Life-Cycle Assesment of different technologies to process sewage sludge

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Challenge the future

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Life-Cycle Assessment of different technologies to process sewage sludge

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List of Abbreviations

ALCA	Attributional Life-Cycle Assessment
CBS	Central Bureau of Statistics, Netherlands
CHP	Combined Heat and Power Generation
CLCA	Consequential Life-Cycle Assessment
daf	Dry ash free basis
db	Dry basis
DS	Dry Solids
EC	European Commission
EPA	Environmental Protection Agency, United States
ER	Equivalence Ratio
EU	European Union
FU	Functional Unit
GT	Gas Turbine
GWP	Global Warming Potential
HHV	Higher Heating Value
ISO	International Organisation for Standardisation
kg	kilogram
kmol (%)	kilo mole percent
kWh	kilowatt-hour
LCA	Life-Cycle Assessment
LCC	Life-Cycle Costing
LCIA	Life-Cycle Impact Assessment
LHV	Lower Heating Value
MJ	Mega Joule
mg	milligram
MSW	Municipal Solid Waste
MW	Mega watt
Nm^3	normal cubic meter, measured at 273K and 1 atm
NOGEPA	Netherlands Oil and Gas Exploration and Production Association
p.e.	Population Equivalents
Plasma G	Plasma Gasification
SCWG	Supercritical Water Gasification
SNB	Slibverwerking Noord-Brabant
SNG	Synthetic Natural Gas
SOFC	Solid Oxide Fuel Cell
STOWA	Foundation for Applied Water Research, Netherlands
tkm	tonne-kilometer
VOC	Volatile Organic Solids
vol (%)	Volume percent
wt (%)	Weight percent
WWTP	Wastewater treatment plant

Abstract

Wastewater production has been steadily increasing globally. Treatment of wastewater and the resulting sludge is crucial for human health and ecosystem conservation. Furthermore, water supply and treatment is an energy intensive sector where the electricity consumption is projected to increase by more than 80% in the next 25 years.

Sludge treatment is one of the few processes in the water sector where energy can be recovered. And with the environmental issues faced by the sludge treatment sector, it becomes essential to not only optimize the energy recovery but also to holistically analyze environmental impacts of sludge treatment technologies. Thus helping to reduce the human and environmental impact to a minimum. In this report, the current sludge treatment chain in the Netherlands was environmentally analyzed along with two promising technologies, plasma gasification and supercritical water gasification for sewage sludge treatment using lifecycle assessment methodology. The end use of the produced syngas in a gas engine and solid oxide fuel cell was also modeled to find their maximum energy generation potential which is also technologically feasible. Furthermore, impact of anaerobic digestion of sludge in combination with the mentioned technologies was also studied resulting in a total of 9 systems.

Literature review and macro modeling of the mentioned systems are used to determine the inputs and outputs over their entire lifecycle. Results from this study show that under the modeled conditions, supercritical water gasification has the potential to considerably reduce environmental impacts since it performed better than other thermochemical conversion technologies in all impact categories. This was primarily due to high energy recovery and relatively clean syngas produced. Furthermore, anaerobic digestion still produced the highest net energy per ton of sludge processed without requiring secondary fuel input. Due to large amount of energy required for drying sewage sludge and material intensive fluegas cleaning, plasma gasification did not offer promising results for sewage sludge treatment. It can still however offer substantial benefits over incineration especially for waste treatment of feedstock with lower moisture content. Solid oxide fuel cell was the environmentally preferred option over gas engine for electricity generation from syngas due to low nitrous oxide emissions combined with higher energy recovery. Results in literature have shown safe short term combination of gasifier-solid oxide fuel cell. However, uncertainties with the respect to its long term reliability still remain.

Energy generation volatility with respect to the scale of anaerobic digestion and feedstock properties for supercritical water gasification performance present two possible sources of uncertainties to the above results. The electricity generation from anaerobic digestion is only economically feasible for capacities of 30,000 population equivalents or higher. Whereas, energy generation and consequently the environmental performance of supercritical water gasification is highly dependent on dry solids content. However, with considerable effort going into research and process optimization, this uncertainty can be eliminated soon.

1 Introduction

1.1. Background of study

Wastewater in general contains variety of different compounds which are synthetic chemicals, pathogens including viruses and bacteria, nutrients, organic matter and heavy metals. Effect of untreated wastewater disposal on public health, wildlife and environmental safety has been emphasized enough over the years [88], [37], [73] & [30]. Wastewater cannot be disposed without treatment in water bodies because of its contents. Organic compounds in the wastewater mainly consist of proteins, carbohydrates and fats. These compounds use dissolved oxygen to undergo oxidation and stabilize. This leads to negative consequences on the aquatic life due to the fact that now there is less oxygen available in the water [53]. Pathogens in water can lead to transmission of waterborne diseases like hepatitis A, dysentery, salmonella and so on.

Wastewater also contains nutrients like nitrogen and phosphorous. These nutrients are essential for the growth of aquatic life however disposal of wastewater can cause large amounts of nitrogen and phosphorous to enter the water bodies [88]. These nutrients may lead to excessive growth of algae, much higher than what can be handled by the aquatic ecosystem. Furthermore, heavy metals also end up in wastewater primarily from industrial discharge. Depending on the type of heavy metal, they can lead to severe consequences such as stagnation of mental development in children, dysfunctioning of internal organs and may also cause cancer, if they end up in human ingestion. Harmful effects are not only limited to humans, but also to plants and animals [40], [75] & [36]. These metals could also end up in food-chain and in turn be harmful to humans.

Hence treatment of wastewater and consequently, appropriate treatment and disposal of sewage sludge arising from wastewater processing is crucial. Modern wastewater treatment plants (WWTP) usually employ a preliminary treatment followed by three treatment stages [31]. In preliminary treatment, coarse particles in water are removed with help of screens followed by primary treatment where suspended solids and organic matter is removed by mechanical processes like sedimentation or floatation. Later comes the secondary stage where biological treatment method is used towards coagulation and removal of biodegradable organics and suspended solids. Finally, for nitrogen and phosphorous treatment, nutrient removal stages are employed. Apart from these stages, treatment plant can also include an “advanced stage” if high quality of water is required [88]. Above treatment stages of wastewater lead to the production of sewage sludge. According to the US environmental protection agency (EPA) it is defined as ’solids obtained as a byproduct during wastewater treatment’. In general, the treatment processes concentrate the contents of the wastewater until a semi-solid mixture of concentration ranging between 0.25 to 12% is formed [88].

According the International Energy Agency, wastewater treatment consumed 200 TWh of energy in 2014 which accounts for about 1% of global energy consumption [47]. In USA and European Union, wastewater treatment formed the largest share of the energy consumption in the water sector due to strict regulations surrounding waste disposal. With the rise of de-

veloping countries in Asia and Africa, as the regulations around wastewater tighten, energy and material consumption for this sector will increase. Moreover, 35% of the wastewater globally is not even collected [47]. Thus the environmental impact around wastewater treatment is only going to increase in the future. Thus it becomes essential for us to quantify environmental impacts to improve current treatment systems and identify hot-spots in the developing technologies.

1.2. Sewage sludge in Netherlands

Wet sludge production decreased four times from 4.86 in 1990 to 1.43 million tonnes in 2000 in the Netherlands. Furthermore, there was a drop in recent years, 2013 and 2014 when it remained constant at 1.27 million tonnes. Sludge production from Wastewater treatment plants (WWTP's) in the Netherlands has deviated slightly but stood fairly steady from 2000 through 2012 [1]. The interesting thing to note here is that the waste water treatment since 1990 has actually increased. Consequently, the dry solids content of sewage sludge has increased. Figure 1.1 shows the treatment statistics in Netherlands trend over the years for wet sludge. A sharp decrease in landfilling and agricultural use from the year 1990 onwards is observed. The use of sludge in Agriculture has to follow strict requirements regarding content organic content, heavy metals, nutrients and odor [98]. Making the agricultural application virtually impossible. On the other hand, landfilling decreased because of the 'landfill tax' introduced by the Dutch government. Incineration has now become the primary technique to treat sewage sludge. As of 2014, almost all of the sewage sludge was incinerated. In the graph, Incineration also includes co-incineration which is usage of sludge as a 'secondary fuel at powerplants' as defined by CBS. These powerplants are coal or municipal solid waste (MSW) incinerators. However, this does not include co-incineration in cement kilns.

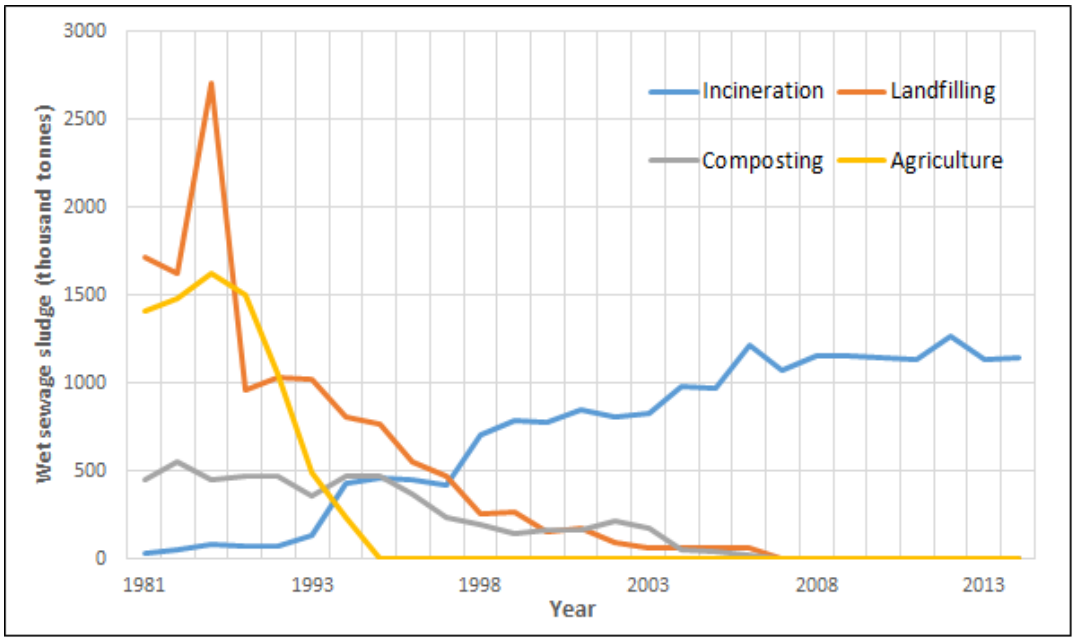


Figure 1.1: Wet sludge treatment in Netherlands over the years [2]

Sludge treatment even in European countries varies a lot. Breakdown for six countries: Germany, UK, Netherlands, Greece, Spain and Sweden for the year 2012 is shown in figure 1.2. For example, in Germany out of about 1,750 thousand tonnes of sludge produced in 2012, only half was incinerated. Composting and agricultural use formed the rest of disposal methods. Whereas in the UK, agricultural use of sludge was the most dominant method. Out of the 1,078.4 thousand tonnes of sludge disposed, 78 % of it was treated and reused in agriculture while the rest was incinerated [1]. Also, we can see here very peculiar Dutch scenario where almost all of the sludge treated was incinerated. In Greece, landfilling and

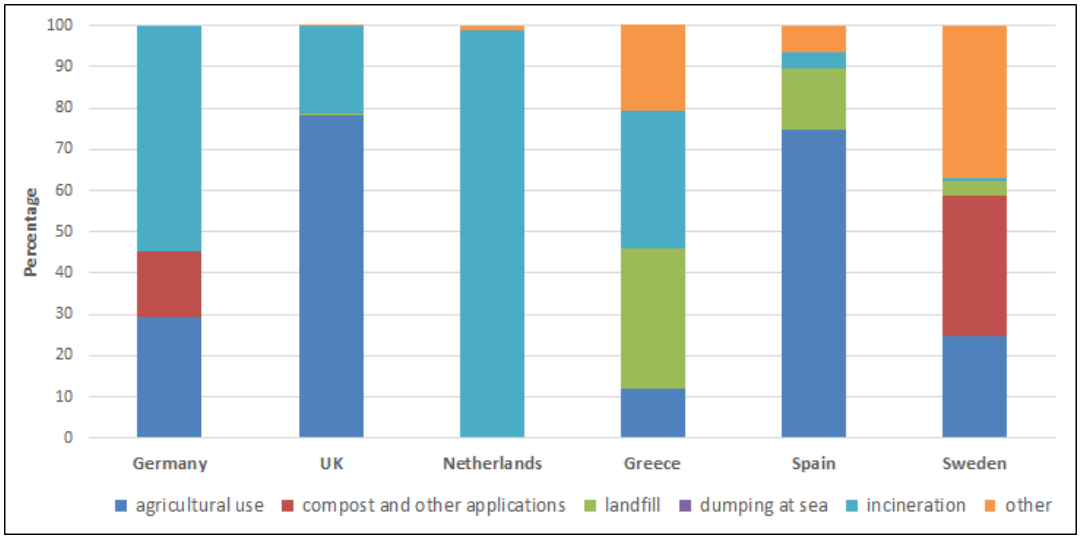


Figure 1.2: Sludge treatment methods in Europe: Country wise breakdown for the year 2012. [1]

incineration were the major technologies contributing 33% each to the treatment of total sewage disposed. Next major share was 'other' which means mechanical separation of solids. For Spain, 75% of sludge was utilized for agricultural applications. Data for composting was not available and perhaps was included in the agriculture use. Incineration was relatively low at 4% while landfilling and other uses stood at 15% and 6% respectively. In Sweden, agriculture use and composting consist of almost 60% of total treatment. Incineration and landfilling were at 3% and 0.7% respectively. The rest was treated by mechanical separation. If we look at sludge treatment data from previous years, the general trend of sewage sludge treatment in Europe is to transition from land application to thermochemical conversion of sludge.

1.3. Dutch regulations around sewage sludge

The first EU directive regarding waste disposal was introduced in 1975 [31]. Today regarding the wastewater treatment, directive 91/271/EEC which was later amended by 98/15/EC came into force in year 2005 [90]. This directive led to a 50% increase in sludge quantity due to its stringent quality standards for wastewater treatment [31]. The Netherlands is only one of the four countries (other countries being Germany, Finland and Austria) which completely comply with these standards, according to a recently published directive implementation report [20].

Apart from directives mentioned above, ones which have the most influence on sewage treatment and management are 1999/31/EC (concerning landfilling of waste), 86/278/EEC (concerning use of sludge in agriculture) and 2000/76/EC (concerning incineration of waste). Also in sludge treatment and disposal standards, even though 80% of legislations and regulations are derived from the EU, Dutch requirements for sludge are much more stringent than the EU directives [19]. For example, if sludge is to be used for agriculture, the allowable heavy metal concentration in Netherlands is almost 10 times stricter than European directives [19]. Regulations of sewage sludge incineration are covered under regulations for waste incineration. The directive 2000/76/EC on waste incineration lays down a requirement of fluegases reaching a temperature of atleast 850° C for atleast 2 seconds in order to ensure complete combustion except when halogenated organic compounds are present [90]. Apart from the general emission requirements as published in NeR (Netherlands emission guidelines for air) the Netherlands does not have any such requirements. Emission requirements can be found here in detail [69].

1.4. The Sludge Chain

A typical sludge treatment chain is shown in figure 1.3. It should be noted that not all the possible techniques for each sludge chain step are listed here. It also possible and not unusual to skip one or more of the mentioned steps. Since sludge is produced at 0.8-1% dry solids (DS) [65], water removal is one of the main focus areas for safe processing of sludge. Two steps in the chain, Thickening and Dewatering are exclusively for water removal. Thickening is a low energy process relying on gravity as a separation force increasing the DS content of sludge from 0.8-1% to 3-7%. Dewatering on the other hand uses mechanical force to remove water and increase DS content usually to around 25% DS.

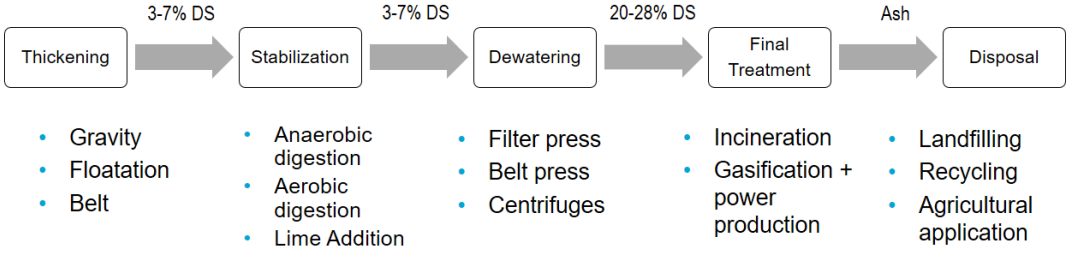


Figure 1.3: A typical sewage sludge treatment chain [31].

Stabilization of sewage sludge is carried out for the following major reasons as described in [97].

- Reduce odor nuisance and prevent rotting of sludge
- Eliminate disease causing micro-organisms
- Improve dewatering characteristics of sludge

However the last point is a topic of discussion amongst the researchers and industry [49]. Choice of stabilization technique depends on the end use of sludge. For example, lime addition results in increasing the pH of sludge to more than 12. As a result, the bacteria cannot survive and the sludge does not putrefy. Lime addition is often used when sludge has to be used for agricultural application [84].

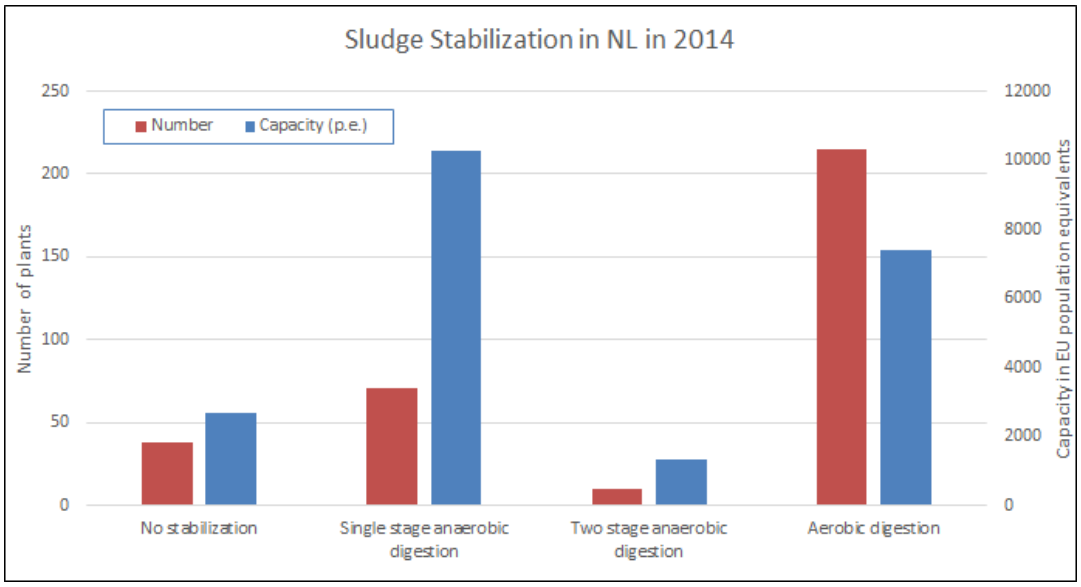


Figure 1.4: Sludge stabilization techniques in the Netherlands in terms of number of units and number of population equivalents for the year 2014 [2]

Digestion in the presence of oxygen (aerobic) or absence of it (anaerobic) is also used as a stabilization method. Popularity of digestion in the Netherlands can be seen in the figure 1.4. Biogas recovered from anaerobic digestion can be use for power generation making it the largest stabilization method in Netherlands in terms of capacity. However it should be noted that anaerobic digestion is only economical for WWTP of capacity more than 25,000-30,000 p.e.[1] according to data by the Central Bureau of Statistics (CBS), Netherlands [2]. Perhaps this is the reason the number of installed units of aerobic digestion plants are high but their overall capacity is lower than anaerobic digestion. Even though aerobic digestion consumes energy, it is economical on smaller scales.

Dewatering techniques in the Netherlands in 2014 are presented in figure 1.5. This data was again collected from CBS [2]. Centrifuges was the most widely applied technique both in terms of number of units and quantity of sludge processed. It uses centrifugal force for removal of water. Whereas belt and filter presses both use mechanical force. A small fraction of sludge is dewatering by atmospheric drying under the categories of lagoons and drying beds. The difference between these two techniques was not specified [2]. In the Netherlands, sludge produced at WWTP after stabilization and dewatering, is transported to the thermochemical treatment [98].

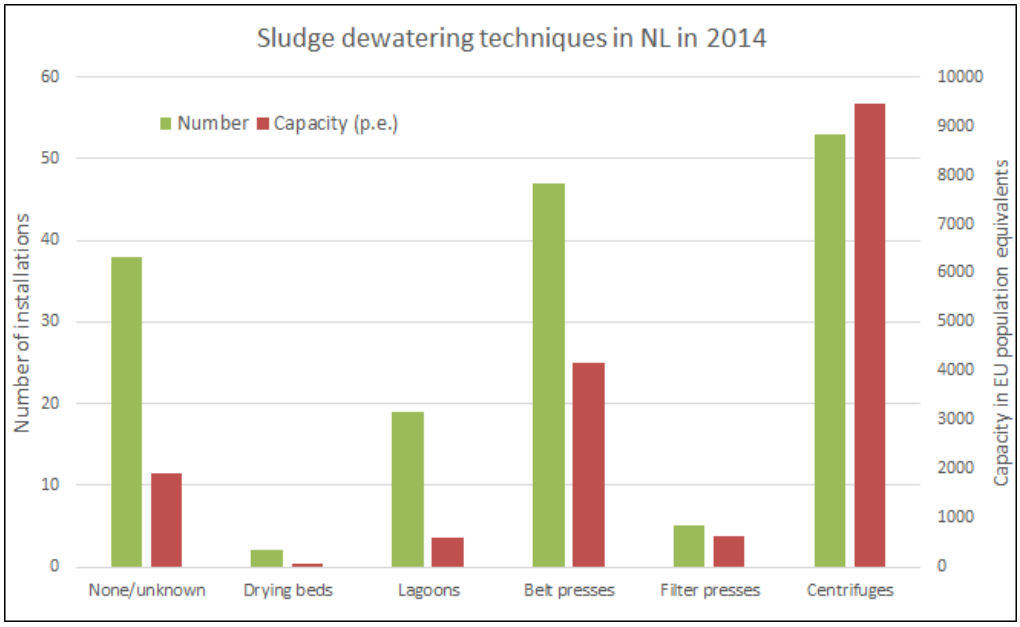


Figure 1.5: Sludge dewatering techniques in the Netherlands in terms of number of units and number of population equivalents for the year 2014. [2]

1.5. Thermochemical conversion of sludge

Even if energy was produced from anaerobic digestion previously, land application of sludge in Netherlands is very difficult. Hence this semi-dried sludge is subjected to thermochemical conversion where the recovered energy is usually used for district heating or generation of electricity. It drastically reduces the environmental impact by avoiding fossil fuel use [55]. A review paper from the university of Wageningen highlights nine different methods of energy recovery from sludge [79]. Out of which thermochemical processes include incineration and novel technologies like gasification/pyrolysis and hydrothermal treatment. Sludge can also be co-incinerated with Municipal solid waste (MSW), coal or used cement/brick kilns. Some of the technologies like anaerobic digestion and incineration are mature while the rest are still in their early development phase or they are not yet as widely accepted commercially .

[1] A population equivalent is defined as 'amount of oxygen demanding substances whose oxygen consumption during biodegradation equals the average oxygen demand of wastewater produced by one person' [6]. It basically defines the capacity of wastewater or sludge treatment plants in terms of the population it caters to.

As mentioned before, the majority of sludge treatment plants are based on incineration. A major problem with incineration is the disposal residual ash [97]. Heavy metals present in the sludge vaporise during combustion and then in the cooler part of the furnace, settle on the surface of ash. Hence ash obtained can contain 78 to 98% of heavy metals originally present in the sewage sludge except Mercury which remains in the flue gases [51] & [72]. The main issue with ash formed is that it is prone to leach heavy metals it contains into the groundwater. Hence for ash disposal, special landfilling sites must be developed or further processing must be done [97]. This issue can be solved if incineration is carried out a temperature higher than the melting temperature of ash. According to various studies, behaviour of ash is straight-forward to predict. The process consist of the following steps, melting at 1500°C, nucleation at 800°C and crystallisation at 1100°C in that order [58] & [85]. It will lead to formation of molten slag which can be cooled after taking out of the furnace to recover energy. After cooling, the ash forms a non-leachable glass like substance. Hence it can be used as glass ceramics. Also, use of ash in bricks and tiles after appropriate thermal treatment can be made [58]. Properties of tiles can be modified by using ash with limestone, silica sand, soda ash and dolomite in varying proportions. More details can be found in this study by Suzuki et al [85].

Gasification

Gasification is defined as "thermochemical fuel conversion technology carried out at high temperature using a gaseous agent to convert a liquid or a solid fuel into combustible gas" [22]. Gasification is not a new technology. In the 19^{th} and beginning of 20^{th} century, coal was heated in the absence of air to obtain carbonized coke and town gas. Coke was then oxidised with limited amount of air to obtain producer gas [63]. Gasification includes hydrogenation, pyrolysis and partial oxidation but currently, the dominant technology is the latter. Partial oxidation, as the name suggests is combusting fuel with a limited supply of air/oxygen. From now on, gasification will be used as a term for partial oxidation interchangeably. Interest in gasification has increased in the last decade due to surging price rises of crude oil and natural gas[39]. Gasification of coal provides wide range of options: power production, chemicals, natural gas and other fuels. Apart from coal, biomass gasification has also garnered attention.

Gasification of sewage sludge

In recent years, gasification has become a very competitive process with combustion. This trend is not unjustified as it has been proven to have many advantages over conventional combustion especially with fuels like biomass and coal [78] & [60]. Same can be said about gasification of MSW according to many studies [59], [71] & [12]. There are 3 main reasons behind this.

Firstly, the level of pollutants is expected to be lower in gasification because of reducing conditions. The high temperatures in gasification completely destroys dioxins and furans [97]. Furthermore, lack of oxygen limits formation of dioxins and furans in the syngas cleaning stage where temperature is lower [59] & [71]. NO_x emissions are also much lower for gasification since these emissions increase with an increase in air to fuel ratio during the combustion process [97].

Secondly, during incineration, excess air is provided to ensure complete combustion of fuel. Whereas in gasification, feedstock is heated at high temperatures with limited supply of oxygen or air (much less than the stoichiometrically required amount) [63]. Hence the volume of gases formed in case of gasification are much lower as compared to incineration. One such study by Arena was conducted on different types of MSW which reports air requirement was 1/3rd of air requirement for incineration [71] & [12]. Equivalence ratio (ER)[2] for gasification in the above study was varied between 0.2 to 0.3. Steier [82] (As cited in [97]) also reported the extent of volume reduction of fluegas in gasification as compared to incineration. A reduction from 24-30m^3 to 1.7m^3 of gas produced per kg dry sludge is detected when pure oxygen is used for gasification instead of incineration. ER for incineration typically for MSW is 1.8-2

[2]Equivalence ratio is defined as the ratio of air supplied to air required for stoichiometric complete combustion of fuel.[22]

while for sludge is 1.25-1.5 (using fluidised bed combustion (FBC)) [64] & [97]. As a result, overall volume of infrastructure and thus the landuse required by the plant is smaller.

And thirdly, the power production compared between the current state-of-the-art gasification and modern incinerators is comparable. For example, MSW combustion has a thermal conversion efficiency of 80% while MSW gasification followed by gas engine or gas turbine is at 70%. The net electric efficiency of MSW combustion is between 19 to 27% whereas for MSW gasification followed by gas engine and gas turbine combined cycle ranges from 13 to 24% and 23-26% respectively [71]. Currently, most gasification plants use steam cycles post syngas cleaning stage. Gasification has a higher power production potential if the syngas produced during gasification is utilised with Integrated gasification combined cycle plant (IGCC), gas engine, gas turbine or solid oxide fuel cell (SOFC). With ample research going on, these two technologies promise a net power production increase by a factor of 2 [71]. Finally, gasification offers greater flexibility as syngas can be used for production of hydrogen or as a precursor to various compounds in the chemical industry [12].

Plasma Gasification

When plasma is added to gasification, we can reap the benefits of gasification along with benefits of melting or vitrification of ash. Due to very high temperatures involving plasma gasification, typically between 2500°C and upto 4500°C (except for reduced pressure RF plasma) [87], ash is taken out as molten slag, energy is recovered by cooling followed by use in building materials. Benefits of this are substantial which include prevention of leaching of heavy metals, avoid production of virgin sand/gravel and no need for expensive disposal technologies [97]. A typical plasma gasification plant facility can be seen in figure 1.6.

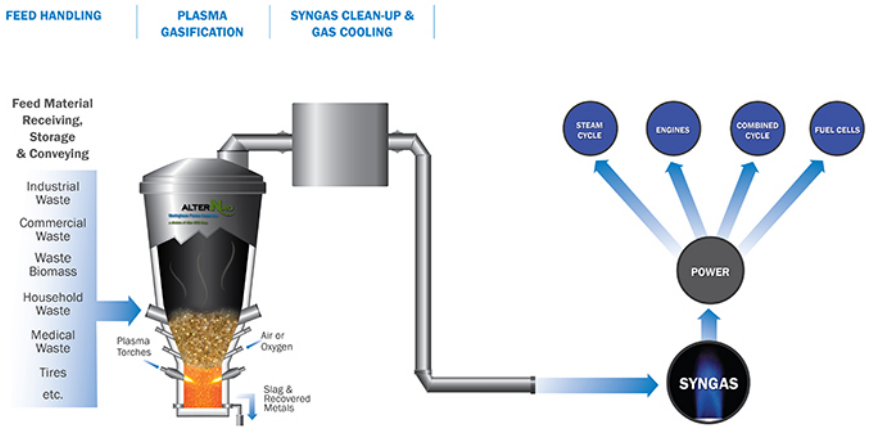


Figure 1.6: A typical plasma waste processing facility by Phoenix Energy, Australia [3]

The gasifier shown in the figure is an updraft fixed bed plasma gasifier. Variety of waste can be used as a feedstock as shown. To explain in simple terms, organic part of the waste is converted into syngas whereas the inorganic part is collected as vitrified slag from the bottom of the reactor. Syngas is cleaned and utilized for generation of power or manufacturing of materials.

Supercritical Water Gasification

Although it was not envisaged at the start of this project, to take supercritical water gasification (SCWG) into account, post the literature review however, the number of benefits it offered for sludge treatment could not be ignored. Main advantage being the ability to process biomass with relatively high moisture content Yoshida et al found that biomass with moisture content more than 30% favored SCWG rather than other gasification technologies. This is very much suited to sewage sludge since sludge is produced at WWTP's with a moisture content of around 99%. Description of SCWG will be detailed in section 3.6.

1.6. Life cycle assessment (LCA)

In the past decade, LCA has been widely applied to various processes, products and systems. This can be concluded by looking at the steady growth of number of articles published on LCA. According to the ISO 14040 standard, LCA can be defined as the ”compilation and evaluation of the inputs, outputs and potential environmental impacts of a product system throughout its lifecycle” [28](as cited in [35]). The important thing to note here is, LCA follows the environmental impacts throughout its lifecycle also known as ’cradle-to-grave approach’. Such an approach avoids ’problem shifting’ and presents a holistic picture of impacts. Right from resource depletion, emission of hazardous substances, land use change to global warming is covered under its scope. Hence LCA is an ideal tool to assess environmental impacts of complex waste management techniques [35]. As we will see in the next section, as much as 35 different studies have applied LCA in its original or modified form to sewage sludge treatment techniques.

1.7. Previous LCA studies

Many LCA studies have been already conducted for various types of technologies for sludge treatment, comparing their environmental as well as economic performances. Yoshida et al recognized 35 studies analyzing the environmental performance of various technologies [103]. However only limited amount of them were based on LCA methodology. Due to the overwhelming amount of studies already conducted, most recent articles which conducted LCA based assessments were considered. A summary of literature survey and their system boundaries/assumptions are presented in tables 1.1 and 1.2 respectively.

Suh and Rousseaux, 2001 [84] studied the environmental impacts of five different alternatives for sewage sludge treatment using LCA. Sewage sludge was considered to be ‘mixed’ and was thickened and dewatered before application of different scenarios. Systems were all possible combinations of one final treatment step supplemented by one stabilization process. Main processes included incineration, agricultural use and landfilling whereas stabilization processes included lime addition, composting and anaerobic digestion. Nine impact categories were considered which are: resource depletion, climate change, human toxicity, fresh water aquatic, marine aquatic and terrestrial ecotoxicity, acidification, eutrophication and photo-oxidant formation. A combination of anaerobic digestion with agricultural application was found to be the most environmentally friendly because of low emissions and low energy requirements. The main reason behind high environmental impact of incineration system followed by landfilling was heavy metals released in air and emissions due to landfilling. The overall worst environmental impact was from the scenario: lime stabilisation followed by agricultural application majorly due to presence of heavy metals in the applied sludge and diesel required by trucks to transport sludge to agricultural fields.

Lundin et al, 2003 [56] conducted an environmental and economic assessment of sewage sludge handling options. Environmental analysis was carried out using LCA and Life cycle costing was used for economic assessment. The considered treatment options were: agricultural application, co-incineration with household waste, incineration with phosphorous recovery and hydrolysis with incineration and phosphorous recovery. Impact categories considered were: acidification, eutrophication, global warming potential and resource depletion. Authors identified energy use, phosphorous depletion and emissions of heavy metals as the most important aspects for environmental performance for sludge handling. Hence the scenario of energy production from incineration along with phosphorous recovery performed the best. On the other hand, agricultural use was environmentally the worst option owing to fossil fuel consumption required for handling and transport of sludge instead of energy recovery. Even though this alternative offered nutrient recycle to the soil, heavy metal emission was a key negative aspect. However, this option was economically the most desirable followed by incineration and further incineration along with phosphorous recovery processes.

Houillen and Jolliet, 2005 [44] studied six sludge treatment scenarios based on LCA. This study considered only energy production and global warming impact of the treatment technologies. Furthermore, nitrous dioxide emissions were excluded from scenarios either due to negligible emissions or lack of available data. The treatment technologies considered

Table 1.1: Summary of literature survey performed on LCA studies of sewage sludge

Sr.no.	Name of study	Technologies considered	Impact categories	Important remarks
1.	Suh and Rousseaux, 2001 [84]	Possible combinations of primary and secondary technologies. Primary: incineration, agriculture & landfilling Stabilisation: Lime, composting & anaerobic digestion	Resource depletion, climate change, human toxicity, fresh water aquatic, marine aquatic and terrestrial ecotoxicity, acidification, eutrophication & photo-oxidant formation	Overall results were heavily influenced by production of energy and end use of heavy metals in sludge. Anaerobic digestion followed by agriculture application performed the best
2.	Lundin et al, 2003 [56]	Agriculture, co-incineration with household waste, incineration with phosphorous recovery & incineration + hydrolysis with phosphorous recovery.	Acidification, eutrophication, global warming potential & resource depletion.	Incineration with phosphorous recovery was optimal due to avoided fossil fuel use and phosphorous recycle. Agriculture application offered nutrient recycle but heavy metal emissions to soil was a key negative aspect
3.	Houillen and Jolliet, 2005 [44]	Innovative techniques (pyrolysis, wet oxidation, incineration in cement kilns) compared to typical processes (agricultural land application, fluidised bed incineration, landfilling)	No impact categories but CO2 emissions & energy production	Energy production technologies performed better than others. Co-incineration in cement kilns had the lowest CO2 emissions.
4.	Hospido et al, 2005 [42]	Anaerobic digestion, pyrolysis and incineration	Eutrophication, statospheric ozone depletion, global warming potential, acidification, photo-oxidant formation, depletion of abiotic resources & human toxicity	Site conditions and sludge properties decide the optimal technologies for treatment. Land application should be done with minimization of heavy metal content and incineration should be combined with recovery of nutrients
5.	Murray et al, 2005 [62]	Extensive 9 scenarios with various combinations of agriculture application, digestion, incineration and co-incineration. See section 1.7 for more details.	No impact categories but air emissions, energy production and economic analysis	Anaerobic digestion followed by agriculture recovered energy and recycled nutrients making it the optimal technology. Ideal processes are considered in studies while in reality this might not be the case.
6.	Hong et al, 2009 [41]	Most complete scenarios considered with primary treatment and end uses. See section 1.7 for more details.	Global warming potential, acidification potential, human toxicity and land use. Economic analysis was also done.	Sludge melting was the environmentally optimal and economically affordable technology.
7.	Stokes and Horvath, 2010 [83]	Anaerobic digestion followed by landfilling/agriculture	No impact analysis but energy use, air emissions and sensitivity analysis	Energy recovery greatly reduces environmental impact of sludge treatment technologies
8.	Xu et al, 2014 [101]	Thirteen different combinations of agriculture, digestion, landfilling, incineration and dewatering. See section 1.7 for details.	Eighteen different impact categories along with LCC analysis	Dewatering, anaerobic digestion and agricultural application performed the best provided heavy metal content in sludge is limited.

Table 1.2: Summary of boundary conditions and major assumptions made in various LCA studies of sewage sludge

Sr. No.	Name of study	Const-ruction of facilities included	Energy Substitution	Sludge Source	Transport distances	Material substitution (Production of materials avoided)
1	Suh and Rousseaux, 2001 [84]	No	Thermal energy recovered was used in the process	Wastewater treatment plant in France	40km for agricultural application, 20km for landfilling & incineration is on site	Land application considered but fertilizers avoided is not
2	Lundin et al, 2003 [56]	No	Excess electricity replaces Swedish electricity mix. Thermal energy used for district heating and replaces primary sources.	Wastewater treatment in Goteborg, Sweden	According to local conditions in Goteborg, Sweden	Fertilizers avoided: Triple super phosphate and ammonium phosphate are considered
3	Houillen and Jolliet, 2005 [56]	Yes	Heat recovered is used in wastewater treatment, sludge drying or cement kilns replacing other fuels	Fixed composition based on data from France & Switzerland	50 km for agricultural application, 100 km each for incineration, wet oxidation & pyrolysis. 70 km & 75 for cement production and landfilling respectively.	Fertilizers, limestone and methanol production avoided is considered
4	Hospido et al, 2005 [42]	No	Heat produced was used for district heating system resulting in gas savings	Fixed composition based on a WWTP (location not specified)	Incineration– 286 t-km, Pyrolysis – 286 t-km, Anaerobic digestion – 105.5 t-km	Crude oil, activated carbon due to char production and fertilizers
5	Murray et al, 2005 [62]	No	Natural gas use avoided due to heat recovery	Composition from 4 WWTP's from Chengdu, China	According to local conditions in Chengdu. Exact values not specified	Fertilizer production & minerals in cement production
6	Hong et al, 2009 [41]	Yes	Waste heat is recovered for electricity production. Excess electricity is reused in the process	Sludge composition from a WWTP plant in Japan	Transportation considered in the study but value not specified	Material substitutions were not considered
7	Stokes and Horvath, 2010 [83]	Yes	Methane recovered was considered for sensitivity analysis	Composition from unnamed WWTP in California, USA	Based on the local conditions of the WWTP. Landfilling – 65 km. Land application –200 km.	Fertilizer substitution was considered
8	Xu et al, 2014 [101]	No	Electricity regeneration replaced Chinese electricity mix	WWTP in China	Around 40 km for all scenarios	Material substitutions were not considered

were innovative techniques (pyrolysis, wet oxidation, incineration in cement kilns) compared to the processes normally used in Europe (agricultural application, fluidised bed incineration, landfilling). Impact categories were not considered but technologies were analyzed only using CO_2 emissions and energy production. Based on CO_2 emissions alone, the oxidation technologies: incineration in fluidised bed, wet oxidation and incineration in cement kilns were better than the others. Use of sludge by incineration in cement kilns in particular had the least CO_2 emissions per ton of dried sludge treated. Whereas, landfilling had the most global warming potential over the entire lifecycle.

Hospido et al (2005) [42] compared environmental performance of anaerobic digestion with thermal processes: pyrolysis and incineration using LCA. Impact categories considered were: eutrophication, statospheric ozone depletion, global warming potential, acidification, photo-oxidant formation, depletion of abiotic resources and human toxicity. Both technologies had positive and negative aspects. For example, anaerobic digestion followed by agricultural application recycled the nutrients (mostly Nitrogen and Phosphorous) into the soil. On the other hand, energy recovery was low and heavy metals present in sludge had negative effects on the environment. Thermal processes provided energy recovery which lowered emissions by offsetting energy production from fossil fuels. However, nutrients were lost into forming other products. Hence no alternative had most favorable results for all categories. The study concluded that the best technology depends on the site conditions to a great extent. Also, land application should be done with minimization of heavy metal content and incineration should be combined with recovery of nutrients.

Murray et al (2008) [62] also conducted a hybrid LCA consisting of economic analysis along with environmental analysis of sewage sludge technologies in a Chinese context. Nine scenarios consisting of various technological options were considered: 1.Dewatering, landfilling, 2. Dewatering, lime stabilisation, agricultural use/cement production, 3. Mesophilic aerobic digestion, dewatering, agricultural use, 4. Aerobic digestion, dewatering, agricultural use 5. Dewatering, heat drying, composting, agricultural use, 6. Dewatering, heat drying, cement manufacturing, 7. Mesophilic anaerobic digestion, dewatering, heat drying, cement manufacturing, 8. Anaerobic digestion, dewatering, heat drying, 9. Dewatering, fluidised bed incineration, cement/clay brick manufacturing. Impact assessment was not carried out but environmental analysis was done by quantifying air emissions and energy production. Anaerobic digestion followed by agricultural application is the recommended pathway according to the authors. Reason behind this is, it offers energy recovery along with nutrient recycling to agriculture. Thus also offsetting the required fertilizers. Few conditions however have to be noted in this ideal process, 1. Lime is not required to stabilize pH of sludge 2. Methane does not leak from the electricity production process.

Hong et al (2009) [41] analyzed almost all possible sewage handling methods environmentally and economically in a Japanese context. These alternatives were divided into primary treatment and end of life application. Primary treatment included dewatering, composting, drying, incineration, incineration with ash melting and dewatered sludge melting. Each of these techniques were studied with or without anaerobic digestion. For the final use, landfilling, construction and agricultural applications were considered for the appropriate methods. For example, composting did not contribute towards construction while ash melting did. Overall, the application of digestion before the primary treatment proved to be beneficial environmentally. Impact categories considered were, global warming potential, acidification potential, human toxicity and land use. “Sludge melting” which basically is combustion of sludge at temperatures of 1300-1800°C has the lowest environmental impact. Economic analysis were carried out using Life cycle costing method. Digestion also performed better as a precursor to all handling processes due to decrease in sludge quantity to be handled. Energy recovery in form of heat and electricity production proved to be crucial in this analysis as technologies implementing this, performed significantly better. Incineration with melting was also economically the optimal technology with sludge melting close to it. Thus, this study established sludge melting as an “environmentally optimal and economically affordable” technology for sludge treatment.

Stokes and Horvath (2010) [83] conducted life-cycle assessment of a wastewater and sludge treatment plant in United states using wastewater energy sustainability tool (WWEST).

Sludge produced in the waste water plant underwent anaerobic digestion followed by dewatering and ultimately disposal partly by landfilling and partly by agricultural application. Biogas produced during digestion was utilized to produce electricity. Impact assessment was not implemented but energy use, air emissions and sensitivity analysis were calculated. Apart from identifying the contribution of each process to ultimate emissions and energy use, sensitivity analysis was also done using a hypothetical scenario. The only thing different in the scenario was to flare the captured methane instead of using it for electricity production. As expected, energy recovery substantially reduces environmental impact in all categories. Hence, this study emphasized the need to recover energy from sludge flow streams.

Xu et al (2014) [101] evaluated 12 different scenarios for sewage sludge treatment environmentally and economically using LCA and LCC respectively. These scenarios include: (a) Thickening, anaerobic digestion and landfilling, (b) Thickening, anaerobic digestion, dewatering and landfilling, (c) Thickening, anaerobic digestion, dewatering and incineration, (d) Thickening, anaerobic digestion, dewatering and agricultural use , (e) Thickening, anaerobic digestion and agricultural use, (f) Thickening, anaerobic digestion, drying and agricultural use. Rest of the scenarios included the above ones but without anaerobic digestion. These scenarios were ranked for 18 different impact categories which are: climate change, ozone depletion, human toxicity, photochemical oxidant formation, particulate matter formation, ionizing radiation, terrestrial acidification, freshwater eutrophication, marine eutrophication, terrestrial ecotoxicity, freshwater ecotoxicity, marine ecotoxicity, agricultural land occupation, urban land occupation, natural land transformation, water depletion, metal depletion, and fossil depletion. Scenarios with anaerobic digestion performed better in all categories except climate change. Anaerobic digestion also had the highest contribution to global warming potential due to direct release of green house gases. Overall, the authors singled out two scenarios which performed better environmentally than the rest. These were, 1. anaerobic digestion, dewatering followed by incineration and 2. thickening, anaerobic digestion and agricultural application. Although to limit heavy metal content in agricultural use, sludge application rate should be less than 0.33 t/hm^2 (tons of sludge per square hectometer) as found by Zhao [100] (cited in Xu et al [101]). LCC results also followed similar projections by favoring scenarios with anaerobic digestion. This was strongly attributed to reduction of sludge treatment volume by 50% due to digestion. Further, scenario with anaerobic digestion, dewatering followed by incineration was again the best performing economically.

Few important points were found and these are highlighted as follows. The optimal technology both environmentally and economically is highly dependent on local conditions like site conditions, sludge quality and distances between utilities. Energy recovery and land application from sludge substantially reduces environmental impact mainly because of fossil fuel use avoided. Incineration should be combined with nutrient recovery and land application should be combined with heavy metal minimization to get the best results.

Furthermore, important assumptions and system boundaries of the above mentioned studies were also noted. Important comments from this are as follows. All papers took into account the energy recovery in terms of heat and electricity which avoided the primary fuel source. It is also interesting to note that when fertilizer avoided due to land application of sewage was not considered, this scenario performed poorly. Unlike energy substitution, replacement of materials was not included by all studies. Construction of infrastructure had a negligible environmental impact but high economic impact. Impact of pathogens and organic pollutants was not considered by most of the studies.

1.8. Research questions

As we see from the literature review, plasma gasification and supercritical waster gasification have not yet been assessed using LCA. Both of these technologies are promising at first glance. However it is essential to investigate environmental impacts of these technologies as compared to the present scenario. Furthermore, both these gasification technologies are relatively new and are currently undergoing constant improvements. LCA will be able to provide valuable insights into improving environmental performance of these technologies while they are still in a developing phase. Thus this study aims to answer the following

research question.

”What combination of the sludge chain technologies amongst the ones considered result in the minimum environmental impact in the Netherlands?”

A number of sub-questions are also presented to help elucidate the main research question and to explicitly state information which might be valuable to improve the novel technologies or the present sludge chain:

1. How do different components namely, anaerobic digestion, thermochemical conversion, and electricity generation from syngas affect the environmental impact of a sludge chain?
2. What are the most environmentally intensive processes in the considered sludge chains and what can be possible recommendations to improve them?

Apart from finding the sludge chain with the lowest environmental impact, the focus of this study is also to identify the environmental ’hot-spots’ not only to improve the current sludge chain but also to recommend improvements for the novel technologies of plasma gasification and supercritical water gasification.

2

Methodology

2.1. General structure of LCA

There are two types of LCA, attributional and consequential. Attributional LCA (ALCA), also known as descriptive LCA is 'defined by its focus on environmentally relevant physical flows to and from a lifecycle and its subsystems' [27]. On the other hand, Consequential LCA (CLCA), also known as change oriented LCA is defined by its aim to describe how environmentally relevant flows will change in response to possible decisions. CLCA differs from ALCA in two ways. First, the co-product allocation[1] performed by using a normative rule in ALCA is avoided by using the system expansion method. System expansion is performed by defining an avoided product and consequently saving the corresponding environmental impact [34]. Secondly, the inclusion of processes outside the system boundaries that are expected to be affected by a change in demand [34]. If CLCA methodology is selected, additional impacts of certain materials in the secondary markets have to be taken into account as well. For instance, if a technology envisages reduced use of limestone, environmental impact due to possible increase in the use of limestone in other markets have to be considered as well. A CLCA is thus more complex since it includes additional concepts like marginal production costs, elasticity of supply and demand, rebound effects, etc [27].

Even though CLCA appears to be a superior methodology, ALCA can be sufficiently used for identifying hotspots and product declarations. Whereas CLCA is preferred for long term strategies such as product development or policymaking [95]. Additionally, due to time constraints of this project, ALCA methodology was a more feasible option.

The primary source of guidance for the methodology of LCA is the Handbook developed by Institute of Environmental Sciences at University of Leiden where a detailed account of this chapter can be found [35]. LCA is carried out in four phases as outlined in figure 2.1. It starts with goal and scope definition followed by inventory analysis, impact assessment and ultimately, interpretation. Usually in a LCA, each of these phases are most probably reiterated based on the feedback from one or more of the following steps. These steps are elaborated as follows:

Goal and Scope Definition

In this phase, the entire structure of LCA is decided based on some initial choices. As a starting point, the goal of the study is formulated in terms of the exact research question, target audience and intended application. Additionally, a description of the scope of research in terms of temporal, geographical and technological coverage along with a level of detail of the study.

Hence defining the research question will be the initial step of a LCA. Then based on its intended application, the type of LCA, consequential or attributional is selected. Intended application of a product, service or a process chain thus also makes the main function clear.

[1] Allocation is a method used in LCA to distribute emissions for processes which satisfy more than one function. It is defined as "partitioning the inputs and outputs of unit processes among product systems" [34]

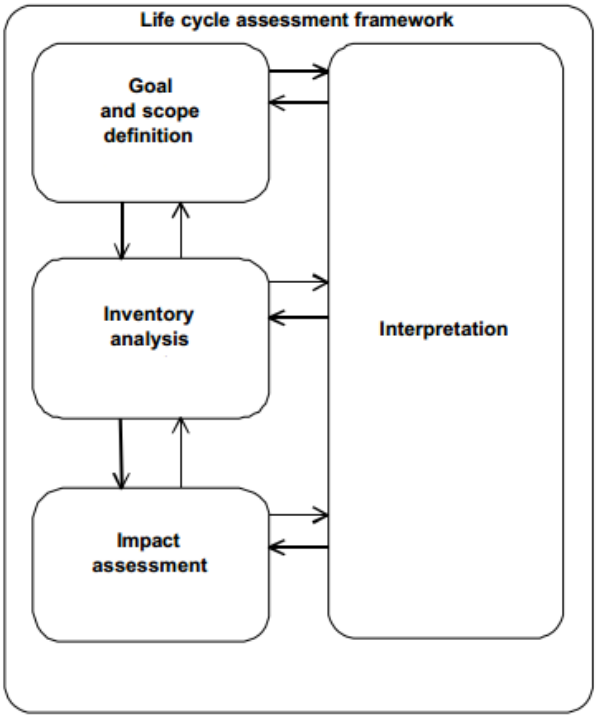


Figure 2.1: Methodology of Life-cycle Assessment [35]

Secondary functions are identified and included through allocation. Thus the reference flow of the system and subsequently the functional unit (FU) is identified. The functional unit enables us to compare different systems which serve the same function and determine the respective functional flows. The next step is defining the temporal, geographical and technological boundaries of the research. The technological boundaries enlist the considered technologies and production stages. Lastly, the temporal and geographical scope illuminates the relevance of the collected data in terms of time and geography.

Inventory Analysis

After enlisting different product systems, process flow diagrams and system boundaries are outlined as a part of inventory analysis. These boundaries are presented as:

- Boundary between the system and environment
- Boundary between the relevant and irrelevant processes (cut-offs)
- Boundary between the system considered and other systems (allocation)

Taking these boundaries into account, flow diagrams are drawn with unit processes and data is collected for each system. Depending on the level of detail, unit processes can be simplified into one process or divided into numerous processes. Each and every flow in the lifecycle of the system is followed until its economic flow is translated into environmental interventions. The flows crossing the boundary between the economy and the environment are termed as environmental flows.

Since LCA involves the collection of large amount of data, data quality is a crucial step. Along with reliability and validity, the data should be representative of the product system. The collected data is structured around unit processes within a certain time period. Multifunctional processes are defined as unit processes performing more than one function, that is, producing more than one good, processing of more than one type of waste or a combination of previous two. For multifunctional processes, allocation performed. Different methods of allocation are available, for example, economic, energy-content, mass, exergy, etc. Typically the inventory analysis phase is the most time consuming and iterative step of any LCA.

Impact Assessment

This phase, also known as lifecycle impact assessment (LCIA) concerns the conversion of inventory analysis results to environmental impacts. A list of impact categories are defined and models for category indicators are selected. The impact categories are related to depletion of resources, impact on human health and impact on ecosystems. Various models for these impact categories are available for selection. Furthermore, category indicators for the selected impact categories such as depletion of abiotic resources, global warming potential, marine ecotoxicity, etc. have to be selected in relevance to the goal and scope of the study.

For example, under CML-IA midpoint approach for characterization, climate change is an impact category and global warming potential is a category indicator. For global warming potential, all the emissions from the inventory analysis to air are considered and multiplied by a characterisation factor to get the indicator result. For example, if 1 kg CO_2 emitted with a characterisation factor of 1, result will be 1 kg CO_2-eq. However these characterisation factors vary for different compounds. If 1 kg of CH_4 is emitted, with a characterisation factor of 25, result will be 25 kg CO2-eq and so on.

A step of normalization is used to better understand the relative magnitude for each indicator of the system under investigation. Normalization helps in checking inconsistencies and significance of results. It is defined as “calculation of the magnitude of indicator results relative to reference information”. For each alternative studied, impact scores are divided by the corresponding normalisation factors. Normalisation data is based on a community (Germany, Asia or World) or a person (Dutch person) or other system over a given period of time. This step is also mandatory if weighting is to be performed.

Weighting is the last but an optional step of impact assessment. Normalised indicator results are multiplied by numerical factors according to their relative importance. These numbers are added to obtain a single score for each system alternative.

Interpretation

Lifecycle interpretation involves the evaluation of results, assumptions and choices in terms of soundness and robustness followed by drawing overall conclusions. Robustness of data, assumptions and cut-offs can be evaluated using the following tools:

- Consistency check

  The objective is to determine whether assumptions, methods, models and data of each alternative are in-line with the goal and scope of study. Based on the knowledge of LCA experts or previously conducted LCA studies on related subjects.

- Completeness check

  The objective here is to ensure all relevant information and data needed for interpretation phase are available and complete. The LCA study is checked for false assumptions, model choices and data through LCA experts or previously conducted LCA studies.

- Contribution check

  In contribution analysis contributions of processes or specific environmental flows to the total environmental score are usually expressed as a total of 100%. Apart from recognising any inconsistencies, this step is also useful to identify ‘hotspots’.

- Perturbation analysis

  Perturbation analysis is the study of effects of small changes within the system on the final results. This analysis can be conducted on levels of inventory table, environmental profile, normalised environmental profile or weighting results depending on data quality.

- Sensitivity and uncertainty analysis

  The objective and procedure of this step is the same as perturbation analysis. However, in this step, uncertainty due to modeling choices like functional unit, allocation, geographical assumption, etc. are varied instead.

Ultimately, conclusions are drawn and recommendations are made for the target audience of the study based on the information and results gathered.

2.2. Goal and Scope definition

The goal of this study is to compare the environmental performance of different sewage sludge process chains. This study also focuses on finding the most environmentally intensive components in the process chain and suggest possible measures to improve the same. Additionally, this study will fill the literature gap with lifecycle analysis study of novel technologies, supercritical water gasification and plasma gasification and potentially assist in their development.

The sophistication of the study of the study lies between a detailed LCA and a simplified LCA. The current scenario of sewage sludge processing chain in Netherlands is considered as a reference case. Even though the technologies of supercritical water gasification and plasma gasification are not relatively mature, emphasis was on collecting commercial or experimental data rather than potential or expected performance values. Hence as much as possible, latest data is collected.

The most common sludge chain scenario in the Netherlands as presented in Chapter 1 is considered to be the system zero. Based on the literature survey and development of novel technologies presented, three components are varied in the sludge chains:

1. Presence of Anaerobic digestion
2. Supercritical water gasification or Plasma gasification
3. Energy production from syngas using Gas Engine or Solid Oxide Fuel Cell (SOFC)

Even though anaerobic digestion is a mature technique, its influence with respect to gasification technologies is still unknown. Additionally, these technologies are operated with different dry solids (DS) content of sludge, for example, for mono-incineration usually sewage sludge at 23-25% DS straight after dewatering is fed in Netherlands. For SCWG, processing of sewage sludge at around 12-13% DS has been demonstrated while for plasma gasification this value is approximately 90%. Hence the dewatering and drying steps of respective chains can change. The 9 systems considered can be listed as follows:

0. Anaerobic digestion → Incineration
1. SCWG → Gas Engine
2. Anaerobic digestion → SCWG → Gas Engine
3. Plasma Gasification → Gas Engine
4. Anaerobic digestion → Plasma Gasification → Gas Engine
5. SCWG → SOFC
6. Anaerobic digestion → SCWG → SOFC
7. Plasma Gasification → SOFC
8. Anaerobic digestion → Plasma Gasification → SOFC

Please note that names of above systems do not completely signify either the chains or system boundaries. They are for representation purposes only and the system components may change drastically. Figure 2.2 shows an illustration of these systems. This is not a flow diagram, it will be presented with inventory analysis in the next chapter.

Even though energy can be recovered, sewage sludge has a negative economic value because of chemicals required at different points of the processing systems and hazardous waste produced. Hence it is classified as a waste. Consequently, the primary function of the above systems is the treatment of sewage sludge. As we have seen in chapter 1, sludge is produced at around 1% DS which increases up to 90% DS before being gasified for the case of plasma gasification. To maintain continuity in the chain and perform easy mass balances, processing of 1 ton DS sludge was selected as the functional unit. Amongst the reviewed literature, the same functional unit was used unanimously.

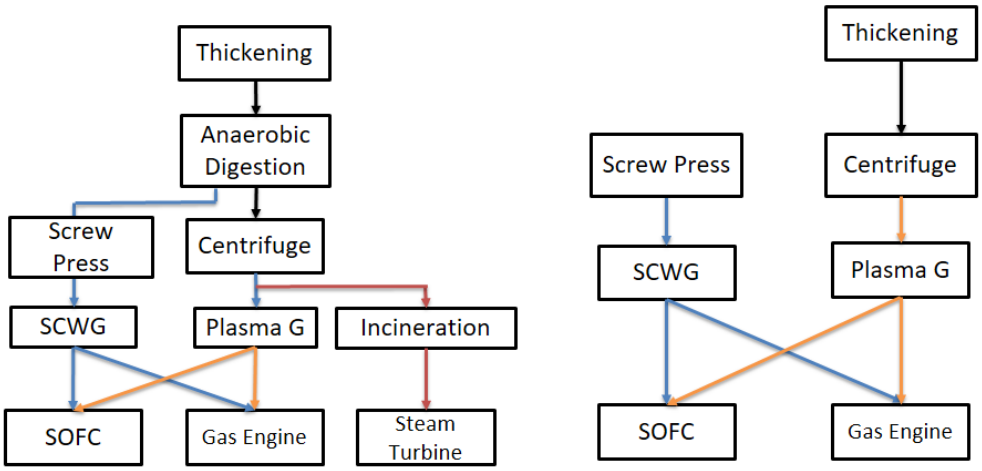


Figure 2.2: Sewage treatment models included in the scope of this study

2.3. Impact Assessment Methodology

The impact assessment method used is CML-IA developed by the University of Leiden. There are two reasons on why this approach was selected. Firstly, it offers a comprehensive overview of impact categories. And secondly, midpoint approach was found to be better suited to the goals and audience of this report. Mainly because endpoint approach involves large uncertainties [15] thus when comparing techniques and identifying hot-spots, midpoint seems an appropriate approach.

In order to accurately calculate the environmental impact of a complex problem like sludge treatment, it is also essential to select the impact categories. The selection of impact categories was based on the results of material flows from inventory analysis and relevance to the goal of this study. It was decided to include all the impact categories as suggested by CML-IA method. Furthermore, these categories are also the most commonly selected impact categories by researchers conducting lifecycle assessments of sewage sludge [103]. The impact categories and the normalisation/weighting steps are elaborated as follows:

Acidification

Acidifying pollutants have a number of impacts on ecosystems, surface waters, soil and biological organisms. As the name suggests, these compounds are transformed into acids and deposited in the environment [13]. This effect is measured in terms of kg SO_2 equivalent units, with the main pollutants being SO_2, NO_x and NH_3. The transportation and energy sectors contribute the most to these emissions. Due to sulphur/nitrogen in fuel or due to high temperature combustion (NO_x formation) [97]. Since the systems considered have combustion technologies, acidification has to be included.

Eutrophication

Eutrophication is defined as the 'excessive richness of nutrients in a lake or body of water leading to dense growth of plant life' [35]. The most important nutrients are Nitrogen and Phosphorous. High concentrations of such nutrients can make fresh water unacceptable for drinking and also cause detrimental effects to the aquatic life. Along with nutrients, some organic compounds released in water bodies can also have the same effect and are thus considered in this impact category. Eutrophication potential is measured in terms of kg PO_4 equivalent units. Water treatment mandatory in case of SCWG makes it important to consider this category.

Climate Change
Climate change is defined as the long term increase of the earth's temperature due to emissions as a result of anthropogenic activities [35]. Public awareness around this impact category is probably the highest because the detrimental long term effects it may have. It is measured in kg CO_2 equivalents for a 100 year time horizon in this context (GWP100). Major contributing emissions are CO_2, CH_4 and N_2O. Even though carbon in sludge is biogenic, emissions from other materials and activities to climate change is substantial.

Ecotoxicity
This category includes impact of toxic compounds on aquatic and terrestrial ecosystems. It is measured in terms of kg 1,4-dichlorobenzene (DB) equivalents. Furthermore, this category is subdivided into: Terrestrial Ecotoxicity, Fresh Water Aquatic Ecotoxicity and Marine Aquatic Ecotoxicity. Heavy metal emissions resulting from hazardous ash disposal may have a significant impact here.

Human Toxicity
Impacts on human health of toxic substances present in the environment are included in this category. It is measured again in terms of kg 1,4-DB equivalents. Emissions to soil, air as well as water are considered however, it is not further divided into specific categories. Heavy metal emissions expected from hazardous waste disposal in the presented systems can present a risk to human health.

Ozone depletion
Ozone depletion refers to weakening of ozone layer due to human activities. As a result, there is a larger proportion of UV-B radiation entering the atmosphere which causes damage to human health and entire ecosystems. This effect is measured in kg CFC-11 equivalents. Naturally CFC emissions and other halogens are the major contributors.

Photochemical oxidation
Photochemical oxidation concerns the formation of reactive compounds due to influence of sunlight on air pollutants. Reactive compounds like ozone and nitric oxides are created from volatile organic compounds (VOC's) and nitrogen oxides. This phenomenon is also known as summer smog since it often occurs in urban areas during summer days. Ozone and peroxyacetylnitrate resulting from this smog are injurious to human health and vegetation. Impact is measured in terms of kg ethylene (C_2H_4)equivalents with major contributors being NO_x, VOCs, CH_4 and CO.

Normalisation and Weighting
Normalization is carried out using data for a person in the EU25 countries as included in the CML-IA method [35]. Weighting factors developed for the Netherlands Oil and Gas Exploration and Production Association (NOGEPA) by a group of scientists have been used here [46]. This method was developed according to ISO14042 standard and the initial weights along with the final weights have been presented in table 2.1. Since resource depletion was not included in the initial weight, an average value was obtained from EPA and BEES weights and then factors were recalculated to 100%. This methodology is inspired by a similar methodology is used for the evaluation of different weighting methods in a report for the European Commission [45]. In the report, more than 1 impact categories were added to the standard NOGEPA weights and adapted factors were calculated in the same way [45].

Table 2.1: Initial and adapted weighting factors used in this study [46].

Impact Categories	NOGEPA [46]	NOGEPA with resource depletion	NOGEPA adapted
Acidification	6	6	5.6
Eutrophication	13	13	12.2
Climate Change	32	32	30
Ecotoxicity Fresh water aquatic Marine aquatic Terrestrial	 6 8 5	 6 8 5	 5.6 7.5 4.7
Human Toxicity	16	16	15
Ozone Depletion	5	5	4.7
Photochemical Oxidation	8	8	7.5
Resource Depletion	-	7.5	7
Total	99	106.5	100

2.4. System Boundaries and Allocation

The system boundaries from the system alternatives introduced in section 2.2 are presented here. These process flow diagrams are simplified versions of sludge chain. Next chapter contains a detailed view of each of these unit processes.

The following list assists in reading the flow diagrams:

- Only few selected inventory results are presented in the figures.
- The reference flow of sludge processing steps is highlighted in red colored arrows.
- Electricity input and output is colored in green. Electricity mix of Netherlands from Ecoinvent database is used.
- All materials whether it be minerals or chemicals, are lumped together for the sake of simplicity for the audience and are presented in orange. Production processes of these materials are obtained from Ecoinvent database.
- Transport processes, colored in black, are also used from Ecoinvent.
- Finally, the wastewater treatment flow considered only for supercritical water gasification, is colored in blue

Allocation

According to ISO standards of conducting LCA, allocation should be avoided wherever possible [35]. In-line with this advice, in most of the systems, allocation is avoided by system expansion. Hence in inventory analysis, majority of the processes contain materials as 'avoided products'.

However, allocation was necessary for the anaerobic digestion process. Since sludge is partially converted to biogas, thus according to the reference flow, it is 'processed' although partially. Allocation method considered here is based on the energy content. With 271 Nm^3 of biogas produced with a calorific value of 22 MJ/Nm^3, the remaining volatile matter decreases from 75% to 63%. Volatile matter is assumed to have a calorific value of 21.318 MJ/kg [65]. Hence emissions are allocated as 39% to the biogas and 61% to the digestate. Biogas will be further processed to recover energy while digestate will be subject to a thermochemical conversion process. Robustness of this allocation method will be checked in chapter 4.

Reference Case: Anaerobic Digestion → Incineration

The most common Dutch sludge chain as found from the literature review is outlined in this section. As presented in figure 2.3, sludge produced from the treatment of wastewater is thickened from 0.81% to 4-5% DS depending on the type of sludge. Anaerobic digestion is the next step which is followed by centrifugal dewatering. Sludge is then transported to a final treatment plant. Here sludge is first pre-heated using residual heat to about 37% DS and then fed to a fluidized bed combustor. Ash residue from sludge along with mineral waste produced from treatment of fluegas is either used as a foundation in Asphalt or transported to germany where it is landfilled in salt mines.

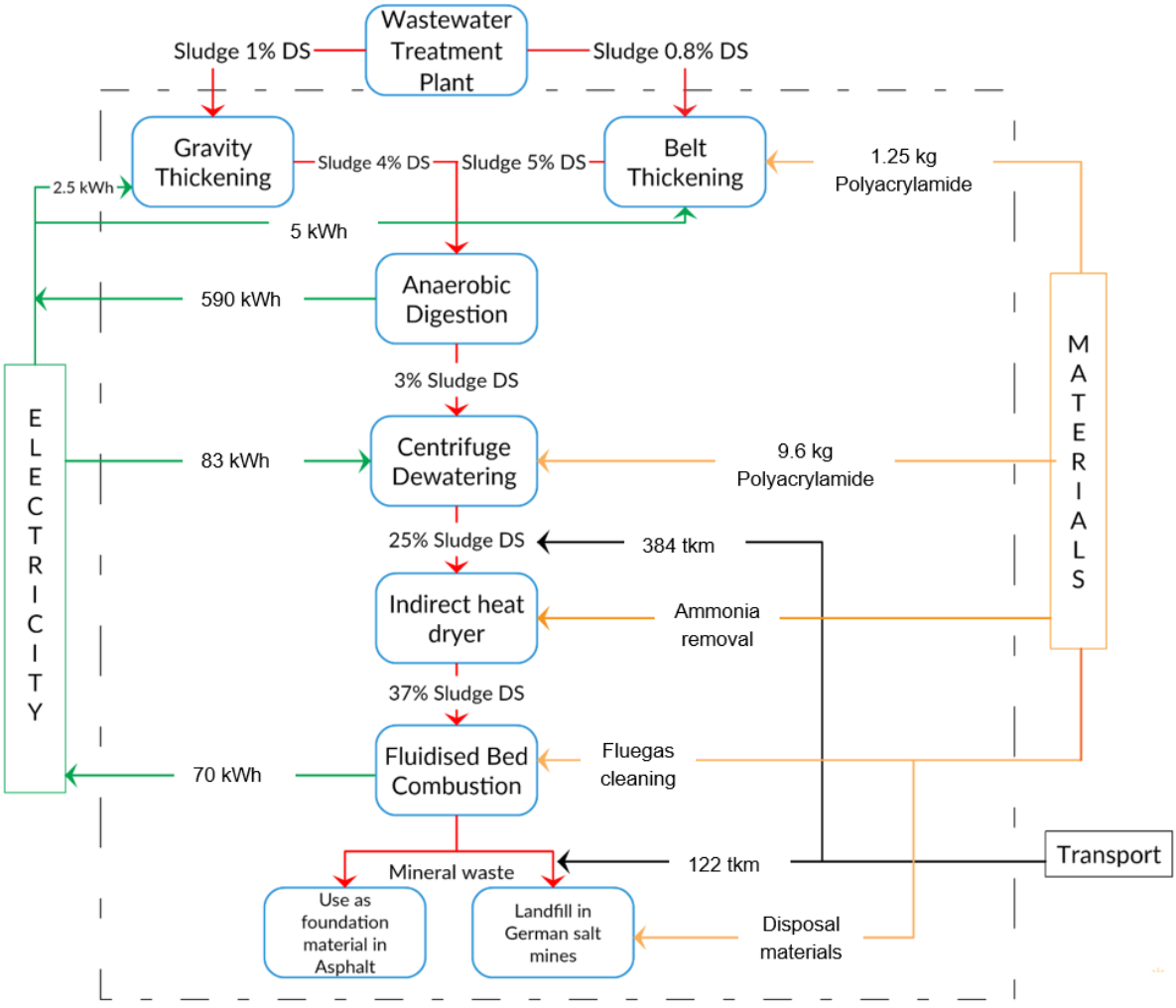


Figure 2.3: System boundary for reference system of Anaerobic Digestion of sewage sludge followed by Incineration.

Systems 1 and 5: Supercritical water gasification of raw sludge followed by Gas Engine/SOFC

This system as outlined in figure 2.4 starts at wastewater treatment plant where primary and secondary sludge are produced. The required dry solids content is however only 13%. Hence dewatering method used here is screw press. This relatively wet sludge is then transported to a SCWG plant where energy is recovered either using a gas engine or a solid oxide fuel cell. SCWG is the only technology where wastewater treatment is considered because an effluent stream with organic matter, salt and ash is produced. In contrast, other technologies require only Nitrogen removal from the condensate is required after which, water is drained to sewage [66]. Ash residue from sludge is again either used in road construction or landfilled in German salt mines.

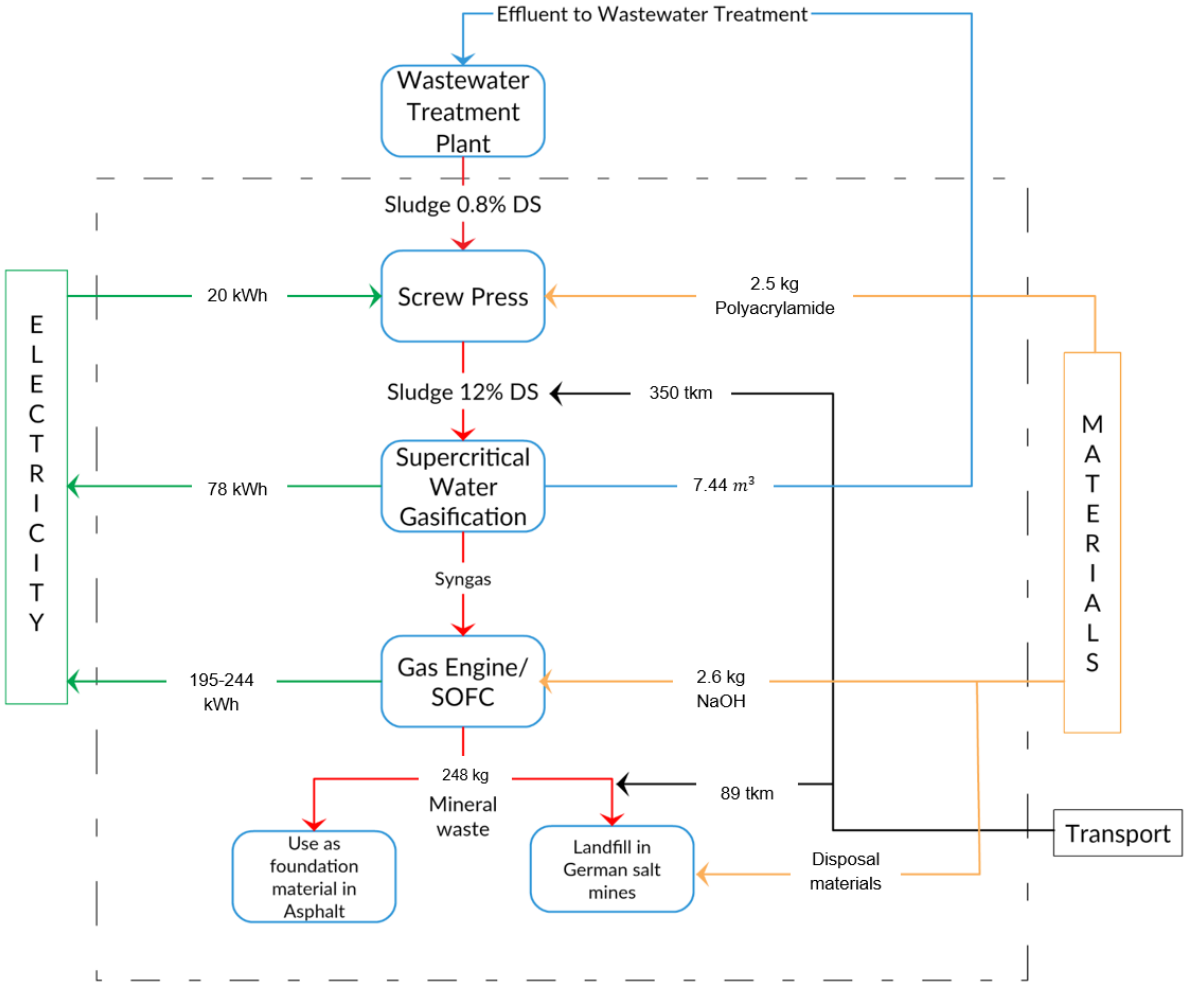


Figure 2.4: Systems 1 and 5: System boundary of supercritical water gasification of raw sludge followed by energy generation Gas Engine/SOFC.

Systems 2 and 6: Supercritical water gasification of digested sludge followed by Gas Engine/SOFC

Effect of anaerobic digestion in combination with SCWG is studied in this section as shown in figure 2.5. Because of addition of an anaerobic digestion step, sludge produced is first thickened and then digested. Heat and electricity are produced from cogeneration of biogas. Electricity is either used for operations in the wastewater plant or exported while the heat is used to maintain temperature of the digester and for desulphurisation unit. Residual thermal energy is lost. Since a dry matter content required again is only 13%, Screw press is used for dewatering instead of energy intensive centrifuges. Energy is recovered from the resulting syngas using gas engine or SOFC. Effluent is again sent to an on-site wastewater treatment plant.

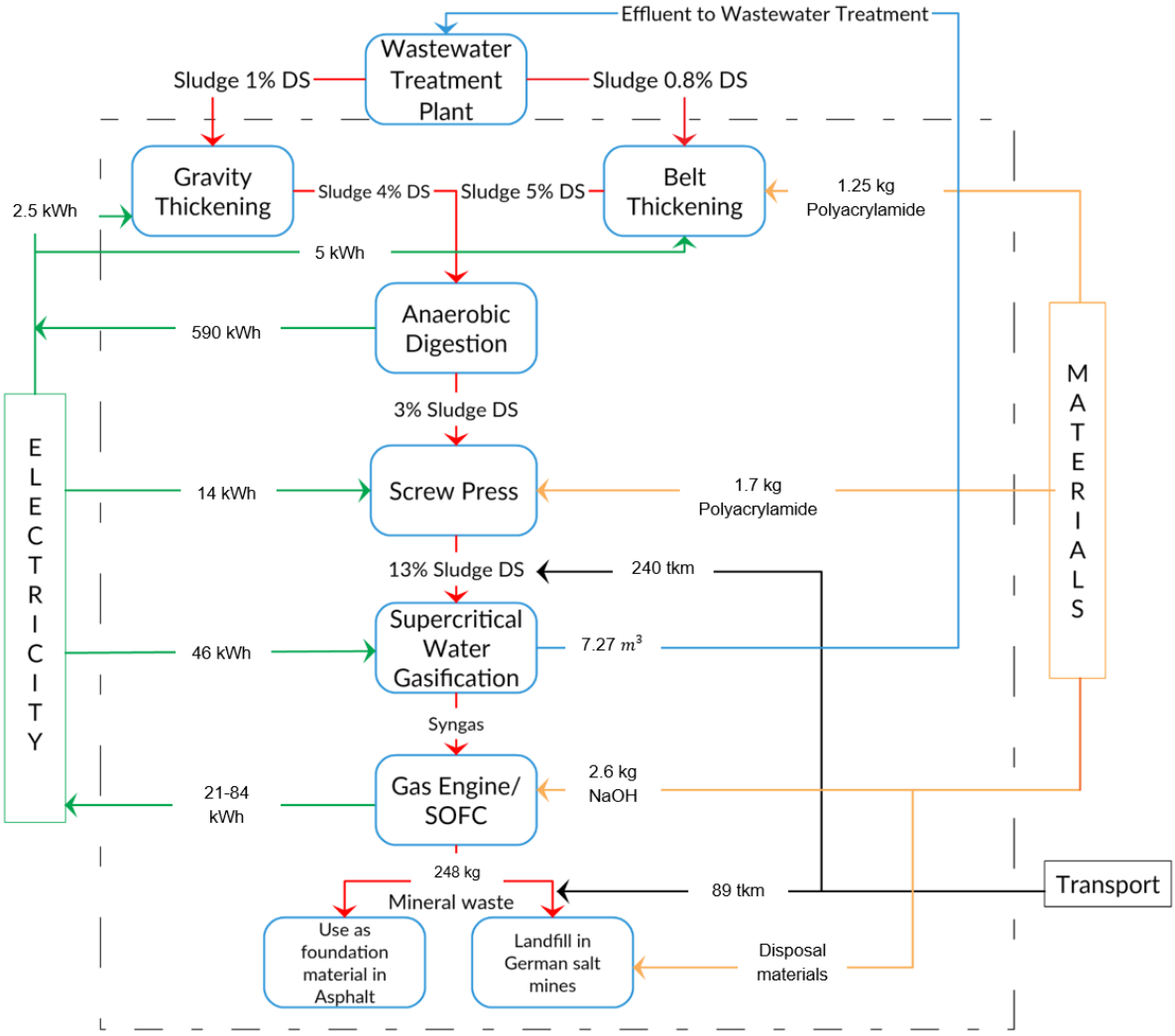


Figure 2.5: Systems 2 and 6: System boundary for supercritical water gasification of digested sludge followed by energy generation in Gas Engine/SOFC

Systems 3 and 7: Plasma gasification of raw sludge followed by Gas Engine/SOFC. Plasma gasification of dry sludge is considered in these systems as shown in figure 2.6. Primary sludge and secondary sludge are thickened using gravity and belt thickening respectively. Thickened sludge is then fed to centrifuges for dewatering. Dewatering results in a dry solids of approximately 25% which is not enough for sludge to be fed to a plasma gasification reactor. Hence an extra drying step is added where sludge is dried to 90% DS using natural gas. Syngas obtained is used to generate energy either in a Gas Engine or SOFC. Hazardous waste produced is drastically reduced in this technology. Majority of the mineral waste is converted to slag which is assumed to be used in manufacturing ceramic tiles. Whereas the hazardous waste is transported to Germany for underground landfill.

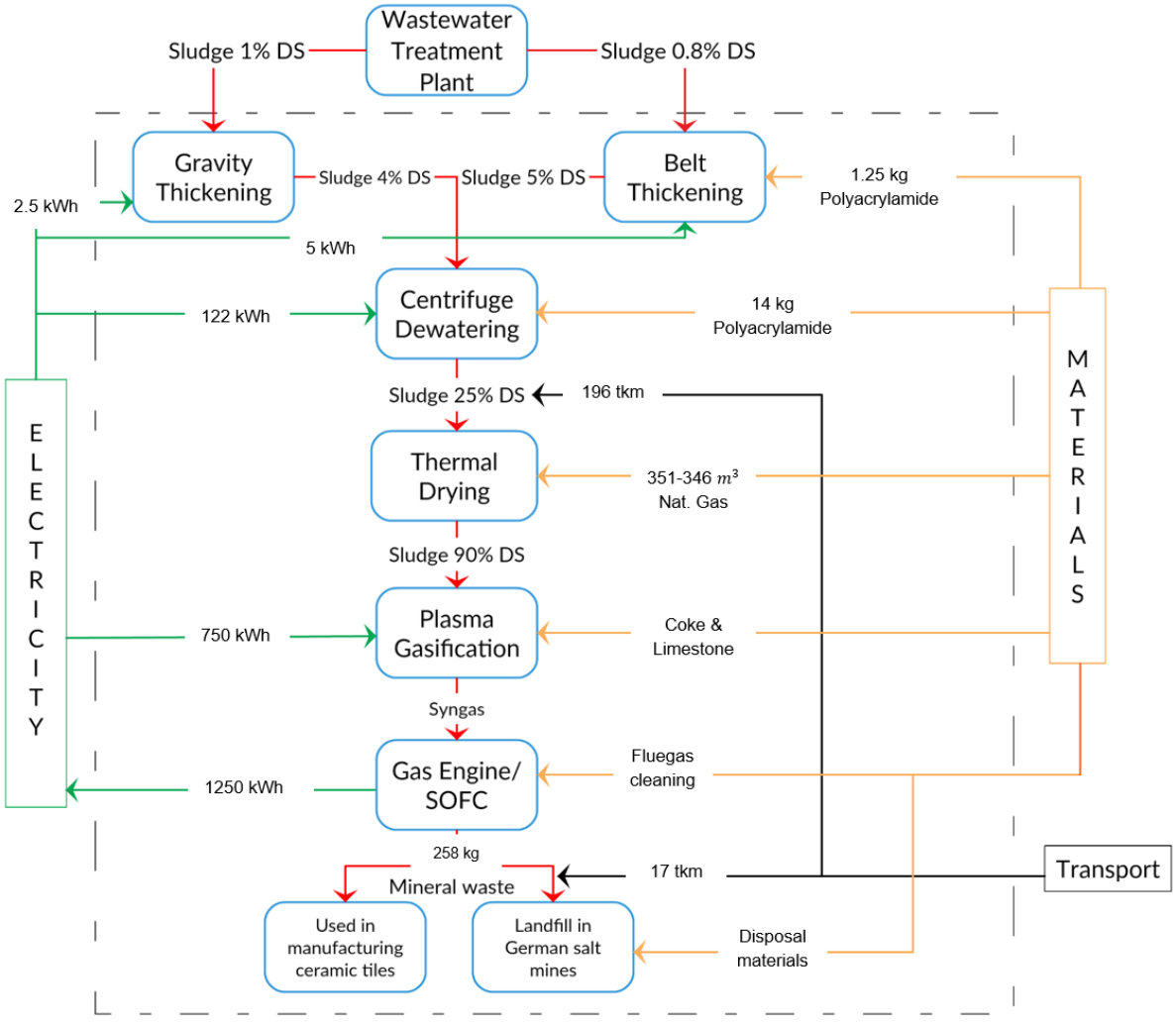


Figure 2.6: Systems 3 and 7: System boundary for plasma gasification of raw sludge followed by energy generation in Gas Engine/SOFC.

Systems 4 and 8: Plasma gasification of digested sludge followed by Gas Engine/SOFC

Plasma gasification with a step of anaerobic digestion is considered in this section as shown in figure 2.7. Gravity and belt thickening similar to previous steps are carried out. Similarly, anaerobic digestion step produces energy while reducing the solids content of the sludge. Dewatering and drying steps are included before sludge is fed to plasma reactor. Resulting syngas and mineral waste are treated in the same way as in the previous plasma system without the digestion step.

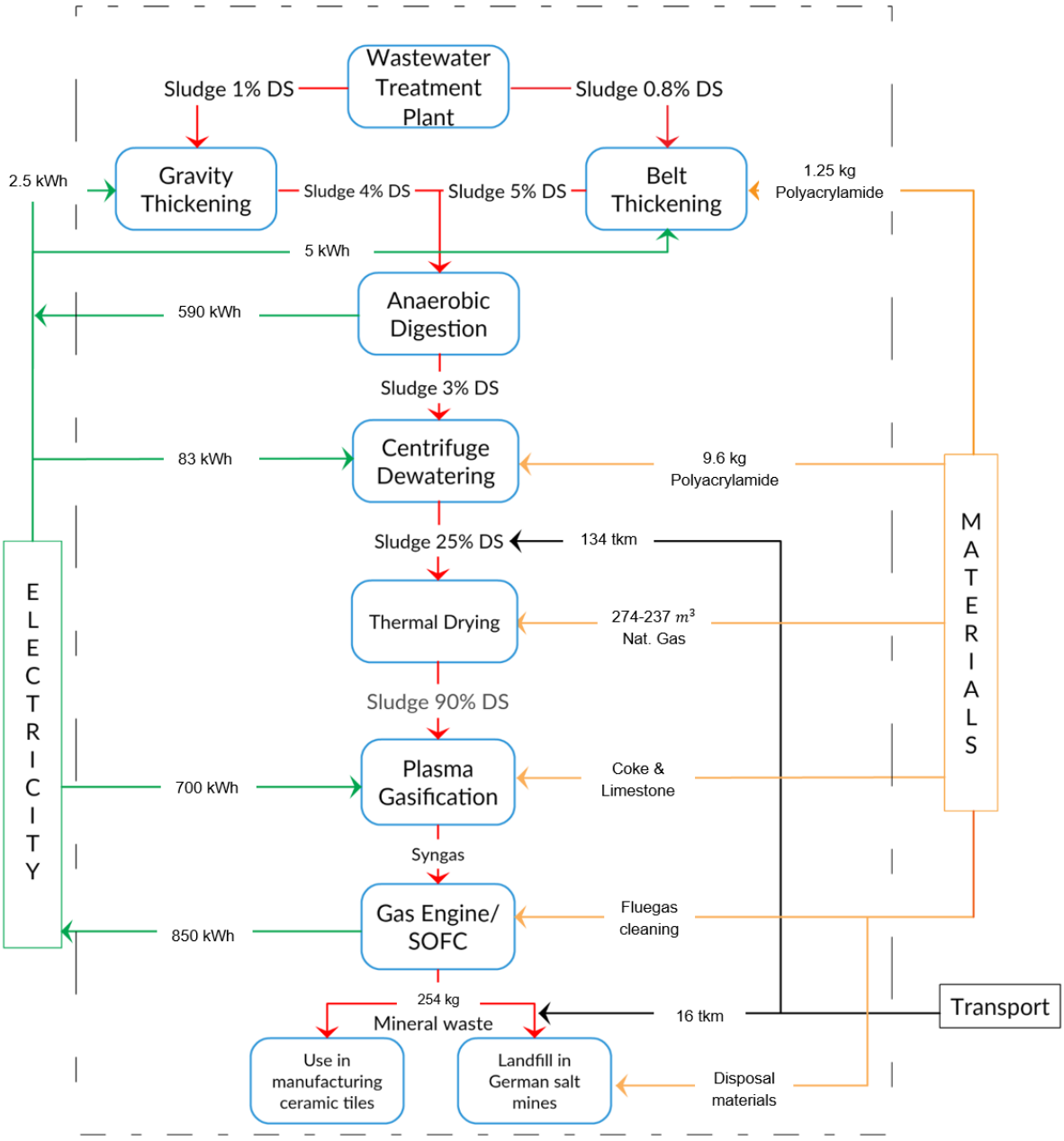


Figure 2.7: Systems 4 and 8: System boundary for plasma gasification of digested sludge followed by energy generation in Gas Engine/SOFC.

3

Inventory Analysis

The second and also the most time intensive step of this study is the inventory analysis. In this step, details of the considered unit processes, assumptions, calculation methods and data regarding all systems is presented. Since there are 9 alternative systems with many unit processes appearing in more than one systems, the data regarding of different systems is presented under the heading of the same unit process. For example, Dewatering presented in section 3.4, at first the technology description is provided followed by calculation details and process data of each of the 9 alternatives and its data sources.

Unit processes were modeled in Excel according to the data collected. This data was scaled according to the FU of 1 ton DS sludge processed and consequently fed to the 9 modeled systems in SimaPro© for impact assessment. Inputs and outputs with a '[E]' symbol next to them mean they are the background processes taken from the Ecoinvent database. The exact input/output data of these processes is modeled but the data related to their lifecycle emissions is provided by Ecoinvent. The full name and details of all these processes can be found in Appendix A.

3.1. Collection of sludge data

Although it is not a unit process, sludge data is still crucial, a separate section is added here to show the contents of sludge. Most of the literature sources don't clarify whether the sludge was subjected to any pretreatment or if they utilize sludge post aerobic or anaerobic digestion or they contain incomplete sludge analysis. Sludge data selected in this report is from a wastewater treatment plant in Oijen since it answers all the above concerns. Four sludge samples were available [48], hence an average sludge composition was is shown in table 3.1. The data was converted into proximate and ultimate analysis with calculation details shown in Appendix B.

Table 3.1: Proximate and Ultimate Analysis of the sludge data.

Primary sludge	**wt. (%)**
Dry matter	1
Moisture	99
Ash (db*)	24.75
Organic (db*)	75.25

Secondary sludge	**wt. (%)**
Dry matter	0.8
Moisture	99.2
Ash (db*)	24.75
Organic (db*)	75.25

Element	**wt. (% daf^)**
C	55.44
H	8.23
N	6.03
S	0.80
O	29.19
Cl	0.30
Total	100

Metal Oxide	**wt. % of ash**
SiO_2	25.75
Al_2O_3	10.49
CaO	10.18
P_2O_5	39.79
Fe_2O_3	9.64
MgO	2.02
Na_2O	0.54
K_2O	1.58

**db - dry basis, ^daf - dry ash free basis*

Primary sludge is produced at 1% DS whereas secondary sludge is produced at 0.8% DS as described in sludge digestion handbook for the Netherlands [65]. Trace metals are also important especially for leaching characteristics but their composition is not modified for further modeling. Data regarding heavy metal content can be found in table B.1. The produced sludge is considered to be a 50-50% mixture of primary and secondary sludge. Primary sludge is produced by screening wastewater whereas secondary sludge is produced from advanced water treatment processes such as activated sludge process [88]. Sludge composition is kept the same for primary and secondary sludge, however the thickening methods and consequently the emissions vary slightly for each of them.

3.2. Thickening

Overview

Thickening is the first water removal step. Depending on the type of sludge, technology for thickening is selected. For primary sludge, gravity thickening is preferred whereas for secondary sludge, mechanical belt thickening is preferred same as at wastewater treatment plant WWTP Amsterdam-West built in 2005 [16].

Gravity thickening works on the principle of sedimentation also known as gravity settling of solids. It consists of a circular tank with thickened solids collected at the bottom and dilute liquid leaving from the overflow. Electricity is required for gentle agitation, hence power consumption is low [94]. In a belt thickener, sludge is fed onto the filter belt and transported towards a discharge point. The water drains off through the filter belt by gravity and the sludge flocs are collected at the end of the belt [7]. Power consumption is thus slightly higher and coagulants are also used for belt thickening.

Inventory

Information about energy and material consumption during thickening is collected from a German thickener manufacturer, Huber Technology [7]. The electricity consumption for gravity and belt thickening is 5 kWh/ton DS and 10 kWh/ton DS respectively. Belt thickening also uses polyacrylamide as a coagulant to facilitate thickening. Since this step is the same for all systems, only one inventory table applies to all systems. These results scaled to the functional unit are shown in table 3.2. Furthermore, this data was also verified from the models used by STOWA [98].

Table 3.2: Inventory table for thickening of mixed sludge.

Economic Output	**Quantity**	**Source**
Thickened Mixed Sludge	1 ton DS	Reference flow
Energy/Material Used		
Polyacrylamide [E] (Belt thickening)	1.25 kg	Huber Technology [7]
Electricity [E] (Gravity thickening)	2.5 kWh	
Electricity [E] (Belt thickening)	5 kWh	
Economic Input		
Primary Sludge	500 kg	Functional Unit
Secondary Sludge	500 kg	

3.3. Anaerobic Digestion

For simplicity of the models for the audience, many unit processes which were modeled differently are presented together. This section is one such example. Apart from anaerobic digestion of sludge, this section also contains information about gas cleaning post-digestion and energy generation through combustion of biogas.

Overview

Anaerobic Digestor

Anaerobic digestion concerns the anaerobic decomposition of organic matter into gas and a stabilized residue with the help of micro-organisms [94]. Along with sludge, anaerobic digestion has also been applied to a variety of organic wastes for a long time. Making this a mature technology with best practices and expected results readily available. Apart from presenting an opportunity to recover energy from combustion of methane rich biogas, it also offers a number of advantages as mentioned in section 1.4. The mechanism of anaerobic digestion can be described as the following successive steps as elaborated in Biogas Handbook [8] by Seadi et al:

1. Hydrolysis - This is a slow rate determining step. Here complex organic matter containing polymers are decomposed into smaller monomers. Thus very little biogas is produced during this step. Carbohydrates, lipids, nucleic acids and proteins, classified as complex compounds are converted to fatty acids, monosaccharides and amino acids at the end of this step.

2. Acidogenesis - Acidogenic or fermentative bacteria convert the products of hydrolysis into acetate, carbon dioxide, hydrogen, volatile fatty acids and alcohols.

3. Acetogenesis - Methanogenic bacteria converts products from previous steps to methanogenic substrates like acetate, hydrogen and carbon dioxide. This step acts as a precursor to methanogenesis and products from acidogenesis which cannot be directly converted to methane are relevant for this step.

4. Methanogenesis - Production of methane and carbon dioxide from intermediate steps occurs in this step. Methanogenic bacteria again facilitates this conversion. This step is critical since it is again a slow process with conditions like pH, temperature, feedstock, etc. heavily influencing biogas production.

Biogas predominantly contains methane and carbon dioxide with small quantities of hydrogen sulfide, hydrogen, water and ammonia. Important factors influencing gas production and composition are temperature, feedstock, pH and toxicity.

Gas Cleaning

Before biogas can be utilized for energy conversion, levels of ammonia, hydrogen sulfide, and water vapor have to be regulated. Even though hydrogen sulfide and ammonia can be combusted along with the biogas, they result in the formation of sulfur dioxide and nitrous oxides [96]. They negatively affect human health and environment. Hence combustion plants have to adhere to set emission limits. In combination with water vapor these compounds may also cause corrosion in pipes and engine parts. Consequently, it is also desirable to remove water vapor as much as possible [98].

Energy production

Combined heat and power generation (CHP) of biogas obtained from anaerobic digestion is a standard practice in many countries including the Netherlands [98]. This is performed by burning biogas in internal combustion engines followed by heat recovery. Thus a high overall efficiency of around 90% can be achieved with a heat efficiency of 45% and electrical efficiency of 37-42% reported [65]. Gas-Otto motors working on Otto cycle commonly utilize diesel as a fuel [8]. These Engines are modified in terms of air to fuel ratio to utilize biogas instead. These engines are also the most commonly used. Electrical efficiencies even upto 45% can be achieved if a steam Rankine cycle is employed. However one drawback of the latter technology is, its financial feasibility only for wastewater treatment plants with a capacity of 100,000 population equivalents or higher [98].

Inventory

Anaerobic Digester

Thickened sludge enters the anaerobic digester. Sludge composition remains the same as in table 3.1, however, the DS content increases to 4.4% and volume decreases by about 80% as seen in table 3.3. Sludge temperature is maintained at 35°C and the retention time is assumed to be 30 days. Electricity is required for mixing to aid digestion. The process of anaerobic digestion is modeled according to the process conditions obtained from sludge digestion handbook by STOWA, Foundation of Applied Water Research [98]. The handbook is based on best practices as well as empirical data collected from the sewage sludge anaerobic digestion plants in the Netherlands. These process conditions were verified using data from CBS [2] and Bolzonella et al [17]. This information can be found in table 3.3. Modeling methodology and process details are explained in Appendix C. Allocation is required only in this unit process. The values assigned to biogas and digestate are based on their respective energy contents with the calculations described in section 2.4.

Table 3.3: Proximate analysis of sludge fed to anaerobic digester by weight basis *(left)* and important process conditions regarding the anaerobic digestion model *(right)*.

Proximate analysis	%
Moisture	95.56
Dry Matter	4.44
Ash	24.75
Organic matter	75.25
Volume decrease due to thickening (as % of initial volume)	80

Important assumptions	Quantity
Digester temperature	35°C
Residence time	30 days
Volatile matter destroyed	40%
Biogas produced per kg volatile matter destroyed	0.9 Nm^3
Allocation (Energy Content) Biogas Digestate	 39% 61%

Results of inventory analysis can be seen in table 3.4. Due to modeling using average values of biogas considered, there is a discrepancy of 3 kg (out of 1000 kg mass flow) in the mass balance between the sludge fed, biogas and digestate out. Additionally, biogas composition used for modeling is also shown in table 3.4.

Gas Cleaning

In this step, hydrogen sulfide, water and ammonia are partially removed from the biogas prior to combustion. Relative humidity is 100% inside the digester. Water vapor is desired to be below than 90% relative humidity at the gas engine input point [8]. This is frequently practiced in the Netherlands by passing the gas pipe underground. As a result, water condenses on the walls and is separated as a condensate [98]. Ammonia is partially removed along with the condensate which is sent back to wastewater treatment plant for treatment [96]. Thus an additional ammonia removal step is not required and is also not commonly practiced in the Netherlands [98].

Sulfur on the other hand, is removed through an additional step. THIOPAQ process which is a proven biochemical sulfur removal process is selected [86]. It involves H_2S absorption into a mild alkaline solution followed by oxidation through microorganisms to obtain elemental sulfur as seen in figure 3.1. The process begins with input of sour biogas into the scrubber where absorption of H_2S takes places at an efficiency above 99% with a minimum value of 95% according to the chemical reaction 3.1.

$$H_2S + NaOH \rightarrow NaHS + H_2O \tag{3.1}$$

The solution so formed is sent to a bio-reactor where bacteria oxidizes the dissolved sulfide into elemental sulfur with a purity of more than 95%. The reactor is supplied with controlled air flow and nutrients. Exact composition of nutrient stream is confidential but it is expected to be a wastewater or a digestate stream based on the desulfurisation techniques reviewed [86]. Reaction of this step is given by equation 3.2.

$$2NaHS + \frac{1}{2}O_2 \rightarrow S^o + NaOH \tag{3.2}$$

The sulfide which is not recovered as elemental sulfur is completely oxidised as sulfate as in the following reaction 3.3.

$$2NaHS + 4O_2 \rightarrow 2NaHSO_4 \leftrightarrow Na_2SO_4 + H_2SO_4 \tag{3.3}$$

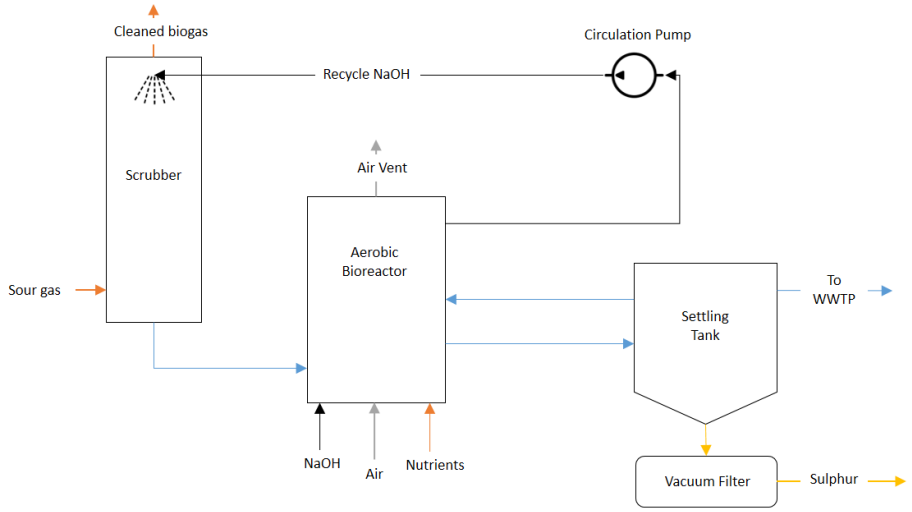


Figure 3.1: Simplified depiction of desulphurization of biogas [18].

Thus NaOH regeneration is not perfect, hence make up stream is constantly added to the bioreactor. This value is 0.44 kg NaOH used per kg sulfur recovered [18]. This relatively pure sulfur is assumed to replace the sulfur produced from conventional processes. Additionally, electricity and heat consumption of 0.151 kWh and 0.685 MJ respectively per Nm^3 biogas processed is assumed [33]. Results from inventory analysis are presented in table 3.5.

Energy Production

Inventory analysis results of this step is presented in table 3.6. It is important to note that carbon emissions are considered biogenic hence they do not influence the global warming results. Emission values of carbon monoxide, nitrogen oxides and sulfur dioxide are taken from Dutch limitations for mid-sized combustion plants [5]. These limits are set at 200 $mg.m^{-3}$, 340 $mg.m^{-3}$ and 10 $mg.m^{-3}$ for nitrogen oxides, sulfur dioxide and carbon monoxide respectively.

Since cogeneration is used, there is also considerable heat production. However, this heat is utilized for maintaining the digester temperature and for the desulfurization unit. Any excess heat if remaining is released to the environment [98]. Thus heat production is not included in the inventory analysis.

Table 3.4: Inventory table for Anerobic Digester *(left)* and biogas composition assumed for modeling *(right)* [98].

Economic Output	**Quantity**	**Source**
Biogas production	312.75 kg DS	Modeling
Digestate production	685.04 kg DS	
Energy/Material Used		
Electricity [E] (Mixing)	32 kWh	Sludge digestion manual [98]
Economic Input		
Thickened Mixed Sludge	1 ton DS	Reference flow

Biogas composition	**vol%**
CH_4	63
CO_2	35
H_2O	1.8
N_2	0.2
Trace	**$mg.m^{-3}$ (ppm)**
H_2	60
H_2S	400
NH_3	500

Table 3.5: Inventory table for unit process of desulphurization of biogas.

Economic Output	**Quantity**	**Source**
Clean Biogas	266.26 Nm^3	Modeling
Avoided Products		
Sulfur [E]	0.001 kg	EPA [18]
Materials/fuels		
Heat [E]	185.56 MJ	EPA [18]
Sodium Hydroxide 50% [E]	0.0033 kg	
Electricity [E]	40.934 kWh	
Economic Input		
Biogas	270.89 Nm^3	Reference Flow

Table 3.6: Inventory table for energy generation from biogas.

Economic Output	**Quantity**	**Source**
Electricity from cogeneration	662.176 kWh	Modeling
Avoided Products		
Electricity [E]	662.176 kWh	Modeling
Emissions to air		
Carbon dioxide, biogenic	521.46 kg	Modeling & Dutch Limitations [5]
Carbon monoxide, biogenic	0.0033 kg	
Nitrogen oxides	0.1392 kg	
Sulfur dioxide	0.000022 kg	
Economic Input		
Clean Biogas	266.26 Nm3	Reference flow

3.4. Dewatering

Overview

Downstream the anaerobic digestion stage, digestate still has to be processed. Even though energy can still be recovered, it has a negative economic value since it is not a profitable process. For thermochemical conversion of sludge, most of the technologies require water removal although at different levels. Same applies if raw sewage sludge is utilized without digestion. Water is contained in sludge in different 4 different ways as described by Werther and Ogada [97]:

- Free water - Water (70-75%) contained between sludge flakes thus can be removed by thickening or clarification.
- Floc water - Water (20-25%) trapped in interstices of floc particles. Usually removed by mechanical dewatering.
- Capillary bound water - Water (1%) attached to sludge flakes due to capillary forces. Mechanical dewatering only with chemical conditioning will help separation.
- Chemically bound water - Part of water (1%) inside the cell mass thus can be removed only by cell destruction i.e. thermal drying

While thickening aims at removing free water, dewatering targets floc water removal. As a result, sludge with a dry matter content of 20-25% is obtained.

Inventory

The most common dewatering techniques in the Netherlands are centrifuges, belt press and filter press as seen in section 1.4. Based on this, centrifuges is the selected technique. Sludge is dried from 4.4% DS to 25% DS based on dewatering trends [49]. Data regarding this unit process is collected from the investigation conducted by STOWA regarding trends in sludge dewatering [49]. For 1 kg of sludge dewatered, 14 g of polyacrylamide and 0.3 g of Iron

chloride are used to facilitate the process [49]. While the electricity consumption is at 0.12 kWh per kg DS sludge processed. This input and output data scaled to the functional unit is shown in table 3.7.

Additionally, screw press is used especially for dewatering sludge before SCWG since a dry matter content of 13% is desired [48]. Electricity requirement is thus predictably lower at 0.02 kWh per kg DS sludge processed. Only Polyacrylamide is required at a rate of 2.5 g per kg DS sludge [7]. Screw press dewatering for digested and undigested sludge is shown in table 3.8.

Table 3.7: Inventory table for dewatering digested sludge *(left)* and raw sludge *(right)* using centrifuges. This data is used by Incineration and Plasma Gasification systems in the next steps.

Economic Outputs	**Quantity**	**Source**
Dewatered digested sludge	685.04 kg DS	Modeling
Materials/fuels		
Electricity [E]	83.57 kWh	Trends in sludge dewatering [49]
Polyacrylamide [E]	9.6 kg	
Iron chloride [E]	0.2 kg	
Economic Inputs		
Digestate from anaerobic digestion	685.04 kg DS	Modeling

Economic Outputs	**Quantity**	**Source**
Dewatered sludge	1000 kg DS	Modeling
Materials/fuels		
Electricity [E]	122 kWh	Trends in sludge dewatering [49]
Polyacrylamide [E]	14 kg	
Iron chloride [E]	0.3 kg	
Economic Inputs		
Sludge from thickening	1000 kg DS	Modeling

Table 3.8: Inventory table for dewatering digested sludge *(left)* and raw sludge *(right)* using screw press. This data is used by Supercritical water Gasification system in the next steps.

Economic Outputs	**Quantity**	**Source**
Dewatered digested sludge	685.04 kg DS	Modeling
Materials/fuels		
Electricity [E]	13.7 kWh	Huber Technology [7]
Polyacrylamide [E]	1.71 kg	
Economic Inputs		
Digestate from anaerobic digestion	685.04 kg DS	Modeling

Economic Outputs	**Quantity**	**Source**
Dewatered digested sludge	1000 kg DS	Modeling
Materials/fuels		
Electricity [E]	20 kWh	Huber Technology [7]
Polyacrylamide [E]	2.5 kg	
Economic Inputs		
Digestate from anaerobic digestion	1000 kg DS	Modeling

3.5. Incineration

Overview

Fluidized Bed Combustion

In fluidized bed combustion, the fluidizing medium (air in this case) coming in from the bottom, lifts the solid fuel particles forming essentially a gas-solid suspension. Additionally, air also provides oxygen for combustion. Werther and Ogada present a variety of reasons why fluidized bed combustion is widely accepted for incineration sewage sludge and also mention case studies [97]. The unit process considered in figure 3.2 is from the sewage incineration plant of SNB at Moerdijk, Netherlands.

Sludge post dewatering enters the system at 25% DS where it is indirectly dried using low pressure steam. Sludge with around 37-40% DS enters the fluidized bed where the sludge is combusted. The fluegas produced is used for producing high pressure steam which is then utilized for electricity generation. Condensing steam is used to preheat the air entering the fluidized bed. Condensate is sent for further treatment before it is reused in the plant or drained in the sewage system. Fluegas leaving from the boiler is sent to cleaning system before being emitted to air.

Gas Cleaning

Fluegases formed during combustion of sewage sludge contain a variety of pollutants. Heavy metals, NO_x, CO, SO_2, HCl, HF, dioxins and furans are the major pollutants which have to be

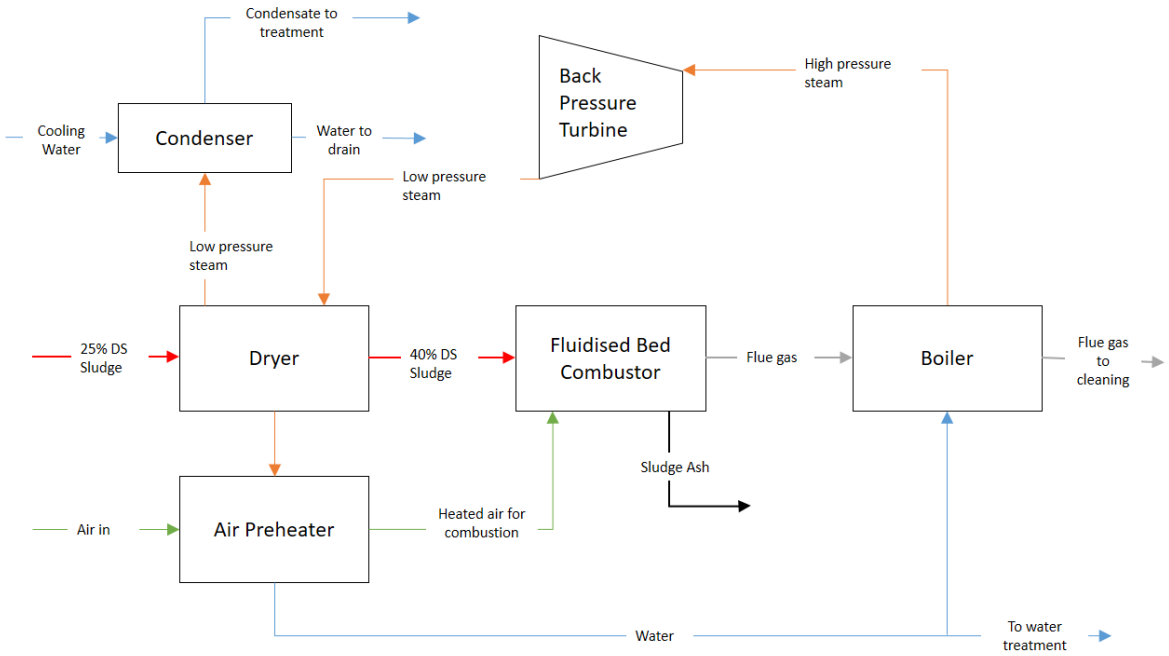


Figure 3.2: Schematic diagram of indirect heating of sludge followed by Fluidised Bed Combustion [98].

removed or regulated before fluegases are emitted to air. Fluegas treatment system selected here is again based on the sludge incineration plant of SNB [66] can essentially be divided into 3 stages:

1. Particle removal using electrostatic precipitator and cyclone separator
2. Water and NaOH scrubbing
3. Adsorption followed by filtration on fabric filters
4. Treatment of produced wastewater

After exiting the fluidized bed combustor, fluegas first passes through a wet electrostatic precipitator to remove all kinds of particulates in the fluegas such as dust, mist, aerosols, ash particles etc with an efficiency of upto 99% [66]. Moreover, upto 80% of dioxins and furans are also removed since they are retained in the flyash [97]. The gas is passed through an electric field, the particles in the gas ionize and are collected at the electric poles [102]. These particles are then washed off using a liquid like water in the case of wet electrostatic precipitation. This is the major difference between a wet and a dry electrostatic precipitator. The wet technique is more suited to fluegas with higher moisture content and since sludge has a DS content of 40% when combusted, this technique is preferred [25].

Next, removal of acid gases, sulphur dioxide and halogens takes place. In fact, lime is added already along with the bed sand in the combustion chamber which absorbs SO_2 according to the following equation:

$$CaCO_3(s) + SO_2(g) \rightarrow CaSO_3(s) + CO_2(g) \tag{3.4}$$

To remove other gases and residual SO_2, two step scrubbing, with water followed by NaOH is carried out. Water essentially washes acid gases like HCl and HF forming a mildly acidic wastewater solution. Residual particles and heavy metals are also washed off by water. Alkaline washing using NaOH targets mainly SO_2 removal [66]. Spent water and NaOH solutions are sent to further treatment as will be discussed in section.

Activated carbon and lime are injected into the fluegas stream. These materials act as adsorbents and show a tendency to accumulate mercury, dioxins, furans, dust particles and

heavy metals. Spent adsorbents are later separated from fluegas stream using fabric filters. Fluegas is finally emitted to air as it now satisfies the environmental regulations.

Wastewater produced in a sewage incineration plant has two sources. First, condensing water from incineration and second, spent solution from fluegas treatment. Condensing water contains Nitrogen impurities (mainly NH_3) while spent solutions as can be extrapolated from the information above contain many more impurities. Heavy metals, acids, dioxins and furans. Treatment of wastewater is done in 2 steps as seen in figure 3.3. Wastewater from the fluegas treatment is fed to a distillation column. The bottom product from this column is centrifuged to get salt residue which is to be landfilled whereas the water is fed back into the system. The distillate is mixed with the sludge condensate and fed to the second distillation column. The bottom product here is discharged to sewage system. The distillate contains an ammonia rich aqueous solution which is a hazardous waste.

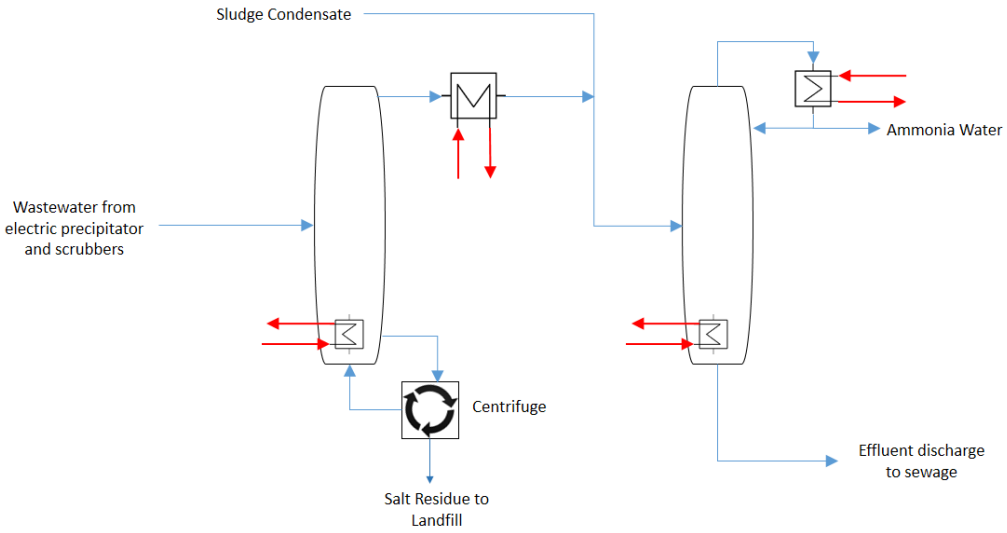


Figure 3.3: Schematic diagram of wastewater treatment in a sludge incineration plant [98].

Inventory

Inventory analysis of fluidized bed combustion and fluegas gas treatment is shown in table 3.9. Main data sources for modeling this process was models from STOWA [98], [99] and annual reports from SNB [66], [67]. Electricity generation was based on the amount of organic content as modeled by STOWA and verified using annual reports of SNB. Sludge considered here was digested, hence if undigested sludge is used, higher electricity yield is expected. This electricity replaces the grid electricity in the Netherlands as the main function of these systems is to process sewage sludge.

Fluegas treatment steps are based on the SNB plant [66]. These values were obtained from the yearly reports in terms of per kg sludge processed or per Nm^3 fluegas produced and then scaled to the functional unit. Transport of 25% DS sludge post-dewatering from wastewater treatment plant to incineration plant is also considered. Transportation methodology will be explained in section 3.10. Half of the waste produced is considered to be used as a foundation material in asphalt whereas half of it is disposed in German salt mines. The reason behind this assumption and environmental intervention as a result of this activity will be presented in section 3.9. Detailed calculations and additional information can be found in Appendix D.

Table 3.9: Inventory table for fluidized bed combustion of sewage sludge and fluegas treatment.

Economic Output	**Quantity**	**Source**
Electricity Generated	70.35 kWh	Stowa model [98]
Avoided Products		
Electricity [E]	70.35 kWh	Stowa model [98]
Energy/Material Used		
Limestone [E]	57.27 kg	SNB report [66]
Sodium Hydroxide 50% [E]	22.05 kg	
Hydrochloric Acid [E]	6.96 kg	
Charcoal [E]	7.8 kg	
Natural gas [E]	4.07 Nm^3	
Economic Input		
Dewatered digested sludge	685.04 kg	Reference flow
Transport [E]	383.76 tkm	Transport 3.10
Emissions to Air		
CO_2, biogenic	1008.27 kg	Modeling
CO_2, fossil	7.5 kg	SNB [66]
CO	0.031 kg	
NO_x	0.27 kg	
Dust	0.0028 kg	
Chlorides	0.0076 kg	
Waste Treatment		
Disposal in German salt mines [E]	153.18 kg	Modeling
Recycle in Asphalt	153.18 kg	

3.6. Supercritical Water Gasification

Overview

Supercritical water gasification system considered here was a pilot scale gasifier at Karlsruhe Institute of Technology (KIT), Germany [48]. A simplified process scheme of the pilot plant also used for modeling this process is shown in figure 3.4. The system can be divided and elaborated in the following steps:

1. **Pump** - Sludge at 13% DS is pumped to a pressure of 280 bar using a high pressure pump. Depending on the sludge, a pre-treatment/screening step might be necessary to prevent blockages from materials like fibers.

2. **Heat Exchanger** - In this step, feed is preheated using the effluent from the reactor to increase the energy efficiency. The heated sludge stream enters the salt and ash removal system at an approximate temperature of 370°C [50].

3. **Salt and ash removal** - At around supercritical point of water, the salts become insoluble in the sludge. Hence it is convenient to separate these salts along with ash using for example a cyclone separator. As a result, corrosion and clogging risks in the supercritical gasifier is prevented. Some installations might include this step after the gasifier which may result in a higher conversion. Since with salt and ash removal some organic content of sludge is also removed.

4. **Heating** - Heat is supplied to the sludge stream to ensure temperature is sufficiently high for gasification reactions to occur. Thermal energy using electricity or heat can be supplied. In many other cases this step is combined with the reactor.

5. **Reactor** - Reactor temperature of more than 650°C is recommended with a residence time between 2-5 minutes is recommended for high conversion.

6. **Gas Separator** - Effluent post pre-heater has a temperature between 60°C and 200°C. Syngas for energy recovery is obtained after the effluent is 'flashed' or subjected to sudden pressure decrease.

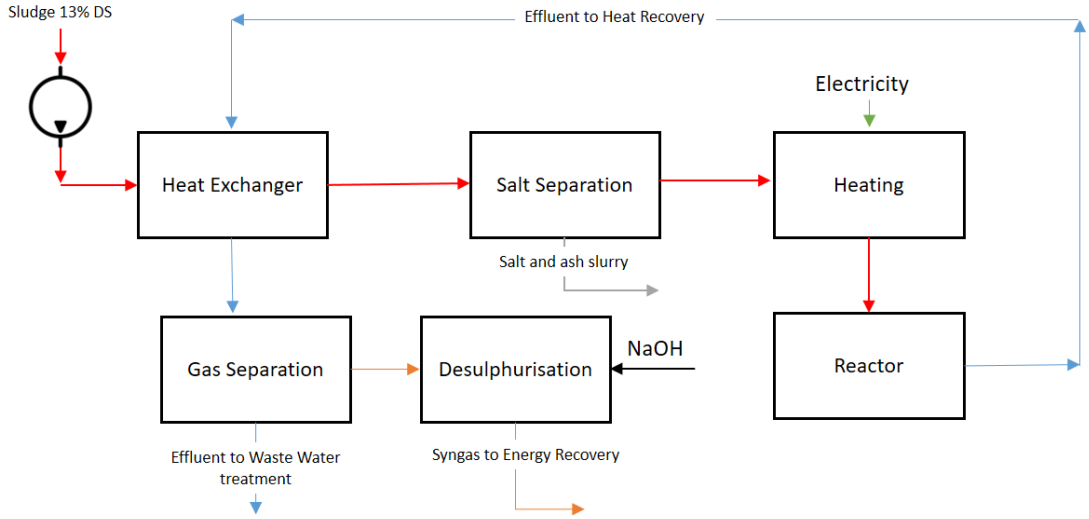


Figure 3.4: Schematic diagram of Supercritical water gasification system [48].

Syngas stream as compared to other thermochemical conversion technologies contains relatively few pollutants. Hence only a desulphurisation step before combustion is required [76]. However, the effluent stream requires more extensive treatment compared to other technologies due to the fact that nitrogen compounds in the sludge fed to the system remain in the effluent [48]. Additionally, it also contains some organic compounds since a carbon conversion of more than 80% is difficult to achieve in practice. For desulphurisation, THIOPAQ is again selected as used in the anaerobic digestion unit process with the same modeling criterion as in section 3.3.

Inventory

The inventory analysis of supercritical water gasification is based on a macro model of the process developed by KIT. Researchers found it to be consistent with ASPEN simulations and preliminary study of the plant [48]. The model was adopted and readjusted with respect to a more technically feasible scenario. Details of which were decided according to information available from literature and conversation with experts from TU Delft [76] and Gensos [81]. The calculation and modeling details for raw sewage sludge and digested sewage sludge is shown in Appendix E. Some important assumptions with respect to the model are shown in table 3.10.

The dry matter content of the input sludge was the most sensitive variable with respect to the energy recovered from this processing technique. Even though drying of wet biomass is not necessary for SCWG, and this is one of its major advantages, it is noticed that the dryer the feedstock gets, more energy can be recovered. This can be attributed to extra thermal energy to be provided to heat the water. The sample calculation shown in the KIT report was presented with a dry solid content of 20%. While in reality, experiments conducted at a dry solids content higher than 12-13% were not successful due to clogging in the reactor. Hence the model had to be adjusted accordingly.

Additionally, the organic conversion was adjusted from 90% to 75% according to the advice from researchers [76] & [81]. In practice, due to presence of organics and nitrogen in the

effluent, an additional wastewater treatment is necessary. Hence unlike other technologies, wastewater treatment as an additional background process is included in this system.

Table 3.11 presents the inventory analysis results for SCWG of raw sludge and digested sludge respectively. SCWG of digested sludge consumes electricity instead of producing. This is due to decrease in organic content due to anaerobic digestion step. Cogeneration similar to one used in section 3.3 is also used here and the heat generated is assumed to replace district heating. It is interesting to note here that the thermal energy produced in case of digested sewage sludge is greater. This is because, the amount of heat required for desulphurisation depends on the amount of syngas produced which is lower in case of digested sewage sludge. The treatment scenario for sewage sludge ash is assumed to be the same as for incineration. Since the produced ash is also expected to be hazardous.

Table 3.10: Important assumptions for the supercritical water gasification model.

Parameter	**Value**
Plant scale in total sludge weight (kg/h)	1000
Dry matter content of sludge (%)	13%
Calorific value of organic matter and syngas (MJ/kg)	22
Electric conversion efficiency (%)	37%
Organic conversion (%)	75%
Gasification pressure (bar)	280
Gasification temperature (°C)	700
Heat loss from system (% of electrical energy in)	18%

Table 3.11: Inventory table for supercritical water gasification of raw and digested sludge followed by combustion of syngas in a gas engine.

Economic Output	**Quantity**		**Source**
	Raw	**Digested**	
Electricity Generated	116.58 kWh	-17.13 kWh	Modeling
Avoided Products			
Electricity [E]	116.58 kWh	-17.13 kWh	Modeling
District heating [E]	612.84 MJ	855.3 MJ	
Sulfur [E]	5.96 kg	5.96 kg	
Energy/Material Used			
Sodium Hydroxide 50% [E]	2.63 kg	2.62 kg	Modeling
Economic Input			
Dewatered sludge	1000 kg	685.04 kg	Reference flow
Transport [E]	350.29 tkm	240 tkm	Reference section
Emissions to air			
CO_2, biogenic	1198.72 kg	697 kg	Modeling and Dutch limitations [5]
CO	0.008 kg	0.006 kg	
NO_x	0.142 kg	0.083 kg	
SO_2	0.204 kg	0.118 kg	
Waste Treatment			
Disposal in German salt mines [E]	123.76 kg	123.76 kg	Modeling
Recycle in Asphalt [E]	123.76 kg	123.76 kg	
Waterwater Treatment [E]	7.44 m^3	7.27 m^3	

3.7. Plasma Gasification

Overview

Plasma gasification system selected here is based on plasma installations in Japan [21]. These plants process Municipal Solid Waste (MSW) or co-process sewage sludge with MSW. A simplified process scheme shown in figure 3.5 can be elaborated as follows.

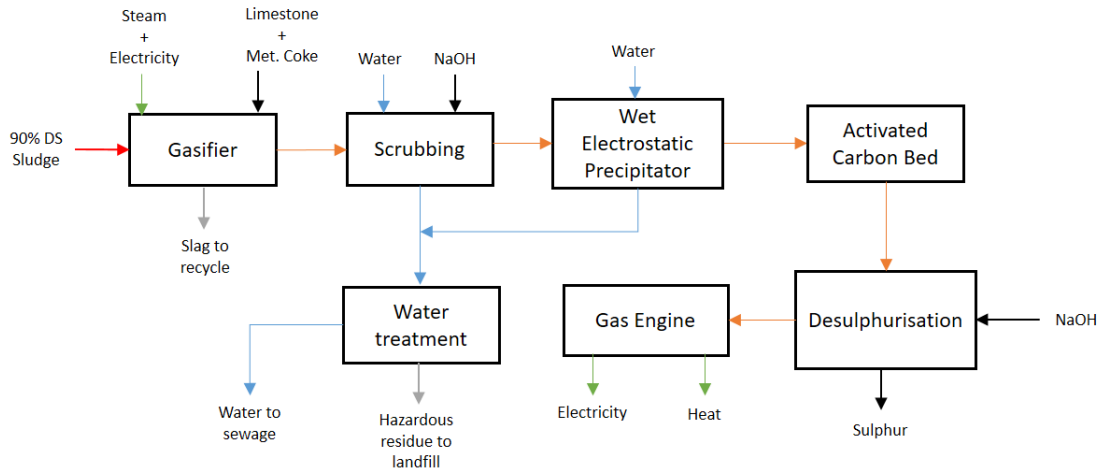


Figure 3.5: Schematic diagram of plasma gasification system [21]

Gasifier

Sewage sludge at 90% DS enters the reactor from the top in most installations (Entrained flow). Plasma torches are placed next to the feed at the top or at the bottom [52]. Electricity is required to power the plasma torches which emit a plume of around 6000°C. Gasification reactions convert the organic fraction of sludge into syngas while inorganic fraction is converted into slag. However, the slag and syngas leave at the temperatures of around 1650°C and 850°C respectively [21]. Limestone and metallurgical coke are also added to the reactor. Limestone is added as a slag fluxing agent. It is used to control the properties of slag and facilitate slag 'tapping' from the reactor. It also ensures full vitrification of sludge ash. Coke on the other hand, provides a constant heat input to the gasification process by absorbing heat from plasma and combusting gradually [21].

Syngas Treatment

Syngas exiting the gasifier may contain a variety of pollutants like organic dust, heavy metals, aerosols, gas phase halogens (HCl/HF), sulphur compounds (H_2, SO_2) and nitrogen species (NH_3). The syngas treatment is done in the following steps:

1. **Scrubbing** - The hot pollutant-laden syngas is washed using water and NaOH in a venturi scrubber. This is done because of three reasons. First, it prevents the formation of toxic dioxins and furans which might have formed if the gas was gradually cooled. Second, there is usually a certain amount of carbon carryover and other particulates in a syngas originating from a plasma reactor. Such particles are washed away using water. Third, as we have seen before, absorbs gas phase halogens like HCl and HF forming an acidic solution. A small amount of NaOH is also added to remove traces of HCl [21].

2. **Wet Electrostatic Precipitator** - This step removes sub-micron particles such as aerosols or zinc oxide by applying an electric field across the passing gas. The working principle of this step has already been explained in section 3.5.

3. **Activated Carbon Bed** - Syngas is then passed through a bed of activated carbon. A sulphur impregnated activated carbon is used by Alter NRG which has a mercury removal efficiency of 95%. Apart from mercury, other residual pollutants like heavy metals, dioxins and furans are also removed in this step.

4. **Desulphurization** - For removal of H_2S, THIOPAQ method described in detail in section 3.3 is used. Produced elemental sulphur is assumed to replace sulphur produced from other methods. Cleaned syngas is then sent for energy generation

Inventory

The inventory analysis of plasma gasification technology is presented in table 3.12. The first thing one can notice is the high consumption of natural gas. It serves the purpose of drying sewage sludge from 25% DS to 90% DS and generation of steam. As syngas is quenched just after leaving the gasifier, the sensible heat of the gas is lost and hence not taken into account for further heat integration.

Cogeneration of heat and power as in the previous sections is considered. An overall efficiency of 66% is assumed inline with a cogeneration unit from Mitsubishi [61]. This value is much lower than the usual cogeneration efficiency of around 90% because there is low pressure steam production at 120°C with this unit. Whereas other units produce hot water at 60-80°C suitable for applications like district heating. Steam produced at an efficiency of 29% is used as an input to the reactor, for all the plasma systems modeled, all the heat is utilized for the steam input to the reactor. Drying of sewage sludge is assumed using a boiler with an efficiency of 95% where calorific value of natural gas evaporates the water in the water in sludge.

Parameters for macro-modeling the reactor were obtained from a plasma gasification pilot scale plant of sewage sludge by Balgaranova, 2003 [14]. The reactor was an entrained flow gasifier with steam as the gasifying agent. Although many parameters such as type of plasma gasifier, downdraft fixed bed, choice of gasification agent can be varied, these parameters were selected because only data related to this particular set was available. Thus gas composition, calorific value of produced gas and amount of gas produced for the model was derived from this study. Calculations and other information about modeling can be found in Appendix F.

Since this study was pilot scale study it had some drawbacks. Data about addition of coke and limestone which as we have seen is essential for successful slag tapping was unavailable. This data was according to MSW plasma gasification of AlterNRG in Japan [21]. Same was done for air pollutants. Since being a pilot scale study, many air borne pollutants might have been below the detection limit. Data about syngas treatment material and energy requirements was obtained from fluegas treatment of incineration plants since the treatment steps were quite similar [66]. However differences like volume of gas and gas composition still remain. It is acknowledged that the collected data might not be the most accurate representation, however, under the given circumstances, this was the best possible approach. It was also discovered that a similar modeling approach was adopted for woody biomass by Nuss et al [68] giving comparable results.

Gasification of digested sewage sludge as compared to raw sewage sludge generates less electricity as expected. Natural gas use does not decrease considerably even though amount of drying and steam required is low. This is because, since less syngas is produced with digested sewage sludge, less heat energy is generated from cogeneration.

Table 3.12: Inventory table for plasma gasification of raw and digested sewage sludge followed by energy recovery in a gas engine.

Economic Output	Quantity		Source
	Raw	Digested	
Electricity Generated	486.1 kWh	144.57 kWh	Modeling
Avoided Products			
Electricity [E]	486.1 kWh	144.57 kWh	Modeling
Sulfur	13.75 kg	9.4 kg	
Energy/Material Used			
Sodium Hydroxide 50% [E]	38 kg	26 kg	SNB and Juniper Report [1]
Natural gas [E]	351 Nm^3	246.61 Nm^3	
Met. Coke [E]	1300 MJ	890.7 MJ	
Limestone [E]	77 kg	52.98 kg	
Hydrochloric Acid [E]	10 kg	6.96 kg	
Charcoal [E]	11.37 kg	7.79 kg	
Economic Input			
Dewatered raw sludge	1000 kg	685.04 kg	Reference flow
Transport [E]	196.16 tkm	134.4 tkm	Reference section
Emissions to air			
CO_2, biogenic	1090.383 kg	746.9 kg	Modeling
CO_2, fossil	539.56 kg	465.91 kg	Juniper Report [1]
CO	0.031 kg	0.021 kg	
NO_x	0.078 kg	0.05 kg	
SO_2	0.0016 kg	0.001 kg	
Dust	0.003 kg	0.002 kg	
Hydrogen Chloride	0.03 kg	0.02 kg	
Waste Treatment			
Disposal in German salt mines [E]	23.75 kg	19.44 kg	Modeling
Recycle in Ceramic Tiles [E]	235.15 kg	235.15 kg	

3.8. Solid Oxide Fuel Cell

Overview

Coupling a SOFC with gasifier can have many problems due to the pollutants contained in the syngas [21]. SOFC's are considered to be relatively tolerant of impurities and flexible to fuel inputs compared to other fuel cells. Because of the nature of sewage sludge, resulting syngas produced contains a variety of pollutants which can be harmful to a SOFC:

- Particulate matter and liquid droplets including carbon dust, heavy metals, tar droplets and aerosols
- Sulphur species: H_2S, COS and SO_2
- Gas phase hydrogen halides: HCl and HF.
- Nitrogen species: NH_3 and HCN.

Hence removal or reduction of these pollutants to often at a level of few ppm is necessary for feasible functioning of SOFC. Coupling of gasifier with SOFC and moreover a SOFC-Gas Turbine (GT) combination for a high energy recovery rate has been studied by various researchers[54], [11] & [89]. The SOFC-GT combinations have been modeled in detail and promise a high energy recovery of up to 60% from syngas. However, they have not yet been demonstrated even on a pilot scale. SOFC have been coupled with gasifiers just on a pilot scale. They have significant technological risk themselves hence, the combination of SOFC-GT was not considered in this report.

Inventory

Literature was studied if the considered cleaning techniques have to be replaced or modified for use with SOFC's. The considered gas cleaning technique is a cold gas cleaning system in line with the one implemented in treatment plants in Japan [21]. Cold gas cleaning systems are comparatively mature and perhaps this might be one of the reasons why they were selected in Japanese plants.

For SCWG, only sulphur removed using THIOPAQ process is required as described in section 3.5. It is a mature process where H_2S removal efficiencies of more than 99.5% have already been proven. Hence a H_2S concentration of a few ppm as required can be achieved which do not cause problems in functioning of SOFC's. As experimentally verified by Arvind et al [10]. Moreover scrubbing of gas using NaOH has already proven to limit SO_x in the incineration and plasma gasification technologies to less than 4 ppm [66] and [21].

Particulate matter removal is not required for systems using SCWG. For plasma gasification and incineration systems, a two step combination of water scrubbing followed by wet electrostatic precipitation has been used in systems described in section 3.5 and 3.7 limiting the dust emissions to 1.5 ppm and 10 ppm respectively [66], [21]. In the latter, a fabric filter was not used, with the use of this cleaning step, it is feasible to limit particulates to less than 5 ppm. There is a lack of concrete information regarding the tolerance of SOFC to particulate matter but it is generally accepted that these levels must be maintained at few ppms [9].

Hydrogen halide removal is done by water and NaOH scrubbing. This technique has helped limit HCl levels to 30 ppm in section 3.7 and 0.9 ppm (0.1 ppm for HF) in section 3.5. PV Arvind et al also demonstrated recently HCl levels upto a few ppms did not affect the short term performance of SOFC [9].

Much of the ammonia for the considered gasification technologies accompanies the water. During thermal drying, when the water is evaporated, ammonia is also evaporated along with it [98]. In the case of supercritical water gasification, it remains in the effluent [48]. However, there is still a considerable amount of ammonia in the syngas. Ammonia although a pollutant for other conversion technologies, is actually a fuel for SOFC's. Ammonia dissociates at the anode into N_2 and H_2, hydrogen oxidizes and nitrogen leaves the fuel cell. Low NO_x concentration of upto 0.5 ppm can still be formed [9].

Since infrastructure processes are not considered in this study, SOFC's with their higher efficiencies and similar or lower emissions compared to gas engines are expected to be promising. With syngas, an efficiency of 41% has been demonstrated and thus the same has been considered in this report [70]. Inventory results for supercritical water gasification and plasma gasification followed by SOFC can be found in tables 3.13 and 3.14 respectively.

Table 3.13: Inventory table for supercritical water gasification followed by energy recovery in a solid oxide fuel cell.

Economic Output	Quantity		Source
	Raw	Digested	
Electricity Generated	244.5 kWh	57.5 kWh	Modeling
Avoided Products			
Electricity [E]	244.5 kWh	57.5 kWh	Modeling
District heating [E]	1303 MJ	1258.1 MJ	
Sulfur [E]	5.96 kg	5.96 kg	
Energy/Material Used			
Sodium Hydroxide 50% [E]	2.63 kg	2.62 kg	Modeling
Economic Input			
Dewatered raw sludge	1000 kg	685.04 kg	Reference flow
Transport [E]	350.29 tkm	240 tkm	Reference section
Emissions to air			
CO_2, biogenic	1198.72 kg	696.97 kg	Modeling
SO_2	0.004 kg	0.004 kg	
Waste Treatment			
Disposal in German salt mines [E]	123.76 kg	123.76 kg	Modeling
Recycle in Asphalt	123.76 kg	123.76 kg	
Waterwater Treatment [E]	7.44 m^3	7.273 m^3	

Table 3.14: Inventory table for plasma gasification followed by energy recovery in a solid oxide fuel cell.

Economic Output	Quantity		Source
	Raw	Digested	
Electricity Generated	620.5 kWh	236.63 kWh	Modeling
Avoided Products			
Electricity [E]	620.5 kWh	236.63 kWh	Modeling
Sulfur [E]	13.75 kg	9.4 kg	
Energy/Material Used			
Sodium Hydroxide 50% [E]	38 kg	26 kg	SNB and Juniper Report [1]
Natural gas [E]	346.2 Nm^3	232.1 Nm^3	
Met. Coke [E]	1300 MJ	890.66 MJ	
Limestone [E]	77 kg	52.78 kg	
Hydrochloric Acid [E]	10 kg	6.96 kg	
Charcoal [E]	11.37 kg	7.79 kg	
Economic Input			
Dewatered raw sludge	1000 kg	685.04 kg	Reference flow
Transport [E]	196.12 tkm	134.37 tkm	Reference section
Emissions to air			
CO_2, biogenic	1090.383 kg	746.9 kg	Modeling
CO_2, fossil	600.8 kg	545.8 kg	Juniper Report [1]
SO_2	0.0016 kg	0.001 kg	
Dust	0.003 kg	0.002 kg	
Hydrogen Chloride	0.03 kg	0.02 kg	
Waste Treatment			
Disposal in German salt mines [E]	28.37 kg	19.44 kg	Modeling
Recycle in Ceramic Tiles [E]	235.15 kg	235.15 kg	

3.9. Treatment of Mineral Waste

Overview

Waste treatment scenario for Incineration and SCWG systems is assumed to be the same. Mainly because ash from SCWG is also expected to be hazardous. However, ash produced from plasma gasification is of a higher quality.

As seen in the waste treatment scenarios, 50% recycle of waste in asphalt and 50% disposal of waste in German salt mines is considered for incineration and SCWG. Even though more applications of waste like use in concrete or road pavements are currently done [67], they were not modeled due to following reasons. Firstly, use in asphalt and german salt mines constituted of more than 70% of waste [67]. Secondly, for construction applications, sludge ash acts as aggregate material, avoiding the production of same material, gravel or sand [24]. Thirdly, for applications like phosphate or nutrient recovery were relatively low (<2%) and data for these processes was not available.

For plasma gasification, around 8% of the total mineral waste produced was hazardous. The source of this hazardous waste was the residue recovered after wastewater treatment. The rest of the waste is slag recovered from the gasifier. Due to high temperatures, recovered slag becomes vitrified, as a result all the heavy metals are trapped in the glass phase. The leaching characteristics of heavy metals is considerably lower. Recovered slag can be used safely as building materials without further isolation [24].

Inventory

Inventory table for mineral waste management for plasma and incineration/SCWG systems can be found in tables 3.15 and 3.16 respectively. For disposal in German rock salt mines, background process of 'Waste for underground deposit' is used from Ecoinvent database. Wastes are stored in steel drums, polyethylene containers or large polyethylene bags. Since these mines have been stable for millions of years, chance of flooding and consequent leaching

from these wastes is quite low. Hence after these wastes are deposited, no emissions from the waste itself occurs, however emission due to transport, manufacture of equipment to store these wastes is taken into account.

For recycling in asphalt, data for leaching values from use of ash in asphalt was not available. Thus data from dutch emission limits for critical building materials was used [93]. Here sludge ash or other mineral waste is used as a construction aggregate replacing 'sand'. Actual leaching values from asphalt are expected to be higher especially since the Dutch government temporarily made an exception in the year 2005 for higher emission values so as to allow use of incineration ash in asphalt [91].

Table 3.15: Inventory table for management of mineral waste of plasma gasification systems.

Waste specification	Quantity		Source
	Plasma raw	Plasma digested	
Landfill of waste	19.44 kg	28.37 kg	Modeling
Recycle of waste	235.15 kg	235.15 kg	
Avoided Products			
Silica Sand [E]	235.15 kg	235.15 kg	VitroArc [38]
Emissions to water			
Cu	0.835 mg	0.835 mg	Critical values for Dutch building materials [93]
Pb	0.003 mg	0.003 mg	
Zn	2.4 mg	2.4 mg	
Cr	0.0013 mg	0.0013 mg	
Mo	0.0004 mg	0.0004 mg	
Ni	0.000095 mg	0.000095 mg	
Cl	3.59 mg	3.59 mg	
Economic Input			
Waste for underground deposit [E]	28.37 kg	19.44 kg	Modeling
Transport [E]	17.08 tkm	14.51 tkm	

Leaching values are considerably reduced by processing ash at a temperature greater than 1500°C as documented by Haugsten and Gustavson [38]. This study was also the source for the leaching data. These values allow use of slag as a class 1 building material in Netherlands. Thus slag produced from plasma gasification replaces the use of 'silica sand' in the manufacture of ceramic tiles.

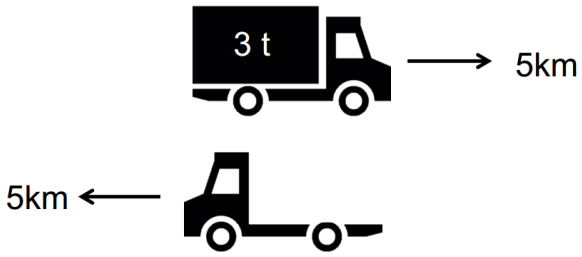
Table 3.16: Inventory table for management of mineral waste of SCWG and Incineration systems.

Waste specification	Quantity		Source
	SCWG	Incineration	
Landfill of waste	123.76 kg	153.8 kg	Modeling
Recycle of waste	123.76 kg	153.8 kg	
Avoided Products			
Silica Sand [E]	123.764 kg	153.8 kg	VitroArc [38]
Emissions to water			
Cu	0.0251 mg	0.0251 mg	Critical values for Dutch building materials [93]
Pb	3.08 mg	3.08 mg	
Zn	0.032 mg	0.032 mg	
Cr	0.022 mg	0.022 mg	
Mo	0.216 mg	0.216 mg	
Ni	0.0003 mg	0.0003 mg	
Cl	532.2 mg	532.2 mg	
Economic Input			
Waste for underground deposit [E]	123.76 kg	153.8 kg	Modeling
Transport [E]	89.04 tkm	121.7 tkm	

3.10. Transport

Overview

Transport is required at two points in the sludge chain. As we have seen in chapter 1, steps until dewatering take place at the WWTP's. Sludge is then transported to a thermochemical conversion destination like mono-incineration, co-incineration with MSW or incineration in cement kilns. The second place where transport is used is at the final disposal of mineral waste. This mineral waste includes sludge ash remaining after thermochemical conversion along with residue resulting from fluegas or wastewater treatment. Data entered in SimaPro for transport processes is in terms of tkm which stands for tonnes kilometer. This can be best explained by the figure 3.6. If a 3 ton load is to be transported for a distance of 5 km, emissions of the truck have to also be accounted for the return journey. The resulting transportation load in tonne-kilometer (tkm) is used to calculate the emissions.



$$\begin{aligned} Transport\ (tkm) &= Total\ load(t) * total\ distance(km) \\ &= (3t + 0t) * (5km + 5km) \\ &= 30tkm \end{aligned}$$

Figure 3.6: Methodology of transport calculation in Ecoinvent [34].

Inventory

For the reference scenario, data regarding sludge transport was available. Two mono sludge incineration plants in Moerdrijk and Dordrecht process almost 50% of the total sludge produced in the Netherlands [98]. One ton functional unit was divided proportionally according to the total annual amount of sludge supplied from wastewater treatment plants to incineration plants in Moerdijk and Dordrecht. Data about sludge supply from waterboards to these locations was available [4]. Thus distances from wastewater plants operating under the corresponding water boards to Moerdijk or Dordrecht was found.

For SCWG and plasma gasification, a plant was assumed to be built at location near the office of water boards. Since water boards operate at a distance close to the corresponding waste water plants they manage. Dewatered sludge from wastewater treatment plants is assumed to be transported to the mentioned location for thermochemical treatment. Solids content for incineration and plasma gasification is 25% DS while it is 13% DS for SCWG.

Sludge ash and other mineral waste as seen from waste treatment in section 3.9, is either utilized as in asphalt production or transported to Germany in salt mines. The location of the salt mine was assumed to be the biggest salt disposal mine in Germany at Tienburg 25, Wunstorf. On the other hand, for ash recycling in asphalt or building materials, it was assumed these materials are acquired and transported by third-party. Hence their transportation was not taken into account. All distances were calculated using Google Maps. Locations of new suggested plants were not optimized according to transport distances.

A sample calculation of wastewater treatment plants under Water Board Schieland for scenarios of SCWG and plasma gasification are shown in table 3.17. The difference between these two scenarios is the amount of sludge produced is because SCWG is carried out at 13% DS while Plasma gasification and incineration is done at 25% DS. The end values of t-km are entered in the SimaPro software. All the distances found are showcased in Appendix G.

Table 3.17: Transportation calculations for Water Board Schieland of a possible sludge treatment scenario using SCWG and Plasma gasification of dewatered raw sewage sludge

Wastewater Treatment Plants in Schieland	**Distance to Water Board Schieland (km)**	**SCWG**		**Plasma Gasification**	
		Sludge (kg)	**t-km**	**Sludge (kg)**	**t-km**
Ammerstol	24.7	27.51	1.36	15.41	0.76
Bergembacht	22.1	27.51	1.22	15.41	0.68
Berkenwoude	18.3	27.51	1.01	15.41	0.56
De groote zaag	9.6	27.51	0.53	15.41	0.30
Groenedijk	8.9	27.51	0.49	15.41	0.27
Haastrecht	29	27.51	1.60	15.41	0.89
Kortenoord	11	27.51	0.61	15.41	0.34
Kralingseveer	4.3	27.51	0.24	15.41	0.13
Stolwijk	29.3	27.51	1.61	15.41	0.90

4

Impact Assessment

Before the start of the chapter, in table 4.1 system description is provided for the major unit processes. The following abbreviations presented are used in the graphs to signify the corresponding systems. Kindly note that these system descriptions do not completely describe the systems but the major processes.

Table 4.1: System details of abbreviations used in this chapter.

Sr. no.	Abbreviation	System
0.	Incineration + Digestion	Thickening→Anaerobic digestion→Centrifugal dewatering →Incineration→Ash landfill/Recycle
1.	SCWG raw sludge	Screw press dewatering→SCWG→Gas engine →Ash landfill/Recycle
2.	SCWG raw FC	Screw press dewatering→SCWG→Gas engine →Ash landfill/Recycle
3.	SCWG digested	Thickening→Anaerobic digestion→Screw press dewatering →SCWG→Gas engine→Ash landfill/Recycle
4.	SCWG digested FC	Thickening→Anaerobic digestion→Screw press dewatering →SCWG→Fuel cell→Ash landfill/Recycle
5.	Plasma G raw	Thickening→Centrifugal dewatering→Plasma G →Gas engine→Ash landfill/Recycle
6.	Plasma G raw FC	Thickening→Centrifugal dewatering→Plasma G →Fuel cell→Ash landfill/Recycle
7.	Plasma G digested	Thickening→Anaerobic digestion→Centrifugal dewatering →Plasma G→Gas engine→Ash landfill/Recycle
8.	Plasma G digested FC	Thickening→Anaerobic digestion→Centrifugal dewatering →Plasma G→Fuel cell→Ash landfill/Recycle

4.1. Characterization results

Results of characterization of all the considered systems are shown in table 4.2. On the first glance, categories of abiotic depletion and ozone layer depletion seem very low especially because of the relative scale. Whereas marine ecotoxicity and global warming impacts seem more significant. However, discussion on the relative importance of these impact categories will be made in the next section. Another two observations can be made from this table. First, systems with fuel cells perform better than one with gas engines. This can be attributed to higher energy production and very low nitrous oxide emissions. Second important observation is the presence of anaerobic digestion. Digestion of sewage sludge as compared to all other technologies offered the highest net electricity recovery of around 660 kWh/ton DS. Moreoever, it did not require any materials apart from NaOH which was used for desulphurisation. Thus, systems with a digestion step perform better than their counterparts.

The current sludge scenario of Incineration + Digestion is competitive with the much-promised gasification systems. Since energy use heavily influences the environmental impact results, these are summarized in table 4.3.

Table 4.2: Characterization results of the modeled sludge chain systems.

Impact Categories	Equival. Units	Incin.+ Digested	Supercritical Water Gasification				Plasma Gasification			
			SCWG raw	SCWG raw FC	SCWG digested	SCWG digested FC	Plasma raw	Plasma raw FC	Plasma digested	Plasma digested FC
Abiotic depletion	kg Sb (x 10^{-4})	3.49	1.43	1.13	0.39	0.2	5.14	4.88	2.78	2.59
Global warming (GWP100a)	kg CO_2	-228.8	-23.5	-144.8	-315.6	-414.4	486.6	458.4	167.2	154.2
Ozone layer depletion	kg CFC-11 (x 10^{-6})	10.5	1.96	-7.06	-15.7	-22.4	74.2	68.8	37.8	32.3
Human toxicity	kg 1,4-DB	27.14	25.03	20.29	9.03	6.26	28.78	24.79	8.55	5.62
Freshwater aquatic ecotoxicity	kg 1,4-DB	0.69	0.86	0.68	0.31	0.20	0.84	0.71	0.2	0.11
Marine aquatic ecotoxicity	kg 1,4-DB	-34157	4377	-14445	-52879	-68955	3391	-14711	-51081	-63771
Terrestrial ecotoxicity	kg 1,4-DB	-0.32	0.059	-0.068	-0.34	-0.45	-0.078	-0.2	-0.42	-0.51
Photo-chemical oxidation	kg C_2H_4	0.089	0.027	0.005	0.0008	-0.012	0.132	0.125	0.074	0.069
Acidi-fication potential	kg SO_2	0.459	0.493	0.031	0.099	-0.173	0.51	0.383	0.157	0.067
Eutro-phication potential	kg PO_4	0.0862	0.13	0.09	0.07	0.047	0.11	0.08	0.038	0.017

Table 4.3: Energy generation and natural gas use of the modeled systems.

Energy production from unit processes	Incin.+ Digestion	Supercritical Water Gasification				Plasma Gasification			
		SCWG raw	SCWG raw FC	SCWG digested	SCWG digested FC	Plasma raw	Plasma raw FC	Plasma digested	Plasma digested FC
Electricity from Digestion (kWh)	589.8	N/A	N/A	589.8	589.8	N/A	N/A	589.8	589.8
Electricity[1] from Thermochemical Conversion (kWh)	70.35	116.58	244.46	-17.13	57.46	486.1	620.5	144.6	236.6
Net Electricity[2] Production (kWh)	660.15	116.58	244.46	572.67	647.26	486.1	620.5	734.4	826.4
Heat from Thermochemical Conversion (MJ)	N/A	612.84	1303	855.3	1258.1	0	0	0	0
Natural gas use (Nm^3)	4.07	N/A	N/A	N/A	N/A	351	346.2	246.61	232.1

[1]Negative value means electricity use

[2]Please note that net electricity production is not of the entire sludge chain but of the considered unit processes

4.2. Normalization

The characterization results by themselves are incapable of providing relative importance of impact categories for a technology or a system. Hence these results are divided by reference information. In this case, default normalization factor of CML method - Emission of an

average person in one year from the EU25 countries. Results of the reference case along with SCWG and Plasma systems with digestion and fuel cell are presented in figure 4.1. To better convey information from this section, only 3 systems are presented. We can see that marine aquatic ecotoxicity and global warming are the most important impacts. With the former being more than 10 times higher than the latter in some systems. Normalization results for the rest of the systems can be found in Appendix H.

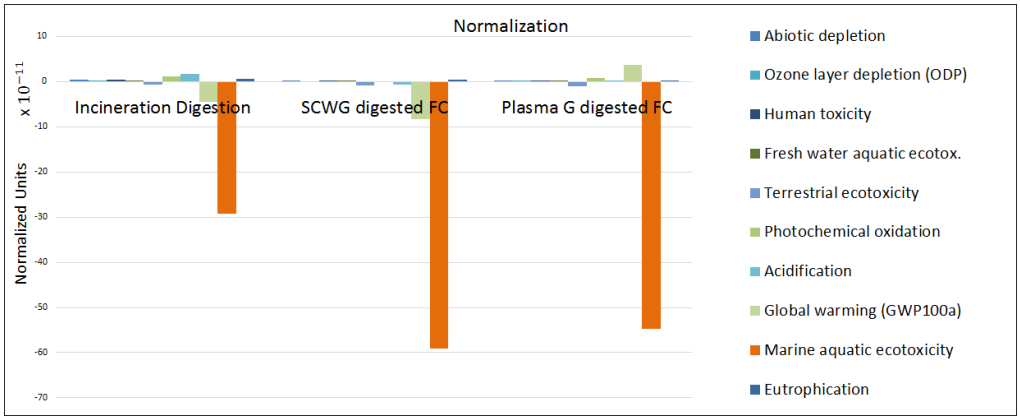


Figure 4.1: Normalized results of three selected systems using normalization factors of EU25 from CML-IA midpoint approach.

4.3. Weighting

Methodology of weighting is explained in section 2.3. Normalization results are multiplied by the adopted weights in table 2.1 and the results are presented in figure 4.2.

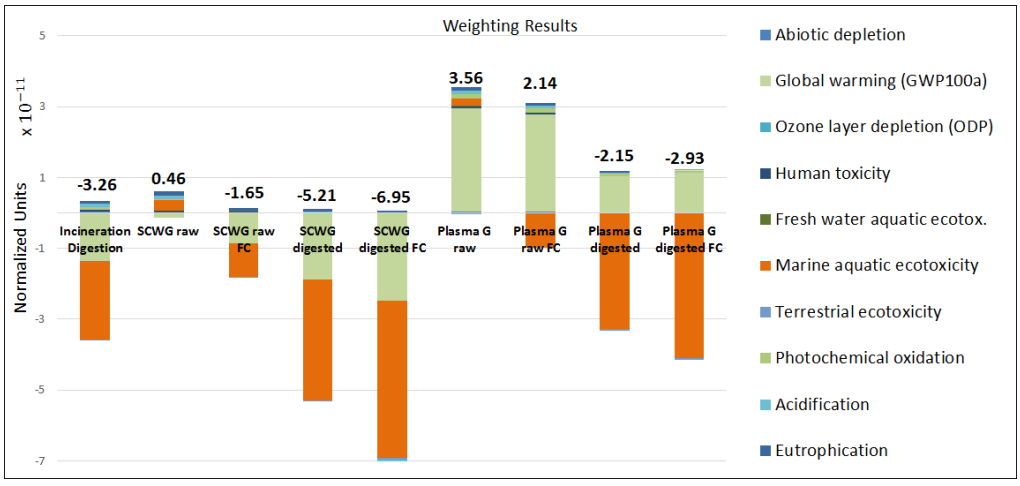


Figure 4.2: Single score results for the modeled systems after weighting is applied.

It can be seen right away that marine aquatic toxicity and global warming are the decisive factors both because of their magnitudes. It is interesting to note that marine aquatic toxicity only has a weight of 7% to the total score, but it still is a decisive factor because of its magnitude. Global warming has a high magnitude and weight of 30% hence it stays relevant. Based on the modeled conditions, supercritical water gasification systems perform better than other systems. Additionally, systems with anaerobic digestion have a lower impact as compared to ones without.

4.4. Contribution Analysis

This analysis calculates contribution of different unit processes to the various impact indicators. Thus all the impact categories considered will be analysed here. Apart from singling out environmental hot-spots, this analysis is also useful to find inconsistencies in data.

Acidification

Contribution analysis of acidification of all the considered systems are presented in figure 4.3. Main contributors to this impact category are SO_2, NO_x and NH_3. On the first glance it is interesting to note that for the system SCWG raw, impact due to thermochemical conversion is larger than any other category. This can be attributed to SO_2 and NO_x emissions during energy conversion. Reason for this being, for the SCWG models with gas engine, dutch limitations for emissions were used. In practice, these plants might operate at emission values lower than prescribed as done in other systems [66] & [21]. Thus, for thermochemical conversion, impact is less for incineration (using fluidized bed combustion) and plasma gasification (gas engine).

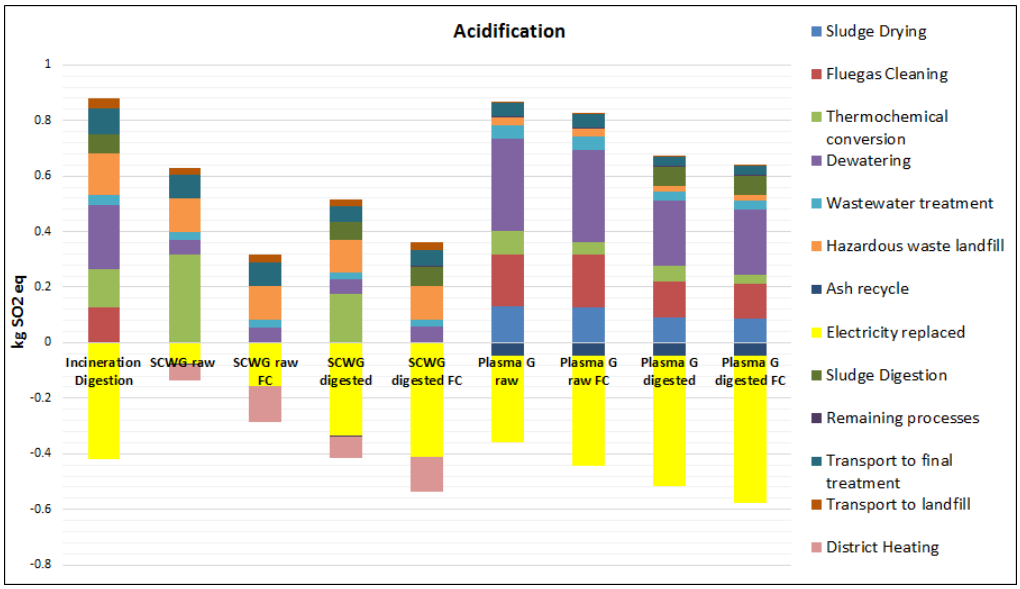


Figure 4.3: Contribution analysis of the modeled systems for acidification.

Moreover, the systems with SOFC perform better due to no NO_x and very small values of SO_2 emitted. We can see the large impact due to electricity and polyacrlyamide use during dewatering. SCWG systems save a substantial amount since they can process sludge at 13% DS. NaOH production and electricity used for the same also contributes to SO_2 and NO_x emissions making fluegas cleaning stage visible in the figure.

Abiotic Depletion

This impact category aims at quantifying the use of abiotic natural resource use by systems. The impact of systems measured here was exclusively in terms of used metals and minerals. This was also the most challenging category to analyze because the impacts were uniformly spread across the extracted materials. The contribution analysis results are quantified in terms of kg Sb (Antimony) equivalents as shown in figure 4.4.

There was no single material highly influencing results in this section unlike in other categories. Hence, contribution analysis in this section is comparatively more general and less specific. Material use in fluegas cleaning systems, in particular NaOH, causes the most material depletion. Energy and materials consumed during NaOH production process are the most likely reasons. Dewatering of sewage sludge also has a large impact which is turn due to consumption of polyacrylamide. Apart from fluegas cleaning, wastewater treatment stands out in all processes except for supercritical water gasification. Main share of this impact is

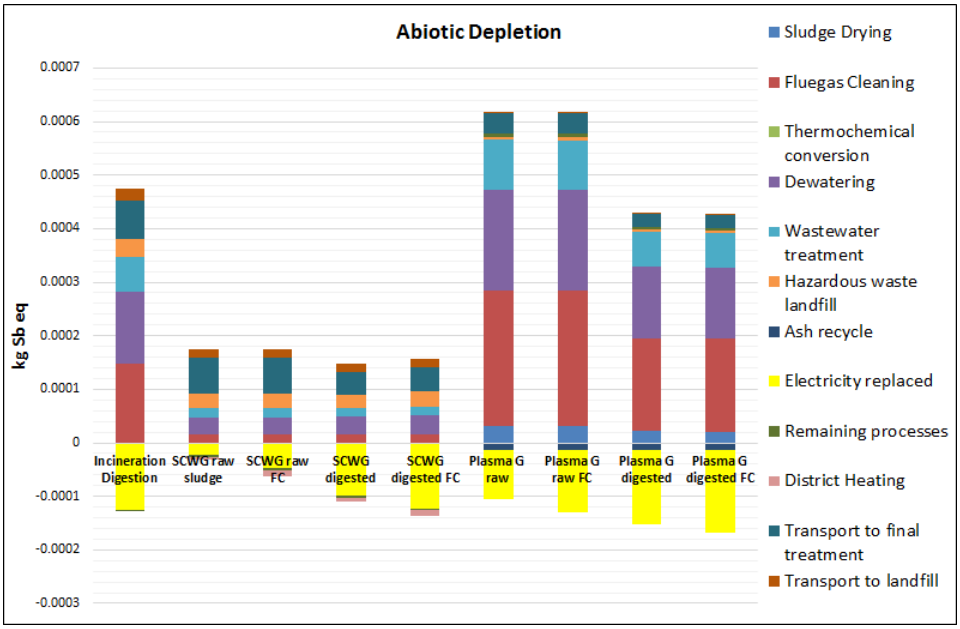


Figure 4.4: Contribution analysis of modeled systems for abiotic depletion.

due to production process of Hydrochloric Acid. HCl at 30% is used in the N-removal process before water is discharged to sewage as done at SNB [66]. Disposal of hazardous waste also makes an appearance due to utilization of iron ore, petroleum and other construction materials for safe landfilling.

Eutrophication

Results from contribution analysis of Eutrophication are presented in figure 4.5. Release of Nitrogen almost single-handedly causes the eutrophication impact of the cosidered systems. Nitrogen is emitted to the environment by ammonium ($NH_4{}^+$) and nitrate ($NO_3{}^-$) release to water and nitrogen oxides release to air.

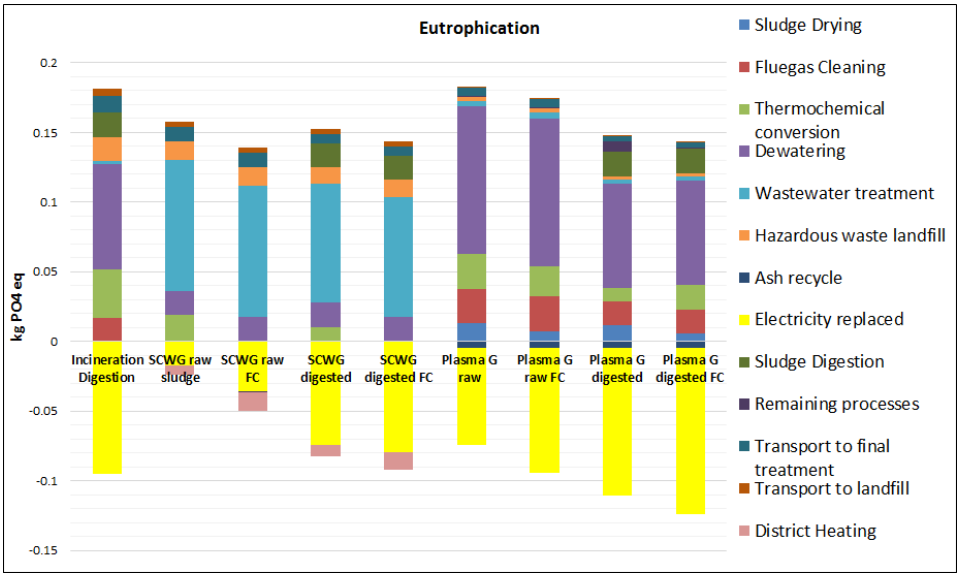


Figure 4.5: Contribution analysis of modeled systems for eutrophication.

A complete wastewater treatment step is taken as a background process for the effluent produced at the end of the supercritical water gasification process. Consequently, not follow-

ing the general trend in this impact category, supercritical water gasification perform almost at the same level as plasma gasification. During treatment of wastewater, ammonium and nitrate is released to the water bodies. Polyacrylamide use during dewatering sewage sludge also causes a substantial impact. Since manufacturing of acrylamide which acts as a precursor to Polyacrylamide involves 0.003 kg of ammonium emissions per kg of product. Energy conversion steps especially systems using gas engine and incineration also contribute to NO_x emissions.

Ecotoxicity

In this impact category, damage potential of emissions of toxic substances to aquatic, marine and terrestrial ecosystems is measured. Thus ecotoxicity is divided into three categories fresh water ecotoxicity potential, marine aquatic ecotoxicity potential and terrestrial ecotoxicity. The ecotoxicity potential is measured in terms of kg 1,4-Dichlorobenzene (DB) equivalents which is an active ingredient of mothballs, fumigants and deodorizers [26]. According to the Normalization results, marine ecotoxicity has the highest relative impact not only in ecotoxicity but overall in all impact categories. Hence contribution analysis of marine ecotoxicity is presented in this section with a graphical overview in figure 4.6. Results of fresh water aquatic and terrestrial ecotoxicity can be found in Appendix H.

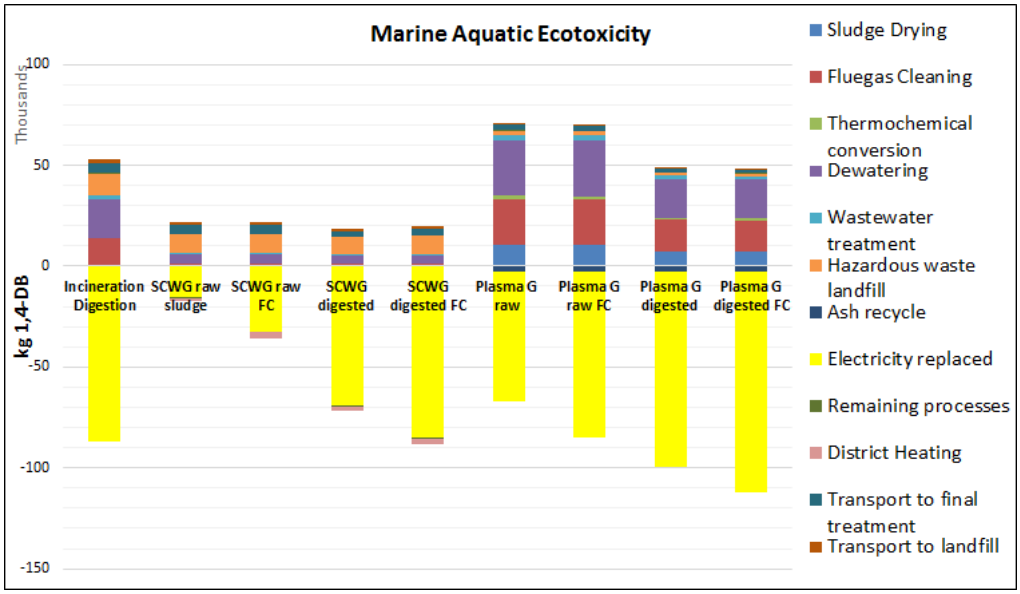


Figure 4.6: Contribution analysis of modeled systems for marine aquatic ecotoxicity.

As we can see in the figure above, this category is also the most sensitive to the replaced electricity in the dutch grid. Taking a look at the background data gives the insight that more than 95% of the impact is due to Hydrogen Fluoride emissions. Hydrogen Fluoride emissions are mainly attributed to Electricity and NaOH production. Other notable emission is Barium to water. These emissions are caused by transportation and polyacrylamide production processes.

Global Warming

Climate change impact is calculated in terms of global warming potential (GWP) with units of kg CO_2 equivalents in figure 4.7. Plasma technologies are at an obvious disadvantage due to large amount of natural gas required either for drying or steam production. Steam acts as a gasifying agent for the gasification reactions in the modeled plasma reactor. Not only the combustion of natural gas, but the extraction processes are also emission intensive.

Dutch electricity which has low share of renewables, when replaced by electricity produced offers negative emissions. District heating replaced by SCWG is substantial but they offer less emission saving because Ecoinvent follows exergy allocation system for cogenera-

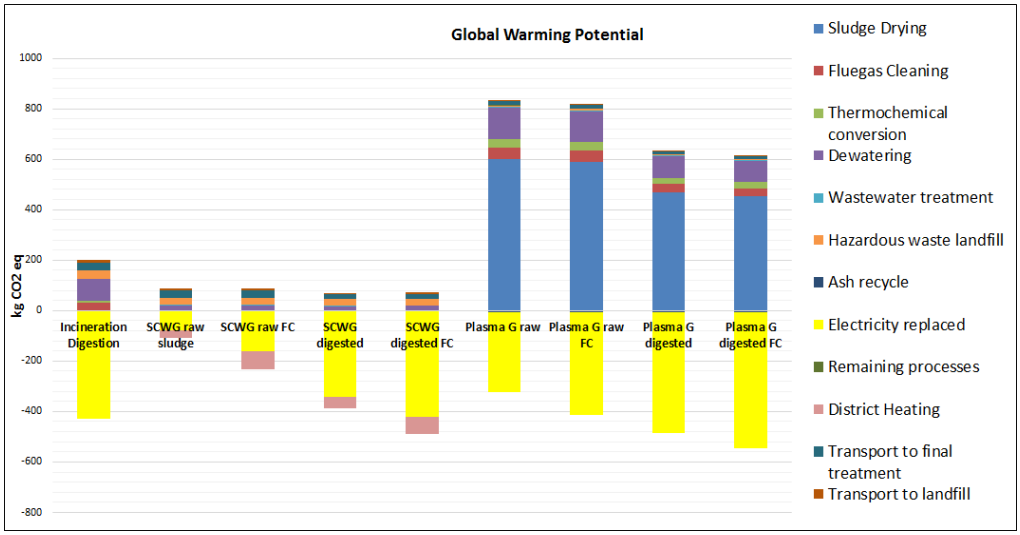


Figure 4.7: Contribution analysis of modeled systems for global warming.

tion. District heating is produced from cogeneration where emissions are mainly assigned to electricity production. Transportation processes also play a substantial role especially for SCWG systems. Since almost twice the amount of sludge transport is required compared to other systems.

Human Toxicity

Human toxicity is also quantified in terms of kg 1,5-Dicholorobenzene equivalents same as Ecotoxicity. Contribution analysis results are shown in figure 4.8.

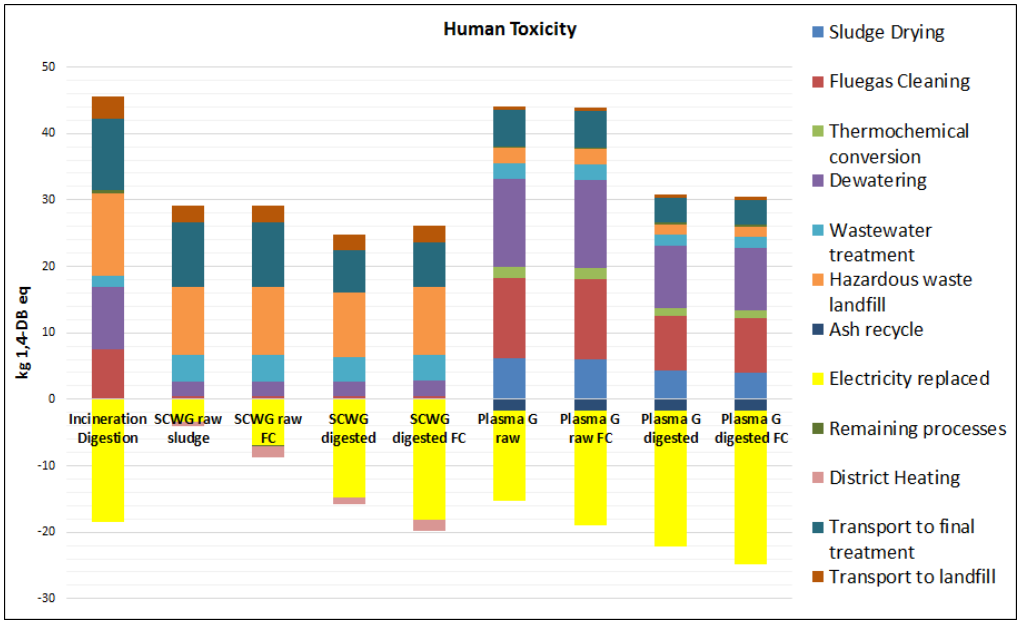


Figure 4.8: Contribution analysis of the modeled systems for human toxicity.

From the background inventory data, it becomes clear that Benzene, Antimony and Chromium emissions account for most of the impact. Hazardous waste landfill forms the majority of Benzene emissions. However it should be noted that benzene or any other substance does not leach from waste. Manufacturing of steel and consequently iron used for safe landfilling are responsible for benzene emissions. Antimony is emitted from transport processes, thus they can be seen to have a substantial impact in the process systems here. The source of Anti-

mony emissions is from brake wear due to abrasion of tyre and brake material. Chromium emissions on the other hand are caused by Polyacrylamide and NaOH production processes. For Polyacrylamide, these emissions were due to use of Ammonia which was in turn produced using steam reforming. Due to NaOH being an energy intensive manufacturing process, large amount of electricity utilized may be a possible reason for high Chromium emissions.

Ozone Depletion

Ozone depletion quantifies the impact on the ozone layer from release of gases from the system processes. From Normalisation step, we can see that ozone depletion has the lowest relative impact as compared to other impact categories. Results from the contributing processes of ozone depletion can be found in figure 4.9.

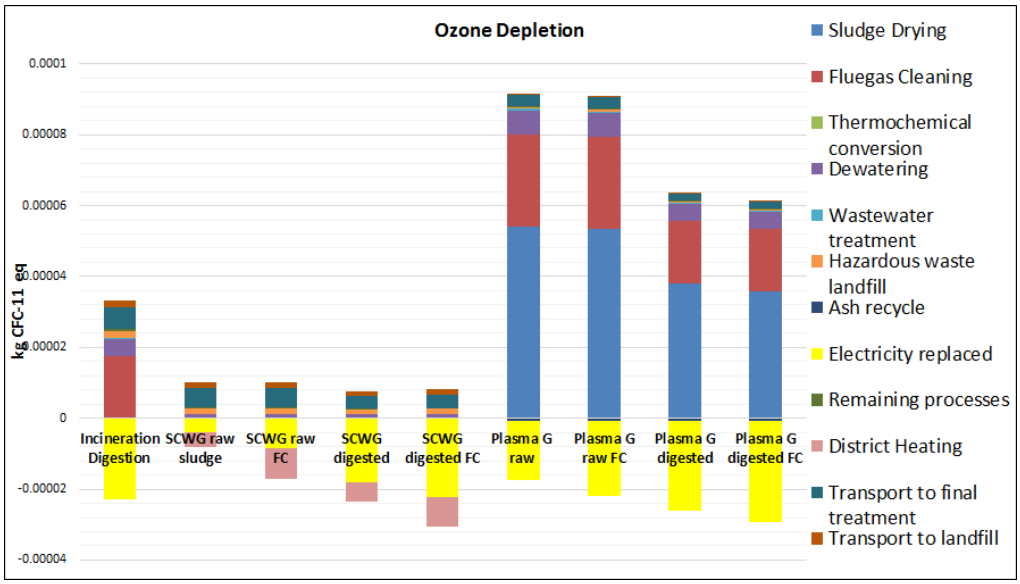


Figure 4.9: Contribution analysis of the modeled systems for ozone depletion.

Tetrachloromethane (CCl_4) and Bromocholorodifluoromethane ($CBrClF_2$) are responsible for the majority of the impact in this section. Bromochlorodifluoromethane is emitted during transportation and extraction of natural gas. Impact of sludge drying and thermochemical conversion for plasma systems is thus high which are natural gas intensive processes. Tetracholoromethane on the other hand is released during production of NaOH during the chlor-alkali electrolysis process. Fluegas cleaning processes can thus be seen with a major share of impact.

Photo-oxidant formation

Due to the effect of sunlight on various air pollutants, reactive chemical compounds are formed which may be harmful for human health and ecosystems. These compounds are called photo-oxidants, their consequent damage potential is measured in this section. Majority of the impact is due to carbon monoxide emission followed by emissions of ethene.

Results of the contribution analysis can be found in figure 4.10. Use of charcoal in fluegas cleaning processes form a major share of carbon monoxide and ethene emissions. During production of 1 kg charcoal, it is estimated there is release of 0.2 kg of carbon monoxide and 0.002 kg of ethylene (biogenic and fossil). Since SCWG employs only desulphurisation, carbon monoxide emissions due to manufacture of charcoal are avoided. Hazardous waste disposal impact is also significant because of carbon monoxide emissions during production of steel and iron.

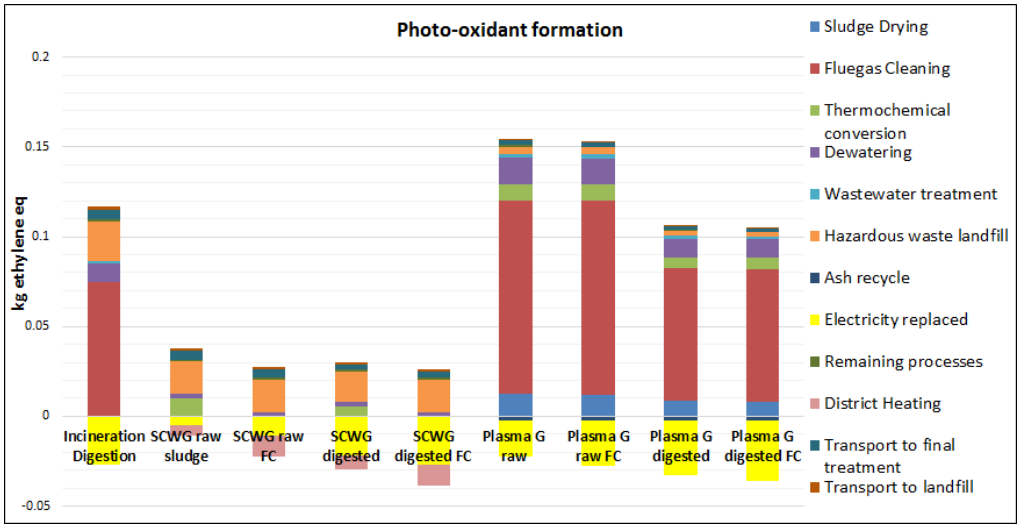


Figure 4.10: Contribution analysis of the modeled systems for photo-oxidant formation.

5

Interpretation

5.1. Comparison with other studies

Also known as consistency check, the objective of this section is to determine whether assumptions, methods, models and data of each alternative are in-line with the goal and scope of study. Based on the literature studied, it was concluded that it is futile to compare only the characterization results of different LCA studies of sewage sludge processing. To elaborate on this topic, Yoshida et al. [103] also published similar conclusions after reviewing 35 studies centered at the same topic. These authors also compared global warming potential of various studies to see the variability in characterization results. Even though GWP results are the least likely to be affected by the choice of impact assessment methods [77], substantial differences were noticed for similar scenarios which can be seen in table 5.1.

Table 5.1: Global warming potential presented in kg CO_2 eq per ton DS sewage sludge processed for selected studies with relevant scenarios [103].

	Drying + Use on Land	**An. Digestion + Use on Land**	**Incineration + Landfill**	**An. Digestion + Incineration + Landfill**
Poulsen and Hansen, 2003 [74]	-	300	-	925
Houillon and Julliet, 2005 [44]	-	-	-55	-
Murray et al, 2008 [62]	326	-359	-	-
Hong et al, 2009 [41]	1600	867	669	882
Hospido et al, 2010 [43]	-	-129/3034	-	-
Xu et al, 2013 [101]	-	4400	-	4000

Apart from the characterization methods, Yoshida et al. [103] found that GWP results are affected by geographical area and local conditions. Geographical conditions can influence transportation distances and weather conditions. Whereas examples of local conditions would be electricity production mix or waste management regulations. For example, in the above table, Hospido et al. concluded that for the scenario of anaerobic digestion followed by land application, the climate change indicator varied from -129 to 3034 kg CO_2 eq depending on the quality of sludge.

Technical assumptions regarding material and energy use also vary considerably between various studies. For example, Poulsen and Hanssen considered 2% methane leak from anaerobic digestion of sludge [74] whereas Murray et al. considered all of the produced methane is captured and utilized for power production [62]. This might explain the variation of GWP

from -359 to 300 kg CO_2 for anaerobic digestion followed by land application in table 5.1. Furthermore, discrepancies also exist in the process data either due to different measured values or the type of technology used.

Hence too much meaning should not be attached to characterization results while comparing different LCA studies. It is thus sensible to compare the important parameters of unit processes along with comparing impact assessment results. Important inventory results along with global warming potential for anaerobic digestion followed by incineration system, are presented in table 5.2.

Table 5.2: Comparison of inventory analysis and global warming potential results from anaerobic digestion followed by incineration system with similar systems in other studies. Results are presented in terms of 1 ton DS of biomass processed.

	Hospido et al, 2005 [42]	Hong et al, 2009 [41]	Xu et al, 2013 [101]	This report
Thickening				
Electricity (kWh)/ Heat (MJ) Consumption	-	179 kWh	39.5 kWh	3.75 kWh
Material Consumption	-	-	4 kg Polyacrylamide	-
Anaerobic Digestion				
Electricity (kWh)*/ Heat (MJ) Consumption	88.56 kWh	150 kWh	-461.5 kWh	-589.24 kWh
Material Consumption	-	-	-	0.003 kg NaOH
Dewatering				
Electricity (kWh)/ Heat (MJ) Consumption	49.09 kWh	70 kWh	61.5 kWh	83.57 kWh
Material Consumption	3.72 kg Acrylonitrile	-	4.31 kg Polyacrylamide	14 kg Polyacrylamide, 0.3 kg Iron Chloride
Incineration				
Electricity (kWh)/ Heat (MJ) Consumption	9.5 kWh -6,284 MJ	-623.7 kWh	-749.5 kWh	-70.35 kWh
Material Consumption	31 kg Fuel oil 273 kg Animal waste 12.2 kg NaOH 4.96 kg Lime 3.72 kg Ammonia	46.5 Nm^3 Natural Gas	45 L Diesel 200 kg Coal 20.25 kg NaOH	4.1 Nm^3 Natural gas 22 kg NaOH 57.3 kg Limestone 7 kg HCl 8 kg Charcoal
Global Warming Potential (kg CO_2 eq)	250[1] 343.8[#]	882	4000	-228.8

*Negative values indicate energy production

[1] System is Anaerobic digestion+Dewatering+Land Application

[#]System is Dewatering+Incineration

As we can see, characterization results vary substantially and so does the energy and material use. Fluidized bed combustion was used in all incineration processes except for Xu et al. where the conversion technology was not specified. It can also be noticed that the energy generated in other studies is more than 10 times as compared to this report. This can be attributed to use of ancillary fuels to aid combustion. Incineration process considered in this study uses natural gas only to start up the process, after which the combustion is self sustaining [66]. Whereas other studies use considerable amount of fossil or biogenic fuels. It should also be noted that, this report presents a better overview in terms of materials used for fluegas cleaning processes. Negligible fossil carbon emission during incineration process in this study leads to a negative GWP score.

The minor difference between Xu et al. and this study for energy production from anaerobic digestion is due to different kind of stirring system used in the digester [101]. Hospido et al. flare the produced biogas instead of reusing it for energy production, thus they have a net energy consumption [42]. For Hong et al, they considered average digestion data for Japan giving a small energy generation. Data from CBS shows that anaerobic digestion installations of capacity less than 25,000 p.e. have a net energy consumption [2].

Dewatering energy data is also comparable. Due to limited description of technologies in scientific journals, the reason behind higher polyacrylamide consumption could not be ascertained. However, the dewatering data in this report has been verified by three sources especially for the Dutch scenario [98], [49] and [2]. However it can be said, polyacrylamide consumption has been rising in the recent recent years in dewatering installations in the Netherlands with a 20% increase in 2010 as compared to 2007 [49]. Similar verification was also done for thickening data with the help of other sources [7], [98]. Use of different agitation systems can explain the difference in energy use above.

For supercritical water gasification, only 3 LCA studies excluding this report were found. Even though the end goal and the feedstock did not match, an attempt was made to compare these studies. A comparison based on the important inventory results and global warming potential can be found in table 5.3.

Table 5.3: Comparison of inventory analysis and global warming potential results from supercritical water gasification system with similar systems in other studies. Results are presented in terms of 1 ton DS of biomass processed.

	Gasafi et al, 2003 [32]	Luterbacher et al, 2009 [57]	Gkranas, 2015 [33]	This report
Supercritical Water Gasification				
Feedstock	Sewage Sludge	Manure	Manure	Sewage Sludge
End goal	Hydrogen	SNG*	SNG	Energy
Organic conversion	100%	100%	100%	75%
Dry solids in feed (wt. %)	20%	27%	20%	13%
Precipitation of inorganics	100%	100%	100%	100%
Material use	320 Nm^3 Nat. Gas	-14.4[1] kg Fertilizer -10.1 kWh Elect.	-14.4 kg Fertilizer -13.4 kWh Elect.	2.63 kg NaOH
Product	60.7 kg H_2	68.7 Nm^3 SNG	62 Nm^3 SNG	244.5[#] kWh Elect.
Global Warming Potential (kg CO_2 eq)	725.47	-1303.9	-97	-144.8

*SNG - Synthetic Natural Gas
[1] Estimated from study
[#]Energy Recovery in a SOFC

Compared to other studies, this report takes into account experimental results. We can see the organic conversion considered in this report is 75% which is a more practical value for current SCWG installations. Same can be said about the dry solids content in the feed which is assumed at 13% instead of 20% in the compared studies. Experiments with DS content above 12-13% were unsuccessful because of reactor clogging [48]. The reason for high value of climate change indicator for Gasafi et al. is because they did not consider avoided hydrogen production from fossil methane reforming. Studies of Luterbacher et al. and Gkranas are fairly comparable in terms of systems considered. However, the huge difference between the CO_2 equivalents stems from an assumption of avoided untreated manure emissions. These values amounted to 0.55 kg CO_2 in Luterbacher et al and 0.06 kg CO_2 in Gkranas. It is also interesting to note that results from Gkranas are based on ideal reactor conditions and also taken into account avoided emissions from untreated manure. Nevertheless, this report promises lower global warming impact possibly since the emission savings from fossil intensive Dutch electricity are substantial.

Only one LCA study by Nuss et al. based on plasma gasification was found in the literature [68]. Important inventory results and global warming potential of this thesis and Nuss et al are compared in table 5.4. Chemical energy of syngas per kg feed is higher for sewage sludge even though woody biomass is expected to have lower ash content. This is possibly due to lower moisture content in the sludge feed. Sludge is dried to 10% DS using natural gas since accurate data about other feed conditions and gasification agents was not available. Also, since steam is used as a gasification agent instead of air, dilution due to nitrogen does not occur. Syngas cleaning system is identical in both studies. NaOH and Coke consumption is slightly more in this report. Data about charcoal and limestone use was not available. However, it was mentioned that these materials are indeed used in the process. Global warming potential of both studies are comparable. The reason for the slight difference

Table 5.4: Comparison of inventory analysis and global warming potential results from plasma gasification system in this study and Nuss et al. Results are presented in terms of 1 ton DS of biomass processed.

	Nuss et al, 2013 [68]	This report
Plasma Gasification		
Feedstock	Woody Biomass	Sewage Sludge
End goal	Energy	Energy
Moisture content feed (wt. %)	25%	10%
Syngas chemical energy (MJ/per kg feed)	7.92	10.9
Material use NaOH (kg per kg syngas) Charcoal (kg per kg syngas) Limestone (kg per kWh) Coke (kg per kWh) Other	 0.02 Not available Not available 0.063 119.5 kg fuel oil	 0.04 0.014 0.16 0.089 23.37 Nm^3 Nat. gas
Net Electricity Produced (kWh)	620	486*
Global Warming Potential (kg CO_2 eq)	-52.17	154.2

*System with energy recovery in Gas Engine

can stem from one of the following reasons: difference in electricity mix between USA and NL, feedstock, choice of gasification agent, geographical conditions and difference in energy conversion process of syngas. The category indicator for Nuss et al had to be converted to kg CO_2 eq per ton of dry biomass.

5.2. Sensitivity Analysis

Sensitivity analysis helps in analyzing robustness of results with respect to uncertainties in LCA modeling. These uncertainties can arise due to effect of different assumptions, lack of representative data and dependency of data on ambient conditions. A number of possible alternatives were considered out of which not all of them had a significant affect on results which should be considered a good sign.

Effect of Functional Unit

Result of changing the functional unit from 1 ton DS processed to 1 MJ energy produced is presented in figure 5.1. Results are presented as a percentage change from the actual functional unit considered. It is interesting to note that only SCWG systems processing raw sewage sludge see an increase in the weighted score. This can be attributed to relatively low energy production from both of these systems. The electricity replaced in the grid increases however, this increase is more than compensated by increase in waste deposited to hazardous waste landfill. Whereas in all the other systems, increase in electricity replaced in the grid causes a decrease in environmental impact. In plasma gasification systems, we can see a decrease in the weighted score due to increase in electricity generation. Consequently, the natural gas use increases but the score decreases because of electricity mix of Netherlands is more energy intensive and hazardous waste produced in plasma gasification is considerably low. Analyzing the contributing processes of the two high impact categories, marine aquatic toxicity and global warming potential in figures 4.6 and 4.7 respectively makes understanding this subsection easier.

If avoided emissions were to be considered as done here [33], the presented results may change drastically. Avoided emissions from raw sludge management was not considered here since there is adequate sludge treatment capacity in the Netherlands. The regulations around wastewater and sludge treatment are quite stringent meaning raw sludge is not disposed to the environment without treatment. Furthermore, as highlighted in section 1.3, landfill or agricultural application of sewage is virtually impossible due the organic and heavy metal limits. Hence it seems to be an unreasonable assumption for the Dutch scenario.

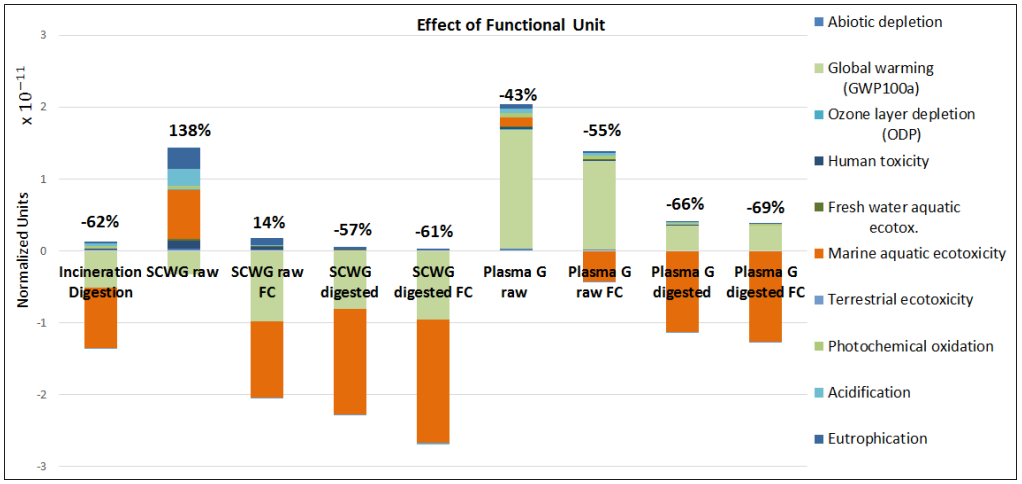


Figure 5.1: Sensitivity of the results with respect to functional unit change from 1 ton DS sludge processed to 1 GJ energy generated.

For LCA studies, functional unit of energy produced should be avoided mainly because of two reasons. Firstly, the aim of sludge treatment chain is to process sewage sludge instead of generating energy. Secondly, since the carbon dioxide emissions from thermochemical conversion of sludge is biogenic, LCA results might favor a technology which produces lower energy per kg DS sludge processed. Or if the functional unit of energy produced is to be used, the assumption of replacing dutch electricity grid should be avoided to obtain credible results.

Choice of Allocation

Allocation was required for the biogas production step. Energy based allocation was considered here with biogas and digestate allocated at 39% and 61% respectively. If economic allocation is considered, biogas has a positive economic value whereas digestate has a negative value. However, all 100% of the allocation cannot be alloted to biogas since aim of digestion step was not to produce biogas but to process sludge. Hence in the digestion of sewage sludge, allocation values are 80% and 20% to biogas and digestate respectively.

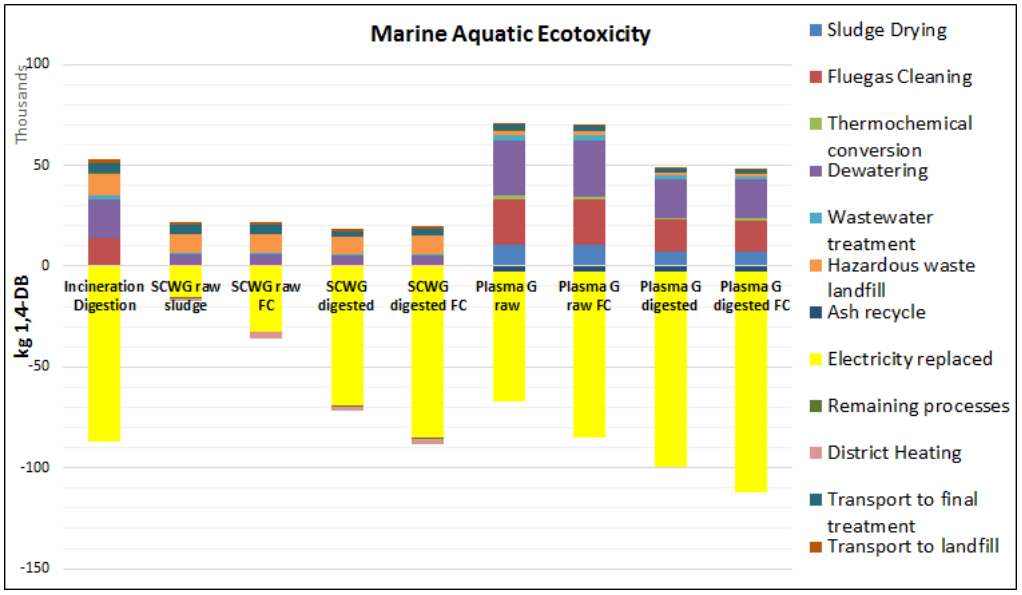


Figure 5.2: Contribution analysis for marine aquatic ecotoxicity when energy based allocation is changed to economic based allocation.

Total environmental impact remained the same which was expected. A change in contribution of different unit processes was expected. However this change was less than 1% of the originally obtained values. Resulting change in contribution analysis in marine aquatic toxicity is shown in figure 5.2. Change in results is so small, it cannot be seen in the figure. Anaerobic digestion is preceded by thickening steps which use very little electricity and materials. This is the probable reason behind almost no sensitivity to the allocation method.

SCWG Feed and Electricity Sensitivity

While working on the SCWG model it was noted that the electricity produced was highly dependent on the DS content of the sludge feed. For example, under the modeled conditions with 75% organic content in raw sludge, net electricity production was -189.3 kWh for 10% DS and 252 kWh for 15% DS. Hence it warranted a sensitivity analysis. The results are shown in figure 5.3 as error bars when the DS content is varied from 10 to 15%. As we can see, the resulting environmental impacts are very sensitive to the amount of dry solids in the feed. Lower environmental impact is obtained with high DS and high electricity production and vice-versa.

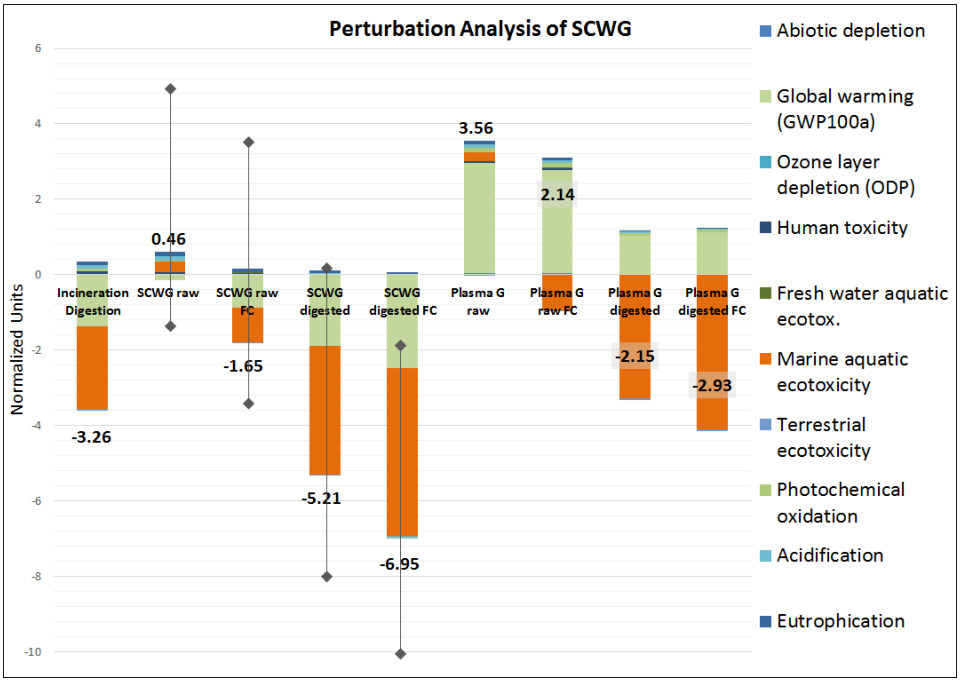


Figure 5.3: Sensitivity/Perturbation analysis of Supercritical water gasification system when the dry solids content in feed is varied from 10% to 15% DS.

The results from this analysis can also help us analyze the sensitivity of results to the amount of electricity produced. Mainly because it is only net electricity production which varies considerably with change in feed DS. There is little change associated with other factors such as material use, waste produced and amount of effluent sent to wastewater treatment. This can be attributed to strong dependence of the impact categories with relatively high values, in this case, global warming potential and marine aquatic ecotoxicity, on the electricity use.

5.3. Conclusions

In this thesis, environmental impacts of various technological options for the sludge treatment chain were studied. Especially options related to energy recovery were explored using the LCA methodology. It is found that results are highly dependent on the energy recovered, particularly electricity. This is because energy generation is an environmentally intensive process and due to exergy allocation, electricity accounts for majority of lifecycle emissions. The research questions outlined at the start of this report will be answered briefly in the following.

1. *How do different components namely, anaerobic digestion, thermochemical conversion, and electricity generation from syngas affect the environmental impact of a sludge chain?*

- Under the modeled conditions, anaerobic digestion results in the highest electricity production per ton DS of sewage sludge processed. Even higher than the considered thermochemical technologies. Thus given the sensitivity of results to electricity generation, sludge chains with anaerobic digestion step performed significantly better. Moreover, due to reduced volume of the sludge post digestion, savings were also achieved in transportation.

- Supercritical water gasification was the most favored thermochemical conversion technology because of two reasons. Firstly, fluegas was relatively clean and only desulphurization step was required before processing it. Secondly, electricity and materials (polyacrylamide and iron chloride) saved due to less degree of dewatering required (13% DS instead of 25%). However, there are two possible sources of uncertainties for this conclusion. First, volatility of energy generation due to dry solids content in feed. Second, feasibility of wastewater treatment step post gasification.

- Even though environmental impacts of plasma gasification systems were the highest, substantial impact was due to drying of sewage sludge. Hence, this technology can still offer significant benefits for feedstocks with lower moisture contents. It was also noticed that LCA did not provide sufficient incentive for the relatively high quality of ash produced.

- The current sludge treatment chain in the Netherlands is environmentally competitive compared to the considered novel technologies. Use of less emission intensive materials for fluegas cleaning and wastewater treatment may offer significant environmental benefits to the present scenario.

- Gasification technologies combined with SOFC's perform better than when gas engines are used. This is because, state-of-the-art gas cleaning systems already remove pollutants at a level which can ensure safe working of SOFC's. Moreover in SOFC's, NO_x emissions are very low and electrical efficiency is comparatively high.

2. *What are the most environmentally intensive processes in the considered sludge chains and what can be possible recommendations to improve them?*

- Syngas and resultant wastewater treatment utilize emission intensive materials such as NaOH and HCl. New technologies for gas cleaning and wastewater treatment can be explored. For example, Ammanox process for N-removal from water.

- Impact of transport was significant especially for SCWG systems since it utilized sludge with 13% DS. Transportation distances were not optimized in this report. This aspect can be looked into for future policy making.

- Mechanical dewatering has significant electricity and material requirements. Apart from the electricity use, polacyrlamide use as a coagulant also had a high environmental impact. This is one of the disadvantages of incineration and plasma systems compared to SCWG. Use of a less emission intensive coagulant can result in environmental benefits.

- Natural gas use for plasma gasification for thermal drying has substantial environmental consequences. Moreoever, steam input required for the plasma reactor utilizes all of the heat co-produced in plasma gasification systems. Use of other gasification agents such as Nitrogen/Air/Oxygen for sewage sludge gasification can be explored but this might be at a cost of lower energy ouput. For thermal drying, utilization of waste heat or better heat integration in plasma system may decrease its impact. Especially, recovery of sensible heat of syngas leaving the reactor by avoiding quenching of gas at the expense of additional dioxin removal stage may have considerable benefits.

What combination of the sludge chain technologies considered result in the minimum environmental impact in the Netherlands?

The sludge chain of Thickening → Anaerobic Digestion → Screw Press → SCWG → SOFC resulted in the lowest environmental impact under modeled conditions and the adapted weighting system. The system had a consistently low indicator scores except for the impact category of Eutrophication. Due to large amount of effluent produced to be treated again in a WWTP. However, careful planning of the treatment facilities is necessary to optimize environmental savings. The following points should taken into consideration:

- Impact of transportation is significant for SCWG systems since the sludge has to be processed at 13% DS. Thus, for 1 ton DS sludge, total weight to be transported for the case of SCWG is almost 80% higher than sludge with 25% DS.
- Elaborate wastewater treatment of effluent post-SCWG is required owing to high amount of organic compounds present.
- Anaerobic digestion is only economically feasible for capacities of more than 30,000 population equivalents or higher.

Hence looking at the above constraints, to obtain maximum environmental benefits, a shift from the current sludge treatment scenario is required. Current scenario involves wastewater treatment at more than 250 WWTPs distributed all over the Netherlands. Sludge after stabilization and dewatering at 25% DS is transported for thermochemical conversion at around 10 plants.

Best possible approach of the above sludge chain would be to encourage large scale water treatment plants with an on-site digestion and supercritical water gasification of the produced sludge. This would ensure feasibility of digestion and also eliminate transportation of sludge a moisture content of 87%. Furthermore, this will also avoid transportation of effluent for treatment post gasification.

5.4. Recommendations

Based on the conducted research and the available information about the sludge treatment, recommendations have been outlined in this section. The suggestions to decrease the environmental impact are included in the previous section. In this section, recommendations for future work based on limitations of this study and LCA in general are presented.

- Apart from the reference scenario of Digestion followed by Incineration, systems with gasification technologies present techno-economic risks. An LCA does not take this into account. A techno-economic feasibility study will further help in decision making regarding these technologies.
- Impact of infrastructure was not considered either in foreground or background processes due to lack of data. Study conducted by Hong et al [41] concluded that infrastructure accounted for less than 2% of total environmental impact however, equipment manufacture played an important role. In future studies, impact of equipment can be investigated.
- Sludge ash contains approximately 10-20% phosphate. This can be recovered and reused for manufacture of fertilizers [29]. Such technologies are already being explored by Dutch sludge mono-incineration companies [67]. It is recommended to include this technology for future LCA studies.
- For hazardous mineral waste, it is assumed there is no leaching of waste due to stability of salt mines. However there have been instances of water flooding these mines [80]. Furthermore, LCA does not sufficiently incentivize production of non-hazardous waste which is easily recyclable. Hence due to these limitations in the LCA methodology itself, supplementing this study with tools such as Risk Assessment or Circularity Indicator might provide a better overview of the systems.
- Based on the observations from this report, plasma gasification may prove to be a promising technology for MSW treatment. Sufficient data although not abundant, is available for conducting a LCA study which may provide promising results [21].
- Results of SCWG are fairly volatile to the DS content of feed sludge. These results could not be verified due to unavailability of data. A SCWG plant of capacity 150 kg/hour is in construction by a consortium of partners [1] in Netherlands [67]. Validation/Modification of SCWG data according to results obtained at this installation will provide more credibility to this report.
- Based on the conclusions from this study and Gkranas [33], it will be interesting to conduct an LCA to include production of synthetic natural gas from sewage sludge as one of the systems.

[1] This consortium consists of the following stakeholders: STOWA, Water Board De Dommel, Water Board Aa en Maas, Procede and Glaesum

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A

Ecoinvent Processes

The details of the background processes used from the Ecoinvent database v3 are presented in this Appendix. These processes were indicated by a symbol of [E] in front of them to signify that they are utilized from the Ecoinvent database. Details of these processes can be found in table A.1 below. Allocation type of recycled content was used, seen as 'Alloc, Rec' in the below table. Background processes for which data was not available for Netherlands, data from other countries was used. Geographical codes in brace brackets {CH}, {GLO}, {RER}, {DE} and {NL} represent Switzerland, Global, Europe, Germany and Netherlands respectively. More information about nomenclature can be found on ecoinvent.org.

Table A.1: Details of background processes used from Ecoinvent v3 database.

Sr. no.	Process used in systems	Ecoinvent Process details
1.	Polyacyrlamide	Polyacrylamide {GLO}|,production | Alloc Rec, S
2.	Electricity	Electricity, medium voltage {NL}| market for | Alloc Rec, S
3.	Heat	Heat, district or,industrial, natural gas {CH}| heat and power co-generation, natural gas, 1MW,electrical, lean burn | Alloc Rec, S
4.	Sulfur	Sulfur {CH}|,petroleum refinery operation | Alloc Rec, S
5.	Sodium Hydroxide 50%	Sodium hydroxide„without water, in 50% solution state {RER}| chlor-alkali electrolysis„membrane cell | Alloc Rec, S
6.	Iron chloride	Iron (III) chloride, 40% in H2O, at plant {RER}| Alloc, Rec, S
7.	Limestone	Limestone, milled, packed, at plant {RER}| Alloc, Rec, S
8.	Hydrochloric Acid	Hydrochloric acid„without water, in 30% solution state {RER}| Mannheim process | Alloc Rec, S
9.	Met. Coke	Coke {DE}| coking |,Alloc Rec, S
10.	Charcoal	Charcoal {GLO}|,production | Alloc Rec, S
11.	Natural Gas	Natural gas, high pressure {NL}| market for | Alloc Rec, S
12.	Recycle in Asphalt	Sand {DE}|,production | Alloc Rec, S
13.	Recycle in Ceramic Tiles	Silica sand {DE}| production | Alloc Rec, S
14.	Wastewater Treatment	Wastewater, average,{CH}| treatment of, capacity 5E9l/year | Alloc Rec, S
15.	Waste Landfill	Hazardous waste, for,underground deposit {DE}| treatment of hazardous waste, underground deposit |,Alloc Rec, S
16.	Transport	Transport, freight, loryy >32 metric ton, EURO6 {RER}| transport, freight, lorry >32 metric ton, EURO6 | Alloc Rec, S

B

Sludge Data

The collected sludge data from wastewater treatment plant in Oijen is presented in table B.1. This data is converted into proximate and ultimate analysis in order to use them for the models using the following steps:

1. Based on the literature study, the major elements are classified whether they make up the organic fraction of sludge (C, H, O, N, S, Cl) or the inorganic fraction (the rest).
2. The oxygen calculated using the difference of all the elements from 1 gram (1000 mg/g DS) of sludge to get a value of 359.5 mg/g DS.
3. Inorganic metals (except trace metals) are present in the ash content of sludge in the form of their respective oxides (CaO, MgO, etc). Oxygen thus contained in the inorganic part is subtracted and oxygen in organic matter is obtained as 218.9 mg/g DS.
4. Finally, the proximate and ultimate analysis were calculated for the corresponding elements. The results can be found in table B.2.

This data is modified solely for the purpose of modeling. Furthermore, the data about trace heavy metals is not modified since it is already available in the required format. Another important note is that the functional unit of this study is 1 ton DS processed. Hence all calculations from now on will be scaled accordingly.

Table B.1: Data of sludge produced from wastewater treatment plant in Oijen, Netherlands [48]

Major elements	Content (mg/g DS)	Trace elements	Content (ug/g DS)
C	415.75	As	0
H	61.75	Cd	0
N	45.25	Cu	1670
S	6	Pb	67.75
P	21	Zn	800
Ca	18	Cr	62.75
K	3.25	Mo	5.75
Mg	3	Ni	9.5
Na	1	Cl	2217.5
Si	29.75	Hg	0.225
Al	13.75		
Fe	16.7		

Table B.2: Proximate and Ultimate Analysis of sludge data

Primary sludge	**(%)**
Dry matter	1
Moisture	99
Ash (db*)	24.75
Organic (db*)	75.25

Secondary sludge	**(%)**
Dry matter	0.8
Moisture	99.2
Ash (db*)	24.75
Organic (db*)	75.25

Element	**(% daf^)**
C	55.44
H	8.23
N	6.03
S	0.80
O	29.19
Cl	0.30
Total	100

Metal Oxide	**% of ash**
SiO_2	25.75
Al_2O_3	10.49
CaO	10.18
P_2O_5	39.79
Fe_2O_3	9.64
MgO	2.02
Na_2O	0.54
K_2O	1.58

**db - dry basis, ^daf - dry ash free basis*

C

Anaerobic Digestion

Biogas Production

Some important assumptions for modeling the digestion process are presented in table C.1. The data sources are already presented in the inventory analysis.

Table C.1: Proximate analysis of sludge fed to anaerobic digester *(left)* and important process conditions regarding the anaerobic digestion model *(right)*.

Proximate analysis	%
Moisture	95.56
Dry Matter	4.44
Ash	24.75
Organic matter	75.25
Volume decrease due to thickening	80

Important assumptions	Quantity
Digester temperature	35°C
Residence time	30 days
Volatile matter destroyed	40%
Biogas produced per kg volatile matter destroyed	0.9 Nm^3
Allocation (Energy Content) Biogas Digestate	 39% 61%

From the assumption of 40% volatile matter destroyed during digestion and 0.9 Nm^3 biogas produced per kg volatile matter destroyed, we get the total biogas produced in equation C.1.

$$\begin{aligned}\text{Biogas produced } (Nm^3) &= \text{Volatile matter in sludge (kg)} * \text{volatile matter destroyed (\%)} \\ &\quad * \text{Biogas produced per kg volatile matter destroyed } (Nm^3) \\ &= 752.47 * 40\% * 0.9 \\ &= 270.89\, Nm^3\end{aligned} \tag{C.1}$$

Average biogas composition is obtained from sludge digestion manual [65] based on scientific and empirical data from sewage sludge treatment plants in the Netherlands. Assuming ideal gas behavior, we know that volume composition is the same as molar composition. Hence from the we get the volume composition of the produced Biogas. From Avogadro's law we know that, 1 kmol of gas will occupy 22.4 Nm^3 of volume. Hence amount of gas in kmol is calculated using the following equations C.2 and C.3. As a result, we get the composition of biogas as shown in table C.2.

$$\text{Kmol of gas} = \frac{\text{Volume of gas (kmol)}}{22.4} \tag{C.2}$$

$$\begin{aligned}\text{Kmol of Methane} &= \frac{170.66}{22.4} \\ &= 7.619\, kmoles\end{aligned} \tag{C.3}$$

Table C.2: Biogas composition in terms of volume and kilomoles

Biogas composition	**Volume %**	**Nm^3**	**Kmol**
CH_4	63	170.66	7.62
CO_2	35	94.81	4.23
H_2O	1.8	4.88	0.22
N_2	0.2	0.54	0.024
Trace	**$mg.m^{-3}$ (ppm)**	**mg**	**Kmol**
H_2	60	162.53	8.13E-05
H_2S	400	1083.56	3.2E-05
NH_3	500	1354.45	7.97E-05

From the above information, elemental composition of biogas is calculated by using the equation C.4 to get the results in table C.3.

$$\text{Amount of carbon in the gas} = \text{Amount of carbon in the form of } CH_4 \text{ and } CO_2 = 7.62 + 4.23 = 11.85\, kmol = 142.22\, kg \tag{C.4}$$

Table C.3: Elemental composition of Biogas

Element	**kmol**	**weight (kg)**
C	11.85	142.22
H	30.91	30.91
O	8.68	138.93
N	0.048	0.68
S	3.19E-05	0.001
Total	51.49	312.74

Using this information, composition of the slurry is calculated. An important assumption here is the inorganic fraction remains in the liquid phase. Hence all the ash and heavy metals stay in the digester except for mercury, arsenic and antimony as discovered in background reports of Ecoinvent [23]. Looking at the sludge composition considered, only mercury is relevant which escapes with the biogas. We already have the ultimate analysis of sludge from which the biogas composition is subtracted to obtain elemental composition of slurry which can be found in table D.1.

Table C.4: Ultimate *(left)* Proximate analysis *(rightt)* of sewage sludge post digestion.

Element	**Kmoles**	**kg**	**wt % (daf)^**
C	22.92	274.98	62.85
H	31.05	31.05	7.10
O	5.05	80.72	18.45
N	3.19	44.73	10.22
S	0.19	6.02	1.38
Total	62.40	437.51	100.00

Content	**kg**	**wt%**
Moisture	21400.61	96.9
Dry matter	685.04	3.1
Ash (db*)	247.53	36.13
Organic (db*)	437.51	63.87

**db - dry basis, ^daf - dry ash free basis*

Ash and heavy metal contents of the sludge at the end of this step is presented in table B.1. Data from the biogas is used for energy generation process while data from the sludge composition is further dewatered and subjected to thermochemical treatment. Ash and trace metal content of the sludge remains unchanged except for mercury which is reduced to zero.

Heat and Electricity input

Energy is required in two forms for an optimized digestion process. First is agitation for mixing and second being heat to maintain the digester temperature around 35°C [65]. First we need to fix the digester dimensions to advance with the calculations. Digester volume of 11,500 m^3 as built at wasterwater treatment plant in Amsterdam west is assumed here [92]. From sludge digestion manual, the ideal height to diameter ratio of 0.8 of a digester is obtained [65]. Tank is assumed to be cylindrical and the dimensions of the tank are calculated using the equation C.5.

$$\begin{aligned} \text{Volume of Tank } (m^3) = 11,500 &= \pi * R^2 * H \\ &= \pi * R^2 * 0.8 * 2 * R \end{aligned} \tag{C.5}$$

Hence we get the values of Radius = 13.18m and Height = 21.1m.

Agitation is done by compressing the gas in the digester and bubbling it from the bottom of the digester. From sludge digestion manual, we get the electrical load for such a system at 5 W/m^3 with the system remaining active for 8 hours each day. Thus the electrical load for the digester capacity mentioned above is 57.5 kWh/hour. The digestion is active for 30 days. Thus, the energy consumption for 30 days can be calculated using the equation C.6.

$$\begin{aligned} \text{Electricity consumed in 30 days (kWh)} &= \text{Electricity consumed in one hour} \\ &\quad * \text{No. of hours activity per day} * 30 days \\ &= 57.5 * 8 * 30 \\ &= 13800\, kWh \end{aligned} \tag{C.6}$$

Since the dry solids content changes from 4.65% to 3.1%, if we consider an average DS content of 3.77%, amount of dry solids estimated inside the digester can be found by assuming the density of the sludge as 1000 kg/m^3 in equation C.7. Density can be assumed at a low value

$$\begin{aligned} \text{Amount of sludge in digester (tonnes)} &= \text{Volume of digester } (m^3) * \text{sludge DS content (wt. \%)} \\ &\quad * \text{Density of sludge } (kg.m^{-3}) * 10^{-3} \\ &= 11500 * 3.77\% * 1 \\ &= 433.91\, tonnes \end{aligned} \tag{C.7}$$

Finally, using about two equations we get an electricity consumption 31.8 kWh for 1 ton of sewage sludge DS processed.

Average monthly temperatures in the Netherlands were obtained from CBS [2] to get an average annual ambient temperature of 10.1 °C and sludge temperature of 15 °C. For simplifying calculations, it was assumed that digestion is a steady state process with mixed sludge continuously entering the digester. Hence the sludge input and output flow rate according to the above calculations will be 4.44 kg/s. In practice, this is more complicated. With primary sludge (50% by wt.) fed in batches and secondary sludge (50% by wt.) fed continuously [65]. Heat input required to the reactor can be given by equation

$$\begin{aligned} \text{Amount of Heat required by the digester} = &\text{Heat required to raise temperature of incoming} \\ &\text{sludge + Heat required to maintain temperature} \end{aligned} \tag{C.8}$$

Similar methodology was also followed by Zupancic and Ros [104]. Heat required to raise the temperature of incoming sludge is given by equation C.9. Where $\dot{m}$ is the mass flow rate of sludge and □T is the difference between digester temperature and incoming sludge temperature. Specific heat of sludge is denoted by C_p = 4.2 $kJ.kg^{-1}.K^{-1}$ and obtained again from Zupancic and Ros [104].

$$\begin{aligned}\text{Heat required by incoming sludge} &= \dot{m} * C_p * \Delta T \\ &= 4.44 * 4.2 * (35 - 15) \\ &= 372.69\, kW\end{aligned} \tag{C.9}$$

Further, if $\dot{U}$ is the overall heat transfer coefficient for heat transfer from sludge to air and A is the surface area of the digester exposed to air, then heat loss from the digester can be calculated using equation C.10.

$$\begin{aligned}\text{Heat loss from digester to air} &= \dot{U} * A * \Delta T \\ &= 0.265 * 12135.6 * (35 - 15) * 10^{-3} \\ &= 80.1\, kW\end{aligned} \tag{C.10}$$

$\dot{U}$ = 0.265 W.m^{-2}.K^{-1} for heat transfer to air and 0.235 W.m^{-2}.K^{-1} for heat transfer to ground. Adding heat loss to the ground, total heat loss meaning heat requirement from digester to maintain a stable temperature amounts to 83.3 kW. Hence the heating load is 456 kW while the heat required for a 30 day period is 1181.92 GJ which amounts to 2363.85 MJ/ton DS sludge.

Biogas Cleaning

Biogas cleaning principle and methodology is already explained in detail in the inventory analysis. Hydrogen sulphide, water and ammonia are removed from the biogas. Hydrogen sulphide removal is done by using THIOPAQ process. Water removal occurs by passing the biogas through an underground pipe, the collected condensate is sent back to WWTP for treatment. Ammonia is partially removed in both of the above treatment systems[96]. Energy and materials are only consumed in the THIOPAQ process with exact values obtained from EPA [18] and Thesis report from Gkranas [33]. Details about the cleaning step are presented in table C.5 followed by composition of cleaned biogas in table C.6.

Table C.5: Important parameters regarding biogas cleaning process

Parameter	Value
Sulphur removal efficiency	99%
Water removal efficiency	95%
Ammonia removal efficiency	95%
Heat use (MJ per Nm^3 Biogas)	0.685
Elect. use (kWh per Nm^3 Biogas)	0.151
NaOH use (kg per kg sulphur recovered)	0.44

Amount of sulphur recovered can be found from the equation C.12.

$$\begin{aligned}\text{Sulphur recovered (kg)} &= \text{No. of moles of } H_2S \text{ in gas} \\ &\quad * \text{Sulphur removal efficiency} * \text{Sulphur Mol.Wt.} \\ &= 3.16 * 10^{-5} * 99\% * 32 \\ &= 0.001\, kg\end{aligned} \tag{C.11}$$

Above equation and tables are then used to scale the energy and material use are then scaled according to the functional unit, results can be found in the inventory table 3.5.

Energy Production

Biogas is utilized in a cogeneration system for heat and electricity production. Important assumptions for the cogeneration system were obtained from digestion manual [65] and are presented in table C.7.

Thus energy recovery is calculated is using the following equation

Table C.6: Composition of cleaned Biogas

Biogas composition	**Kmoles**	**volume %**
CH_4	7.62	64.1
CO_2	4.23	35.61
H_2O	0.0109	0.0916
N_2	0.0242	0.2035
Trace		
H_2	8.1267E-05	0.00068
H_2S	3.18694E-07	2.68112E-06
NH_3	3.98368E-06	3.3514E-05

Table C.7: Important values to model energy production from biogas

Parameter	**Value**
Biogas calorific value	22 MJ/Nm^3
Electrical energy efficiency	40%
Heat energy efficiency	45%

$$\begin{aligned}\text{Electricity recovered (kWh)} &= \text{Biogas produced (Nm}^3) * \text{Biogas calorific value (MJ/Nm}^3\\ &\quad * \text{Electric energy efficiency} * 0.278\\ &= 270.89 * 22 * 45\% * 0.278\\ &= 662.18\, kWh\end{aligned} \tag{C.12}$$

Same was done for heat generation. However, according to multiple sources, it was confirmed that heat produced from CHP unit is only utilized to maintain the digester temperature and the rest of the energy is lost [98], [65]. Hence only electricity output is entered in the inventory analysis.

To calculate emissions from the CHP, first emission limits of SO_2, NO_x and CO obtained from Dutch limitations for mid-sized combustion plants. These values are 200 mg.m^{-3}, 340 mg.m^{-3} and 10 mg.m^{-3} respectively calculated at 3% Oxygen content. First, carbon dioxide and water vapor in the fluegas is calculated by the following chemical equation

$$CH_4 + 2O_2 \rightarrow CO_2 + 2H_2O$$

The carbon dioxide and water vapor amounts to a total of 15.67 kmol with partial amount coming from the content already present in the biogas. The total amount of While carbon monoxide and nitrogen oxide content in the fluegas was calculated according to the emission limit values, sulphur dioxide was calculated according to the following chemical reaction.

$$H_2S + 2O_2 \rightarrow SO_2 + H_2O$$

Amount of oxygen (3%) and nitrogen in terms of kmol in the fluegas using oxygen balance in the following equation

$$\text{Volume \% oxygen in fluegas} = \frac{\text{kmol oxygen}}{\text{kmol oxygen + kmol nitrogen + kmol rest of the fluegas}}$$

$$0.03 = \frac{x}{x + 3.762 * x + 15.67}$$

Solving for x, we get kmoles of oxygen and nitrogen as 0.544 and 2.063 respectively. Using the values obtained for NO_x and CO, total composition of fluegas was calculated as shown in table C.8. These values are already scaled to the functional unit. It is also important to note that carbon emissions here are biogenic.

Table C.8: Composition of fluegas leaving the CHP unit.

Gas	**kmoles**	**Nm^3**	**volume %**
N_2	2.06	46.22	11.29
CO_2	11.85	265.47	64.82
O_2 (3% excess)	0.55	12.29	3.00
H_2O	3.82	85.58	20.89
CO	0.00011	0.0024	0.00059
NO_x	0.003	0.068	0.017
SO_2	3.19E-07	7.14E-06	1.74E-06

D

Incineration

Sludge Input

Incineration is carried out in a fluidized bed combustor. Dewatered sewage sludge from anaerobic digestion with the composition shown in table enters the incineration plant.

Table D.1: Ultimate *(left)* Proximate analysis *(rightt)* of sewage sludge post digestion.

Element	Kmoles	kg	wt % (daf)^
C	22.92	274.98	62.85
H	31.05	31.05	7.10
O	5.05	80.72	18.45
N	3.19	44.73	10.22
S	0.19	6.02	1.38
Total	62.40	437.51	100.00

Content	kg	wt%
Moisture	2055.12	75
Dry matter	685.04	25
Ash (db*)	247.53	36.13
Organic (db*)	437.51	63.87

**db - dry basis, ^daf - dry ash free basis*

The detailed process of incineration is already described in inventory analysis, hence in this section, only the calculations will be presented.

Energy Production

Data for energy production is obtained from reports from STOWA and publicly available reports from SNB. Important energy generation data can be found in table D.2. The net energy produced is also calculated which equals 70.35 kWh. Please note that the net electricity produced also includes fluegas and wastewater treatment.

Table D.2: Energy generation from fluidized bed combustion of digested sewage sludge

Parameter	Value
Electricity for FBC (kWh/kg D.S.)	0.31
Electricity wastewater treatment (kWh/kg D.S.)	0.007
Electricity produced (kWh/kg O.C.[1])	0.66
Net Electricity produced	70.35

[1] *O.C. - organic content*

Material Use

Description of materials and their purpose is also explained in the inventory analysis. Limestone, NaOH and charcoal are used for fluegas treatment. Natural gas is used during the initial stages of the combustion process whereas HCl is used in the waste water treatment process. The material use data was received from mono-sludge combustion operating on

fluidized bed combustion [98], [66]. The collected material use data and the scaled results are presented in table D.3.

Table D.3: Material use by sewage sludge incineration process.

Materials used	Unit per kg sludge DS processed	Total amount[1]	Units
Limestone	0.0836	57.27	kg
NaOH 50%	0.0322	22.045	kg
HCl 30%	0.0102	6.96	kg
Charcoal	0.0114	7.793	kg
Natural gas	0.0059	4.07	Nm^3

[1] *Amount scaled to the functional unit of 1 ton DS of sewage sludge processed.*

Waste Produced

Total waste produced is also collected from annual publicly available reports from SNB [66], [67]. Results can be seen in table D.4.

Table D.4: Waste produced in the incineration process

Waste produced	Weight (kg per kg sludge DS processed)	Total amount (kg)
Incineration ash	0.4055	277.81
Fluidized Bed sand	0.015	10.3
Spent adsorbent	0.012	8.19
Salt	0.017	11.65
Total	0.45	307.94

Out of the total waste produced, 50% is assumed to be recycled in asphalt as a foundation material and 50% is disposed in German salt mine. The justification and details of this assumption can be found in inventory analysis, section 3.9. Emissions from the salt mines is only due to transport and material/energy use for landfill maintenance. No leaching is considered since waste is sufficiently isolated from the environment. These parameters are included in the background process of Ecoinvent. Incase of recycle in asphalt, data from dutch emission limits for critical building materials was used [93]. Actual leaching values from asphalt are expected to be higher especially since the Dutch government temporarily made an exception in the year 2005 for higher emission values so as to allow use of incineration ash in asphalt [91].

Table D.5: Amount of heavy metal leached from asphalt recycle.

Element	Leachability (mg/kg)	Amount in 1 ton DS sludge (kg)	Leached (mg)
As	0.5	0	0
Cd	0.005	0	0
Cu	0.03	1.67	0.0251
Pb	91	0.068	3.083
Zn	0.08	0.8	0.032
Cr	0.7	0.063	0.022
Mo	75	0.0058	0.216
Ni	0.07	0.0095	0.00033
Cl	480	2.218	532.2
Hg	2.9	0	0

The amount of heavy metals leached to the environment is calculated using the equation D.1. The result is multiplied by 50% since half the sludge ash is sent to german salt mines.

An example of copper leaching is shown below. Other heavy metals are calculated using the same methodology.

$$\text{Amount leached (mg)} = \text{Amount of element in sludge (mg)} * \text{Leachability (mg)} * 50\% \quad \text{(D.1)}$$

$$\text{Amount leached Copper leached(mg)} = 1.67 * 0.03 * 50\% = 0.0251\, mg$$

E

Supercritical Water Gasification

Sludge Input

For supercritical water gasification, sludge DS content of 13% DS is considered. Hence the input is from screw press dewatering rather than centrifuges as considered for other systems. The sludge composition for raw sludge entering the supercritical water gasification system is presented in table E.1.

Table E.1: Ultimate *(left)* Proximate analysis *(rightt)* of sewage sludge post digestion.

Element	Kmoles	kg	wt % (daf)^
C	34.77	417.20	55.44
H	61.97	61.97	8.23
O	13.73	219.65	29.19
N	3.24	45.41	6.03
S	0.19	6.02	0.80
Total	113.96	752.47	100.00

Content	kg	wt%
Moisture	6692.31	87.00
Dry matter	1000.00	13.00
Ash (*db)	247.53	24.75
Organic (*db)	752.47	75.25

**db - dry basis, ^daf - dry ash free basis*

Results for the modeling of supercritical water gasification model were obtained from a pilot plant investigation of this technology using sewage sludge as a feedstock [48].

A simplified energy and mass balance of this system can be found in figure E.1. This model was developed by researchers of Karlsruhe Institute of Technology, Germany based on the empirical results and empirical simulations. The model was adapted and modified for this thesis to obtain the desired results. The following parameters were modified:

- Organic conversion - Changed from 95% to a more realistic value of 75%
- Dry solids in feed - Since the experiments above the dry solids content of 12-13% DS were unsuccessful due to reactor cloggin, DS content was modified from 20% to 13%.

Mass Balance

Feed

The capacity of the plant is 1000 kg/h of sludge. With respect to the DS and organic wt. %, dry solids and organic contents are calculated. Density is assumed to be at 1000 kg.m−3 but in practice, it is expected to be slightly higher. Temperature of feed sludge is 15°C while it pumped to a pressure of 28 MPa.

Salt and Ash Stream

The salt stream mass balances are calculated using the assumptions made in table E.2. These assumptions are recommended by the source report of this model. Firstly, the total stream separated is calculated as 7% of the feed. This salt stream contains 50% of the ash

in the feed. Ash is also accompanied by 200 g/kg ash of organics. Using this information, the mass flow rates of under the salt stream are calculated.

Table E.2: Important assumptions for modeling supercritical water gasification

Parameter	Value
Fraction of feed separated	7%
Precipitation of inorganics	100%
Ash fraction in stream	21%
Organics in ash	200 g/kg ash

Gas

Total mass flow rate of the gas stream is calculated using the organic conversion using equation.

$$\begin{aligned}\text{Mass flow rate of gas (kg/h)} &= \text{Feed rate of organics (kg/h)} * \text{organic conversion (\%)} \\ &= 97.5 * 75\% = 73.13\, kg/h\end{aligned}$$

Effluent

The mass flow rates of effluent is calculated by the following equations

$$\text{Water in effluent} = \text{Water in feed} - \text{Water in salt stream}$$

$$\text{Ash in effluent} = \text{Ash in feed} - \text{Ash in salt stream}$$

$$\text{Organics in effluent} = (1 - \text{organic conversion}) * \text{organics in feed} - \text{organics in salt stream}$$

It is also assumed that 100% of the ash in the effluent is separated partially by collecting reactor bottoms and partially by screens. The 'total out' box is made only to verify mass flow rates.

Energy Balance

Energy input to the process is required for pumping power and for heating. Since the feed enters at 15°C and the gasification occurs at 700°C and 28MPa. The required pump has a rated power of 16kW. The heat input to the reactor is calculated using equation E.1. Electrical heating is considered here.

$$\text{Energy input} = \text{Enthalpy of leaving streams} - \text{Enthalpy of feed entering} - \text{Heat Loss} \quad \text{(E.1)}$$

Heat loss is assumed to be 23kW which is approximately 18% of the heat input. Enthalpies of the input and output streams were calculated by adding enthalpies of the dry matter and water. Enthalpy of dry matter was calculated as in equation **??** whereas for water it was calculated as in equation E.2 and E.3 respectively.

$$\begin{aligned}\text{Enthalpy of dry matter (kW)} &= \text{Heat Capacity (kJ.kg}^{-1}\text{.C}^{-1}\text{)} * \text{Temperature (C)} \\ &* \text{Mass flow rate (kg.s}^{-1}\text{)}\end{aligned} \quad \text{(E.2)}$$

$$\text{Enthalpy of water (kW)} = \text{Specific enthalpy of water (kJ.kg}^{-1}\text{)} * \text{Mass flow rate (kg.s}^{-1}\text{)} \quad \text{(E.3)}$$

The specific heat capacities and the enthalpies were readily available from the document itself.

Energy Production

Gas is obtained with an average composition and calorific value as shown in table E.3.

Table E.3: Properties of gas produced from supercritical water gasification of raw sludge

	Vol %	kmol
H_2	20	4.55
CO	0.4	0.09
CO_2	34	7.74
CH_4	33	7.51
C_2H_4	0.5	0.11
C_2H_6	12	2.73
C_3H_6	0.2	0.05
C_3H_8	0.5	0.11
H_2S	0.82	0.19

HHV	22 MJ/kg
Chemical energy	446.875 kW
Avg. Mol wt	24.796 kg/kmol
Molar flow	22.76 kmol

Gas cleaning step of only desulphurisation is required. THIOPAQ process as used in anaerobic digestion is assumed. Hence modeling method is not described here. However, material and energy requirements can be found in the inventory analysis in section 3.6. Electrical and thermal efficiency was assumed to be 37% and 29% respectively for CHP. These values were 41% and 35% for SOFC. Calculations are carried out using the above values. The calculation steps will be repetitive, hence they are not shown here. The results can be found in the inventory analysis in section 3.6 for gas engine and 3.8 for SOFC.

Fluegas

Fluegas composition is calculated in exactly the same way as done for anaerobic digestion by obtaining Dutch emission limits for mid-sized combustion plants. Methodology is not presented again but the fluegas composition can be seen in table E.4.

Table E.4: Fluegas composition exiting from gas engine of a supercritical water gasification system of raw sludge.

	kmol	Volume (Nm^3)	kg
CO_2	21.51	481.78	946.36
H_2O	9.32	208.73	167.73
O_2	1.08	24.17	17.26
N_2	4.06	90.92	113.65
	mol	**Nm^3**	**g**
SO_2	25.17	0.56	161.12
CO	0.29	0.01	8.06
No_x	2.45	0.05	112.78

Ash Disposal

Ash disposal scenario is considered to be the same as in incineration since major differences between ash properties are not expected. Leaching of heavy metals from the ash recycle in asphalt is presented in table E.5.

Digestion scenario

The difference in the digestion scenario is the sludge composition, mainly the organic content. Hence the energy recovery is lower. The model for digested sewage sludge can be seen in figure E.2. The results from this system can be found in the inventory analysis. Calculation steps are identical to the ones highlighted in this appendix.

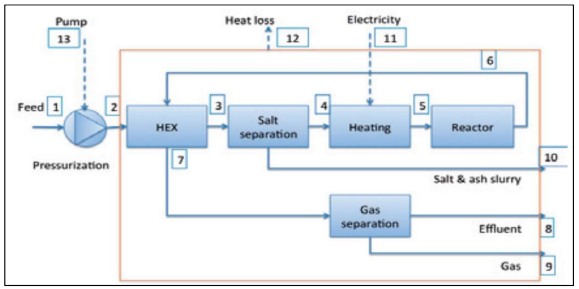
Table E.5: Amount of heavy metal leached from asphalt recycle.

Element	Leachability (mg/kg)	Amount in 1 ton DS sludge (kg)	Leached (mg)
As	0.5	0	0
Cd	0.005	0	0
Cu	0.03	1.67	0.0251
Pb	91	0.068	3.083
Zn	0.08	0.8	0.032
Cr	0.7	0.063	0.022
Mo	75	0.0058	0.216
Ni	0.07	0.0095	0.00033
Cl	480	2.218	532.2
Hg	2.9	0	0

Summary		
Energy input	112.45579	kWe
Pump	16	kWe
Total E in	128.45579	kWe
Energy production		
HHV gas	446.875	kW
LHV/HHV	93%	
Electric efficiency	37%	
Electricity production	153.76969	kWe
Net Electricity	25.313901	kWe
	194.72231	kWh/ton dm

Calculation of Electricity requirement:
Electricity requirement 11 =
sensible heat in (8 + 9 + 10) - sensible heat in 2 + Heat loss 12

Dry matter content	13%	
Organic content	75%	
Heat loss	23	kW



Feed		
Total mass	1000	kg/h
Water	870	kg/h
Ash	32.5	kg/h
Organic	97.5	kg/h
Total mass	0.2777778	kg/s
Water	0.2416667	kg/s
Ash	0.0090278	kg/s
Organic	0.0270833	kg/s
Dry matter	13	%
Ash content	25	%
Density	1000	kg/m3
Temperature	15	deg C
Pressure	28	Mpa
Flow	1	m3/h
	0.0002778	m3/s
Specific enthalpy wate	89.4	kJ/kg
Heat capacity dry matt	2	kJ/kg/C
Sensible heat		
Water	21.605	kW
Dry matter	1.0833333	kW
Heating value		
Organic matter	22	MJ/kg om
	595.83333	kW

Effluent		
Organic conversion	75%	
Temperature deg C	60	
Total mass	855.295	kg/h
Water	817.92	kg/h
Ash	16.25	kg/h
Organic	21.125	kg/h
Total mass	0.2375819	kg/s
Water	0.2272	kg/s
Ash	0.0045139	kg/s
Organic	0.0058681	kg/s
Specific enthalpy water	274.6	kJ/kg
Heat capacity dry matte	2	kJ/kg/C
Sensible heat		
Water	62.38912	kW
Dry matter	1.2458333	kW
Heating value		
Organic matter	22	MJ/kg om
	129.09722	kW

Salt & Ash		
Fraction of feed	7%	
Ash fraction in stream	50%	
Organics in ash	200	g/kg ash
Total mass	70	kg/h
Water	52.08	kg/h
Ash	16.25	kg/h
Organic	3.25	kg/h
Temperature	450	Deg C
Total mass	0.019444	kg/s
Water	0.014467	kg/s
Ash	0.004514	kg/s
Organic	0.000903	kg/s
Specific enthalpy wate	2875	kJ/kg
Heat capacity dry matt	2	kJ/kg/C
Organic matter	22	MJ/kg om
Sensible heat		
Water	41.59167	kW
Dry matter	4.48	kW
Total	46.07167	kW
Heating value		
Organic matter	19.86111	kW

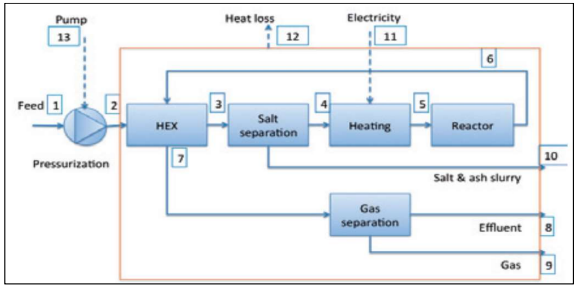
Gas		
Total mass	73.125	kg/h
(equal to from organic mass)		
Total mass	0.020313	kg/s
Temperature	60	deg C
Specific enthalpy wate	2875	kJ/kg
Heat capacity dry matt	2	kJ/kg/C
Organic matter	22	MJ/kg om
Sensible heat		
Dry matter	2.4375	kW
Heating value		
Gas	446.875	kW

Total out		
Total mass	998.42	kg/h
Water	870	kg/h
Ash	32.5	kg/h
Organic	97.5	kg/h
Total mass	0.277339	kg/s
Water	0.241667	kg/s
Ash	0.009028	kg/s
Organic	0.027083	kg/s
Sensible heat		
Water	103.9808	kW
Dry matter	8.163333	kW
Total	112.1441	kW
Heating value	595.8333	kW

Summary		
Energy input	112.16979	kWe
Pump	16	kWe
Total E in	128.16979	kWe
Energy production		
HHV gas	381.33333	kW
LHV/HHV	93%	
Electric efficiency	37%	
Electricity production	131.2168	kWe
Net Electricity	3.0470133	kWe
	23.438564	kWh/ton dm

Calculation of Electricity requirement:
Electricity requirement 11 =
sensible heat in (8 + 9 + 10) - sensible heat in 2 + Heat loss 12

Dry matter content	13%	
Organic content	64%	
Heat loss	23	kW



Feed		
Total mass	1000	kg/h
Water	870	kg/h
Ash	46.8	kg/h
Organic	83.2	kg/h
Total mass	0.2777778	kg/s
Water	0.2416667	kg/s
Ash	0.013	kg/s
Organic	0.0231111	kg/s
Dry matter	13	%
Ash content	36	%
Density	1000	kg/m3
Temperature	15	deg C
Pressure	28	Mpa
Flow	1	m3/h
	0.0002778	m3/s
Specific enthalpy wate	89.4	kJ/kg
Heat capacity dry matt	2	kJ/kg/C
Sensible heat		
Water	21.605	kW
Dry matter	1.0833333	kW
Heating value		
Organic matter	22	MJ/kg om
	508.44444	kW

Effluent		
Organic conversion	75%	
Temperature deg C	60	
Total mass	857.44	kg/h
Water	817.92	kg/h
Ash	23.4	kg/h
Organic	16.12	kg/h
Total mass	0.2381778	kg/s
Water	0.2272	kg/s
Ash	0.0065	kg/s
Organic	0.0044778	kg/s
Specific enthalpy water	274.6	kJ/kg
Heat capacity dry matte	2	kJ/kg/C
Sensible heat		
Water	62.38912	kW
Dry matter	1.3173333	kW
Heating value		
Organic matter	22	MJ/kg om
	98.511111	kW

Salt & Ash		
Fraction of feed	7%	
Ash fraction in stream	50%	
Organics in ash	200	g/kg ash
Total mass	70	kg/h
Water	52.08	kg/h
Ash	23.4	kg/h
Organic	4.68	kg/h
Temperature	450	Deg C
Total mass	0.019444	kg/s
Water	0.014467	kg/s
Ash	0.0065	kg/s
Organic	0.0013	kg/s
Specific enthalpy wate	2875	kJ/kg
Heat capacity dry matt	2	kJ/kg/C
Organic matter	22	MJ/kg om
Sensible heat		
Water	41.59167	kW
Dry matter	4.48	kW
Total	46.07167	kW
Heating value		
Organic matter	28.6	kW

Gas		
Total mass	62.4	kg/h
(equal to from organic mass)		
Total mass	0.017333	kg/s
Temperature	60	deg C
Specific enthalpy wate	2875	kJ/kg
Heat capacity dry matt	2	kJ/kg/C
Organic matter	22	MJ/kg om
Sensible heat		
Dry matter	2.08	kW
Heating value		
Gas	381.3333	kW

Total out		
Total mass	989.84	kg/h
Water	870	kg/h
Ash	46.8	kg/h
Organic	83.2	kg/h
Total mass	0.274956	kg/s
Water	0.241667	kg/s
Ash	0.013	kg/s
Organic	0.023111	kg/s
Sensible heat		
Water	103.9808	kW
Dry matter	7.877333	kW
Total	111.8581	kW
Heating value	508.4444	kW

F

Plasma Gasification

Sludge Drying

Sludge is available at 25% DS from the dewatering unit. However, plasma gasifier requires a dry solids content of 90%. The water is removed from the sludge using natural gas. It is assumed that the lower heating value of natural gas removes the latent heat of water with a boiler efficiency of 95%.

$$\text{Heat required (MJ)} = \text{mass of water (kg)} * \text{sensible heat (kJ.kg}{-}1.\text{C}{-}1) * \Delta T * \text{latent heat (kJ.kg}{-}1) * \frac{10^{-3}}{0.95} \tag{F.1}$$

$$\text{Natural gas required (Nm}^3) = \frac{\text{Heat required (MJ)}}{\text{calorific value Natural gas (MJ/Nm}^3)} \tag{F.2}$$

However, this approach gave a Natural gas use approximately 70-80% lower than actual values. This is because the heating requirement for sludge heating does not only depend on water evaporation [97]. Hence according to a study from STOWA, natural gas use of 0.36 Nm^3/kg sludge for achieving a DS content of 90% from 25% was assumed [98].

Sludge Input

Finally, the sludge input to the plasma gasification reactor will have the composition as in table F.1.

Table F.1: Ultimate *(left)* Proximate analysis *(rightt)* of sewage sludge feed into plasma gasification reactor.

Element	Kmoles	kg	wt % (daf)^
C	34.77	417.20	55.44
H	61.97	61.97	8.23
O	13.73	219.65	29.19
N	3.24	45.41	6.03
S	0.19	6.02	0.80
Total	113.96	752.47	100.00

Content	kg	wt%
Moisture	72.16	10.00
Dry matter	1000.00	90.00
Ash (*db)	247.53	24.75
Organic (*db)	752.47	75.25

**db - dry basis, ^daf - dry ash free basis*

Gas Produced

Data about gas and energy produced is obtained from two sources. First is a scientific experimental investigation of sewage sludge plasma gasification [14] while the second one is a commercial plasma gasification plant operating in Japan [21]. One ton of sludge using steam as a gasification agent, will produce 1080 Nm^3 of gas with the composition in table F.2. We

know that 1 Nm^3 of gas will contain 22.4 kmol of gas. Thus volume and no. of moles of each component is calculated.

For the next step, calorific value of the gas is calculated using the equation F.3. Where x is the volume % or mol % while k is the calorific value in MJ/kmol. The results can be seen in table F.2.

$$\text{Calorific value of gas (MJ/kmol)} = \sum_{i=1}^{i} x_i.k_i \tag{F.3}$$

Table F.2: Properties of syngas generated from plasma gasification of sewage sludge.

Gas Comp.	Vol %	Nm^3	kmol	kg	wt %	MJ/kmol	MW
H_2	46	496.8	22.18	44.36	5.5	239.92	2
CO	48	518.4	23.14	648.00	80.8	283.00	28
CO_2	3.4	36.72	1.64	72.13	9.0	0.00	44
N2	1.7	18.36	0.82	22.95	2.9	0.00	28
NH_3	0.0	0	0.00	0.00	0.000	0.00	17
H_2S	0.9	9.72	0.43	14.75	1.8	517.44	34
Total	100	1080	48.21	802.19	100	250.86	16.638

Energy content of the sludge is calculated according to equation obtained from sludge digestion manual [65]. where H denotes the energy content. H_{water} is the latent heat of water which equals 2.258 MJ/kg. Whereas OC is organic weight % of sludge on dry basis and DS is weight % of sludge. The values of energy contents entered below are also obtained from the manual [65].

$$\begin{aligned} H_{sludge} &= (OC * H_{organic}) * DS - H_{water} * (1 - DS) \\ &= (75.25\% * 21.318) * 90\% - 2.258 * (1 - 90\%) \\ &= 14.21\, MJ/kg \end{aligned} \tag{F.4}$$

Thus for a capacity of 1 ton sewage sludge DS per hour, chemical energy in feed and syngas is calculated using equation F.5.

$$\begin{aligned} \text{Feed power in (MW)} &= \text{Mass flow rate of sludge (kg/s)} * \text{Energy content of sludge (MJ/kg)} \\ &= \frac{1072.16}{3600} * 14.21 \\ &= 4.23\, MW \end{aligned} \tag{F.5}$$

Syngas power output is calculated to be 3.36MW using a similar methodology with the energy content of the gas calculated from the table F.2.

Energy Production

Using the same energy generation efficiencies for gas engine and SOFC as used in supercritical water gasification. Results can be seen in table F.3. Heat is not inventoried here since it is utilized to produce steam. In some plasma gasification models, there is a negligible amount of heat residue which is not considered. Steam flow rate to the reactor is required to be 1.5 times the mass flow rate of sludge. Hence, if heat is insufficient, natural gas is used. Plasma torch power is assumed to be 600 kW which is the same as sludge gasification in Japan for a similar capacity of 1 ton DS/hour [21]. Results of other scenarios can be found in the inventory analysis.

Material use

Syngas cleaning system followed the same steps as incineration gas cleaning system. Hence material and energy use data except for coke and limestone is the same and scaled accord-

Table F.3: Energy generation from plasma gasification of raw sewage sludge.

Parameter	Value
Syngas power	3.36 MW
Electrical efficiency	37%
Heat efficiency	29%
Electricity produced	1.243 MW
Plasma power	0.6 MW
Plant operations	0.15 MW
Wastewater treatment	7 kWh/ton ds
Net electricity	486.1 kWh/ton ds

ingly. Limestone and coke data was available from the same plant report used above [21]. The results can be found in table F.4.

Table F.4: Material use for plasma gasification process, fluegas cleaning and wastewater treatment.

Materials used	kg per kg sludge DS processed	Total amount per ton DS fed (kg)
Limestone	0.077	57.27
NaOH 50%	0.0322	22.045
HCl 30%	0.0102	6.96
Charcoal	0.0114	7.793
Metullargical Coke	0.0433	4.07

Fluegas Composition

Flue gas composition was also obtained from the plant report [21]. With the exception of CO_2 and H_2O which was calculated from the sludge composition. The results can be found in table F.5.

Waste disposal

Waste disposal for plasma gasification is slightly different since due to extraordinarily high temperatures involved, ash becomes vitrified and hence the heavy metals leaching is considerably reduced. Enabling the use of ash for manufacturing of building materials. The details of leaching is obtained from VitroArc [38] and calculation steps were similar to the ones used before with the leachability (mg/kg) being the only change. Results can be found in table F.6. A small amount of fly ash is also disposed in hazardous waste landfill in Germany.

Table F.5: Composition of fluegas exiting the plasma gasification plant

Contaminants	mg/Nm^3	kg
CO_2	-	1090.38
H_2O	-	399.21
Dust	3	0.003240
HCl	30	0.032400
SO_2	1.5	0.001620
NO_2	72	0.077760
CO	29	0.031320
PCDD/PCDF	0.00064	6.912E-07

Table F.6: Leaching from plasma ash when recycled in building materials

Element	Leachability (mg/kg)	Amount in sludge (per ton ds)	Amount leached (mg)
As	0.009	0	0
Ba	0.3	0	0
Cd	0.003	0	0
Co	0.014	0	0
Cr	0.02	0.063	0.001255
Cu	0.5	1.67	0.835
Hg	0.0001	0.000225	2.25E-08
Mo	0.07	0.0058	0.000403
Ni	0.01	0.0095	0.000095
Pb	0.04	0.068	0.00271
Sb	0.06	0	0
Se	0.0003	0	0
Sn	0.01	0	0
V	0.008	0	0
Zn	3	0.8	2.4
Br	0	0	0
Cl	1.62	2.2175	3.59235

G

Transport

Data about sludge processed by each municipality was collected from a study from STOWA [98]. This data was proportionally converted into kg of sludge to be transported from WWTP to thermochemical treatment plant. It should be noted that amount of sludge for SCWG (7143 kg) is different for plasma and incineration systems because SCWG requires sludge at 13% DS whereas the latter systems require it at 25% DS (4000 kg). The results can be found in table G.1. Results for digested sewage sludge were also calculated using the same principle.

Table G.1: Amount of sludge to be processed by each water board

Water Treatment Board	Sludge DS (kton/year)	Percent (%)	Incineration/Plasma (kg)	SCWG (kg)
Delfland	22.5	13.00	519.93	928.45
Schieland	6	3.47	138.65	247.59
Rijnland ZH deel	13.3	7.68	307.34	548.82
Rijnland NH deel	8.2	4.74	189.49	338.37
Hollandsche Delta	24.5	14.15	566.15	1010.98
Zuiderzeeland	7.5	4.33	173.31	309.48
Brabantse Delta	11.5	6.64	265.74	474.54
Nieuwveer	4.6	2.66	106.30	189.82
De Dommel	20.8	12.02	480.65	858.30
Aa en Maas	20.6	11.90	476.03	850.05
Alm en Biesbosch	1	0.58	23.11	41.26
Zeeuwse Eilanden	6.2	3.58	143.27	255.84
Zeeuws Vlaanderen	2.6	1.50	60.08	107.29
Vallei en Eem	5.4	3.12	124.78	222.83
Stichtse Rijnlanden	8.7	5.03	201.04	359.00
Regge en Dinkel	9.7	5.60	224.15	400.26
Total	173	100	4000	7142.86

For the reference scenario, data regarding sludge transport was available. Two mono sludge incineration plants in Moerdrijk and Dordrecht process almost 50% of the total sludge produced in the Netherlands [98]. One ton functional unit was divided proportionally according to the amount of sludge supplied from wastewater treatment plants to incineration plants in Moerdijk and Dordrecht. Data about sludge supply was available from waterboards to these locations [4]. Thus distances from wastewater plants operating under the corresponding water boards to Moerdijk or Dordrecht was found. For SCWG and plasma gasification, a plant was assumed to be built at location near the office of water boards. Since water boards operate at a distance close to the corresponding waste water plants they manage. These distances were found using google maps and can be seen in tables G.2, G.3, G.4 and G.5.

Using the presented data, results are calculated. Methodology is already explained in 3.10. The final results can be seen in table G.6.

Table G.2: Water boards (in bold) and wastewater treatment plants (WWTP) under them with their corresponding distances to Incineration plants (column on the leftt) and distance between the water board and WWTP (column on the right).

Zuiderzeeland	**Distance to Incineration, Dordrecht (km)**	**Distance from WWTP to Water Board (km)**
Almere	94.5	28.7
Dronten	125	20
Lelystad	117	2.5
Tollebeek	143	32
Zeewolde	103	28.3
Delfland		
De groote lucht	41.1	18.3
Nieuwe waterweg	54.9	18.9
Harnaschpolder	46.9	5.9
Houtrust	59	17.2
Schieland		
ammerstol	43.9	24.7
Bergembacht	41.3	22.1
Berkenwoude	37.4	18.3
De groote zaag	28.8	9.6
Groenedijk	28	8.9
Haastrecht	48.1	29
kortenoord	35.5	11
kralingseveer	23.5	4.3
stolwijk	48.4	29.3
Rijnland		
Aalsmeer	92.6	92.6
Aardam West	62	62
Alphen Kerk en Zanen	56.3	56.3
Alphen Noord	56.3	56.3
Bodegraven	48.5	48.5
Gouda	45.2	45.2
Haarlem Schalkwijk	104	104
Haarlem Waarderpolder	104	104
Hazerswoude Dorp	55.7	55.7
Heemstede	99.4	99.4
Hoogmade	72.7	72.7
Katwijk	74.3	74.3
Langeraar	65.3	65.3
Leiden Noord	75.8	75.8
Leiden Zuid-West	67.4	67.4
Leimuiden	69.9	69.9
Lisse	89.5	89.5
Nieuwe Wetering	48.8	48.8
Nieuwveen	66.2	66.2
Noordwijk	78.3	78.3
Noordwijkerhout	83	83
Rijnsaterwoude	66.9	66.9
Rijsenhout	89.1	89.1
Stompwijk	56.1	56.1
Velsen	112	112
Waddinxveen Randenburg	41.2	41.2
Woubrugge	65.4	65.4
Zwaanshoek	93.8	93.8
Zwanenburg	103	103

Table G.3: Water boards (in bold) and wastewater treatment plants (WWTP) under them with their corresponding distances to Incineration plants (column on the leftt) and distance between the water board and WWTP (column on the right).

Hollandse Delta	**Distance to Incineration, Dordrecht (km)**	**Distance from WWTP to water Board (km)**
Barendrecht	21.5	5
Den Bommel	50.4	34.3
Dokhaven	29.6	13.4
Dordrecht	0	19
Goedereede	65.8	49.5
Goudswaard	45.8	29.5
Heenvliet	41.2	24.9
Hellevoetsluis	49.8	38.8
Hoogvliet	33.3	15.7
Middelharnis	62	45.7
Numansdorp	39.1	22.8
Ooltgensplaat	52.3	36
Oostvoorne	58.3	42
Oud-Beijerland	31.4	15.1
Oude-Tonge	54.4	38
Piershil	42.7	26.4
Ridderkerk	15.3	5.3
Rozenburg	42.7	26.4
Spijkenisse	37.3	21
Strijen	19.8	21.8
Zuidland	45.6	32.4
Zwijndrecht	7.5	11.9
Regge en Dinkel	**Distance to Incineration, Moerdijk (km)**	**Distance from WWTP to Water Board (km)**
Almelo-De Sumpel	199	1.3
Almelo-Vissedijk	200	4.2
Den Ham	196	20.8
Denekamp	220	34.2
Enschede	213	28.1
Glanerbrug	219	33.9
Goor	188	15.7
Haaksbergen	204	26.3
Hengelo	202	17.3
Hengevelde	194	21
Losser	217	31.4
Nijverdal	191	16.5
Oldenzaal	211	25.6
Ootmarsum	222	23.7
Rijssen	188	12.6
Tubbergen	211	16.4
Vriezenveen	208	10.3
Vroomshoop	206	16.8
Westerhaar	216	16.8
Vallei en Eem		
Amersfoort	97.6	54.3
Bennekom	112	54.7
Ede	112	36.2
Nijkerk	108	43.5
Renkum	107	48.1
Soest	95	59.9
Veenendaal	104	57.8
Woudenberg	95.3	55

Table G.4: Water boards (in bold) and wastewater treatment plants (WWTP) under them with their corresponding distances to Incineration plants (column on the leftt) and distance between the water board and WWTP (column on the right).

Stichtse Rijnlanden	**Distance to Incineration, Moerdijk (km)**	**Distance from WWTP to Water Board (km)**
Breukelen	90	31.2
Bunnik	78.9	7.9
De Bilt	80.2	15.9
De Meern	75.5	17
Driebergen	82.4	11.9
Houten	68	4.6
Leidsche Rijn	80.2	21.2
Lopik	62.4	22.2
Maarssenbroek	83.8	23.5
Montfoort	73.5	23.4
Nieuwegein	64.7	10.4
Oudewater	67.2	36.5
Rhenen	88.8	31.2
Utrecht	79.9	15.7
Wijk bij Duurstede	91.6	16.2
Woerden	66.5	30.1
Zeist	83.8	13.6
Brabantse Delta		
Baarle-Nassau	43.5	19
Bath	54.3	55.4
Chaam	34.6	14.2
Dinteloord	27.9	39.6
Dongemond	23	19.1
Halsteren	42.2	46.8
Kaatsheuvel	42.6	36.8
Lage Zwaluwe	7.2	21.7
Nieuwveer	13.5	11.4
Nieuw-Vossemeer	41.5	51.1
Ossendrecht	49.8	47.9
Putte	54.4	50.7
Riel	49.5	53
Rijen	37.6	14.1
Waalwijk	37.5	37.1
Waspik	30.1	24.2
Willemstad	27.1	41
De Dommel		
Biest-Houtakker	59	28.8
Boxtel	63.7	2.6
Eindhoven	82.1	25.9
Haaren	64	9.5
Hapert	77.3	30.9
Sint-Oedenrode	89.9	13.4
Soerendonk	104	45.1
Tilburg	54.6	20.3
Aa en Maas		
's-Hertogenbosch	57.6	7.3
Aarle-Rixtel	95.3	37.3
Asten	108	53.3
Dinther	72.5	14.6
Land van Cuijk	114	47.3
Oijen	77.2	25.7
Vinkel	69.4	13.7

Table G.5: Water boards (in bold) and wastewater treatment plants (WWTP) under them with their corresponding distances to Incineration plants (column on the leftt) and distance between the water board and WWTP (column on the right).

Schelderstromen	**Distance to Incineration, Moerdijk (km)**	**Distance from WWTP to Water Board (km)**
Breskens	132	58.9
Camperland Polder	81	21.8
The Verseput	66.4	42.2
Groede	132	59.2
Hulst	92.5	54.7
Kloosterzande	100	52.7
Mastgat	53.2	52.9
Oostburg	130	57
Retranchement	141	68.5
Sintmaartensdijk	56.1	70.5
Sint Philipsland	43.8	67.4
Terneuzen	104	31.8
Tholen	45.2	63.7
Waarde	62.5	38.5
Walcheren	98.2	8.8
Westerschouwen	74.8	32.5
Willem Annapolder	70.9	29.2

Table G.6: Final transport results for all the systems

	Sludge Transport (t-km)	**Ash Transport (t-km)**
Incineration	383.76	121.66
SCWG	350.29	89.04
SCWG digested	239.95	89.04
Plasma	196.16	17.09
Plasma digested	134.37	14.51

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Impact Assessment Graphs

Contribution Analysis of fresh water aquatic ecotoxicity

Barium, Copper, Nickel and Vanadium are the major emission sources contributing to this impact category. This impact category is measured with the units of kg 1,4-DB (dichlorobenzene) equivalents. As we can see from figure H.1, the unit processes of dewatering, wastewater treatment, transportation, electricity production and hazardous waste landfill are the main contributors. Dewatering impact is mainly due to production of polyacrylamide which is in turn due to Barium and Nickel emissions due to natural gas use. Due to brake and tyre abrasion, Barium and Copper is released in the environment. Hence transportation process makes an appearance in the graph. Wastewater treatment process releases copper in the water bodies causing supercritical water gasification systems attain a reasonable impact value. Ultimately, hazardous waste landfill also cause fresh water toxicity due to emissions of Nickel, Copper and Barium during the steel manufacturing process.

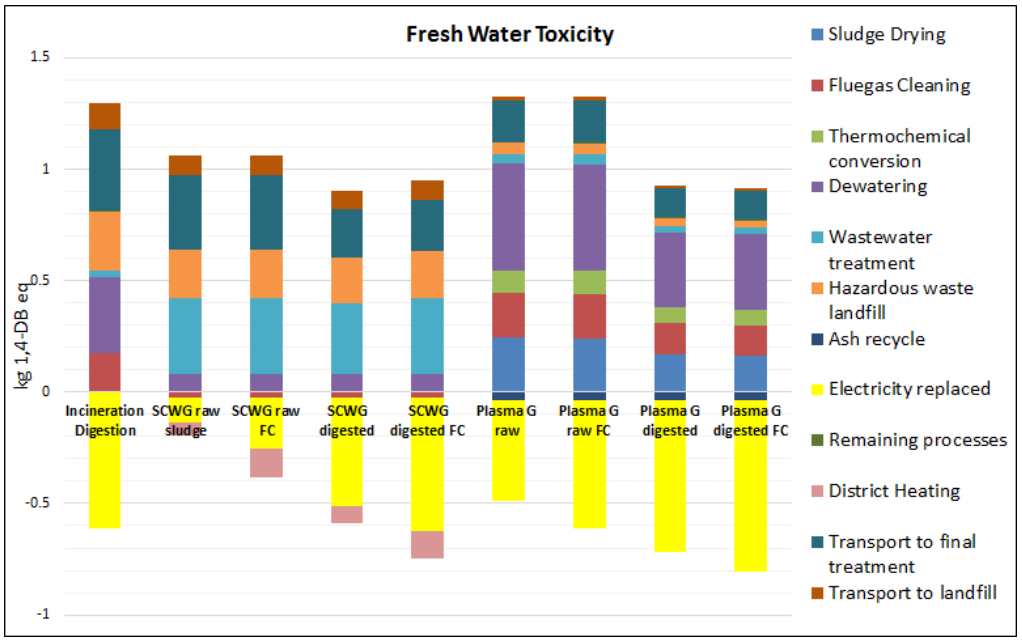

Figure H.1: Contribution analysis for fresh water aquatic ecotoxicity

Contribution Analysis of terrestrial ecotoxicity

Terrestrial ecotoxicity is caused almost entirely by the release of chromium to soil. Just like other ecotoxicities, this impact category is also measured with the units of kg 1,4-DB (dichlorobenzene) equivalents. Chromium emissions are caused by Polyacrylamide and NaOH production processes. For Polyacrylamide, these emissions were due to use of Ammonia which was in turn produced using steam reforming. Due to NaOH being an energy intensive manufacturing process, large amount of electricity utilized may be a possible reason for high Chromium emissions.

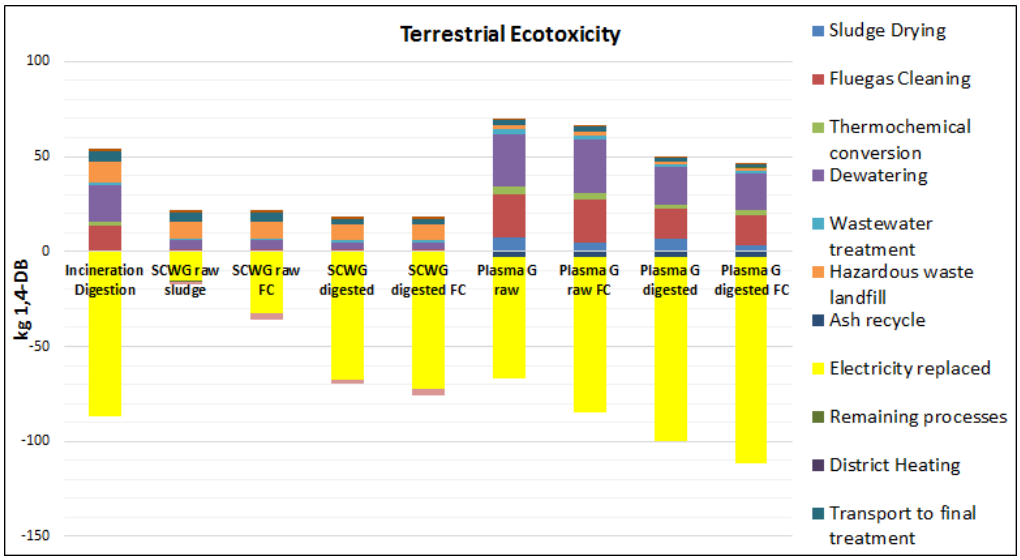

Figure H.2: Contribution analysis for terrestrial ecotoxicity

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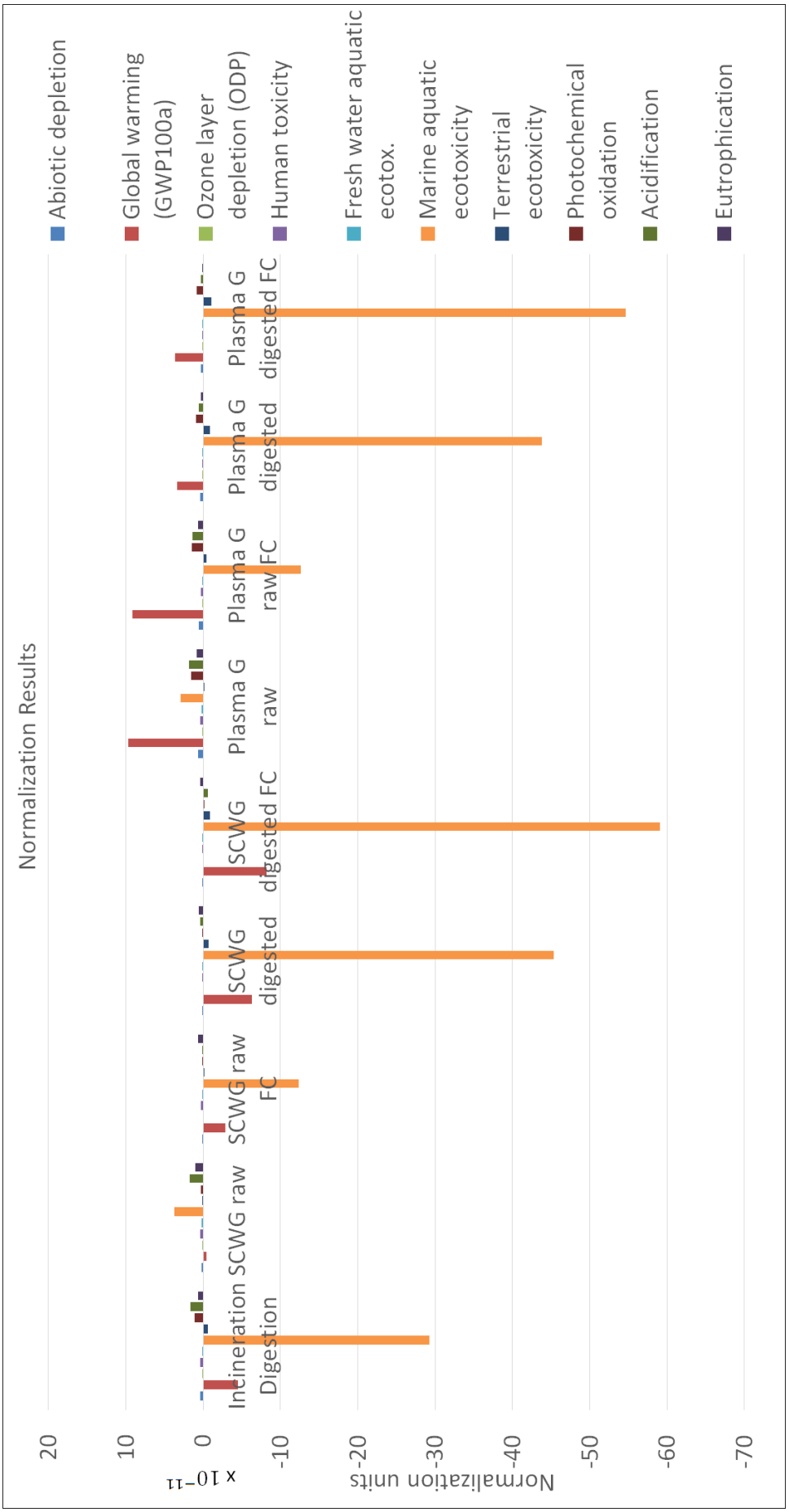

Figure H.3: Normalization results for all the considered systems