach new users

how to perform assessments. To accompany this software the database Ecoinvent was used, due to its good integration with openLCA and large library of processes. It also has the benefit of having emissions divided into short-term and long-term categories, where long-term covers the time span 100-60 000 years [27]. As mentioned previously, there are other software programs which are better suited to LCAs of waste management systems, such as EASEWASTE. This is however unavailable to those who have not taken a course in how to use the software, which excludes it from the possible choices.

3.4 Selection of impact categories and assessment methods

For this study, six midpoint impact categories have been selected based on relevance to the system and which impact categories are common in similar studies. The climate change impact category is included in practically every MSW management study [23, 24, 4, 21, 28, 29, 19], with acidification being almost as common [20, 24, 4, 21, 28, 29, 19]. Ozone depletion and human health indicators are quite common as well [24, 4, 21, 29, 19]. Respiratory inorganics or particulate matter categories, which were deemed to be of particular interest at the start of the study, are not found as frequently but are still present in multiple relevant studies [24, 29, 19].

The life cycle impact assessment (LCIA) methods for these categories have been selected based on the Product Environmental Footprint (PEF) guidelines by the European Commission Joint Research Centre [30]. These methods are available in openLCA under the name "EF method (adapted)". The six selected midpoint impact categories and their respective units are:

- Climate change [kg CO_2-eq]
- Respiratory inorganics [Disease incidences]
- Acidification [mol H+ eq]
- Ozone depletion [kg CFC-11 eq]
- Cancer human health effects [CTUh]
- Non-cancer human health effects [CTUh]

3.5 Long-term emissions in LCA

The question of how to handle long-term emissions has long been debated amongst LCA practitioners and academics [31]. The problem stems from the fact that landfills release emissions over a very long period of time, at a rate that varies depending

on several factors, such as pH and oxidation rates [32]. This poses a problem since LCA uses time integration, which means all emissions are regarded as occurring as a pulse at one point in time [33], regardless of when they actually occur. Due to this there is no established method for how to model long-term emissions. This has led to many practitioners either:

1. Choosing to only consider emissions that occur withing the near future, such as 100 or 500 years, and disregarding the rest;

2. Integrating over a very long or even infinite length of time to include the entirety of potential emissions;

3. Accounting only for emissions that occur until the concentrations in the leachate from the landfill reach levels equal to the background concentrations.

Each of these has its benefits and downsides. The first approach obviously neglects all emissions occurring over a longer period of time, which usually results in an underestimation of the total impact, but has the benefit of not trying to model a very long, uncertain timeframe. The second approach does take all potential emissions into account, but the result is an overestimation of the actual impact, since not all emissions are occurring immediately. On the other hand, no emissions are ignored, and basing regulations on this method will ensure that results can never be worse than expected. The last approach might have some merit for certain elements, according to the Declaration of Apeldoorn from 2004 [34]. The Declaration argues that the oceans are deficient in essential metals, and as such emissions that fall within or below this "window of essentiality" should be disregarded. However, this would of course only apply to certain metals, and be very region specific.

In general, to ignore emissions that fall below a certain threshold is conflicting with the principle that an LCA should include any and all emissions. There is also the fact that heavy metal emissions do not always follow a linear trend, and emissions might increase after a certain period of time [35].

In regards to the second method, to compensate for the overestimation of emissions, concepts such as positive or negative temporal discounting can be introduced [36]. In these cases future emissions are weighted either lower or higher, depending on the outlook of the LCA practitioner. Weighing future emissions lower (positive temporal discounting) is based in the argument that in the future we will be better at handling waste and pollution, and therefore future emissions will have less impact. On the other hand, negative temporal discounting is based on the argument that in the future we will have higher amounts of pollution than today, and additional pollution should therefore be weighed heavier [32]. However, discounting is rarely applied in LCA studies [36], due to the idea that discounting implies that current problems will be dealt with by future generations. Shifting the burden of emissions, whether they be higher or lower, in time contradicts the purpose of an LCA study, which is to identify problematic processes so that they can be changed or amended.

In addition, the selection of a discount rate to use is very subjective, and will have large effect on the overall results [35].

Evidently none of these methods are a perfect way of handling long-term emissions. Attempts have been made to simulate landfill conditions and leachate rates in order to obtain better data [35], but even then there are many uncertainties that cannot be accounted for. How will technology develop in the future? How will future societies manage landfills? How will the receiving environment around the landfill change? Will an earthquake or other natural disaster hasten the release of emissions, perhaps releasing all remaining pollutants at once? Such things are impossible to predict with any great accuracy, which makes the long time perspective very difficult to deal with.

One method of dealing with this that has been put forward is the concept of stored toxicity. In this technique, all potential emissions occurring after 100 years are grouped into a new category called stored toxicity. This then represents the potential impact that would result from all of the remaining emissions in a landfill being released [31]. Emissions occurring during the first 100 years are treated as normal, together with all the other emissions found in the life cycle of the process. This procedure does not use any discounting for the stored emissions, but instead uses the normal characterisation factors for toxicity categories [35] to convert emissions into category indicators. This proposed method is a step towards distinguishing the two separate time regions of near future where emissions can be monitored, and the far future where the uncertainty is very high. However this method is not without its issues either, as it requires very region-specific data due to soil properties that affect metal toxicity varying depending on location [35], and does not completely escape the large uncertainties of future modeling that are so problematic.

Another potential solution could be time dependent toxicity impacts, where traditional characterisation factors are rejected and emissions are instead calculated as a function of time. One such method is suggested by Shimako et al., where a so called dynamic LCA is developed based on the USEtox model. In this so called dynamic LCA, both the inventory and impact assessment are time dependent [37]. This enables LCA to take into account events that are time-sensitive, and be used as a tool to investigate when, for example, the best time would be to stop or start an activity that affects the environment, such as water use or deployment of a pesticide. It can also distinguish the fate of persistent and non-persistent pollutants separately, which conventional methods cannot do. Nevertheless, this method also has limitations, and more intricate systems such as specific climate factors and interactions with local ecosystems cannot be taken into account. It should also be noted that when this method is applied to longer time spans, the results grow closer and closer to the results of traditional methods [37].

Due to all of these complications and the lack of a clear answer to how to best approach this dilemma, there are some who suggest that no business or policy decisions should be based on the currently available methods of evaluating long-term

emissions without careful consideration [34]. In this study, the long-term approach was deemed the most appropriate since there were several long-term emissions in the system. This method also provides the most complete estimation, albeit an overestimation, which seems fitting for comparing several scenarios, as is done in this study.

4
Life cycle inventory

In this chapter the methods use to calculate the inputs for openLCA as well as the modifications to the used database processes are described. The inventory results are also presented.

The LCA was done using the openLCA software, version 1.10, along with the database Ecoinvent 3.6. To model the processes that are compared in this study, existing processes in Ecoinvent were copied and modified according to either the known inventory data, displayed in section 4.1, or the calculated values, displayed in section 4.1.1. In the cases where no data was obtained, estimations have been made based on experience and the default data available in Ecoinvent.

4.1 Foreground system

In this section all of the inventory data obtained from various sources is shown. The nominal design data can be found in Table A.1 and A.2 in Appendix A.

The gathered inventory of yearly input and output for the Swedish waste incineration plant is presented in Table 4.1. These were received through private communication with operators at the Swedish waste incineration plant [38]. The data is for the year 2018.

Table 4.1: Inventory data for the Swedish waste incineration plant inputs and outputs.

Specification	Input	Output
Waste	200 000 tonne/year	-
$Ca(OH)_2$	2749 tonne/year	-
PAC	45 tonne/year	-
HCl	66 tonne/year	-
NaOH	61 tonne/year	-
Ammonia	1008 tonne/year	-
Slag	-	38 000 tonne/year
Fly ash	-	7500 tonne/year
Condensate	-	60 000 m^3/year
Electricity	-	125 GWh/year
District heat	-	450 GWh/year

The air emission data for the Swedish waste incineration plant is presented in Table 4.2. These were also received through private communication with operators at the plant [38]. All data is based on 2018 mean values. The flue gas flow is 130 000 Nm^3/h, as specified in A.1.

Table 4.2: Inventory data for the Swedish waste incineration plant air emissions.

Emission	Amount
SO_2	4.1 mg/Nm^3
NOx	72 mg/Nm^3
Dust	0.2 mg/Nm^3
HCl	0.0 mg/Nm^3
CO	2.7 mg/Nm^3
TOC	0.6 mg/Nm^3
NH_3	0.0 mg/Nm^3
HF	0.005 mg/Nm^3
Dioxins and furans	0.0018 ng/Nm^3
Hg	0.2 $\mu g/Nm^3$
Cd+Tl	2.2 $\mu g/Nm^3$
Sb+As+Pb+Cr+Co+Cu+Mn+Ni+V	28 $\mu g/Nm^3$

The water emission data for the Swedish waste incineration plant is presented in Table 4.2. These were received through private communication with operators at the plant as well [38]. All data is based on 2018 mean values.

Table 4.3: Inventory data for the Swedish waste incineration plant water emissions.

Emission	Amount
pH	8
NH_4-N	0.08 mg/l
As	<0.0002 mg/l
Pb	<0.0005 mg/l
Cd	<0.0001 mg/l
Co	<0.0005 mg/l
Cu	<0.0005 mg/l
Cr	<0.0005 mg/l
Hg	<0.0001 mg/l
Ni	<0.0005 mg/l
Tl	<0.0001 mg/l
Zn	<0.002 mg/l
Dioxins and furans	0.00005 ng/l

The raw gas measurements are displayed in Table 4.4. Due to the received data being several hundred entries long, only the average values are presented here. The measurements were taken during the year 2018. The data was obtained through private communication [38].

Table 4.4: Inventory data for the Swedish waste incineration plant raw gas measurements.

Measured value	2018 average
SO_2 in flue gas from boiler	685.44 mg/Nm^3
HCl in flue gas from boiler	1296.57 mg/Nm^3
H_2O in flue gas from boiler	18.74 vol%
O_2 in flue gas from boiler	5.08 vol%
CO in flue gas from boiler	9.98 mg/Nm^3
HT steam flow	24.58 kg/s
Dry gas flow	111 935.73 Nm^3/h

The data for production of NaOH is presented in Table 4.5. This inventory data is from private communication with Nouryon [39], and included in this footprint are all major steps from the raw material extraction until the product is ready to deliver. The footprint is based on the production sites in Germany and the Netherlands. In this process, chlorine and hydrogen are also produced, and allocation between these have been allocated on a mass ratio basis. The co-generated steam and process heat has been allocated based on exergy.

Table 4.5: Inventory data for the production of 1 tonne of 50 % caustic soda lye.

Emission	To air [kg]	To water [kg]
Carbon dioxide	340	-
NMVOC	$6.38 \cdot 10^{-2}$	-
Nitrogen dioxide	$8.06 \cdot 10^{-5}$	-
Sulfur dioxide	0.198	-
Mercury	$3.74 \cdot 10^{-6}$	$2.22 \cdot 10^{-5}$
Chloride	-	8.86
Sulfate	-	0.354

The data for the production of H_2O_2 is presented in Table 4.6. The data is from a publicly available EPD of Nouryon's production process [40]. In this inventory, all major processes between natural resource extraction and transport to the customer have been included. In the process, waste heat is produced and sold, and the allocation between heat and H_2O_2 has been done on an economical basis. The data was gathered during the year 2016 from plants in Sweden and Norway.

Table 4.6: Inventory data for the production of 1 tonne of hydrogen peroxide, 100 % concentration.

Emission	To air [kg]	To water [kg]
Fossil CO_2	464	-
SO_2	0.29	-
CH_4	1.6	-
NOx	0.57	-
NMVOC	0.52	-
CO	0.24	-
Particulates	0.22	-
N	-	$1.6{\cdot}10^{-4}$
P	-	$3.83{\cdot}10^{-3}$
COD	-	2.0
BOD	-	0.03

4.1.1 Calculations

The first step of the calculations was to recalculate all inventory data, given in Tables 4.1 to 4.3, into kg per kg waste, since this is the reference amount used in openLCA.

The data in Table 4.1 was easily converted by simply dividing each input or output with the annual waste input, 200 000 tonnes/year. The air emissions per kg waste were calculated using the flue gas flow rate, 130 000 Nm^3/h, and a assumed operational time of 8000 hours/year. An example is given, using the annual SO_2-emissions to air, in Equation 4.1.

$$\frac{\frac{4.1 \text{ mg } SO_2}{Nm^3} \cdot \frac{130000 \text{ } Nm^3}{\text{hour}} \cdot \frac{8000 \text{ hours}}{\text{year}}}{\frac{1{\cdot}10^9 \text{ mg}}{\text{tonne}} \cdot \frac{200000 \text{ tonne waste}}{\text{year}}} = \frac{2.132 \cdot 10^{-5} \text{ kg } SO_2 \text{ to air}}{\text{kg waste}} \tag{4.1}$$

Similarly, the emissions to water were calculated by using the annual condensate output, 60 000 m^3/year.

4.1.1.1 Dry process scenario

For the dry system, the same amount of dry input chemicals as for the standard operating scenario of dry and wet combined was used, as specified in Table 4.1. However, since dry systems do not reach as high efficiencies due to kinetics, the emissions are modeled according to the data in Table A.2 instead, which are the nominal emission values for the baghouse filter outlet.

Since the dry system lacks a wet scrubber and thus a condensing step, less heat is generated for the district heating. This is adjusted using values for how much heat the wet scrubber contributes to the total annual heat production [11].

4.1.1.2 Wet process scenario

For the scenario with wet scrubbing only, the chemical consumption was calculated in the same way as in the dry scenario, but with the reactions in Equation 2.3 and 2.6 instead. The amount of chemicals needed for the water treatment, where HCl (aq) is neutralized, was calculated with Equations 2.4 and 2.5. Here it was assumed that the water stream diverted to the cleaning process contains 5 weight% HCl, which is neutralized using $CaCO_3$ until the pH of the solution reaches a value of 1, and then $Ca(OH)_2$ is used until the pH of the solution reaches 9. To achieve this the water stream was increased until the HCl that needs to be treated constitutes 5 %. The current concentration was then calculated by dividing the moles of HCl by the volume of the 5 % stream. The difference in concentration between that and pH=1 equals the needed amount of $CaCO_3$, and the same was then done using $Ca(OH)_2$ and pH=9. It was assumed that since HCl is a strong acid it is completely ionized in the water. It was also assumed that the fly ash is separated using a baghouse filter which does not require any extra input of electricity. The extra amount of CO_2 produced by this reaction was also accounted for.

4.1.1.3 Sulfuric acid production scenario

If SO_2 is treated with H_2O_2 instead of NaOH, sulfuric acid is produced, which could potentially be recycled in some other process, which is what was investigated in this scenario. The H_2O_2 demand was calculated stoichiometrically in the same way as in the previous scenarios, but with Equation 2.7 being the relevant reaction instead. The produced amount of H_2SO_4 was also calculated from the stoichiometry. However, the produced acid is only at 20 % concentration [11], and needs to be increased to 98,3 % which is the concentration of commercial grade H_2SO_4. This can be achieved through evaporation of the solution using 20 bar(g) steam, with a consumption rate of 1.5 kg steam per kg evaporated solute [11]. The mass flow of commercial grade H_2SO_4 was calculated by reducing the flow by a factor of $\frac{20}{98.3}$. The mass that needs to be evaporated was then calculated as the difference between the old mass flow and the new, and the required steam as 1.5 times this amount. Instead of adding this steam as a new required input, it was taken from the WtE plant, since it produces steam. This consumption of steam led to a net decrease in output of electricity and heat, which was calculated using the heat of evaporation of the steam in kJ/kg [41] multiplied by the calculated mass flow. However, as one kW of steam does not equal 1 kW of electricity due to the efficiency of the condensing turbine, the ratio of power to heat was for the Swedish waste incineration plant was calculated. Since the plant only produces heat for 5000 hours per year, 5000 hours of electricity production was compared to 5000 hours of heat production to determine the electrical efficiency. This was then used to determine the output reduction when using steam to increase the H_2SO_4 concentration.

4.1.1.4 Calcium chloride production scenario

The amount of calcium chloride produced was calculated using the reactions described in Equations 2.4 and 2.5, with the same calculation steps as in section 4.1.1.2. Once the produced $CaCl_2$ amount had been determined, the weight percent was determined by comparing the mass flow with the water stream mentioned in section 4.1.1.2. Commercially $CaCl_2$ is sold at 40 % concentration [11], and the obtained solution was only 7.5 %, so the concentration was increased using the same method of evaporating with steam as in section 4.1.1.3.

4.2 Background systems

In this section the processes used to model the background systems and the changes made to processes obtained from Ecoinvent via openLCA are described, with the processes providing the chemicals listed in Table 4.1 divided into their respective production chains of upstream processes. In all processes where electricity is used the provider has been changed to the Swedish market for the relevant voltage.

The production chains for NaOH and H_2O_2 have not been modeled since the obtained inventory data covers the entire process from cradle to gate. Therefore these processes were created manually and emissions added according to Tables 4.5 and 4.6. The production chains of the recycled materials (H_2SO_4 and $CaCl_2$) were also modeled to give a fair estimate of the potential impact reduction.

4.2.1 Waste incineration

The process used as a base scenario for the WtE facility is based on a Swiss technology mix encountered in 2010. It is stated to be well applicable to modern incineration practices in Europe, North America and Japan, and its recommended use is for average municipal waste mixtures. It is modelled for a lower heating value of 11.7 MJ/kg, which corresponds well with the inventoried data for the Swedish plant which is 11 MJ/kg. It should be noted that the long-term emissions of the slag and residues going to landfill are modeled as elemental flows out of this process.

The providers of all input chemicals detailed in Table 4.1 has been changed to the modified processes, as well as the input amount. The input of $FeCl_3$, polyelectrolyte and the heavy metal precipitation agent TMT15 have been removed, as they were modeled for use in a water treatment method that is not used at the Swedish waste incineration plant. Likewise, the input of chromium oxide flakes and TiO_2 have been removed since they were for use in a SCR, while the Swedish plant uses a SNCR. An input of GAC has been added, as a stand-in for PAC since there is no process producing PAC in Ecoinvent. The amount of all emissions which are included in Table 4.2 and 4.3 have been changed. The process burden of constructing landfill space for APC residues has been changed to reflect the correct amount, and transport from

the Swedish waste incineration plant to the landfill has been added. Lastly, negative inputs of heat and electricity has been added to give credit for the avoided impacts of producing these. Due to the selected allocation method not giving credit for heat or electricity produced by waste incineration, the carbon dioxide emissions for these generated utilities has been added manually. The amount of CO_2 saved by heat generation has been set to 44 g CO_2/kWh heat [42], and the value for the electricity to 50 g CO_2/kWh [43]. The electricity value is based on the Nordic electricity mix, since Sweden exports and imports electricity from its neighboring countries which should be accounted for.

4.2.2 Quicklime production chain

The quicklime production chain starts with the quarrying of calcite. The process accounts for the land use, dust emissions, blasting, transports within the quarry and heating for the quarry infrastructure. The next step in the process is the production of crushed and washed limestone. The process includes both infrastructure and equipment use, with an expected lifespan of 25 years. The original process mixed limestone from several countries, but was changed to include only limestone originating from the modified calcite quarrying process described above. The crushed limestone is then taken to be processed into quicklime, which consists of two steps. The first step gives loose quicklime in pieces through calcination of crushed limestone. In this process heavy fuel oil is used, which was changed to natural gas instead, based on contact with personell at the Swedish company SMA Mineral [44]. In the second step the quicklime in pieces is milled into a finer form. This is then transported to the waste incineration plant for use.

In several of the processes it was assumed that half of the electricity demand would be provided by a nearby hydro plant. This might be accurate in some cases, but not all, and since the production mix of Sweden already consists of a very high percentage of hydroelectric power this was changed to be provided by the regular Swedish network.

4.2.3 Hydrochloric acid production chain

For the production of hydrochloric acid a model of the Mannheim process was selected, based on a report indicating that the Swedish chemical company Kemira co-produces hydrochloric acid and sodium sulfate [45], which is what is done in the Mannheim process. The material input for this process is sulfuric acid and sodium chloride powder. While consumption of raw materials, chemicals and emissions are taken into account, production of solid wastes are omitted. The electricity input was changed to a Swedish market provider and the transport of sulfuric acid was removed, since Kemira produces sulfuric acid as well.

The upstream processes providing the raw materials were modified as well. The production method of sulfuric acid was assumed to be the contact method, since this is the most common current method [46]. In this process the only raw material

input is sulfur, and so the only change made was the electricity provider. For the sodium chloride powder, it was assumed that it is produced by underground and solution mining. The input of milled quicklime was changed to the process described in section 4.2.2. Lastly, transport from the production site to the Swedish waste incineration plant was added.

4.2.4 Ammonia production

The ammonia production process method was assumed to be through steam reforming since this is by far the most prevalent method. The original process used natural gas from many different sources, so these streams were summed up and changed to the Swedish market for natural gas.

4.2.5 GAC production

Since the Ecoinvent database seems to lack an entry for powdered activated carbon, granular activated carbon (GAC) was used instead. This process produces GAC from hard coal, which is provided from a European market provider. This market represents the different sources of hard coal import into Europe as well as the average transport distances. Since Sweden does not have any coal mines [47], this was assumed to be accurate.

4.2.6 Calcium carbonate production chain

The calcium carbonate process results in precipitated calcium carbonate, which is what is used industrially. The model is based on stoichiometry and literature data, with inputs of heat, electricity, water and nitrogen being based on industry averages. One of the inputs to the calcium carbonate production is packed hydrated lime. In this process the source of the required heating was changed from the Swiss market to the European one. The production of the packaging that is used to pack the hydrated lime was also changed: the kraft paper now comes from a kraft paper production that uses Swedish sustainable forestry as well as sulfuric acid and milled quicklime from the processes mentioned in previous sections. The hydrated lime that is packed has also been changed to receive loose quicklime in pieces from these aforementioned processes. In addition, both the hydrated lime process and the process supplying packaging make the assumption mentioned in section 4.2.2 about nearby hydro plant supplying 50 % of the electricity. This has therefore been changed.

4.2.7 Calcium chloride production chain

The production chain for $CaCl_2$ is in many ways similar to the calcium carbonate chain. The chain starts with the quarry operation described in section 4.2.2. The limestone is then delivered to a crusher, and from there it goes to a mill. The Ecoinvent processes for both of these include all stages of each process as well as heating and energy consumption. The milled lime is then packed, using the same provider of packing as in the $CaCO_3$ chain, and lastly delivered to the $CaCl_2$ production

plant. The production process is modeled after the Solvay process, which is the most common method of sodium carbonate production in most of the world, and which produces calcium chloride as a byproduct. The ammonia input to this process has also been changed to the modified ammonia production process described in section 4.2.4. It should be noted that the majority of the ammonia used in the process can be regained with only minor losses, and that the consumption is based on a data survey from 2002.

The database inventory for this process is based on a German report on the best available technology, with some modifications. The dataset accounts for raw material use, auxiliaries, infrastructure, energy and land use. Emissions to air and water are also considered, as well as waste generation.

4.2.8 Transport

Usually an input in a process in Ecoinvent comes from a market provider, which mixes the average sources of a product flow and includes an average transport distance. Since this study focused on a specific region, market processes were for the most part replaced with the direct producer of the required input. The exception would be if there was a Swedish market provider available. Since replacing the market provider also removed the calculated average transport, this was added in manually, based on estimations of the distance between the Swedish waste incineration plant and Swedish suppliers. The fly ash was assumed to be transported to the landfill in the former limestone quarry on Langøya island near Oslo, Norway. The slag is transported to a local recycling company that recovers metal scrap and gravel from the slag which can then be reused, and the non-recoverable portions are then landfilled as well. The transport was assumed to be a lorry with a 16-32 tonne carrying capacity, and meeting the emission standard EURO V. This emission standard was chosen based on a report from the Swedish traffic agency Trafikanalys [48] from 2016, which states that most of new heavy cargo trucks are EURO VI compliant. Since trucks are expected to have a lifetime of around 10 years [48], it seems reasonable to assume that the trucks used for transport would be at least EURO V. A table of the assumed transport distances is presented in Table 4.7.

Table 4.7: Assumed transport distances within the modeled processes. Every entry is transport by truck unless otherwise specified.

Object transported	**Distance [km]**
Limestone to waste incineration plant	29
NaOH to waste incineration plant	258
GAC to waste incineration plant	544
$CaCO_3$ to waste incineration plant	29
HCl to waste incineration plant	11.5
H_2O_2 to waste incineration plant	258
Ammonia to waste incineration plant	10.4
Residues to landfill	449 + 14 by barge

4.3 Uncertainty analysis

The uncertainty analysis was performed using openLCA's built in function for Monte Carlo-simulation. This is a very common method for estimating uncertainty in LCA analysis [49], based on uncertainty values for each flow or species in the system. In Ecoinvent all flows have inherent uncertainty values based on the data quality, and a distribution type which can be changed. In this study the uncertainty analysis was done using a logarithmic uncertainty distribution.

5

Life cycle impact assessment and discussion

In this chapter the results for each selected impact category and process case is shown and discussed, as well as the top contributing processes for each category. The results have been complied into bar charts using Microsoft Excel for easier comparison, and are also presented in normalized form in section 5.6. The results of the uncertainty and sensitivity analysis are also included. The results have been calculated **including long-term emissions**, which means they are probably overestimated as discussed previously. The reasons for selecting this approach are discussed in section 3.5.

In the contribution analysis, the impact from direct emissions is displayed as well as the four highest contributing processes. For the processes other than the direct impacts, the contribution is calculated based on all upstream processes, which means all contributing processes are inputs into the WtE-facility, one "level" upstream. The negative contributions are already included in the final result presented in the bar chart and should not be reapplied by the reader. It should also be noted that the percentages in the contribution tables are based on the gross emitted carbon before consideration of offsets caused by system expansion.

5.1 Climate change

The impact assessment model used to evaluate climate change is the global warming potentials of the 2013 IPCC assessment report [30]. The results are shown in Figure 5.1. For this category, a graph showing the sum of the indirect emissions is also presented in Figure 5.2.

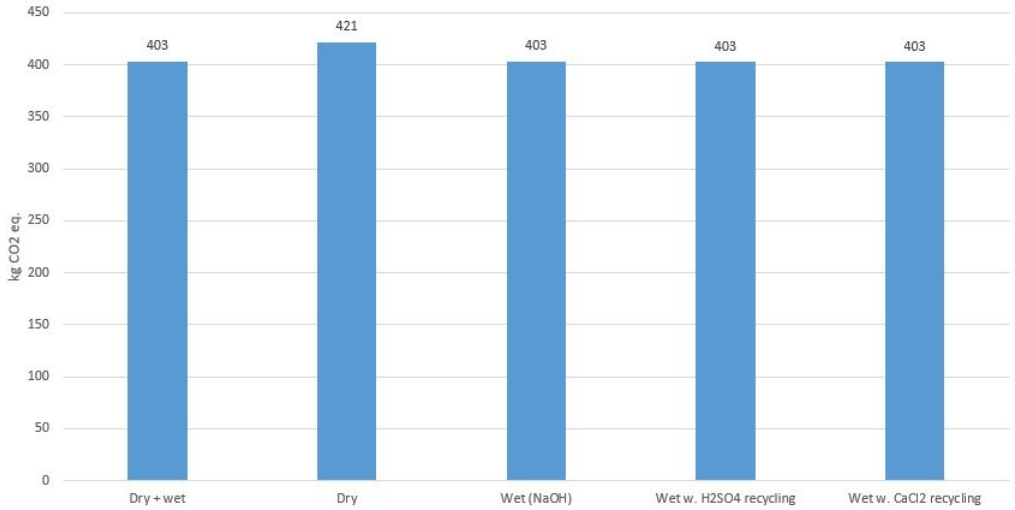

Figure 5.1: The impact analysis results for the climate change impact category.

The climate change impact is quite similar for most of the scenarios, with all scenarios except the dry one having almost the same result. The reason the dry scenario has a higher impact originates for the most part in that without a flue gas condensation stage, which is found only in the wet system, less district heat is generated and therefore the dry system does not get as much credit for saved CO_2.

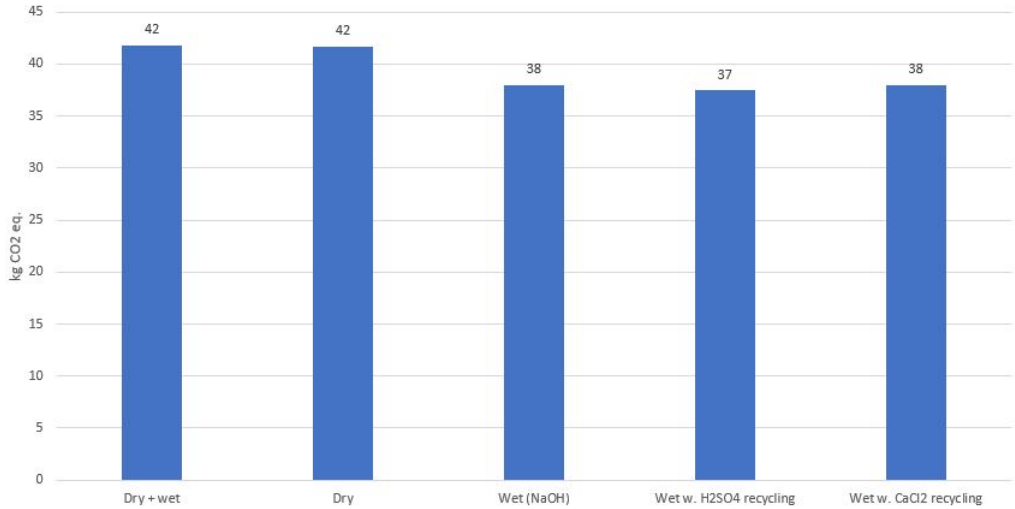

Figure 5.2: The impact analysis results of the indirect emissions for the climate change impact category.

As for the emissions not originating from the waste incineration plant, the three wet alternatives have a lower impact than the dry and dry + wet systems, due to $CaCO_3$ having less impact as seen in Table 5.1. Of the wet systems, the sulfuric acid recycling case is the lowest by 1 kg CO_2-equivalent.

5.1.1 Process contributions

For all scenarios, the highest impact stemmed from direct emissions from the incineration plant. Following this, the largest contributions came from the production of quicklime or calcium carbonate, production of ammonia, heating requirements and lastly the construction of the facility itself. The exact percentages are shown in Table 5.1. The highest contributing process being the direct emissions is not surprising since a waste incineration plant obviously releases a lot of CO_2. The high contribution from quicklime production can be explained by carbon dioxide being formed in the calcination process. As for the calcium carbonate, lime is used in this process which leads to a high contribution as well. It can be noted that the recycling of H_2SO_4 and $CaCl_2$ does not have a noticeable effect on the overall CO_2-emissions. The generation of usable heat and power does however decrease the overall impact by a significant amount.

Table 5.1: Impact contributions of each process to the climate change category.

Contributing process	**Dry+wet**	**Dry**	**Wet**	**Wet+H_2SO_4**	**Wet+$CaCl_2$**
Direct emissions	89.6 %	90.1 %	90.6 %	90.7 %	90.6 %
$Ca(OH)_2$ prod.	3.8 %	3.6 %	-	-	-
$CaCO_3$ prod.	-	-	2.5 %	2.5 %	2.5 %
Ammonia prod.	2.2 %	2.1 %	2.2 %	2.2 %	2.2 %
Heating requirements	1.7 %	1.6 %	1.7 %	1.7 %	1.7 %
MSWI facility constr.	1.0 %	1.0 %	1.0 %	1.0 %	1.0 %
Recycling impact	-	-	-	-0.0 %	-0.9 %
District heating impact	-24.5 %	-19.2 %	-24.6 %	-24.6 %	-23.9 %
Power generation impact	-7.8 %	-7.4 %	-7.8 %	-7.8 %	-7.6 %
Other	1.7 %	1.6 %	2.0 %	1.9 %	2.0 %

5.2 Respiratory inorganics

The EF impact assessment model uses the UNEP model from 2016 to evaluate the impact of particulate matter [30]. The results are shown in Figure 5.3.

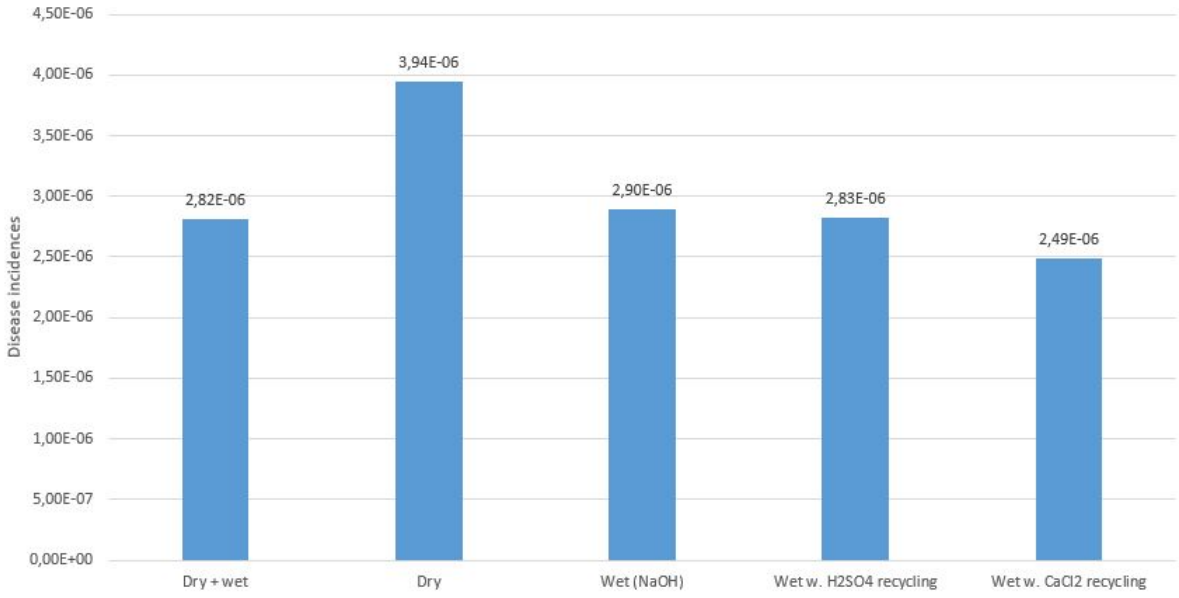

Figure 5.3: The impact analysis results for the respiratory inorganics impact category.

As in the previous category, the dry scenario has the highest impact. The difference between the other scenarios are however a bit more apparent, with both recycling cases having a degree of improvement over the standard wet scenario. There is also a difference between the wet system and the combined system, with the combined being lower. It is also interesting to note that the benefits of recycling H_2SO_4 is not enough to make the wet system have a lower impact than the combined system.

5.2.1 Process contributions

The direct emissions from the WtE facility play a lesser role in this impact category, though still a significant one. As the results in Figure 5.3 indicate, the benefits of recycling are a bit more clear in this case, showing a decreased impact contribution for H_2SO_4 and an even more significant decrease for the $CaCl_2$ scenario. The highest upstream process contribution comes from the production of ammonia, due to emitting both particulates and nitrogen oxides. This is also the reason the $CaCl_2$-recycling is so impactful: ammonia is an input when making in the calcium chloride production process.

The construction of the facility and the slag storage will naturally involve result in a lot of particle emissions as well. The transport by truck is also a major factor in the overall impact, being responsible for 5-8 % of the overall impact.

Table 5.2: Impact contributions of each process to the respiratory inorganics category.

Contributing process	Dry+wet	Dry	Wet	Wet+H_2SO_4	Wet+$CaCl_2$
Direct emissions	49.4 %	63.9 %	48.0 %	46.9 %	39.5 %
Ammonia prod.	14.5 %	10.4 %	14.1 %	14.5 %	16.4 %
MSWI facility constr.	10.9 %	7.8 %	10.6 %	10.9 %	12.3 %
Truck transport	7.2 %	5.1 %	7.3 %	7.2 %	8.5 %
Slag storage constr.	4.7 %	3.3 %	4.5 %	4.6 %	5.3 %
Recycling impact	-	-	-	-2.2 %	-16.4 %
Other	13.4 %	9.5 %	15.5 %	15.9 %	18.1 %

5.3 Acidification

The impact assessment method used to evaluate acidification in the EF model is the method by Seppälä et al. and Posch et al. [30]. The results are shown in Figure 5.4.

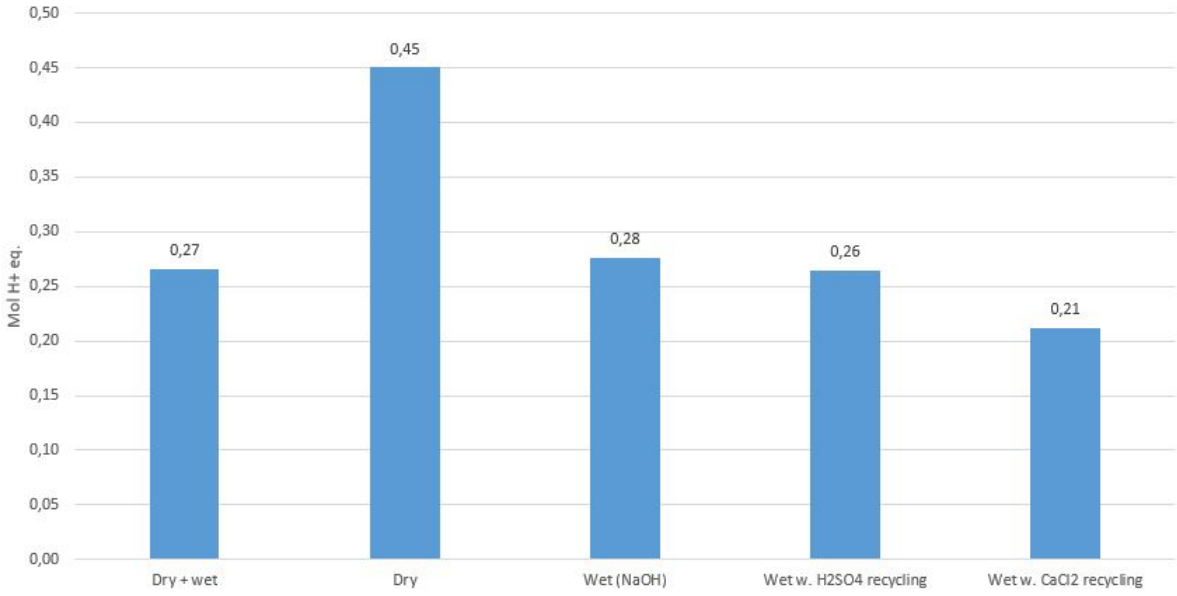

Figure 5.4: The impact analysis results for the acidification impact category.

As with the other indicators, the dry system has the highest impact, followed by the wet system. It should be noted however that the difference between the wet, dry + wet and H_2SO_4 recycling cases are very small, although in this category the H_2SO_4 has a lower impact than the combined system. The $CaCl_2$ recycling scenario has the lowest impact, as in the previous category, by a quite significant margin.

5.3.1 Process contributions

As seen in Table 5.3, the direct impacts of the wet scenario is actually lower than for the dry + wet scenario, but due to the $CaCO_3$ production chain the overall impact

category result is slightly higher. As with the respiratory inorganics category, the ammonia production has the highest contribution, which explains why the $CaCl_2$ recycling exhibits such a significant impact decrease. The benefit of recycling H_2SO_4 is larger than in the two previous categories but still quite small compared to the $CaCl_2$ scenario.

The construction of the plant itself has a slightly lower impact than the ammonia and calcium carbonate production chains, which makes it the third most contributing process. The production of activated carbon and $Ca(OH)_2$ are also important contributors.

Table 5.3: Impact contributions of each process to the acidification category.

Contributing process	**Dry+wet**	**Dry**	**Wet**	**Wet+H_2SO_4**	**Wet+$CaCl_2$**
Direct emissions	61.8 %	77.6 %	59.6 %	58.0 %	47.3 %
Ammonia prod.	7.6 %	4.5 %	7.3 %	7.6 %	9.5 %
$CaCO_3$ prod.	-	-	7.4 %	7.7 %	9.7 %
MSWI facility constr.	6.3 %	3.7 %	6.1 %	6.3 %	7.9 %
GAC prod.	4.8 %	2.8 %	4.6 %	4.8 %	6.0 %
$Ca(OH)_2$ prod.	4.6 %	2.7 %	-	-	-
Recycling impact	-	-	-	-4.0 %	-30.3 %
Other	14.9 %	8.8 %	15.0 %	15.5 %	19.6 %

5.4 Ozone depletion

For the the ozone depletion impact assessment the EF model uses the characterisation factors ozone-depleting substances as suggested by the World Meteorological Organisation in 2014 [30]. The results are presented in Figure 5.5.

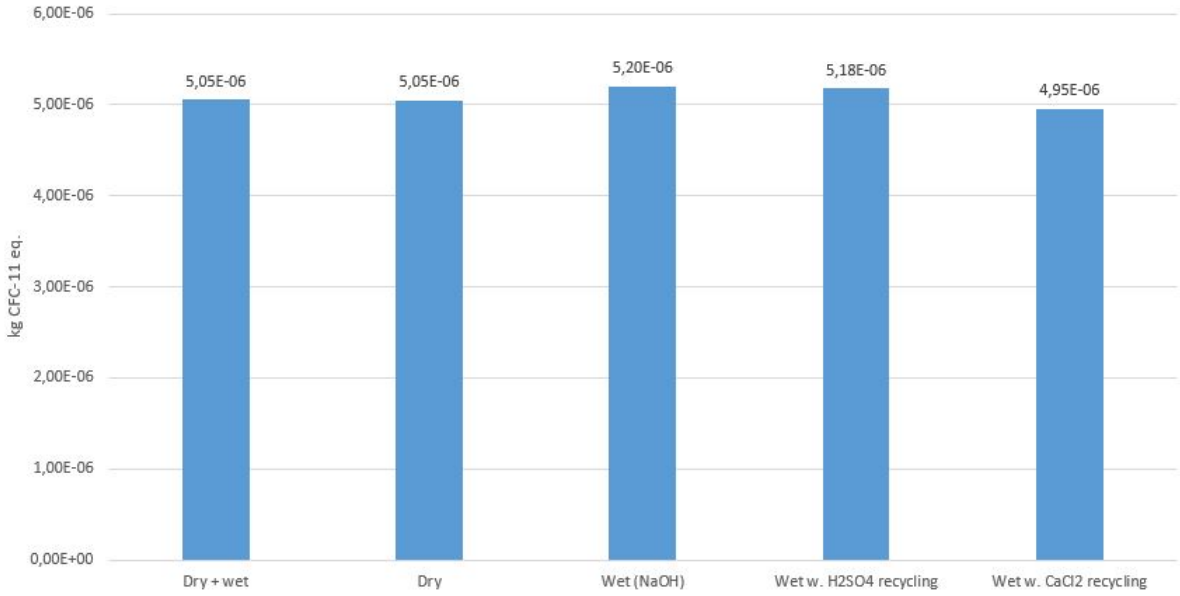

Figure 5.5: The impact analysis results for the ozone depletion impact category.

As with all previous indicators, $CaCl_2$-recycling scenario has the lowest impact, with the two dry scenarios in the middle and the two other wet scenarios having the highest impact. As shown in Table 5.4, this stems largely from the difference in contribution of calcium carbonate compared to quicklime.

5.4.1 Process contributions

Contrary to the previous categories, the direct emissions from the waste incineration process do not contribute to the ozone depletion category at all. In the two recycling cases, the waste incineration plant actually has a negative contribution, due to the "saved" environmental load of producing H_2SO_4 and $CaCl_2$. The ammonia production is once again the most contributing process, followed by heating and transport. The two calcium-based inputs are also significantly contributing to the overall emissions. In this category the H_2SO_4 recycling has a practically negligible effect.

Table 5.4: Impact contributions of each process to the ozone depletion category.

Contributing process	Dry+wet	Dry	Wet	Wet+H_2SO_4	Wet+$CaCl_2$
Direct emissions	0.0 %	0.0 %	0.0 %	-0.1 %	-5.1 %
Ammonia prod.	34.8 %	34.8 %	33.8 %	34.0 %	35.5 %
Heating requirements	19.9 %	19.9 %	19.3 %	19.4 %	20.3 %
Truck transport	13.1 %	13.1	13.2 %	12.8 %	13.9 %
$CaCO_3$ prod.	-	-	11.5 %	11.6 %	12.1 %
$Ca(OH)_2$ prod.	9.3 %	9.3 %	-	-	-
Recycling impact	-	-	-	-0.1 %	-5.1 %
Other	22.9 %	22.9 %	22.3 %	22.4 %	23.4 %

5.5 Human toxicity

The EF LCIA model uses the same method for evaluation of both cancerous and non-cancerous human health effects [30], namely the USEtox 2.1 model developed by Rosenbaum et al. [50]. The impact results for both categories are displayed in Figure 5.6.

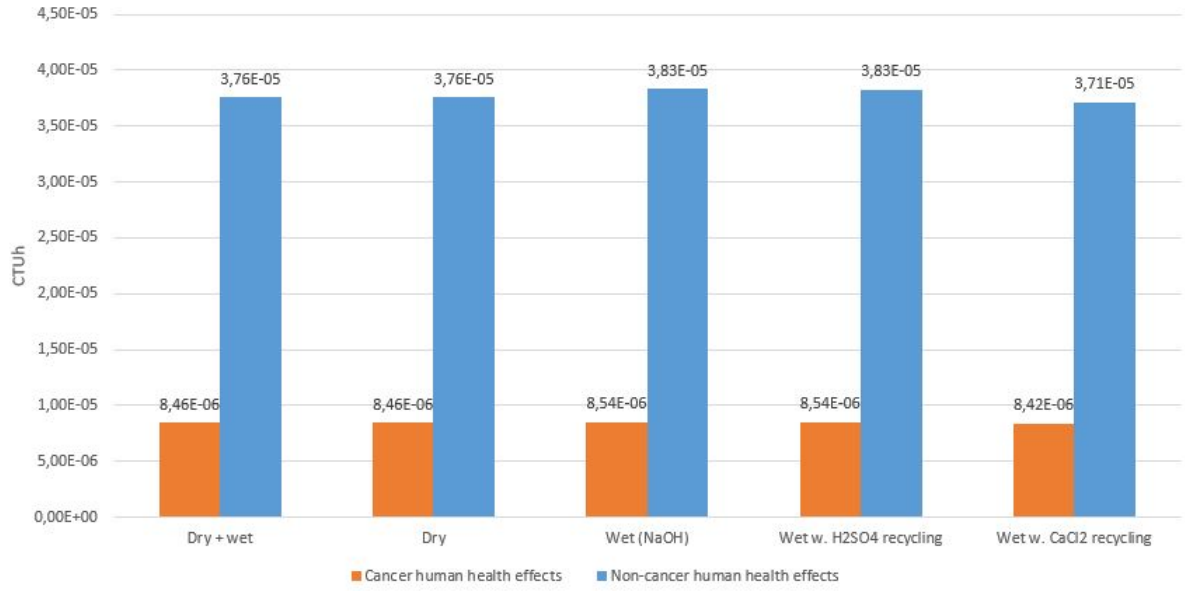

Figure 5.6: The impact analysis results for the human health impact category, both cancerous and non-cancerous.

The results for both the cancerous and non-cancerous human health-categories follow the same overall trend as the ozone depletion category. A result of note is that the sulfuric acid recycling does not seem to have any noticeable effect on the human health impact of the system, as the results are the same as for the wet only scenario.

5.5.1 Contribution analysis

As in the climate change category, the majority of human health impacts originate from the incineration plant, which is probably why there is so little difference between the cases. The difference here would be that this large contribution presumably stems from long-term landfill emissions instead of emissions to air or water from the plant. Of the contributing processes, the facility construction and the treatment of hydrated waste cement have the highest contributions in both categories. The facility construction will be very similar for all cases and has not been adjusted between scenarios in this study. The hydrated waste cement process refers to the method used in landfills of solidifying residues using cement. The making of this cement is what gives this process its relatively large impact.

Table 5.5: Impact contributions of each process to the human health, cancerous category.

Contributing process	**Dry+wet**	**Dry**	**Wet**	**Wet+H_2SO_4**	**Wet+$CaCl_2$**
Direct emissions	90.3 %	90.3 %	89.4 %	89.4 %	89.3 %
MSWI facility constr.	4.2 %	4.2 %	4.2 %	4.2 %	4.2 %
Hydrated waste cement	3.9 %	3.9 %	3.9 %	3.9 %	4.0 %
Ammonia prod.	0.5 %	0.5 %	0.5 %	0.5 %	0.5 %
$CaCO_3$ prod.	-	-	1.0 %	1.0 %	1.0 %
Truck transport	0.3 %	0.3 %	-	-	-
Recycling impact	-	-	-	-0.1 %	-1.4 %
Other	0.7 %	0.7 %	0.9 %	0.9 %	0.9 %

For the cancerous human health effects, only the $CaCO_3$ has a contribution higher than one percent besides the aforementioned facility and cement contributions. For the non-cancerous, all input processes have a contribution higher than 1 %, and all except the MSWI facility contribute more to the overall impact compared to the cancerous category. The cement and facility contributions have also switched order, which means the hydrated waste cement production contributes more to non-cancerous impacts and the facility construction contributes more to the cancerous human health effect indicator. For both categories, the recycling seems to have a relatively small impact.

Table 5.6: Impact contributions of each process to the human health, non-cancerous category.

Contributing process	**Dry+wet**	**Dry**	**Wet**	**Wet+H_2SO_4**	**Wet+$CaCl_2$**
Direct emissions	88.3 %	88.3 %	86.5 %	86.6 %	86.1 %
Hydrated waste cement	5.6 %	5.6 %	5.4 %	5.5 %	5.6 %
MSWI facility constr.	2.2 %	2.2 %	2.2 %	2.2 %	2.2 %
Ammonia prod.	1.3 %	1.3 %	1.2 %	1.2 %	1.3 %
$CaCO_3$ prod.	-	-	2.1 %	2.1 %	2.2 %
Truck transport	1.2 %	1.2 %	-	-	-
Recycling impact	-	-	-	-0.1 %	-3.3 %
Other	1.5 %	1.5 %	2.6 %	2.5 %	2.7 %

5.6 Normalized results

The results for each category are also presented in normalized form in Figure 5.7. The weighting set used is the global normalization factors as recommended for use with the EF method by the European Commission. The factors are based on data gathered during 2010 at a global scale [51]. The results indicate the impact of the studied system relative to the global impact, and can be used to estimate the relevance of the impacts in the the different categories. The results are given in person equivalents (PE) per functional unit, which in this case is 1 tonne of municipal solid waste.

Based on Figure 5.7, the largest relative impacts are in the two human health effect categories, followed by the climate change category. The impact in respiratory inorganics, acidification and ozone depletion are all very low in comparison. It should however be noted that while the impact of the human health effects are quite high, these are the two categories with the lowest reliability due to incomplete supporting inventories [51].

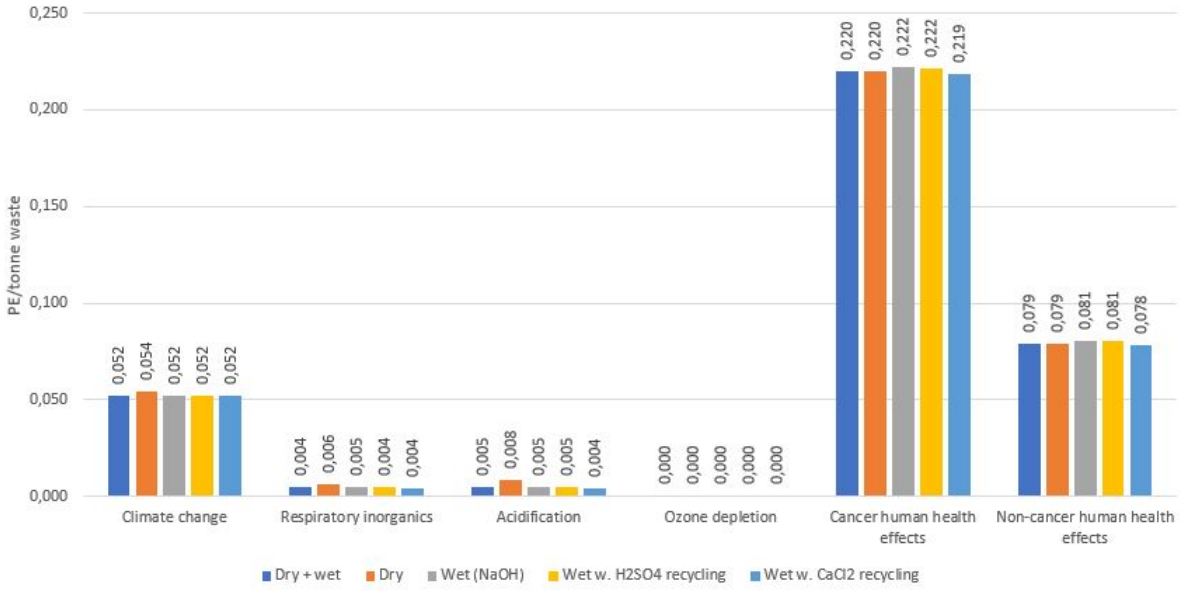

Figure 5.7: The impact analysis results for the human health impact category, both cancerous and non-cancerous.

5.7 Uncertainty analysis

As described in section 4.3, the uncertainty analysis was performed through a Monte Carlo simulation, which was run for 1000 iterations. There has been much debate regarding how many iterations should be performed to obtain good results, with some claiming the more iterations the better [52] and others stating that more iterations do not actually lead to better accuracy, only better precision [53]. Due to this, a thousand iterations was selected as it should provide a good enough sample size while also not taking an extreme amount of time to simulate. Even so, 1000 iterations took an unexpectedly long time to process, and therefore was only performed for the base scenario with a dry + wet system due to time constraints. However, as the contribution analysis shows, many of the major contributing processes are the same for all scenarios, and so even an analysis of only one scenario should give an indication of the uncertainty of all of them.

The negative results in some of the categories can be explained through processes with negative inputs, such as the generated electricity or heat. Since the Monte Carlo simulation picks a value from within the uncertainty range for all flows, the overall impact could in some iterations be very low, and lesser than the negative input. This would result in a overall negative impact, in the same manner as in Figure 5.5.

Table 5.7: The LCIA result, Monte Carlo mean value, standard deviation and relative standard deviation for the dry + wet scenario.

Impact category	LCIA result	Mean	SD	RSD
Climate change	403	431	96	22.3 %
Respiratory inorganics	$2.82 \cdot 10^{-6}$	$2.74 \cdot 10^{-6}$	$1.20 \cdot 10^{-6}$	43.9 %
Acidification	0.27	0.44	0.30	67.8 %
Ozone depletion	$5.05 \cdot 10^{-6}$	$8.51 \cdot 10^{-6}$	$3.00 \cdot 10^{-6}$	35.2 %
Cancerous health eff.	$8.46 \cdot 10^{-6}$	$1.06 \cdot 10^{-5}$	$1.13 \cdot 10^{-5}$	107.0 %
Non-cancerous health eff.	$3.76 \cdot 10^{-5}$	$8.68 \cdot 10^{-5}$	$8.54 \cdot 10^{-5}$	101.6 %

The summarized results of the Monte Carlo analysis is presented in Table 5.7. The table contains the calculated result of the dry + wet scenario, as well as the mean value of all the Monte Carlo iterations, the standard deviation (SD) and the relative standard deviation (RSD), which is the SD divided by the mean. The detailed results for each of the selected impact categories are shown in Figures 5.8 through 5.13.

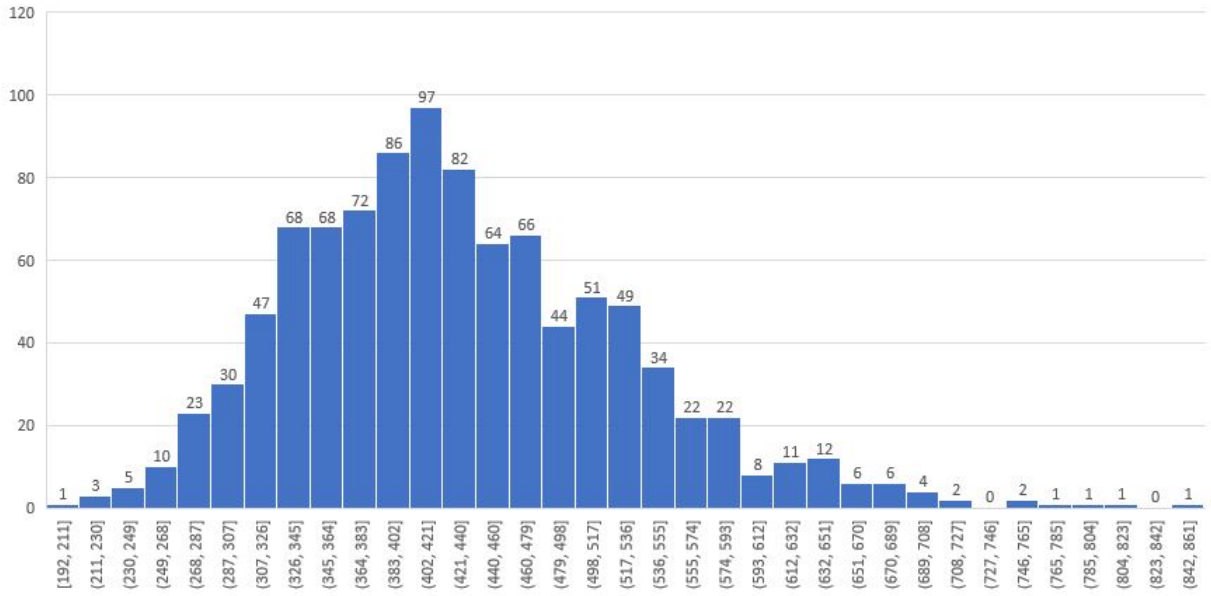

Figure 5.8: The results of the Monte Carlo analysis of the dry + wet system for the climate change impact category.

The probability distribution for the climate change impact category is quite even. As shown in Table 5.7, the climate change category has the lowest RSD, and it can also be noted that the calculated result is found in the interval with the highest probability. The mean is a little higher than the calculated result due to multiple higer-value outliers.

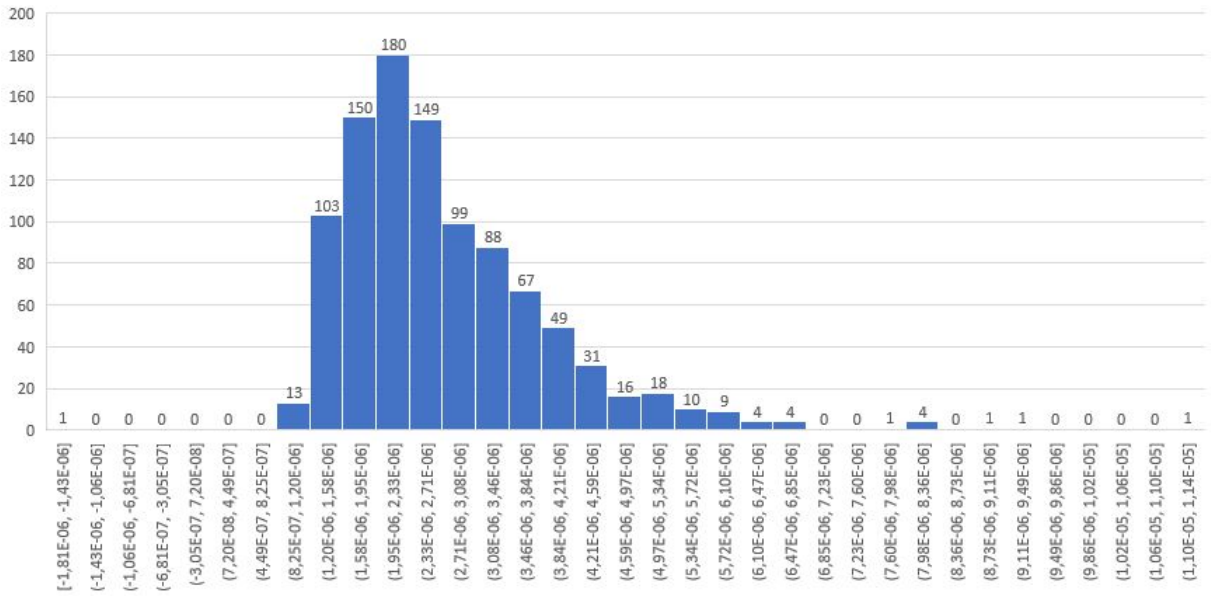

Figure 5.9: The Monte Carlo analysis results for the respiratory inorganics impact category.

The respiratory inorganics category has a LCIA result that is a little higher than the mean, with the distribution in Figure 5.9 indicating that the result should perhaps be somewhere in the range of 1.58E-6 - 2.71E-6.

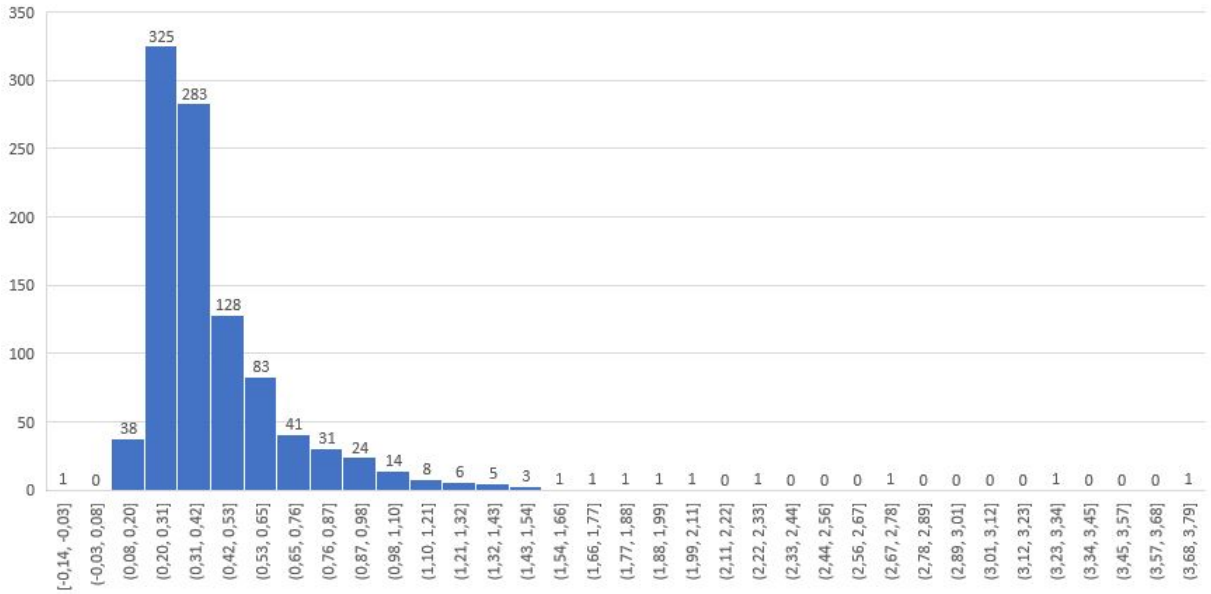

Figure 5.10: The results of the Monte Carlo analysis of the dry + wet system for the acidification impact category.

The results for the acidification impact category has some very distant high-value outliers, which might be the reason the mean is marginally higher than the LCIA result. The LCIA result does lie withing the interval with the highest occurrence.

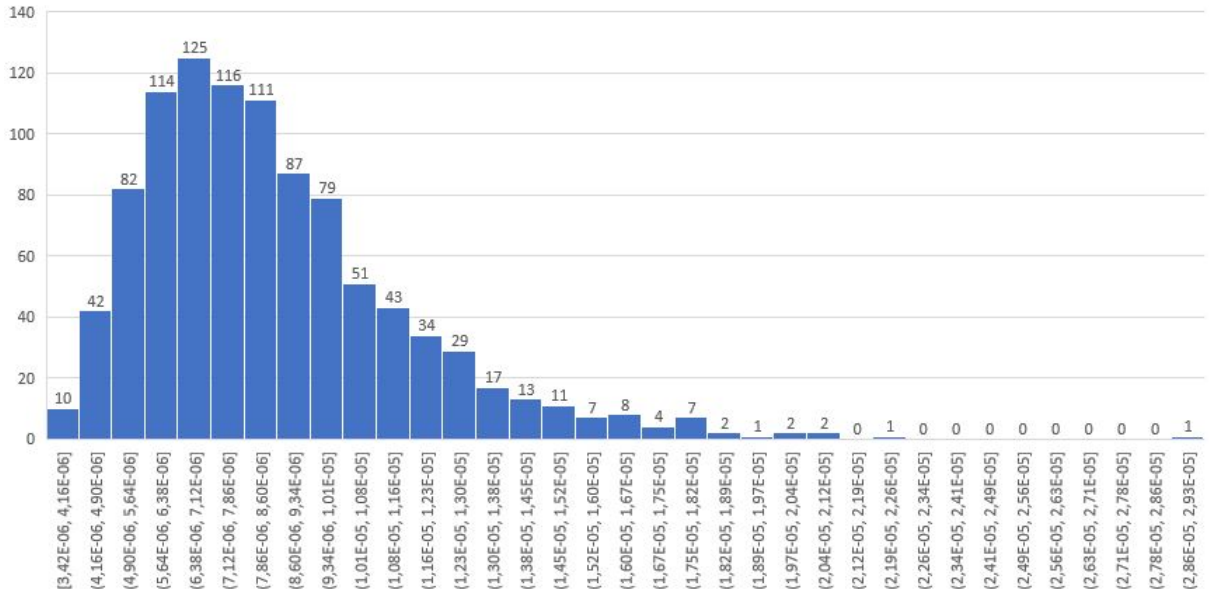

Figure 5.11: The results of the Monte Carlo analysis of the dry + wet system for the ozone depletion impact category.

The ozone depletion category is the one with the second-lowest RSD, although it is the first Monte Carlo result where the calculated LCIA result is not within 1 SD of the mean. The true result is therefore likely to be a bit higher, closer to 8.51E-6.

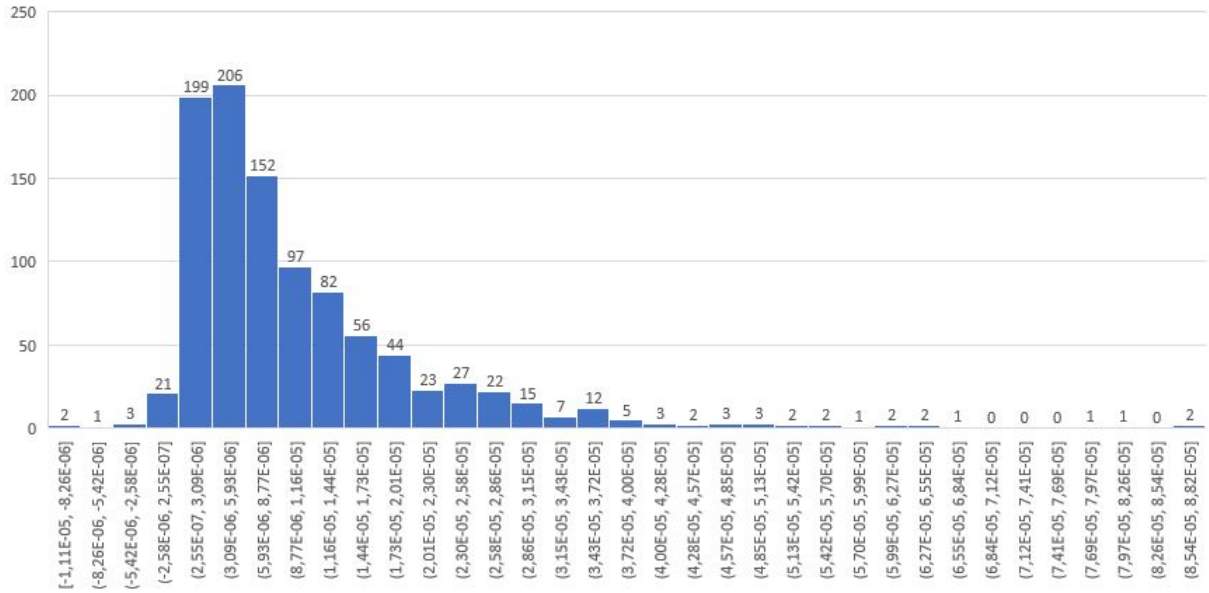

Figure 5.12: The results of the Monte Carlo analysis of the dry + wet system for the human health, cancerous impact category.

The cancerous human health results have a very high RSD, over 100 %, but the LCIA result is still within an interval with a relatively high probability (152/1000). Even so, based on the mean value the true value is possibly a bit higher than the calculated result.

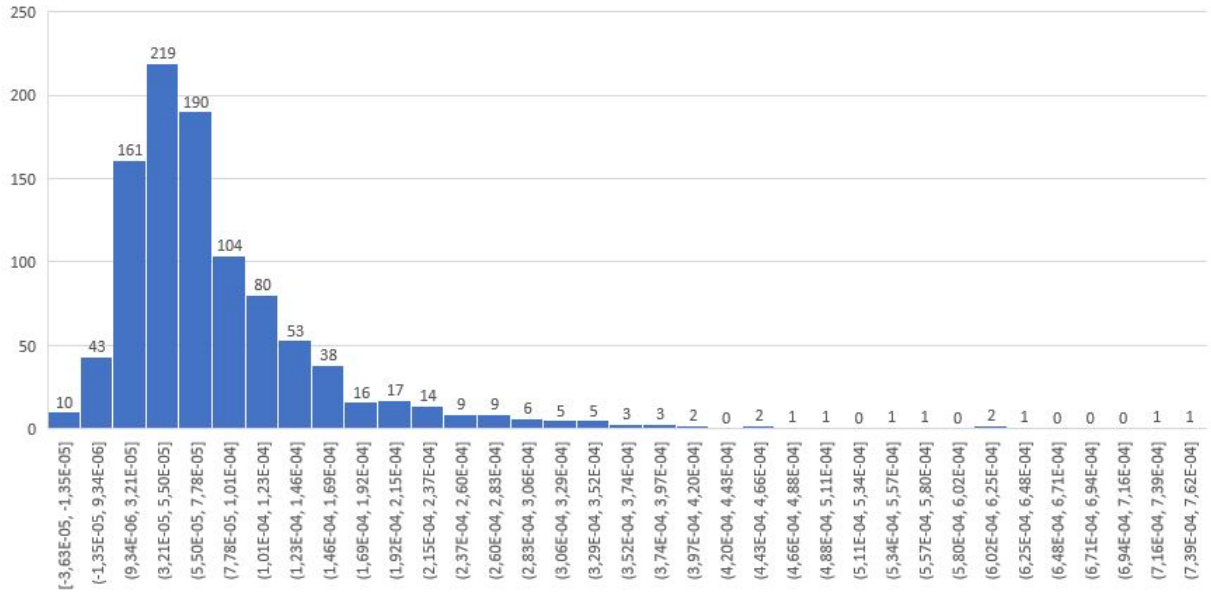

Figure 5.13: The results of the Monte Carlo analysis of the dry + wet system for the human health, non-cancerous impact category.

The non-cancerous human health result shown in Figure 5.13 resembles the result of the cancerous human health category, with a similar distribution and a few high outliers, as well as a RSD above 100 %. Based on the distribution and the mean the true result is probably higher than the LCIA result as well.

5.8 Final discussion

In general, it seems that the best choice of technology depends on which impact category one deems to be the most important. The best overall results, if recycling is disregarded, seem to come from the systems combining a wet and a dry stage. Having only a dry stage results in significantly higher impacts in the climate change, respiratory inorganics and acidification categories. Having only a wet stage gives a result that is almost as good as the dry + wet technology in most categories, although a little bit higher in ozone depletion. If recycling can be implemented, in the manner modeled in this study, the $CaCl_2$ recycling scenario would be the best in each category. The recycling of H_2SO_4 has overall a very low effect.

The results for the climate change category aligns fairly well with the results of Boesch et al. [23], where results show that 425 kg CO_2-equivalents are generated in the incineration process. Regarding which system has the best environmental performance, in the study by Dal Pozzo et al. (2018) [13] the combined dry and wet system is the best option in the categories global warming and acidification, but the wet system is the better choice in the ozone depletion category and the dry system has the best performance with regards to ozone depletion. The study by Dong et al. does not include combined systems, but claims that dry systems have lower environmental impacts than wet systems, with semi-dry in between. These differing results may stem from inventory data or alternative model selections, as these have shown

to have a very significant influence on the results, as high as 1400 % [4]. The results can also vary with waste composition, which was not included in this study. Nevertheless, dry technologies seem to have some merit, as the market share of dry and semi-dry systems have been increasing as of late due to their simpler operation and lower complexity [13], and multi-stage dry systems also seem very promising [14, 17].

Regarding the recycling scenarios, the reason the production of H_2SO_4 makes such a little difference is probably due to the high purity required for commercial grade, 98.3 %. This means that you get both a high cost of increasing the concentration up from 20 %, and a much smaller amount of product. In comparison, the $CaCl_2$ concentration only needs to be increased from 7.5 % to 40 %, making it much more viable as a recycling option. Among the input processes, ammonia production frequently ranks as the highest contributor to the environmental impacts. This means that the system is quite sensitive to changes in ammonia input, and the results in several categories will change significantly if the ammonia input is changed. It also means that if one were to attempt to improve the system impact on respiratory inorganics, acidification or ozone depletion, improving the ammonia production process would be an advisable place to start, if possible. By contrast, the choice of technology will not have a large effect on climate change or human health, as the vast majority of emissions affecting these categories originate from the waste incineration facility.

Based on the contribution tables, the incineration facility is responsible for a large share of the impact in all categories except ozone depletion. Especially in the climate change and human health categories, where roughly 90 % of the emissions originate from the waste incineration facility. Besides the ozone depletion category, the one with the most upstream contribution is the respiratory inorganics, where 40-60 % of the impact comes from input processes.

The results of the Monte Carlo analysis indicate that the LCIA results in most categories is quite close to the true value, even if there is a large spread in the results for the acidification and human health categories. Regardless, since many of the processes which have significant contributions to the impact categories are common to all scenarios and it is just the input amount changing, their impact relative to each other should be correct. The most noteworthy exception would be the calcium carbonate process, which is found only in the wet systems and has a sizeable contribution to several categories. It should also be mentioned that the impact results for the activated carbon input should in reality be a somewhat higher, since GAC has been used as a replacement for PAC, which would require additional energy to mill.

6
Conclusion

The findings of this study indicate that if calcium chloride can be recycled from the cleaning process, then that will be the technology choice with the least impact in all categories reviewed in this report. The recycling of sulfuric acid will reduce the impact of wet systems, but not enough to make it the next lowest after $CaCl_2$ in all categories, only in the acidification category.

Disregarding recycling scenarios, the combined dry + wet system seems to be the best option from an environmental standpoint. The dry system has the highest results in climate change, respiratory inorganics, and acidification, but shares the lowest result in ozone depletion and human health effects with the dry + wet system. The wet system has results that are slightly higher but very close to the dry + wet system in all categories except climate change, where it has the same result.

The processes with the most influence on the impact results would seem to be the ammonia production, the quicklime production and the calcium carbonate production. These three are also the chemicals with the highest consumption rates, so their contribution might have been a foregone conclusion, although it should be noted that the consumption rate of ammonia is the least of these three and yet often has a higher contribution than the other two.

Overall, based on the results of the Monte Carlo analysis, the uncertainty of the modeled system is in some cases quite high, but the LCIA results are, according to the probability distributions, quite close to the expected results. However, due to the high RSD in comparison to the relatively small differences between the scenarios in each category, the results can be considered indicative of the differences between scenarios, but not robust. The final conclusion is that if $CaCl_2$ recycling can be implemented, that would give the least environmental impact in all categories, out of the investigated technology alternatives. If not, then a combined system of dry and wet technologies together is preferred. The dry and wet systems each have their respective categories where they are the next best alternative, although the dry system has a significantly higher impact than all other alternatives in the categories climate change, respiratory inorganics and acidification.

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A

Appendix 1

In this appendix the nominal design data for the Swedish waste incineration plant APC system is presented in Table A.1 and A.2.

Table A.1: Nominal design data for the Swedish waste incineration plant at the boiler outlet.

Design variable	**Specification**
Flue gas flow	130 000 Nm^3, wet gas
Temperature	160 °C
Pressure	-1500 Pa
Humidity, wet gas	15 vol%
O_2	6 vol%, wet gas
NOx as NO_2	100 mg/Nm^3, 11 % O_2 dry gas
NH_3	15 mg/Nm^3, 11 % O_2 dry gas
Dust	2000 mg/Nm^3, 11 % O_2 dry gas
HCl	1200 mg/Nm^3, 11 % O_2 dry gas
SO_2 + SO_3	350 mg/Nm^3, 11 % O_2 dry gas
HF	12 mg/Nm^3, 11 % O_2 dry gas
Sb+As+Pb+Cr+Co+Cu+Mn+Ni+V	12 mg/Nm^3, 11 % O_2 dry gas
Cd+Tl	1 mg/Nm^3, 11 % O_2 dry gas
Hg	0.2 mg/Nm^3, 11 % O_2 dry gas
Dioxins and Furans, dry gas	5 ng/Nm^3, 11 % O_2 dry gas

Table A.2: Nominal design data for the Swedish waste incineration plant at the baghouse filter outlet.

Design variable	**Specification**
Flue gas flow	135 200 Nm^3, wet gas
Temperature	140 °C
Static pressure	-3976 Pa
Humidity, wet gas	15.7 vol%
O_2	6.4 vol%, wet gas
Flue gas density	1.267 kg/Nm^3
CO	25 mg/Nm^3, 11 % O_2 dry gas
NOx as NO_2	100 mg/Nm^3, 11 % O_2 dry gas
NH_3	15 mg/Nm^3, 11 % O_2 dry gas
Dust	2 mg/Nm^3, 11 % O_2 dry gas
HCl	11 mg/Nm^3, 11 % O_2 dry gas
$SO_2 + SO_3$	25 mg/Nm^3, 11 % O_2 dry gas
HF	1 mg/Nm^3, 11 % O_2 dry gas

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