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MASTER'S THESIS – MASTER ENERGY SCIENCE

REDUCING GREENHOUSE GAS EMISSIONS FROM MUNICIPAL SOLID WASTE INCINERATION BY CARBON CAPTURE AND ENHANCED RECYCLING

A TECHNO-ECONOMIC ANALYSIS

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| MASTER'S THESIS ENERGY SCIENCE

Preface

This thesis, conducted in fulfilment of my master's degree in Energy Science at Utrecht University, is based on research contributing to the MIDDEN project. An initiative from the Dutch Environmental Assessment Agency (PBL) and TNO, MIDDEN aims to build an open-source knowledgebase for decarbonisation options for industry, that is recognised by all stakeholders. At the time of writing, the MIDDEN project has published 40 reports on a variety of industries, including the chemical industry, steel manufacturing and many others. This research contributes to the MIDDEN project in the form of a report, and is accompanied by a database that quantifies the decarbonisation options in terms of costs and emissions.

Although the majority of the research was done during times of COVID-restrictions, I received great support and guidance from my fellow interns, project members and most of all, my research supervisor at PBL. I would like to thank dr. Robert Koelemeijer the effort he has put in overseeing the successful completion of this project, and for the assistance and knowledge needed to do so. My university supervisor prof. dr. Gert Jan Kramer has also been a positive influence on this research. His insights and guidance have helped me to make sense of the large amount of information I was presented with, and distil the most relevant parts into this thesis.

In conducting the research, experts from governmental organisations, universities and industry were consulted through emails, calls and interviews. For their contribution to the project, I would like to thank Liane Schoonus (Vereniging Afvalbedrijven), Hans Wassenaar (AVR), Carina Oliveira (TNO), dr. Li Shen (Utrecht University), Robin Hamerlinck (Ministry of Infrastructure and Water Management), Bas van Huet (Rijkswaterstaat) and Mike Muller (PBL).

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Abstract

The Dutch waste incineration industry is one of the most efficient in the world, extracting 36% of the energy in the waste as useful heat or electricity on average, while reducing the volume of the solid municipal waste by around 90% (Rijkswaterstaat, 2020). Consequentially, the industry is also one of the Netherlands' largest emitters of greenhouse gases. In 2018, the industry emitted 7.5 Mt of CO_2-equivalent emissions, of which 3.0 Mt was non-biogenic in origin. In order to achieve the climate ambitions of the Dutch government of 60% GHG emission reduction and net-zero emissions in 2050, the waste incineration industry is under pressure to reduce its emissions (Rijksoverheid, 2018).

This research, conducted in the context of the MIDDEN project, attempts to compare different technologies, configurations and pathways for the reduction of GHG emissions related to the currently incinerated waste stream in the Netherlands. The two main categories that were investigated were carbon capture technologies and enhanced recycling technologies. Emission reduction options are compared on the basis of their technology readiness level, potential feedstock, total CO_2 emission reduction potential and estimated CO_2 avoidance cost. The net emission reduction potential is estimated by an ex-ante comparative life cycle assessment. Validation of the results is done via a Monte Carlo analysis of uncertainty.

Carbon capture for waste incinerators is already an established technology, which is currently applied at two major waste incinerators in the Netherlands. This report identified three configurations for carbon capture technologies and assessed their potential for emission reduction and associated cost. CO_2 avoidance cost ranged from €159-224 per tonne CO_2, depending on the configuration chosen. Furthermore, policy issues that hinder implementation of carbon capture are addressed in the discussion.

Enhanced recycling aims at decreasing the total volume of incinerated waste, while reducing net GHG emissions. Enhanced recycling technologies identified by this research with the potential of reducing GHG emissions are waste plastic pyrolysis, EPS solvolysis, PET depolymerisation and gasification of municipal solid waste. Findings include limited total GHG emission reduction potential for enhanced recycling (154 kt CO_2 annually) compared to carbon capture technologies (5,130 kt annually). A configuration combining recycling (gasification) and CCS was found to result in 1,043 kt CO_2 reduction annually. In part, this is due to the limited availability of the feedstock for these processes. Another factor is the emissions from the alternative processes themselves. CO_2 avoidance cost results for enhanced recycling technologies were inconclusive, as uncertainty was found to be too large.

Significant short-term CO_2 emission reduction is most likely achieved through large-scale implementation of carbon capture technologies in combination with waste incineration and/or gasification. Carbon capture and storage proves to be the most potent short-term solution to waste incineration GHG emissions. In the long-term, GHG emission reduction must be achieved by full separation of the plastic waste from the residual fraction and subsequent recycling of this plastic material. The methods and practices to achieve this, however, are not available at this time.

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1   Introduction

The Sixth Assessment Report, published by the IPCC in August of 2021, reaffirmed the notion that anthropogenic climate change is factual, measurable and poses a serious threat to billions of people around the world (IPCC, 2021). The report concluded with high certainty that action to curb greenhouse gas emissions (GHG) needs to be taken at a swift pace. As a means of accomplishing this, global, European and national emission reduction targets have been set (United-Nations, 2015). The Dutch government has set the target of reducing its carbon dioxide emissions by 49% by 2030 compared to 1990 and 95% by 2050 in the Dutch Climate Accord (Rijksoverheid, 2018). Ambitions in the coalition agreement published in late 2021 aim for 60% reduction by 2030 (Coalition Agreement, 2021). In the Urgenda court ruling, the Dutch government was legally bound to reduce CO_2 emissions by 25% in 2020 relative to 1990 emissions. More recently, the responsibility of industry has come into the judicial arena when the Dutch court ruled in favour of the activist group Milieudefensie that Shell has an obligation to reduce its scope 1 emissions by 45% before 2030, when compared to 2019 emissions. This may indicate that even sectors that are most difficult to reduce their emissions must take their responsibility and abide by the emission reduction targets.

The challenges accompanied by the emissions reduction targets are especially great in the Netherlands. Due to its geographical location, exceptional infrastructure and large supply of natural gas, the Netherlands have historically been a favourable area for industrial companies to settle. As a result, 31% of Dutch GHG emissions originated from the industrial sector in 2018. Roughly 17% of these industrial emissions are caused by the waste management industry, totalling just over 5% of total Dutch GHG emissions (CBS, 2021). The largest emitters in this industry in terms of annual CO_2-equivalent emissions are the waste incineration plants (WIPs).

In order to aid the decarbonisation of industry, the government set up the Manufacturing Industry Decarbonisation Data Exchange Network, or the MIDDEN initiative, in 2018 (PBL, 2018). MIDDEN is a joint venture between the Dutch Environmental Assessment Agency (PBL) and research institute TNO. The project aims aggregate data and information on industrial processes, as well as possible means of decarbonisation for these processes. The focus is on sectors with large, concentrated emissions of GHG. This research contributes towards the MIDDEN program through an investigation into possible methods to reduce GHG emissions for Dutch municipal solid waste incineration.

1.1   Waste incineration in the Netherlands

The 13 WIPs in the Netherlands processed 7.5 megatonnes (Mt) of waste in 2018 through incineration, generating 7.8 Mt of CO_2-eq emissions (Rijkswaterstaat, 2020). 36.6% of these emissions were based on fossil carbon, the remaining 63.4% were of biogenic origin (RIVM, 2021). Following IPCC guidelines, biogenic carbon emissions that cause a net change in the biogenic carbon stock have already been accounted for in other sectors (particularly in the sector land-use, land-use change and forestry (LULUCF), within and outside the Netherlands) (Eggleston et al., 2006). For this reason, counting these biogenic emissions here would lead to double counting. Therefore, this report prefers the use of fossil carbon emissions wherever possible. Long term storage of biogenic CO_2 thereby causes net negative emissions. Hereafter, this methodology is referred to as "IPCC

accounting" or "IPCC guidelines". The fossil CO_2 emissions of the waste incineration industry were 3.0 Mt in 2018, as reported in the National Inventory Report to the UNFCCC (RIVM, 2021).

In the process of incinerating emissions, 12 out of the 13 Dutch waste incinerators produce useful heat and electricity. In total, 12.1 PJ of electricity and 14.9 PJ of useful heat were exported. At an average LHV of the waste of 10 GJ/tonne, the average electric and useful thermal efficiencies of WIPs in the Netherlands is 16% and 20% respectively (Rijkswaterstaat, 2020). There currently does not seem to be a sustainable alternative for the disposal of waste. Landfilling causes the release on large amounts of methane, which exacerbates climate change (Kumar et al., 2004; Scharff, 2014).

Dutch waste incinerators are among the world's first plants to implement large scale carbon capture facilities to mitigate the emissions from waste incineration (Wienchol et al., 2020). These carbon capture facilities allow for either storage or utilisation of the CO_2 by sequestering the CO_2 from the flue gas after incineration is the waste. However, as the Netherlands proposes to transition to a circular economy by 2050, governmental institutions call for the reduction of total incinerated waste by increasing recycling efforts (PBL, 2021). Current literature on increasing recycling for the incinerated waste stream is highly fragmented and mainly focuses on highly specific streams (Broeren et al., 2019; Brouwer et al., 2018). In an absence of literature on the subject, this report aims to provide a comparison of the two main proposed methods of decreasing GHG emissions from waste incineration: carbon capture or enhanced recycling.

1.2 Research questions

The problem definition and research gap presented above allow for the definition of the following research question:

1. What are the relevant material and energy flows in the current waste incineration process and how do they affect possible GHG emission reduction efforts?
2. What is the potential for carbon capture technologies in reducing GHG emissions for municipal solid waste incineration, and what are the associated costs?
3. Which technologies exist that can enhance recycling efforts for the currently incinerated waste stream, while also reducing net GHG emissions?
4. What is the potential for enhanced recycling techniques to reduce GHG emissions related to municipal solid waste incineration, and what are the associated costs?

The first research question intends to provide the necessary background knowledge of the waste incineration industry and identify the origin of the GHG emissions in terms of the composition of the waste. Additionally, any avoided emissions arising from the exports of useful heat and electricity were accounted for by means of energy balances. The second research question is focused on carbon capture technologies, and constitutes a techno-economic analysis of the processes and costs involved. The third and fourth research questions attempted to provide a non-exhaustive but comprehensive overview of the enhanced recycling technologies available, and to what extent they can contribute to the reduction of GHG emissions.

2   Methods

This section details the research methods that were used to provide answers to the research questions listed in Section 1.2. RQ1 was answered with a literature analysis of the current waste incineration industry. The answer to RQ2 was gathered from desk research and expert interviews, as were RQ3 and RQ4. For RQ4, a simplified LCA methodology was applied to find the net GHG emissions relative to the reference. Finally, this section discusses the methods applied to assess the associated costs of technology implementation. Figure 1 gives an overview of the steps taken in the research, as well as the corresponding methods and deliverables.
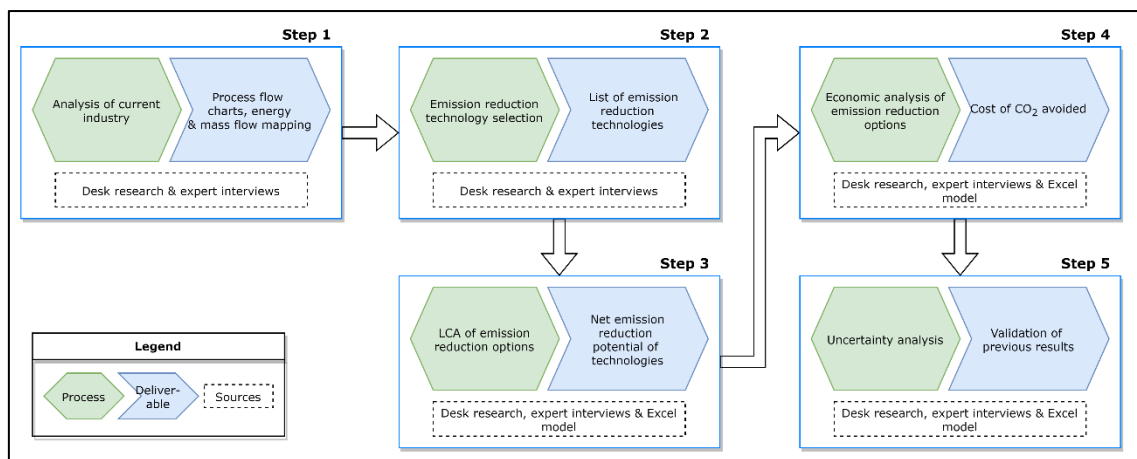
Figure 1. Research methods and deliverable for each research step conducted.

For all of the methods outlined in this chapter, desk research and expert interviews were the main sources of information. Desk research made use of publicly available information from scientific literature, government publications and databases. Additionally, reports from research institutes such as TNO and CE Delft were used, as well as other organisations such as the knowledge centre for sustainable packaging (KIDV). Search engines used were Worldcat, GoogleScholar and Google. On various occasions, (publicly available) literature was provided by interviewees or experts who were contacted over the course of the research.

Expert interviews were a vital resource in both gathering and verifying information. Interviews were arranged jointly by the author and the supervisors, and followed a semi-structured approach. Most interviews were held via online meetings, although input was also gathered via emails, phone-calls and informal conversations. The interviews mostly supplied the research with guidance, an no confidential information was disclosed. As a result, no transcripts were made and this research will not reference these interviews. Experts that gave input for this research were:

- Dr. Li Shen, Assistant Professor at Utrecht University on Circular Economics
- Bas van Huet, Waste Management Advisor at the Department of Waterways and Public Works
- Robin Hamerlinck, Team leader Market Based Instruments at the Ministry of Infrastructure and Water Management
- Liane Schoonus, Policy Officer at the Dutch Association of Waste Companies
- Hans Wassenaar, Senior Project Manager at Dutch waste incinerator AVR

- Carina Oliveira, Chemical Engineering Specialist at TNO
- Olaf Van Hunnik, Waste and Materials advisor at the Department of Waterways and Public Works
- Mike Muller, Researcher on Subsidies and Circular Economics at PBL

2.1 Analysis of current waste incineration process

The primary research step conducted was desk research and expert interviews to form a clear picture of the current waste incineration process, in order to answer RQ1. The main deliverables of this step are the energy- and mass balances. These balances were used in further analyses to show the impact of GHG emission reduction technologies on the current waste incineration process. Aggregated and company specific data used in setting up the mass- and energy balances were gathered from company websites and annual reports, scientific literature and, as a valuable resource, the annual report from the Department of Waterways and Public Works on the Dutch waste management (Rijkswaterstaat, 2020).

2.2 Technology selection & data collection

The selection of possible technologies to be included in this research is based on expert interviews, literature research and consultation with university- and PBL supervisors. Review articles on the subject of reducing waste-related GHG emissions and enhancing recycling efforts were included. Prior to the research, a list of prerequisites for inclusion of technologies in the research was established. These are:

1. The technology must have a potential feedstock that is currently incinerated
2. The process must be proven to be technically feasible with a TRL>5
3. Data must be available for a cost estimation

The prerequisites were verified in the order in which they are listed. If technologies were found to fail on any of these criteria, they were excluded from further research, thus limiting the amount of inessential research that was done. The technology readiness level (TRL) is defined by the U.S. Department of Energy (2010), and is used to compare the stage of implementation a technology is in, and how market-ready a technology is. The individual levels and brief descriptions are shown in Table 1.

Once prerequisite information was collected, the data collection phase of the research began. For the subsequent phase of the LCA, a list of necessary parameters were compiled for each technology. The exact parameters depend on the type of technology (alternative or retrofit) system boundary (Cradle-to-Gate, Cradle-to-Grave, etc.) the type of product produced (fuel substitute, chemical feedstock or pure CO_2).

Table 1. Technology readiness levels and their descriptions (U.S. Department of Energy, 2010).

Category	TRL	Description
Demonstration	9	Normal commercial service
	8	Commercial demonstration, full scale deployment in final form
	7	Sub-scale demonstration, fully functional prototype
Development	6	Fully integrated pilot tested in relevant environment
	5	Sub-system validation in relevant environment
	4	System validation in laboratory environment
Research	3	Proof-of-concept, component level
	2	Formulation of the application
	1	Basic principles, observed, initial concept

2.3 Ex-ante comparative life cycle analysis of technologies

An ex-ante life cycle analysis was performed for each of the selected technologies to assess the net GHG emissions they would cause upon implementation. An ex-ante or anticipatory life cycle analysis is an impact assessment of an emerging technology on the environment over its life cycle, before it is fully implemented. Tsoy et al. (2020) note that ex-ante LCA is generally a challenging exercise due to the lack of full-scale demonstrations of the technology. This leads to the use of lab- or pilot scale data, which often does not take efficiency gains into account which only present themselves after full-scale industrial implementation. Tsoy et al. (2020) therefore recommend the use of energy- and mass balances to accurately assess the minimum energy requirements of the system, providing a more accurate assessment of the projected emissions. Whenever this data was available, mass- and energy balances were constructed in this research. When dealing with proprietary technology with few detailed resources, construction of mass- and energy balances was often not possible. In these instances, uncertainty increases, which is discussed in Section 4.4.

General steps in LCA are the goal and scope definition, life cycle inventory, impact assessment and interpretation (Nieuwlaar, 2013). The goal for this research is to assess the potential GHG emission reduction from the implementation of technologies for management of the currently incinerated waste stream. Therefore, the only impact category of interest is the global warming potential (GWP) of the technology. To limit the amount of data needed for the LCA, emissions are compared with the reference system. This excludes all processes from the system boundary that do not differ from the reference system. The exact scope is highly dependent on the nature of the technology, but some general rules apply. First, the scope of the new technology must include the same (or substituted) steps as the reference system. Second, any step that is identical to the reference is omitted from the scope (this includes transport over similar distances, combustion of fuels with identical emission factors or disposal stages of identical products). Lastly, due to the prerequisite for technology selection that the feedstock is currently incinerated, the reference process is always (at least partly) incineration in a Dutch waste incinerator. The functional unit (FU) for each of the systems in the LCA is 1 tonne of waste feed. If the process is fed with part of a waste stream, the residual stream is assumed to be incinerated and the emissions and benefits from this incineration are included in the scope. Due to availability of data, only level 1 and level 2 energy demands are included in the scope,

meaning energy requirements for capital equipment and infrastructure used are not included. Additionally, transport energy requirements are excluded since they are presumed to be similar in both the reference system and the proposed systems.

The benefits from waste incineration are in the form of useful heat and electricity that are exported to industry, the built environment or the electricity grid. To assess the (negative) environmental impact of these benefits, Equation 1 and Equation 2 are used.

Equation 1. Avoided emissions from electricity exports for waste incinerators.

$$E_e = \eta_e * \left(\sum_{i=1}^{n} LHV_i * m_i + LHV_{i+1} * m_{i+1} + \cdots + LHV_n * m_n \right) * EF_{grid}$$

Equation 2. Avoided emissions from useful heat exports for waste incinerators.

$$E_h = \eta_h * \left(\sum_{i=1}^{n} LHV_i * m_i + LHV_{i+1} * m_{i+1} + \cdots + LHV_n * m_n \right) * EF_{heat}$$

Here, E_e and E_h are the (negative) emissions related to the exports of electricity and useful heat respectively. η_e and η_h are the average electrical and useful thermal efficiencies of Dutch waste incineration, which are 16% and 20% respectively. LHV_i is the lower heating value of component i, m_i is the mass of component i. The calculation accounts for the individual component of the waste in order to assess the impact of removing certain components from the waste for alternative processes. EF_{grid} and EF_{heat} are the respective emission factors used for the electricity grid and useful heat.

The emission factor for electricity is based on the Climate and Energy Outlook (KEV) 2021 with the Reference Park Method. This method uses the average emission factor of the marginal electricity production unit in a year. The choice of this method, as opposed to the Integral Method, is based on the report by Harmelink et al. (2012). They argue that this method is preferred when analysing the effect on CO_2 emissions caused by changes in the use of electricity production installations. The unweighted average of this emission factor over the years 2019-2030 is taken, which is 0.40 kg/kWh (PBL, 2021b). For heat, the reference emission factor is a gas boiler with 90% conversion efficiency, making the emission factor 0.23 kg CO2/kWh. These emission factors are also used to assign emissions to electricity and heat demand from processes, if these exist. In the case where a product is substituted, be it a chemical feedstock or an energy carrier such as a fuel, the emissions from production (and combustion in case of fuel) of the reference product were included in the scope.

The impact indicator global warming potential (GWP) is expressed as kg or tonne CO_2-eq emissions. The CO_2 equivalence of GHG emissions is calculated with the 100-year equivalence parameters defined in the 5th assessment report by the IPCC (AR5) (Pachauri et al., 2014).

2.4 CO$_2$ avoidance cost calculation

As mentioned, a large cause of uncertainty in ex-ante LCA is the size of the plant once the implementation phase is reached. This uncertainty in emission carries on as an uncertainty in capital and operational expenses. In order to estimate the cost of differently sized plants relative to a base

case while accounting for economies of scale, the cost-to-capacity method is used, shown in Equation 3 (Reilly, 2015).

Equation 3. Cost-to-capacity scaling method.

$$\frac{C_2}{C_1} = \left(\frac{Q_2}{Q_1}\right)^x$$

Here, C_1 is the known cost of the base scale plant, Q_1 is the known capacity of the base scale plant. C_2 and Q_2 represent the cost and capacity of the differently sized plant and x represents the scaling factor. For any plant capacity Q_2 and scaling factor, the corresponding cost C_2 can be estimated. The scaling factor ($0<x<1$) thus represents the advantages of economies of scale. In this research, a scaling factor of 0.7 is chosen for all technologies, which is consistent with other research on plastic recycling plants (Oliveira & Van Dril, 2021). This method is applied to both the capital expenditure (CAPEX) and the fixed operational expenditure (OPEX) of the plant.

A frequently used metric to assess the economic feasibility of GHG emissions reduction measures is the CO_2 avoidance cost. The CO_2 avoidance cost can be calculated in a number of ways depending on the inputs and outputs of the processes that are assessed. For industry, the CO_2 avoidance cost can be calculated using Equation 4 and Equation 5, adopted from Roussanaly (2019).

Equation 4. CO_2 avoidance cost.

$$C_{CO_2 \text{avoided}} = \frac{\text{AIC} + \text{OPEX} + C_{\text{feed}} - \text{Revenue} + \text{LOI}}{Annual\ CO_2\ \text{avoided}}$$

Equation 5. Annualised investment cost.

$$AIC = CAPEX * \alpha = CAPEX * \frac{r}{1 - (1+r)^{-n}}$$

Here, $C_{CO_2 \text{avoided}}$ is the cost of CO_2 avoided, *AIC* is the annualised investment cost of the decarbonisation technology, which is calculated by multiplying the total CAPEX by the annuity factor α, which is determined by the discount rate r and the plant lifetime n. For all plants, lifetime is assumed to be 15 years and the discount rate is set at 5%. Since the reference investment costs for the existing WIP infrastructure) have already been made, these investment costs are not included. *OPEX* is the annual operating costs of the decarbonisation technology and the revenues are composed the sales price of the products. C_{feed} is the feedstock cost. Feedstock costs are assumed to be zero for MSW. Although WIPs do receive a fee from municipalities, it is assumed that an identical fee would apply to chemical recycling plants. *LOI* is the loss of income relative to the reference scenario. In the case of waste incineration as the reference, LOI would equal the revenue gained from sales of heat and electricity generated by the WIP. Both CAPEX and OPEX depend on plant size and are assumed to scale with a scaling factor of 0.7, as described above.

2.5 Uncertainty analysis

To assess the rigidity of the results obtained from the life cycle GHG emissions analysis and the calculation of the CO_2 avoidance cost, an uncertainty analysis was performed in the form of a Monte Carlo analysis. The Monte Carlo method relies on simulations that generate semi-random values based on probabilistic distributions of input parameters (McMurray et al., 2017). This type of uncertainty analysis is especially useful when uncertainties are large, non-Gaussian distributed and data are correlated.

The methodology for the Monte Carlo analysis followed that of McMurray et al. (2017), and was performed with @RISK 8.0 Course Version software. Steps taken in the uncertainty analysis were: 1) parameter selection, 2) fitting probability density functions (PDFs) to parameter ranges, 3) running and combining Monte Carlo simulations and 4) error analysis with 90% confidence interval and interpretation of results with a sensitivity report. The uncertainty analysis was split in two parts. First, uncertainty in net emission reduction (LCA) was investigated. Hereafter, the results were combined with an Monte Carlo analysis on financial parameters to find the confidence interval for the CO_2 avoidance cost.

Parameter selection was done by reviewing the total set of input parameters for each LCA and CO_2 avoidance cost calculation, and assessing the possible range for the parameters by searching for alternative values in literature. When alternative values could not be found, a ±20% range was taken unless otherwise indicated in Section 4.4. Parameters included in the analysis were: electricity grid emission factor, heat emission factor, emission factors for virgin product production, emissions from processes, material efficiencies, CO_2 capture rates, plant size, feedstock costs, product price, fuel costs such as hydrogen and natural gas, discount rate, electricity price and equipment lifetime. Although many parameters are process specific, some parameter uncertainties are generic and apply to multiple technologies. These uncertainties are discussed below.

The range for the electricity grid emission factor was chosen to be 64-130 kg CO_2/GJ_e. Minimum and maximum values were taken from the Climate- and Energy Outlook by PBL (2020) which are based on the highest and lowest values in 5-year estimated using the Integral Method. Heat emission factor depends on the reference heat source used. The base case is a natural gas boiler with a 90% efficiency. The minimum emission factor was based on the emission factor of a heat pump with a coefficient of performance of 3, while the maximum emission factor was chosen to be 50% above the natural gas boiler emission factor. Range was 21-95 kg CO_2/GJ_{th}. Prices for natural gas and electricity are highly uncertain. The range for these values is based on projections by PBL (2020) and were 16-32 €/m^3 and 32-68 €/MWh for natural gas and electricity respectively. The discount rate was ranged from 2-8% with a preferred value of 5%. Equipment lifetime was varied from 10-20 years with a preferred value of 15 years. For all these parameters, the PDFs type was triangular. This choice is based on IPCC methodology, which prefers this distribution type for ranges where preferred values are known, as well as an upper and lower bound (Eggleston et al., 2006).

For many cost parameters such as feedstock and product costs, no preferred values could be ascertained, only a range. Therefore, the PDFs for these parameters were chosen to be uniform, following Eggleston et al. (2006). Tsoy et al. (2020) note that ex-ante LCA results are often highly uncertain due to an over- or underestimation of the benefits of industrial scale applications of new

technologies. To assess this, the plant scale was varied over a large range. The range is specific to the technology used, but in general the upper bound is the total annually available feedstock in the Netherlands, and the lower bound is a small scale-up plant of 5-10 kt feed annually.

The uncertainty analysis of carbon capture technologies was limited to a PDF for the CO_2 avoidance cost, since an LCA was not performed in a similar manner to the enhanced recycling techniques. The capture rate, defined as the CO_2 captured relative to the amount of CO_2 in the flue gas at a given point, was varied between 80-90%. The base value for the capture rate and other parameters are further elaborated in Section 4.1.

The Monte Carlo simulations were run 10,000 times to allow for convergence of the parameters. The final error analysis consisted of determining the 90% confidence interval for the net emission reduction per tonne feed, as well as the CO_2 avoidance cost in €/tonne CO_2 avoided. Results were interpreted with a tornado-diagram showing the most sensitive parameters.

3   Analysis of current waste incineration process

This section describes the current processes in waste incineration in the Netherlands. Figure 2 offers a guide to the aspects discussed in this section. In order to characterise not only the process but also the mass- and energy flows, this section first discusses the composition and characteristics of Dutch municipal solid waste. Hereafter, the incineration process and subsequent flue gas treatment are discussed, as well as cogeneration and accompanying efficiencies and the energy balance. Finally, an overview of the residual products and the mass balance are presented.
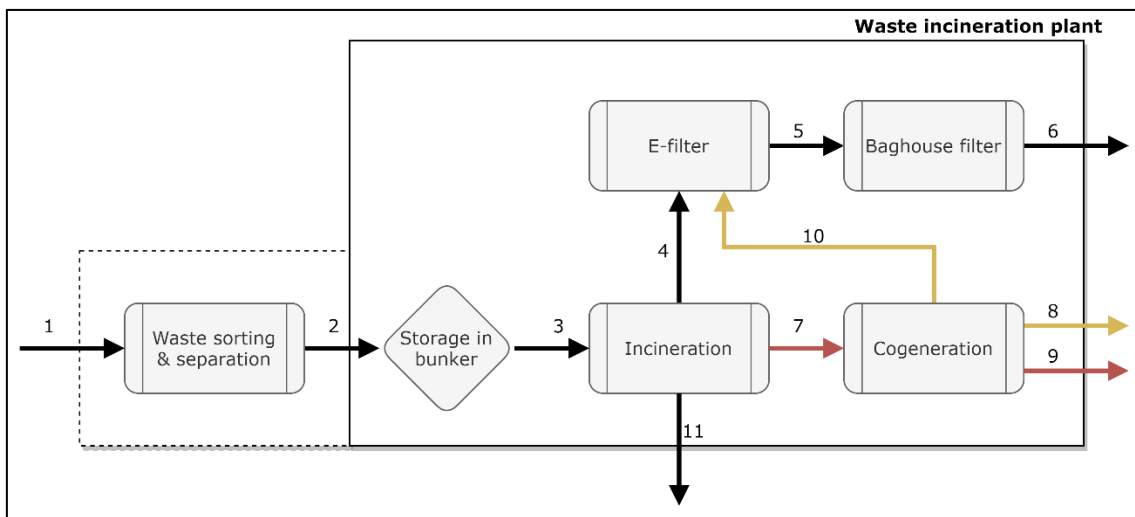
Figure 2. Process flowchart of waste incineration.

Flow number	Flow type	Description
1	Mass	Input waste prior to sorting/separation.
2	Mass	Input waste after sorting/separation.
3	Mass	Input waste after bunker storage.
4	Mass	Unfiltered flue gas.
5	Mass	Filtered flue gas.
6	Mass	Vented flue gas.
7	Heat	Heat flow from incineration of waste.
8	Electricity	Net electricity export after self-consumption.
9	Heat	Useful heat export, may be steam of condensate depending on application.
10	Electricity	Self-consumed electricity for on-site processes.
11	Mass	Bottom ash.

3.1 Municipal solid waste

Flow 2 in Figure 2 depicts the final waste stream that is incinerated. This stream is composed of many different streams and starts off at the initial disposer. This initial disposer is most often a household or business. Bijleveld et al. (2021) aggregated data on the 13 main waste types in the Netherlands and to what extent they are recycled. Table 2 shows abbreviated data on processing of the 13 main waste types in the Netherlands.

Table 2. Types of municipal solid waste and typical processing (Bijleveld et al., 2021).

Waste stream	Typical processing method
Paper/cardboard	85% pre-sorted and recycled up to 7 times. Rest is incinerated with energy recovery.
Plastics	PET bottles 99% pre-sorted. Part of HDPE, LDPE and non-bottle PET is pre-sorted in PMD, rest is post-separated or incinerated with energy recovery.
Drink cartons	Pre-sorted in PMD. Cardboard is recycled, LDPE layer is incinerated with energy recovery.
Metal packaging	Pre-sorted and recycled or sent to incinerator where 85-98% is recovered from bottom ash and recycled. Some metals are lost in the bottom ash.
Glass	Pre-sorted in curb side containers. 86% is recycled, some material ends up at the incinerator and is lost in the bottom ash.
Stone/rock	Pre-sorted at municipal recycling centre, or ends up in residual waste.
Textiles	45% is pre-sorted in curb side containers or collected in bags and 80-90% of this fraction is recycled. 55% ends up in residual waste and is incinerated with energy recovery.
Woody waste	77% recycled as wood chips, 23% incinerated with energy recovery.
Kitchen/yard waste	Pre-sorted partly, also ends up in residual waste. Pre-sorted fraction is 70% digested to biogas with composting of residual fraction, and 30% direct composted. Unsorted fraction is incinerated with energy recovery.
Bulky yard waste	Pre-sorted at municipal recycling centre and composted or incinerated with energy recovery.
Frying fats/oils	Pre-sorted at recycling centre and recycled as fuel oil. Partly lost to sewers or in residual waste. Residual waste fraction is incinerated.
Diapers/sanitary material	New method available for recycling, currently most material is incinerated with energy recovery.
Electric & electronic devices	Pre-sorted at municipal recycling centre, or ends up in residual waste where it is incinerated. Metals are 85-98% recovered from bottom ash.

After recycling efforts described in Table 2 have taken place, a major part of the initially disposed waste stream has been recycled in some way. In 2019, roughly 80% of domestic waste disposed in the Netherlands was recycled (PBL, 2021). However, it must be noted that large amount of recycling in the Netherlands are labelled as low-quality recycling. This means materials are repurposed in a way that most of the value of the material is lost, and subsequent recycling is often not possible. Examples include the mechanical recycling of mixed plastic waste into roadside traffic poles and the use of waste granulate as a filler material in infrastructural works (PBL, 2021). Another note is that

the high recycling rate in the Netherlands mostly stems from reuse of construction and demolition waste, paper and glass (PBL, 2021).

Municipal solid waste collection

The composition of waste is highly variable and depends on the method of collection, the specifics of the area, the time of year (in terms of moisture content), the effectiveness of recycling efforts prior to collection, post-collection separation efforts and more (Corsten et al., 2013; Gradus, 2020; Rijkswaterstaat, 2020). Dutch waste collection is, for a major part, the responsibility of the municipality in which the waste is generated. Waste collection techniques are diverse throughout the Netherlands, but the most dominant systems in place concerning MSW collection are pre-collection separation (kerbside or drop-off) and post-collection separation (Gradus, 2020). Figure 3 shows the development of pre-collection separation and non-separated waste collection from 1995-2020. The fraction of pre-sorted waste increased from 45% in 2000 to 59% in 2020 (CLO, 2021).

In pre-collection separation, the disposer (end-consumer) has the responsibility of sorting the waste, while post-collection separation schemes involve a sorting plant that attempts to extract recyclable materials from the MSW after collection. Both techniques have their advantages and disadvantages. Pre-collection separation of waste is applied in almost all municipalities, but often only for organic waste, glass and paper/board waste. Plastics, cans and beverage cartons (PMD-waste) are collected separately in many municipalities, although some large municipalities like Amsterdam and Utrecht have recently decided to move to post-collection sorting. Pre-collection sorting has the potential for creating so-called "monostreams", which are waste streams that contain one specific material or narrowly defined group of materials. These monostreams are valuable to recyclers as they greatly improve the recyclability of the stream. However, pre-collection sorting has also been found to be highly contaminated with other waste, dirt, food residue and moisture (Gradus, 2020). When the contamination exceeds a certain level, often set at 10%wt, freights are rejected.
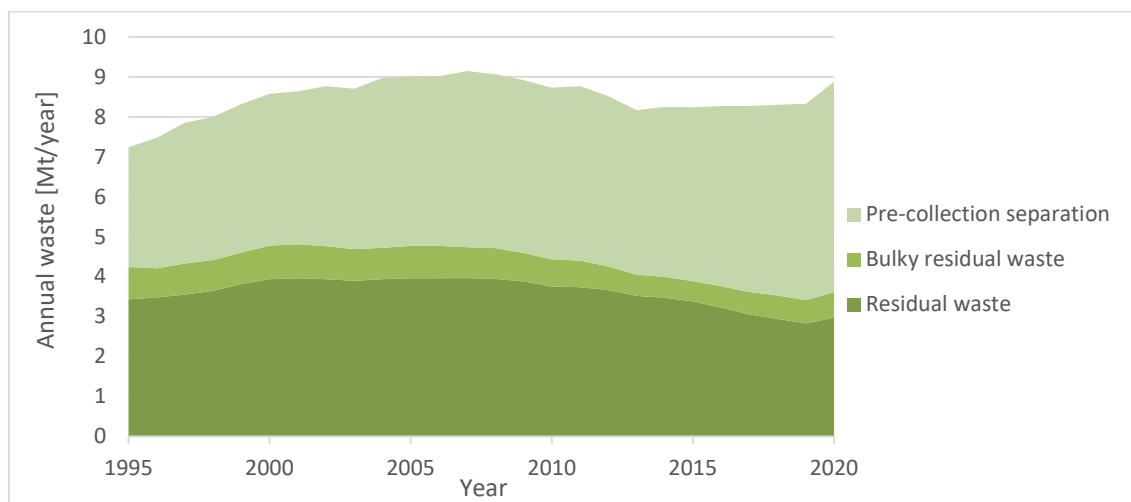
Figure 3. Municipal solid waste collection in the Netherlands, 1995-2020 (CLO, 2021).

Post-collection separation relies on sorting machines to sort out materials. The Netherlands currently has a total processing capacity of 1.7 Mt household waste in separation plants (Kerstens & Blanksma, 2019). The National Test Centre Circular Plastics develops new techniques for, among other things, separation machines, with the goal of maximizing the amount of recycled plastic (NTCP, n.d.). The

estimated efficiency of these machines, expressed as the percentage of successfully separated material, is roughly 50%, and it is not expected to rise significantly (Kerstens & Blanksma, 2019).

Due to the aforementioned variability in collection techniques, one single characterisation of MSW in the Netherlands is not possible. However, many studies have been conducted on the composition of solid municipal waste that ends up in incinerators in the Netherlands and the EU 27 (Corsten et al., 2013; Lu et al., 2017; Makarichi et al., 2018). Table 3 shows the typical composition for solid municipal waste that enters an incinerator. It shows the fractions of materials as weight percentages. The origin of waste incinerated in Dutch waste incinerators in 2018 is shown in Figure 4. Here, ''Business waste'' is from the trade, service and government sectors. ''Other'' includes composting residues, industrial waste, hospital waste and residue from municipal waste services. Data is from Rijkswaterstaat (2020).

Table 3. Typical composition of incinerated waste in the Netherlands and minima and maxima found in EU statistics.

Type of waste	Typical share (wt%)	Minimum (wt%)	Maximum (wt%)
Organic	30	28	35
Paper	25	20	30
Plastic	12	11	19
Glass	5	5	5
Textiles	4	4	4
Metal	3	3	4
Other	21	8	22
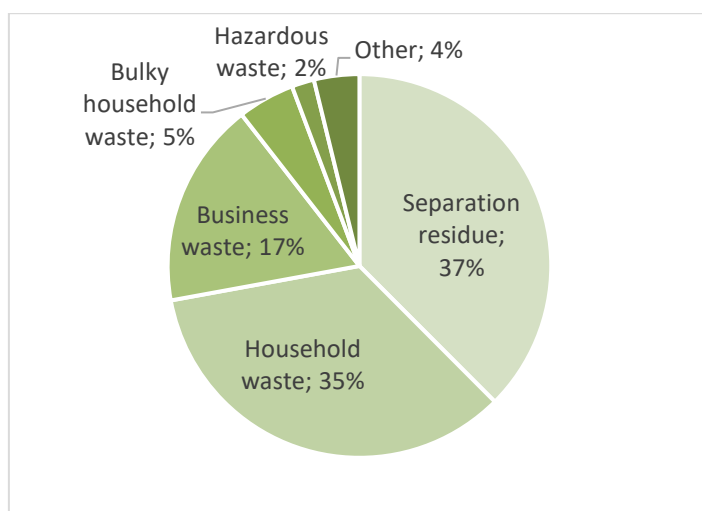
Figure 4. Origin of waste incinerated in the Netherlands in 2018.

Plastic waste in the Netherlands

The Netherlands processed 1,650 kt of plastic waste in 2017, of which 30% was packaging waste (Snijder & Nusselder, 2019). The rest consists of plastic waste from clothing, utensils, construction material and waste from agriculture, fishing and industry. 80-90% of fossil CO_2 emissions from waste incineration are estimated to originate from plastic incineration (Larsen & Astrup, 2011; Yang

et al., 2012). Packaging waste was incinerated at a rate of 52%, while 48% was recycled in some way.

Table 4 shows the mass balance for Dutch plastic packaging waste, adapted from Snijder and Nusselder (2019). The Dutch government has set a goal of using at least 60% recycled or biobased material in packaging material by 2030. Currently, the amount of recycled material in plastics is around 20% (Krebbekx et al., 2018).

Table 4. Plastic packaging waste processed in the Netherlands in 2017, values in kt plastic (Snijder & Nusselder, 2019).

Collection stream	Recycled	Incinerated	Total
PET deposit system	25.1	<0.5	25.6
Consumer pre-sorted	102.4	76.8	179.2
Business pre-sorted	87.0	0	87.0
Residual waste	25.6	179.2	204.8
Total	240.1	256	496.1

In order to reduce the emissions related to the currently incinerated waste stream, it is necessary to assess the feasibility of reducing the amount of plastics incinerated. For packaging waste, Table 4 shows that there are 2 streams that come in to consideration for incineration reduction. These are the pre-collection sorted consumer plastics and the sortable plastic fraction in residual waste.

The fraction of pre-sorted plastic packaging waste that is incinerated consists mostly of recycling refuse. Recycling refuse is the result of either unsuitability of the feedstock for the recycling methods in place or intolerable levels of contamination of the feedstock (Gradus, 2020). For an optimistic assessment of the availability of plastic packaging waste for chemical recycling, the entire stream mass of 76.8 kt is assumed to be available. The plastic packaging fraction in residual waste originates from a lack of effective separation techniques in place, be it post-collection or pre-collection separation. The estimated efficiency of plastic post-collection separation is 50% (Kerstens & Blanksma, 2019). This brings the potential yield of plastic packaging material from residual waste to 89.6 kt per year, and the total potential for alternative recycling methods to 166.4 kt per year. The potential is shown graphically in Figure 5.

It should be noted from Figure 5 that the potential for non-packaging plastic waste ("other") is not included. The stream consists of plastic waste from clothing, utensils, construction material and waste from agriculture, fishing and industry. The exclusion of this stream is related to its lack of coverage in literature. The composition and recyclability are therefore difficult to assess. What can be said about the "other" stream is that its size is roughly 1154 kt annually. From the average composition of incinerated MSW it is estimated that 12% is composed of plastics, totalling 935 kt per year. 256 kt of this amount is plastic packaging waste, meaning 679 kt of "other" plastic waste is incinerated annually. 475 kt of "other" plastic waste is thus not incinerated but most likely recycled, landfilled or exported.
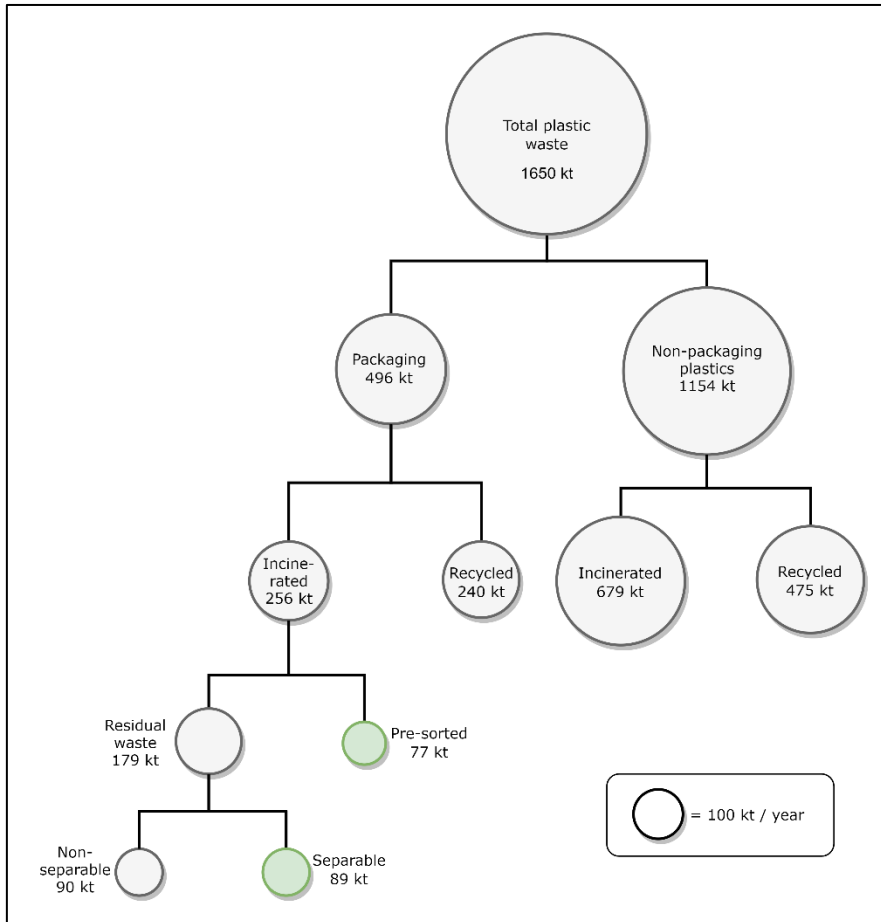
Figure 5. Mass balance for total plastic waste in the Netherlands, green highlights potential for alternative processing methods of currently incinerated plastics.

3.2 Waste incineration

Waste is incinerated in a continuously operated furnace in all Dutch plants except Zavin, which operates on an intermittent basis due to the specialized nature of its waste input. All WIPs use a "moving grate" process to efficiently incinerate the waste at a constant flowrate (van Blijderveen, 2012). The movement of the grate also ensures proper mixing of the waste during incineration. Figure 6 shows a schematic of this kind of furnace. Waste is inserted in the funnel on the left (stream 3 in Figure 2), after which the grate slowly moves the waste it into the furnace. The grate itself is commonly made from two rows of bars that move independently of each other, although a roller track type conveyor is also used in some situations (van Blijderveen, 2012). The resulting heat is transported to the cogeneration plant. Air is introduced at the bottom of the grate. If the LHV of the waste is below the threshold for proper incineration, this air can be preheated to aid the incineration process (Makarichi et al., 2018).
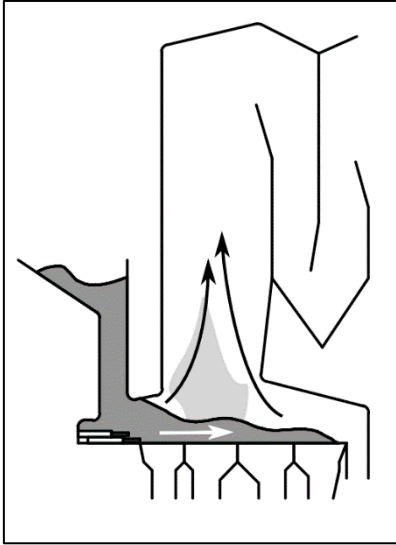
Figure 6. Process scheme of a moving grate furnace for waste incineration (van Blijderveen, 2012).

Waste incineration furnaces are limited in terms of their mechanical and thermal throughput. Since the furnace has to be operated continuously, they have both a maximum and minimum throughput in order to supply the furnace with enough fuel to keep it burning without overloading the grate or overheating any part of the incinerator. The implications of these limits is that the furnace has a "window" in which it operates, boxed in by thermal and mechanical maxima and minima, and further limited by the boundaries posed by the LHV of the input waste. This window is shown in a furnace diagram, see Figure 7 (Rijkswaterstaat, 2020).

Figure 7 shows the furnace diagram of a hypothetical furnace with typical mechanical and thermal limitations. The upper and lower horizontal lines indicate the thermal maximum and minimum of the furnace, in this example being 40 and 95 MWth input capacity respectively. The vertical left and right lines delimit the mechanical minimum and maximum load, respectively, which are 18-34 tonne/hour here. The blue and yellow lines indicate the highest and lowest LHV of the waste this furnace is able to process. It is visible in Figure 7 that an LHV of 5 GJ/tonne barely falls within the limits of the furnace. For this reason, WIPs often maintain a minimum LHV of the waste of around 5 GJ/tonne until they need to aid the incineration process with preheated air injection or high-LHV fuel addition such as coal (Lu et al., 2017; Makarichi et al., 2018).
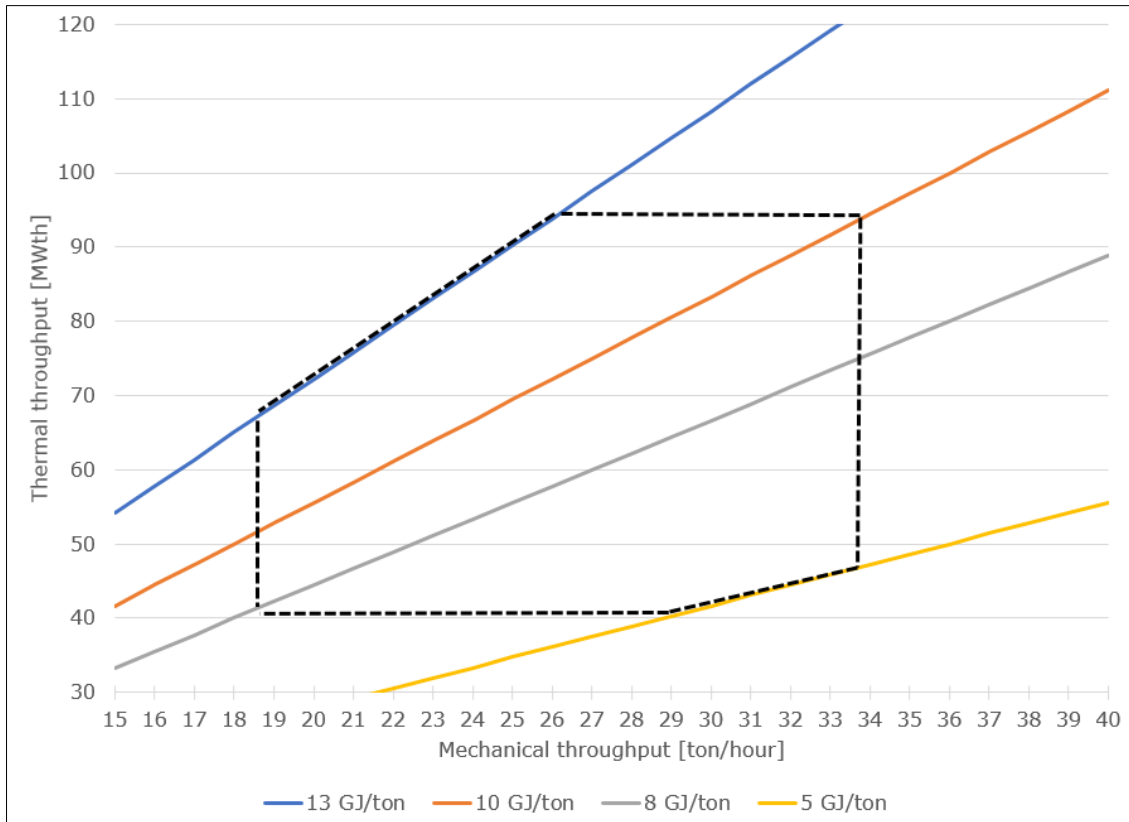
Figure 7. Furnace diagram of a waste incinerator.

3.3 Flue gas treatment

The flue gas from the waste incineration process is indicated with streams 4, 5 and 6 in Figure 2. Due to the wide variety of substances present in the input waste, untreated flue gasses from waste incinerators can contain harmful emissions including but not limited to: heavy metals, dioxins, furans, carbon monoxide, hydrogen chloride, hydrogen fluoride, sulphur dioxide and nitrous oxides. The emissions of Dutch waste incinerators are limited by the European Waste Incineration Directive, which includes limiting values for all of the aforementioned pollutants (EC, 2000). To limit the emission of these pollutants, WIPs use electrostatic precipitators (E-filters) and baghouse filters, often combined with the use of calcium carbonate or sodium bicarbonate as part of a multistage scrubbing process (Makarichi et al., 2018). Additionally, combustion control using data from heat sensors to adjust the air supply is used to limit the formation of pollutants.

In terms of energy consumption, the E-filter uses the bulk of the power. Rijkswaterstaat reports that, on average, 20% of the electricity produced at the WIP site is used for self-consumption, and the bulk of this consumption is caused by the E-filter and ancillary processes on the waste incinerator site (Rijkswaterstaat, 2020). This energy demand is shown in stream 10 in Figure 2.

3.4 Cogeneration, efficiencies & energy balance

The plant efficiencies of Dutch waste incinerators are not published and were inferred from the R1-coefficient and the LHV of the incinerated waste. The R1-coefficient is a performance measure for the useful energy recovery from waste incineration plants. The R1-coefficient is calculated using Equation 6 and Equation 7 (EC, 2008).

Equation 6. R1-coefficient.

$$\eta_{R1} = \frac{E_p - (E_f + E_i)}{0.97 * (E_w + E_f)} * CCF$$

Equation 7. Annual energy production coefficient.

$$E_p = 2.6 * E_{el} + 1.1 * E_h$$

Here, η_{R1} is the R1-coefficient, E_p is the annual energy production as heat or electricity in GJ/year. Electricity production (E_{el}) is multiplied by 2.6 and heat production (E_h) by 1.1. The factor 2.6 is related to the average electrical efficiency of an EU coal plant (38%), while 1.1 relates to the average thermal efficiency of an EU heat plant (91%) (EC, 2008).

E_f is the annual energy input needed for the production of steam for the WIP in GJ/year. E_w is the annual energy contained in the waste, computed with the lower heating value (LHV), in GJ/year. E_i consists of all other energy imports apart from E_w and E_f in GJ/year. 0.97 is the factor accounting for energy loss in the bottom ash, and through radiation. The CCF is the climate correction factor which accounts for reduced maximum conversion efficiency at higher ambient temperatures. For EEW Delfzijl, the CCF is set to 1.143, while all other plants get a CCF of 1.069. η_{R1} is calculated annually to ensure that the R1-certification is still relevant (Rijkswaterstaat, 2021). Assuming E_f and E_i are 0, which holds for plants that do not import energy for the process of waste incineration, the energy content of the waste is calculated using the following equation.

$$E_w = \frac{E_p * CCF}{0.97 * \eta_{R1}}$$

The resulting plant-level LHVs are shown in Table 5. For the purpose of validation of the above described method, the total weighted average was compared to the NIR average, which are both equal al 10.0 GJ/tonne. Zavin C.V. is excluded from these calculations since they do not deliver any heat or electricity and therefore lack the R-1 status. Also, characterization of Zavin waste input is more difficult since this plant only processes hospital waste (Rijkswaterstaat, 2020).

Table 5. Plant level LHV for input waste for 2018, Zavin C.V. excluded.

Plant name	LHV of waste (GJ/tonne)
Afval Energie Bedrijf (Amsterdam)	9.5
ARN B.V.	12.8
Attero BV (Moerdijk)	10.0
Attero Noord BV	9.3
AVR Afvalverwerking BV (Duiven)	9.2
AVR NV (Rijnmond)	9.1
EEW Energy from Waste Delfzijl B.V.	8.6
HVC (Alkmaar)	9.9
HVC Afvalcentrale Dordrecht	13.1
Reststoffen Energie Centrale (REC)	13.9
SUEZ ReEnergy Roosendaal	10.0
Twence BV	11.3
Total calculated weighted average	10.0

With the LHV of the input waste per plant known, the thermodynamic efficiency can be calculated with Equation 8.

Equation 8. Thermodynamic efficiency for waste incinerators.

$$\eta_i = \frac{E_{el} + E_h}{E_w}$$

E_{el} is the net electricity delivered to the grid and E_h is the heat delivered to external parties, these flows are shown in streams 8 and 9 in Figure . The efficiencies of the WIPs are shown in Table 6. Since there is a strong correlation between the relative amount of heat produced and the plant efficiency, the Heat/Power ratio or HP ratio is also given in Table 6. This is simply the ratio of useful heat exported to gross electricity production. Finally, Figure 8 shows the energy balance of the average Dutch waste incineration plant.

Table 6. Plant level thermodynamic efficiencies, total heat & power production and the Heat/Power ratio.

Plant name	Net Efficiency[a]	Gross electricity production (TJ)[b]	Net heat delivered (TJ)[b]	HP ratio[c]
Afval Energie Bedrijf (Amsterdam)	27%	3,370	1,112	0.33
ARN B.V.	44%	2,221	1,618	1.34
Attero BV (Moerdijk)	38%	608	813	0.73
Attero Noord BV	24%	1,404	327	0.23
AVR Afvalverwerking BV (Duiven)	31%	518	698	1.35
AVR NV (Rijnmond)	47%	1,498	4,489	3.00
EEW Energy from Waste Delfzijl B.V.	55%	666	1,257	1.89
HVC (Alkmaar)	25%	1,620	293	0.18
HVC Afvalcentrale Dordrecht	37%	486	971	2.00
Reststoffen Energie Centrale (REC)	70%	508	1,705	3.36
SUEZ ReEnergy Roosendaal	25%	1,004	101	0.10
Twence BV	36%	1,235	1,519	1.23
Total	36%	12,115	14,903	1.23

a)   Own estimation based on calculations outlined above.
b)   Data from Rijkswaterstaat (2020).
c)   Own calculation, ratio of gross electricity production and net heat delivered.
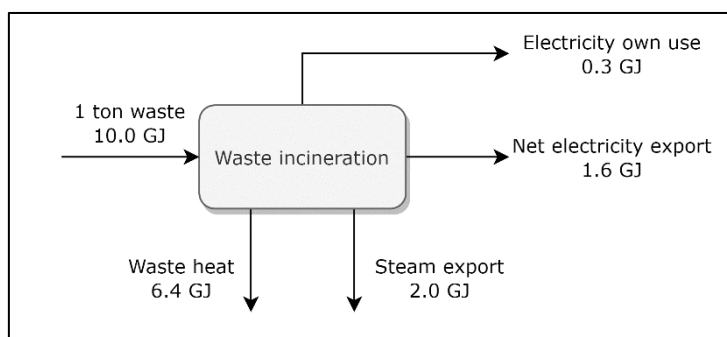
Figure 8. Energy balance for average Dutch waste incinerator.

3.5   Residual products & mass balance

The non-energetic solid products from the incineration process are indicated as stream 11 in Figure 2. The main products are fly ash and bottom ash. Of the 7.479 kt of input waste in 2018, 1.855 kt of bottom ash were produced, or roughly 25% of the input mass (Rijkswaterstaat, 2020). This includes a ferrous metal fraction (126 kt, 7% of bottom ash) and non-ferrous metal fraction (34 kt, 2% of bottom ash). Up until 2021, bottom ash was primarily used as a filler under newly constructed roads. Since then, the Dutch government has implemented a law to forbid this type of utilization of bottom ash in order to prevent contamination of the soil and groundwater (Ministerie van Infrastructuur en Waterstaat, 2021). A transition period has commenced which ended on January 1st 2022 to process the current bottom ash stockpiles. Since 2020, other applications of the bottom ash have been found, such as filler material in foundations and concrete structures (BodemPlus, n.d.).

The second-largest fraction of residue is fly ash. 99 kt of fly ash was produced in 2018 (1% of total input mass) (Rijkswaterstaat, 2020). 48 kt was landfilled while the remaining 51 kt used for other useful applications, mostly as a filler material in, for example, asphalt.

Figure 9 shows the mass balance for the incineration of 1 tonne of MSW in a typical Dutch waste incinerator. The typical composition of the waste shown in Table 3 is assumed. Amounts for bottom ash, fly ash, metal content and CO_2 emissions are sourced from Rijkswaterstaat (2020) and RIVM (2021). The amount of water formed during combustion is calculated from an average hydrogen content of dry waste of 4% (Arena et al., 2015), and is based on the assumption that all hydrogen is oxidised to water during combustion. This assumption is supported by the fact that fly- and bottom ash are found to contain no traceable amounts of hydrogen (Saikia et al., 2015). On average, 12%wt of MSW is found to be oxygen (Arena et al., 2015). This oxygen is assumed to form water during combustion. The residual oxygen necessary for oxidation of the hydrogen comes from the atmospheric input (0.20 t). A similar assumption is made for the carbon in MSW, as close to 100% of this carbon oxidises to CO_2. The moisture content of the MSW is calculated by subtracting the fly ash, bottom ash and CO_2 from the total input mass, finding a moisture content of 29.5%.
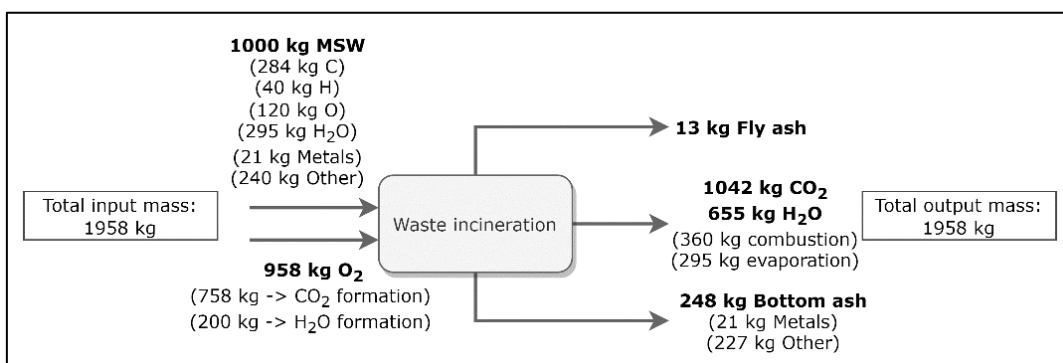
Figure 9. Mass balance for incineration of 1 tonne of waste in an average Dutch waste incinerator.

4   Emission reduction technologies

After finalising the analysis of current waste incineration in the Netherlands, the research moves to answering RQ2. The first step was technology selection, which is done based in initial criteria from Section 2.2. The reference process for the feedstock used by technology must be incineration, and the TRL must be 5 of higher. For these reasons, supercritical water gasification was not included. For MSW pyrolysis and gasification, the exclusion process was more complex. Therefore, these technologies are discussed separately below. This section first evaluates carbon capture technologies for waste incineration, after which it discusses the setup and results of each LCA. Each technology is introduced with its basic principles, whereafter the mass- and energy balances are discussed and the net emissions are estimated. Lastly, the economic analyses for all technologies are presented, as well as the uncertainty analysis.

Table 7. Emission reduction technology selection.

Category	Technology	Potential feedstock	TRL	Included	TRL Source
Carbon capture	CCS	MSW	7	Yes	a)
	CCU	MSW	9	Yes	a)
	CCUS hybrid	MSW	7	Yes	a)
MSW options	Pyrolysis	MSW without glass/metal/stone	5-7	No	b)
	Low-medium temperature Gasification	MSW without glass/metal/stone	8	Yes	c)
	Supercritical water gasification	Biomass	3-4	No	d)
Plastic options	Pyrolysis	Mixed plastic waste, recycling refuse	5-7	Yes	b)
	Gasification	Mixed plastic waste, recycling refuse	5-7	No	b)
	PET depolymerisation	PET trays	5	Yes	b)
	EPS Solvolysis	HBCD-containing EPS	5	Yes	b)

a) Lensink and Schoots (2020)
b) Broeren et al. (2019)
c) Enerkem (n.d.)
d) IEA Bioenergy (2020)

4.1   Carbon capture, storage and utilisation

In order to reduce the net CO_2 emissions to the atmosphere, the CO_2 can be captured. This can be done in a number of ways, the most common methods being: 1) post-combustion carbon capture using a chemical absorption solvent, 2) pre-combustion carbon capture or 3) oxy-fuel carbon capture. Globally as of 2020, there were 4 waste incineration plants using carbon capture technology to reduce their atmospheric CO_2 emissions. These are Klementsrud CHP in Norway, Saga City plant in Japan and AVR Duiven and Twence in the Netherlands (Wienchol et al., 2020). Twence and AVR Duiven captured 3.6 kt and 30 kt of CO_2 in 2020 respectively (AVR, 2021; Twence, 2020). Both of the Dutch plants use a method of post-combustion carbon capture using amines.

The general process for flue gas carbon capture with amines is shown in Figure 10. The input flow (1) consists of the scrubbed flue gasses, which enter the absorber unit containing the amine solvent, most commonly monoethanolamine (MEA). The CO_2 in the flue gas dissolves in the solvent and the flow is pre-heated in a heat exchanger (2-3). This preheated rich solvent stream enters a stripper unit, where the flow is heated (4), releasing the CO_2 in near pure form (5) (Oh et al., 2016). The CO_2 can then be compressed and optionally liquefied, depending on the mode of transport. The hot lean solvent flows from the stripper unit to the heat exchanger, where it is cooled and travels back to the absorber unit (7-8). In the absorber, excess flue gasses containing some undissolved CO_2 need to be vented (9). This limits the CO_2 capture rate of the process to 85% (Ho et al., 2019). The process consumes heat and electricity. Heat is needed for the separation of CO_2 from the solvent, and electricity for pumps, compression, liquefaction and ancillary processes. In total, the process consumes around 212 kWh of electricity and 670 kWh of heat per tonne CO_2 captured (Lensink & Schoots, 2020).
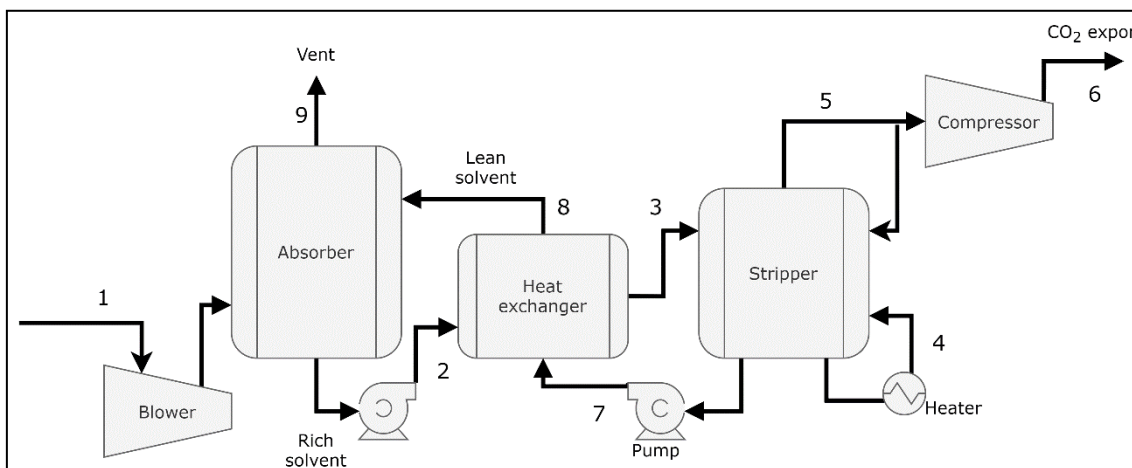
Figure 10. Process flowchart of chemical based absorption carbon capture (Mumford et al., 2015; Tang & You, 2018).

Flow number	Description
1	Flue gas from incineration process.
2	Cold solvent with dissolved CO_2.
3	Partially heated solvent with dissolved CO_2.
4	Fully heated solvent towards stripper.
5	CO_2 stripped from solvent.
6	Compressed CO_2.
7	Hot lean solvent.
8	Partially cooled lean solvent.
9	Undissolved CO_2 vented to atmosphere.

There are two major alternatives to the post-combustion carbon capture shown in Figure 10. These are pre-combustion carbon capture and oxy-fuel carbon capture. In pre-combustion carbon capture with Integrated Gasification and Combustion Cycle (IGCC), the municipal solid waste is fed into a gasification chamber where syngas (mostly H_2, CO, CO_2, CH_4) is generated and extracted. This gas is then treated in a carbon capture unit similar to that shown in Figure 10, whereafter the H_2-rich

stream can be combusted for energy generation purposes (Descamps et al., 2008; Hossain et al., 2020). The capture rate of this kind of setup is estimated at 50-60% (Lensink & Schoots, 2020).

In oxy-fuel carbon capture processes, the MSW is combusted in a similar fashion to normal WIP processes, but the air that is normally introduced into the process is replaced with a stream of near-pure oxygen. In doing so, the process temperature is usually higher, resulting in more complete combustion of the MSW. Additionally, the flue gas does not contain the large fraction (ca. 80%) of N_2 that it normally does, thereby creating a high-CO_2 flue gas, also containing H_2O as a by-product of combustion (Wienchol et al., 2020). CO_2 capture rates vary widely, but can be as high as 90% (Letcher, 2018).

For the pre-combustion and oxy-fuel techniques, the technology readiness is estimated at level 7, since no qualified demonstration at scale could be found at this time for waste incinerators. For this reason, combined with a lack of technical and economic data on the application of these techniques, and the lower capture rates, pre-combustion- and oxy-fuel carbon capture were not analysed further.

4.1.1   Carbon capture & storage

Once the CO_2 is captured, long-term underground storage is a viable method of reducing atmospheric emissions. In the Netherlands, the Porthos project (Port of Rotterdam CO_2 Transport Hub and Offshore Storage) is planned to start storing industrial CO_2 from various sources, mostly located in and around the Port of Rotterdam industrial cluster (Porthos, 2021). The gas will be stored in a depleted natural gas field, roughly 20km offshore. The project is planned to accept up to 2.5 Mt of CO_2 annually in the first few years of operation, after which up to 5 Mt can be stored at Porthos each year (Ministerie van Economische Zaken en Klimaat, 2019). Since Porthos is not the only offshore CCS project that the Dutch government intends on support, the total offshore CCS potential is assumed to be large enough to provide storage for CO_2 from waste incineration for 20-30 years. In this sense, CCS should be viewed as a transition technology.

4.1.2   Carbon capture & utilisation

When CO_2 is captured it may also be applied in a useful way, which is called carbon capture and utilisation (CCU). Examples of CCU that are currently applied in the Netherlands are the artificial fertilisation of crops in greenhouse horticulture and the production of sodium bicarbonate. Ho et al. (2019) also notes enhanced oil/gas recovery, methanol production and urea production as utilisation options for captured CO_2. The two technologies that are currently applied in the Netherlands are briefly discussed here, whereafter an assessment of the TRL and total potential of CCU is addressed.

Greenhouse horticulture fertilisation
In order to increase crop yield, the greenhouse horticulture industry artificially increases the CO_2 concentration inside the greenhouses. This is a form of artificial fertilisation. The atmospheric CO_2 concentration of roughly 400 ppm is increased to anywhere between 500-1000 ppm, resulting in an

increase in crop yield[1] (Lensink & Schoots, 2020). For around 2/3rd of the greenhouse area in the Netherlands, gas-powered CHP units generate the necessary CO_2 for this fertilisation. In this situation, heat is used whenever necessary, while most of the electricity generated is sold to the grid. As such, the CHP units are not just a means of generating CO_2, they also supply heat and generate revenue from electricity sales. Around 1/3rd of greenhouse area is heated and fertilised by gas boilers, which do not generate electricity. It has been shown that supplying greenhouses with 1 tonne of CO_2 from WIPs decreases the greenhouses own emissions by 0.93 tonnes (Van der Velden & Smit, 2020), although not all of this emission reduction can be attributed to CCU, since there is still a demand for heat, and the electricity that was previously sold to the grid must be generated elsewhere. Accounting for all these factors, the authors of the SDE++ 2021 Eindadvies estimate a total avoidance of 0.80 tonnes of CO_2 per tonne delivered to the greenhouse.

In terms of the attribution of the reduced emissions, CCU differs from CCS in another way. Since the CO_2 delivered to greenhouses is released shortly after delivery, either by venting from the greenhouse or after harvest of the crop, the only CO_2 reduction that is realised is the reduced use of natural gas at the greenhouse. This removes the incentive for the waste incinerator to pay for the handling of the CO_2 after capture. Instead, the CO_2 is sold on the market, in this case to greenhouses. Moreover, demand for CO_2 fluctuates throughout the year peaking in the summertime due to growth season of the produce (Lensink & Schoots, 2020). This allows for a scenario where part of the captured CO_2 from waste incineration is utilised for greenhouse fertilisation, while the remainder is stored via the Porthos infrastructure.

The total potential for horticulture fertilization with CCU is estimated to be 2.5 Mt per year in the Netherlands, of which roughly 0.6 Mt is currently being met annually (Glastuinbouw Nederland, n.d.). The technology is currently applied at large scales, thus the TRL is 9.

Sodium bicarbonate production

Apart from delivering captured and liquefied CO_2 to greenhouses, Twence also uses part of its CO_2 to produce sodium bicarbonate, roughly 2 kt annually. In this process, aqueous sodium hydroxide (NaOH) is used as a feedstock, which is exposed to scrubbed flue gas from the waste incinerator. The sodium hydroxide reacts with the CO_2 in the flue gas to form sodium bicarbonate (better known as baking soda). The reaction is endothermic and requires 522 kWh of heat per tonne sodium bicarbonate, capturing 0.52 tonnes of CO_2 per tonne sodium bicarbonate produced (Shim et al., 2016). In a laboratory setting, the process was found to have a CO_2 capture rate of 95-99%. The overall reaction equation is shown below.

Equation 9. Reaction equation for sodium bicarbonate production from sodium hydroxide and CO_2.

$$NaOH(aq) + CO_2(g) \rightarrow NaHCO_3(aq)$$

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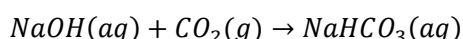
[1] *The actual increase in crop yield varies between crops and is also found to be dependent on lighting conditions, temperature, and other factors (Dieleman & Meijnen, 2003). The extent of the increase in crop yield is debatable, since a reduction in CO_2 dosage of 50% was found to correlate with a 1,5% reduction in crop yield for tomatoes (de Gelder, 2012).*

Twence uses the sodium bicarbonate for flue gas scrubbing (Twence, 2020), but the product is versatile in its application and has a global market size of 1.6 billion USD and an expected annual growth rate of 0.8% (More, 2021). If used as a CO_2 emissions reduction technology, this type of CCU requires that the product displaces another product that produces more net emissions. It is unclear whether this can be the case for sodium bicarbonate production, since the conventional process can also incorporate industrial CO_2, negating any emissions reduction effects.

The total potential for this technology is difficult to assess since it is not clear whether the produced sodium bicarbonate is of food-grade quality. The process is applied currently, albeit at a sub-scale level. The TRL is estimated at 5-7, depending on the product application.

Alternative uses for captured CO_2

Captured CO_2 may also be used for a number of other purposes. Among these are enhanced oil/gas recovery, methanol production and urea production (Ho et al., 2019). Each technology may have its own requirements for the CO_2 feed in terms of pressure, temperature and purity. Communication with waste incinerators brought to our knowledge that despite adequate CO_2 purity standards, food-grade application is an unsuitable option for waste incineration CCU plants. This is because, due to the variety of the waste feed, contamination of the CO_2 can never be fully ruled out by all parties. Any resulting downstream issues when utilising the CO_2 may cause liability issues for the waste incinerator supplying the CO_2. In the long-term future, it is expected that CO_2 can act as a feedstock for many organic compounds, but as of yet this technology is underdeveloped.

The TRL of the alternative CCU technologies varies greatly given the parameters involved and uncertainty regarding suitability of the captured CO_2. For this reason, CCU for greenhouse fertilisation is assumed to be the most likely application for CO_2 from waste incinerators. The analysis from here will include a CCU/CCS hybrid option, where 50% of the captured CO_2 is used for greenhouse fertilisation in the growth season, and 50% is stored during the winter months.

4.1.3 Techno-economic parameters of carbon capture plants

Table 8 summarises the techno-economic parameters for the three identified CCU and CCS variants at a plant scale of 100 kt CO_2 captured per year. The amount of CO_2 captured for utilisation is based on an installation that operates only during the summertime, when greenhouse CO_2 demand is high. The CCS and CCUS hybrid variants operate year-round, where the hybrid variant stores any excess CO_2 that is not utilised.

Capital expenses (CAPEX) are the sum of the investment costs for the carbon capture unit, as well as compression, liquefaction and transportation. Operating expenses are the sum of the fixed annual O&M, variable O&M, transportation and energy costs, as well as the processing charge incurred when delivering CO_2 to Porthos. This charge is set at € 54.7/t CO_2 delivered. Revenue from the sale of CO_2 to greenhouses is estimated at € 60/t CO_2 in consultation with industry. Any benefits gained in the form of reduced carbon taxes for CCS options are not taken into account in these calculations.

Table 8. Techno-economic parameters for reference carbon capture facilities in three configurations (Lensink & Schoots, 2020).

Parameter	CCU only	CCS only	CCUS hybrid
Operational hours (hours/year)	4000	8000	8000
Captured CO2 for utilisation (kt CO2 captured/year)	55	0	55
Captured CO2 for storage (kt CO2 captured/year)	0	100	45
CAPEX (M€)	56.38	56.38	56.38
Annual cost of capital (M€/year)	5.80	5.80	5.80
Fixed O&M cost (M€/year)	1.70	1.70	1.70
Variable O&M cost (M€/year)	1.26	2.29	2.29
Transport costs (M€/year)	1.16	2.10	2.10
Porthos processing charge (M€/year)	0	5.74	2.46
Revenue from CO2 sales (M€/year)	3.30	0	3.30

The net fossil CO_2 emissions resulting from these three carbon capture configurations are shown in Figure 11. The functional unit is 1 kg of MSW incinerated in a Dutch waste incinerator. The emissions due to heat and electricity consumption are based on an electricity consumption of 212 kWh/tonne CO2 and a heat consumption of 670 kWh/tonne CO2 captured, and are (partially) compensated with the local electricity and heat generation from incineration (Lensink & Schoots, 2020). Generation efficiencies for an average Dutch WIP are 16% and 20% for electricity and heat respectively, and the LHV of MSW is 10 MJ/kg. The direct fossil emissions result from a capture rate of 85% and a fossil carbon content of 36.6%. Hence, negative emissions due to biogenic carbon storage is based on the inverse of this, meaning a 63.4% biogenic carbon content. Avoided CO_2 at the greenhouse (for CCU configurations) is based on the avoidance rate of 0.80, detailed above. Since the majority CO_2 delivered to the greenhouse is biogenic in origin, thereby not counting towards fossil CO_2 emissions, and the emissions avoided at the greenhouse are 100% fossil (as they are assumed to originate from natural gas combustion), the avoided emissions at the greenhouse are larger than the total fossil CO_2 captured. Finally, the fossil CO_2 delivered to the greenhouse is eventually emitted as it is not stored long-term, meaning that the fossil part of the emissions for the reference incineration and the CCU configuration are identical.
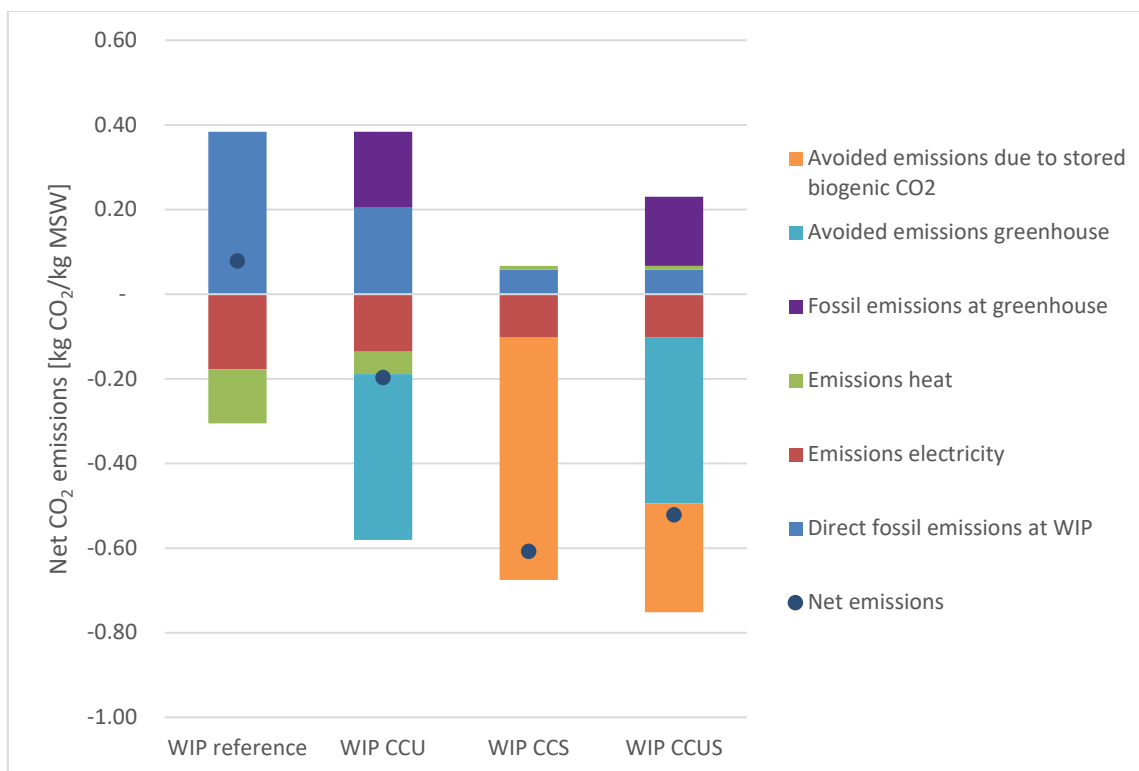
Figure 11. Net CO_2 emissions of three carbon capture configurations compared to the reference following IPCC guidelines.

4.2 Ex-ante comparative life cycle analysis results

The comparative LCA was used to estimate the net emission reduction of various enhanced recycling techniques. Figure 12 shows the overview of various recycling pathways, and how they interact with the existing material production chain. In this case, the plastic production chain is shown, but for base chemicals and chemical feedstocks, other production chains are conceivable. Additionally, since the status of fuel substitution pathways as recycling techniques is debatable, these are not shown in this figure. The appendix contains a list of planned projects of enhanced waste recycling technologies, see Table 19.
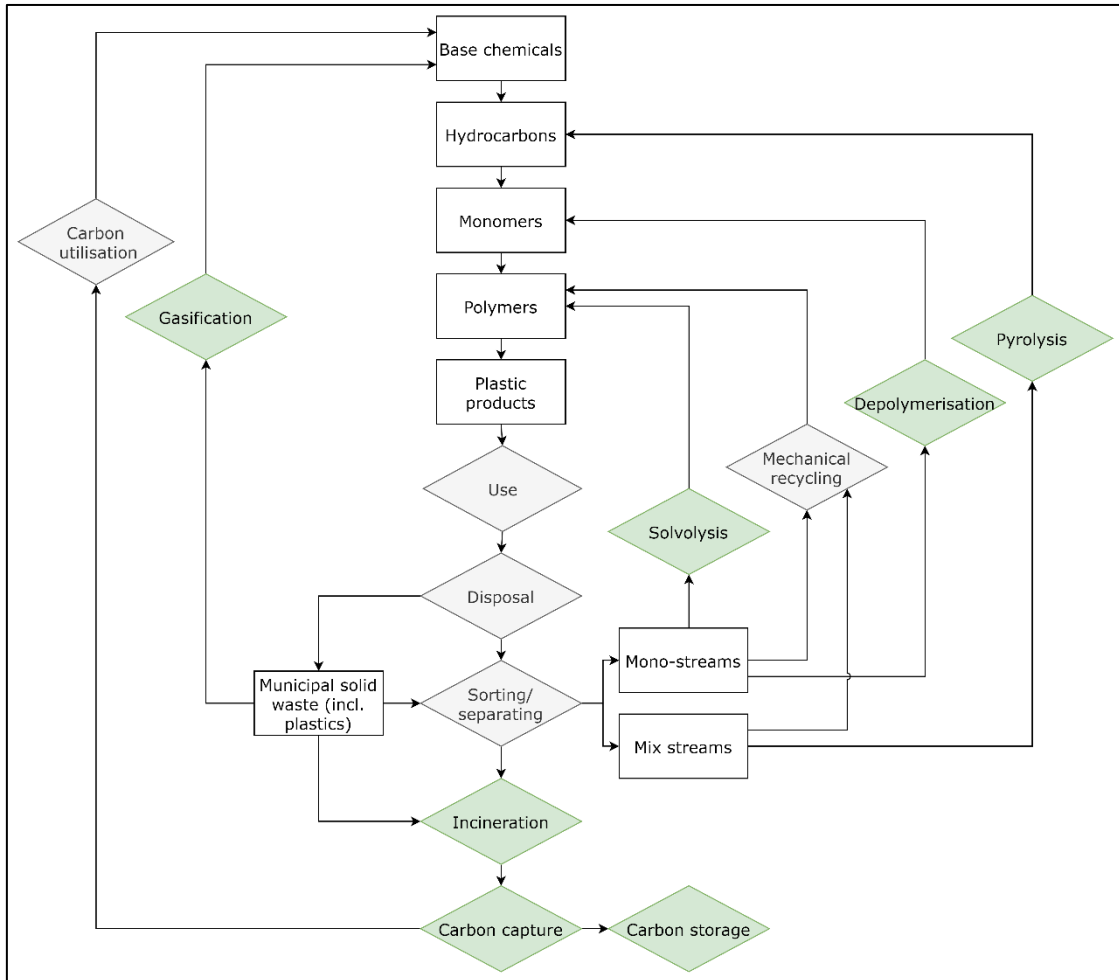
Figure 12. Overview of recycling technologies relative to the plastic production chain.

The green elements in Figure 12 highlight pathways that are discussed in this report in detail. This section will first discuss techniques that deal with (relatively) untreated and unsorted municipal solid waste streams, since these are the most widely available and omnivorous in terms of the process feed. Next, more specific and niche techniques will be discussed, such as depolymerisation and solvolysis of specific plastics.

4.2.1 Municipal solid waste gasification

Gasification involves thermochemical treatment in a reactor to produce syngas, a mixture of gaseous compounds, generally with a high H_2 and CO content (Arena, 2012). This can be done in the presence of air (oxygenated or standard conditions) or steam. The advantage of steam and oxygenated gasification is that the resulting syngas contains less or no nitrogen gas, making the syngas product more concentrated. The process is endothermic and needs a supply of external energy or the combustion of part of the produced gas in order to sustain the reaction (Arena, 2012). Most commonly, gasification is applied to solid biomass, but some plants are known to deal with MSW as well. The main disadvantages of MSW gasification are the variability of the feedstock, incombustible materials such as metal and glass, and heavy contamination of the process outputs. In part, these problems can be alleviated by sorting, separating, drying and pelletising the MSW to create Refuse Derived Fuel (RDF) pellets. These pellets have a higher calorific value due to concentration of the combustible materials and low moisture content.

Energy generation

Most gasification plants for MSW are designed for energy production through the combustion of the gases that are produced. Panepinto et al. (2015) notes that there were 5 plants worldwide with more than 10 kt of MSW per year capacity as of 2015. These plants were all Waste-To-Energy plants with an average net efficiency of 16%. This efficiency is substantially lower than Dutch WIPs, which have an average net efficiency of 36% (Rijkswaterstaat, 2020). Two large factors limiting the efficiency of gasification energy plants are the full utilisation of the on-site heat production and the higher self-consumption of the electricity that is produced (26% for gasification versus 20% for WIPs).

Fuel substitution

An alternative to gasification for energy generation is the utilisation of syngas to create fuels or chemical substitutes. AlMohamadi (2021) modelled a Waste-to-Fuel plant processing 580 kt of MSW per year at 7000 operational hours per year. The plant produced 61.8 wt% syngas which finally produces 9.5 wt% gasoline (a mixture of paraffins, olefins and aromatics). The energetic content of the gasoline product was 32% of the feedstock on LHV basis. The remainder of the energy in the feedstock was used for process heat, and steam and power production. The energy was extracted by combustion of the solid fraction and some gases, and was enough to make the process self-sufficient. The MSW used by the process of AlMohamadi (2021) is similar to Dutch MSW, so no residual streams need to be incinerated. Residuals such as metal, glass, tar and char are assumed to be landfilled with negligible GHG emissions resulting from this.

Before assessing the net emissions from the gasoline produced, the system boundary must be set. Figure 13 shows the system boundary for both the reference system and the gasoline pathway. The combustion process is included in the scope since the higher LHV of the produced gasoline substitute must be taken into account for a fair comparison.
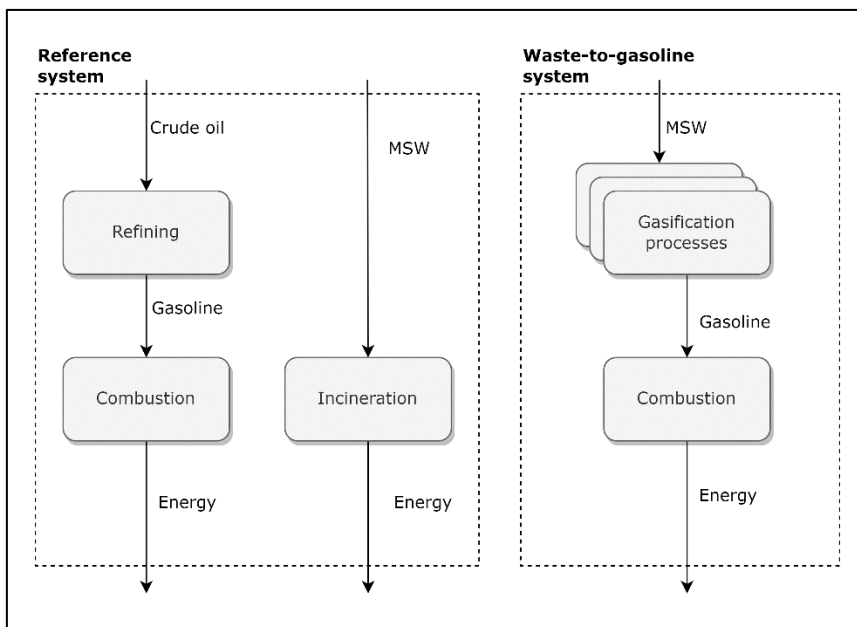
Figure 13. Cradle-to-Grave scope for gasification pathway producing gasoline.

AlMohamadi (2021) does not mention the carbon content of the gasoline. It is assumed that the carbon content is identical to regular petroleum-based gasoline. The net emissions of the produced gasoline are slightly lower per kg than regular gasoline due to the higher LHV (45.1 MJ/kg versus 43

MJ/kg for regular gasoline). This leads to a 5% lower carbon footprint of the gasoline substitute per unit energy. When comparing this pathway to the incineration of MSW, this 5% reduction in emissions for the gasoline is not enough to compensate for the increased net emissions due to the absence of heat and electricity exports. This leads to higher net emissions for the gasification pathway, as is shown in Figure 14. Emissions resulting from gasoline production are based on the emission factor for crude oil refining in the Netherlands, which is 0.328 kg CO_2/kg crude oil (Jing et al., 2020). This value includes electricity use at the refinery, as well as natural gas and hydrogen consumption and the resulting emissions from these energy sources. AlMohamadi (2021) mentions the processes used are currently at laboratory scale, making the TRL 4-5. Since Figure 14 shows that the net CO_2 emissions for the gasification and gasoline substitution pathway are higher than the emissions incurred from the reference system, this method is not analysed further.
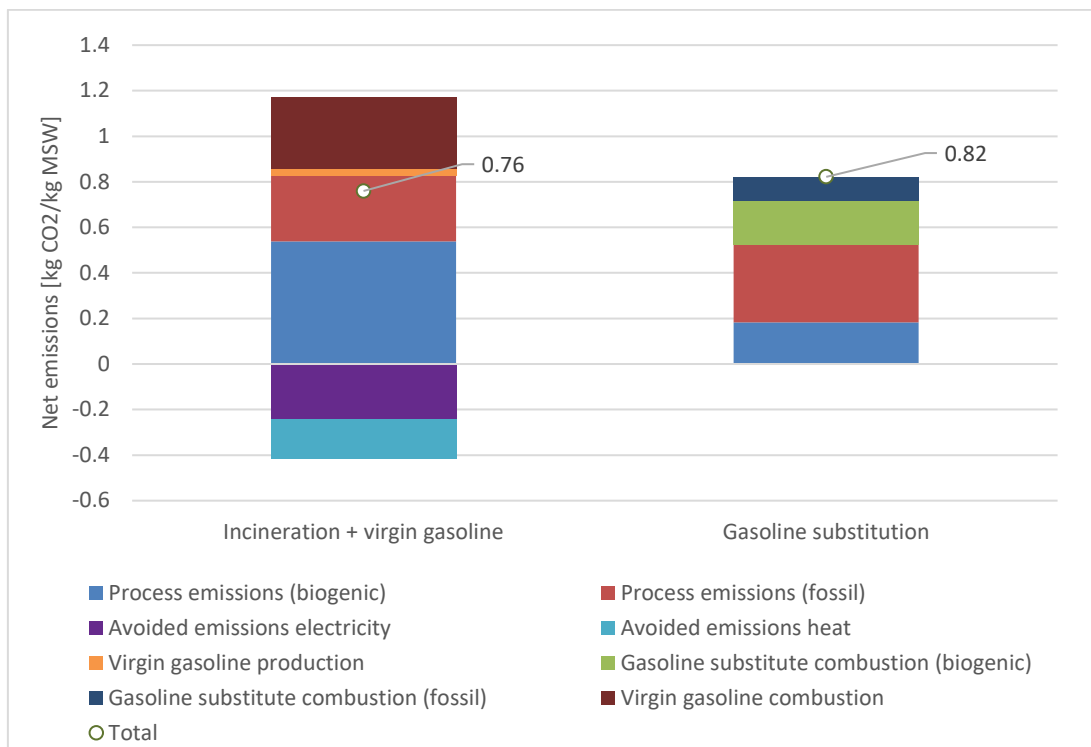
Figure 14. Net emissions for waste-to-gasoline pathway.

Methanol production

Iaquaniello et al. (2017) developed a model for techno-economic evaluation of a process for converting MSW to methanol. The model is based on the process employed by Enerkem, a Canadian biofuel production company who have a full-scale commercial plant in Alberta, Canada. The plant is operational since 2017 and processes 100 kt per year of refuse-derived fuel (RDF) (Enerkem, n.d.). Enerkem is also planning to construct a 360 kt per annum plant at the Botlek industrial cluster in the Netherlands. The Enerkem process is shown in Figure 15. The process requires a further 0.08 MJ for the air separation unit to supply the oxygen for the system. The system has a 39% carbon efficiency from RDF to methanol. If MSW is included, the carbon efficiency is 16%. Dotted lines indicate the possibility for liquefaction of the pure CO_2 outlet for storage.

The input for the process is typically not MSW but a derivative called RDF. To make 1kg of RDF, the glass, metal and moisture fractions must be removed from MSW and a higher concentration of paper, plastic and high-energy organics must be attained. The largest change in concentration is

needed for plastics, which go from 12% in MSW to 31% in RDF. At a sorting efficiency of 75% (which is on the high end, but achievable according to Kerstens and Blanksma (2019)), this requires 3.44 kg of MSW. It is assumed that all other fractions can be sufficiently sorted to achieve RDF purities with the 3.44 kg of MSW as a feedstock. Separation, milling and drying operations consume 1.8 MJ of electricity per kg RDF (Grzesik & Malinowski, 2016). At this ratio, there is roughly 1,700 kt of RDF available from domestic MSW in the Netherlands annually.
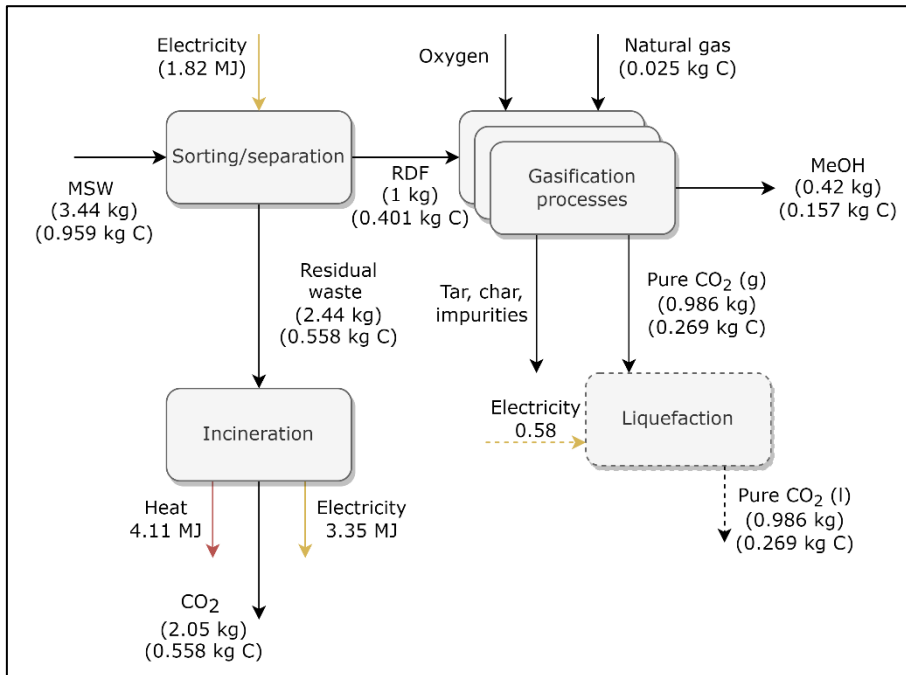
Figure 15. Process flowchart of Enerkem MSW-to-Methanol process.

The residual waste is assumed to be incinerated in a standard Dutch WIP. The RDF is gasified at >1000 °C in a high-oxygen environment to increase product purity. The resulting syngas does not have a sufficiently high H_2-content for methanol production. Hence, the syngas is purified and the flow is split. A fraction undergoes a water-gas shift to increase the H_2-content, after which the CO_2 is removed with an amine-based carbon capture process. This CO_2 is thus high purity and can be liquefied and stored, which is discussed at the end of this section. The hydrogen from the WGS is sent to the methanol synthesis reactor. The other fraction of syngas is sent directly to the methanol reactor. Residual gases from the methanol reactor are sent to a pressure-swing absorption process to remove any H_2. The purge gas is recycled into the gasification reactor. In this way, the only carbon coming out of the process is in the form of CO_2 or methanol (Iaquaniello et al., 2017). Key parameters of the process are shown in Table 9.

Table 9. Technological parameters for MSW-to-Methanol process in a 100 kt input per year plant .

		Parameter	Unit	Figure
Input		MSW	kt/year	344
		RDF	kt/year	100
		Electricity	TJ/year	190
		Natural gas	kt/year	2.87
Output	**from**	Methanol	kt/year	42
gasification		Pure CO_2 [a, b]	kt/year	99
		Residual waste	kt/year	244
Output	**from**	CO_2 emissions[a]	kt/year	205
incineration		Electricity[c]	TJ/year	335
		Heat[c]	TJ/year	412

a) CO_2 emissions are the sum of biogenic and fossil emissions.
b) Gasification emissions calculated assuming all carbon from natural gas and RDF is converted to methanol or CO_2.
c) Residual waste assumed to be incinerated in average Dutch WIP with 16% and 20% electrical and thermal efficiencies respectively.

Iaquaniello et al. (2017) note that tar formation, char build-up and contamination of the syngas with impurities are constant issues with RDF gasification due to feedstock variability. Especially tar and char formation cause significant downtimes for continuously operated reactors like in the Enerkem process. To combat these downtimes, Enerkem uses a layout with three parallel gasification and purification lines. For a 100 kt RDF input per year plant, each line has a capacity of 10 tonnes per hour. Due to expected downtime, the annual average capacity factor for the 3 lines is 38%. This increases investment and operational costs. The economic breakdown of a 10, 100 and 1000 kt plant is shown in Table 10. Costs based on economic figures from Iaquaniello et al. (2017) for a 100 kt plant, scaled up and down with a scaling factor of 0.7.

Table 10. Economic parameters for a 10, 100 and 1000 kt per year plant for MSW-to-Methanol system.

Parameter	Unit	10 kt plant	100 kt plant	1000 kt plant
CAPEX	M€	37.71	189.00	947.24
CAPEX per kt MeOH/year	M€/kt MeOH/year	3.77	1.89	0.95
OPEX incl electricity and NG	M€/year	3.57	17.90	89.71
CAPEX CO_2 liquefaction	M€	5.90	29.56	148.15

The net emissions for the Enerkem process are calculated using the system scope shown in Figure 16. The reference system includes the industry standard for methanol production, which is steam methane reforming. Additionally, MSW is incinerated in an average Dutch WIP with electrical and thermal efficiencies of 16% and 20% respectively. The MSW-to-MeOH system first converts the MSW to RDF, whereafter methanol is synthesised and the residual waste is incinerated in an average Dutch WIP. Iaquaniello et al. (2017) states that the methanol produced has similar characteristics to methanol from steam methane reforming. For this reason, the scope ends at the methanol output.

Carbon capture with gasification

The setup described by Iaquaniello et al. (2017) includes a carbon capture facility using amines, effectively creating a high-purity CO_2 stream, which is very suitable for storage or utilisation. For this assessment, it is assumed that the CO_2 is stored via the Porthos infrastructure. Therefore, CCU or CCU/S hybrid options are not included. This is done to limit the number of configurations assessed in the remainder of the research. Investment and operational costs of running the carbon capture facility are included in the cost figures presented in Table 10. Additionally, energy consumption from Table 9 includes energy consumption for the carbon capture process. What is not included is the Porthos processing charge, the investment costs of liquefaction and the energy consumption related to liquefaction of the CO_2. It is assumed that the facility has an 85% capture rate.
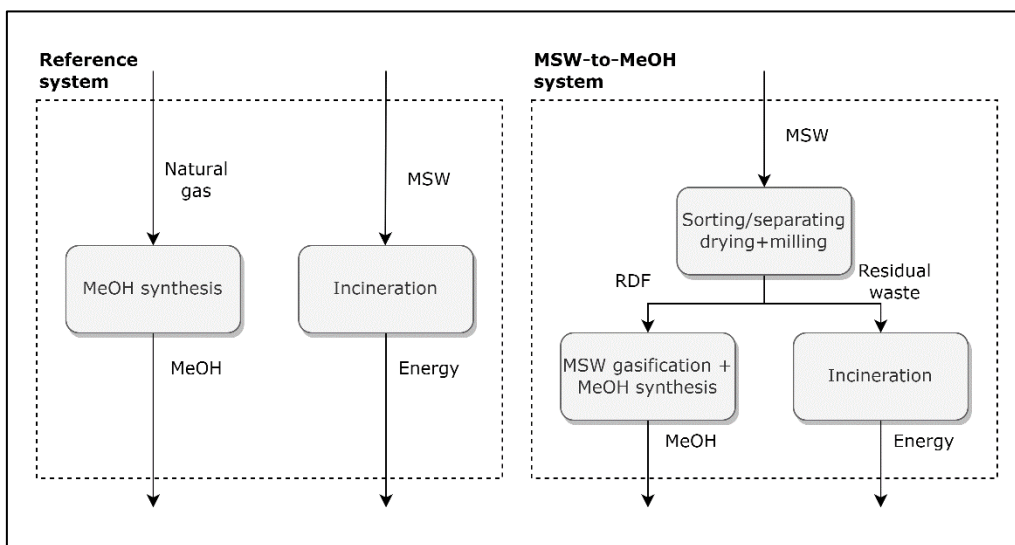
Figure 16. Cradle-to-gate scope for net CO2-eq emission calculations for the Waste-to-methanol system and the reference system.

Net CO_2 emissions of MSW-to-MeOH system

The net CO_2-eq emissions for both systems are shown in Figure 17. Reference emissions for methanol production are the average of the two most common methanol production methods 1) steam methane reforming and 2) partial oxidation of residual oil. This average emission factor is 0.76 kg CO_2/kg methanol. Avoided emissions from power are based on a predicted emission factor for 2019-2030 for Dutch electricity of 0.40 kg CO_2/kWh. Avoided emissions from heat based on emission factor of a 90% efficient natural gas boiler, 0.23 kg CO_2/kWh.
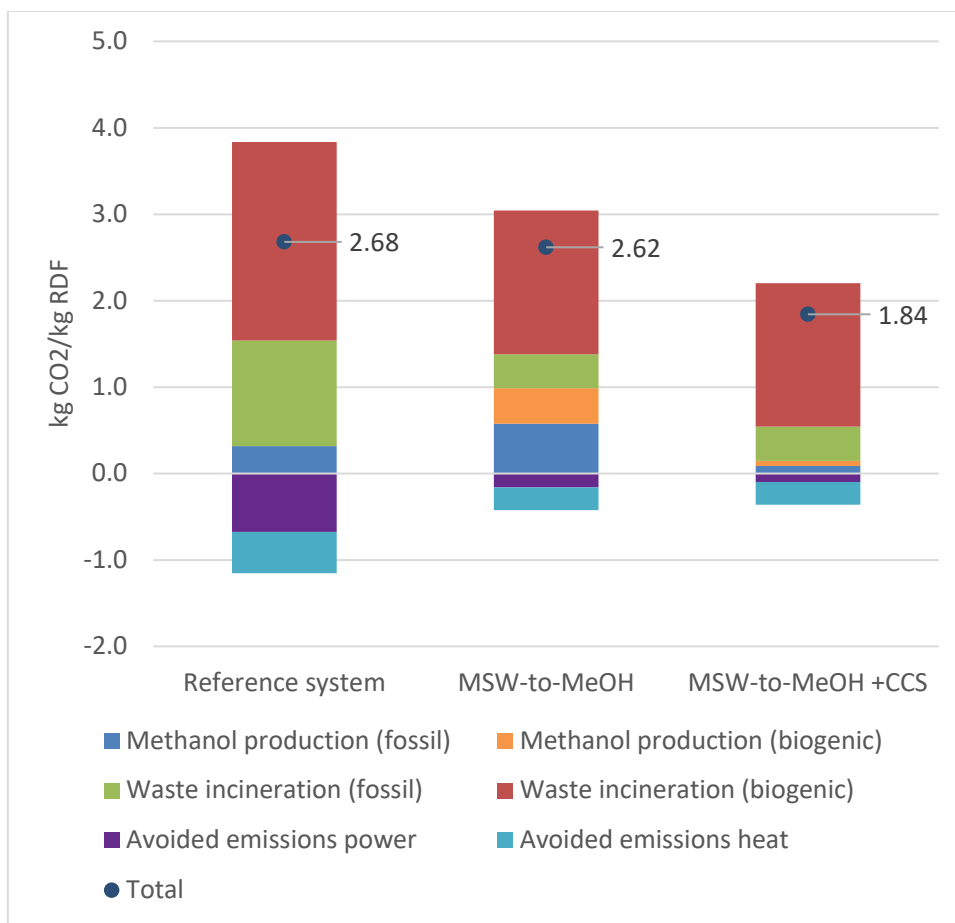
Figure 17. Biogenic and fossil CO2-eq emissions associated with the reference system and the MSW-to-Methanol pathway.

The CO_2 emissions related to methanol production and waste incineration shown in Figure 17 are split between fossil and biogenic emissions. The split is based on the fossil and biogenic carbon content of the RDF and the MSW it is made from. Following IPCC guidelines, biogenic CO_2 emissions that cause a net change in the biogenic carbon stock have already been accounted for in other sectors, and can thus be viewed as net-zero emissions. The net emissions for the reference system and MSW-to-MeOH system are impacted by this method of emission calculation, as is shown in Figure 18. It shows that when only counting fossil emissions, the reference is less carbon intensive than the MSW-to-MeOH system. This is mostly due to the larger avoided emissions in the waste incineration case. Storage of biogenic CO_2, following the IPCC guidelines, is counted as a negative emission. This causes the net emissions for the gasification + CCS configuration to be below zero. The resulting emission reduction per kg of RDF is estimated at 0.61 kg CO_2. Considering a total available RDF feed of roughly 1,700 kt per year, the total emission reduction potential of gasification combined with CCS is estimated at 1,043 kt CO_2 per year.
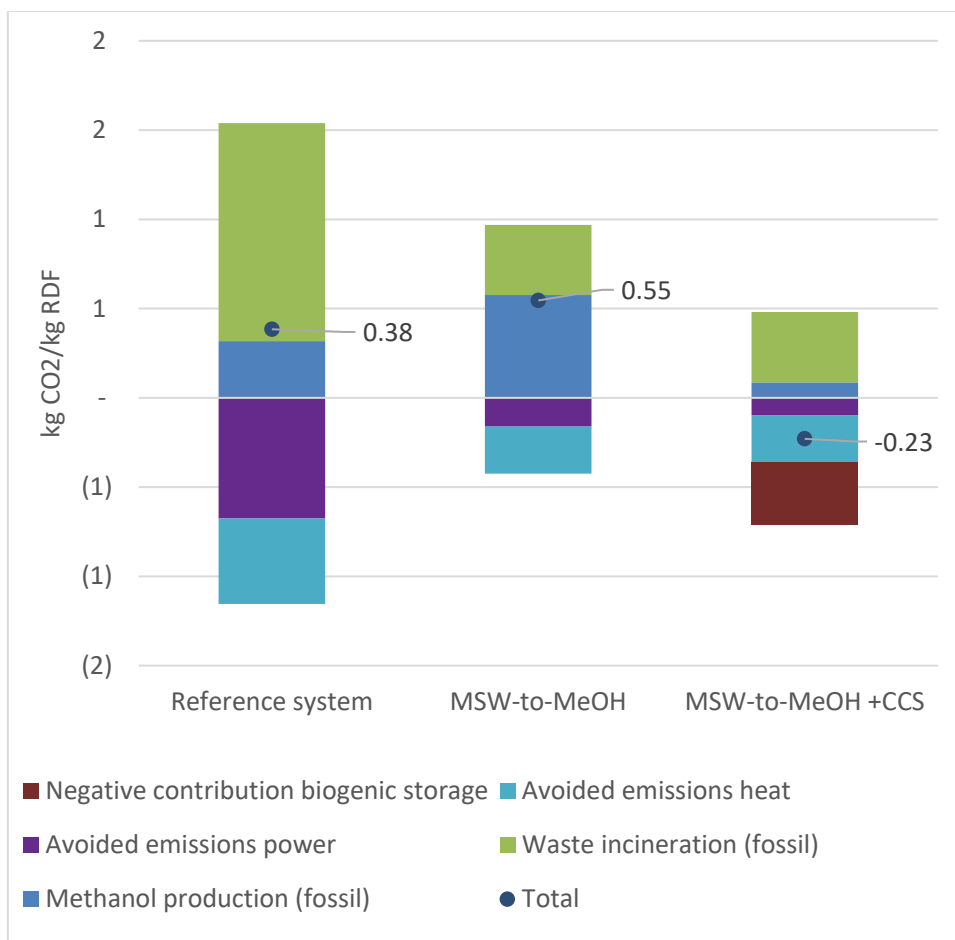
Figure 18. Net CO2 emissions associated with the reference system and the MSW-to-MeOH pathway following IPCC guidelines.

4.2.2 Municipal solid waste pyrolysis

This section discusses the use of pyrolysis as an alternative to MSW incineration. Pyrolysis is a method of thermochemical conversion of material by subjecting it to high temperatures (400-900 °C) in the absence of oxygen. The process produces three main groups of products: pyrolysis oil, gaseous products and solid char. The relative amounts of these products depend heavily on several process parameters such as the temperature, residence time and the heating rate (Chen et al., 2014). Pyrolysis processes can be applied to many feedstocks, most commonly biomass. The use of various types of waste streams has also gained attention over the past years, among which unsorted municipal solid waste.

The three most suggested applications of MSW pyrolysis products are: 1) as a chemical feedstock, 2) on-site energy generation or 3) the production of transport fuels (Buah et al., 2007; Chhabra et al., 2021; Czajczyńska et al., 2017; Wang et al., 2015). Although the chemical feedstock route is often suggested, no research could be found on the theoretical or practical testing of such a pathway.

Research on the use of MSW pyrolysis in on-site energy generation has been reviewed by Rollinson and Oladejo (2019). It was found that multiple issues are associated with this pathway. One of the main issues is the heterogeneity of the feedstock. The continuous variations in the composition and moisture content of the waste lead to high levels of uncertainty in the products of pyrolysis and the

energy requirements for the necessary processes. Additionally, the pyrolysis of unsorted MSW leads to high concentrations of toxic chemicals in the products, which may render these unusable for energy generation. Finally, Rollinson and Oladejo (2019) showed that articles often exclude energy requirements for heating and drying of the waste, energy losses in the system and auxiliary systems such as flue gas scrubbing. When the consumption related to these aspects is included, the process is shown to be a net energy consumer.

For the fuel production pathway, Wang et al. (2015) designed a theoretical pathway to create diesel, gasoline and hydrogen. The char was to be sold to coal-fired powerplants to be used as an energy source. The paper did not address the quality of the products obtained however, and research suggests that contamination in these fuels would be too extensive for them to be used in internal combustion engines (Rollinson & Oladejo, 2019). To conclude, MSW pyrolysis was found to be unsuitable as an alternative to waste incineration due to its high energy intensity, unpredictability and contamination of end-products.

4.2.3 Plastic waste gasification

Plastic gasification involves a similar procedure to MSW gasification, and its technical feasibility has been proven in experimental setups and on larger scales (Arena et al., 2011; Dogu et al., 2021). The syngas produced can be used for a number of processes. In terms of technical feasibility however, there are definite drawbacks to using plastic waste for gasification instead of more common feedstocks such as biomass or MSW. The high carbon content of the plastic waste combined with the high viscosity in its molten state and poor thermal conductivity cause excessive tar build-up in the reactor (Bai et al., 2020).

Gasification attempts to fully break down the polymers into molecules such as methane, carbon monoxide and hydrogen. Compared to plastic pyrolysis, the products of the gasification are less complex, lower in energy content and have a lower economic value (TNO, 2018). Making use of as much of the polymer bonds as possible preserves the energy content and economic value of the plastic. For these reasons, plastic pyrolysis is preferred over gasification in this research.

4.2.4 Mixed plastic pyrolysis

Plastic waste pyrolysis shows potential for producing chemical feedstocks and fuel substitutes (Al-Salem et al., 2017; Al-Salem et al., 2009). This is related to the relatively high energy content of the waste streams, combined with an often low oxygen content of the feedstock. This section covers the a techno-economic analysis of the potential for using plastic waste pyrolysis for chemical feedstock production (naphtha) and for producing a fuel substitute (heavy fuel oil, HFO).

Chemical feedstock production

There have been several operational pilot plants in the past for the production of chemical feedstocks from plastic waste through pyrolysis. Among them was a plant in Ludwighafen (Germany) and one Grangemouth (Scotland). Both were operational in the 90's and have since been decommissioned for a variety of reasons (Al-Salem et al., 2009). In the Netherlands, the so-called Pyrolyseproeftuin Zuid-Nederland was set up in 2016 and was operational until 2021 (Joppen, 2017). Several small-scale pyrolysis demos have been tested there, using both plastic waste and biomass as inputs, and producing pyrolysis oil as well as gas and solid pyrolysis products. Currently, SABIC and Plastic Energy

have started construction on a joint-venture pilot plant at the Chemelot site to pyrolyse mixed waste plastics for the production of a chemical feedstock. The plant will be operational in 2022 and process roughly 20 kt per year of plastic waste to produce TACOIL, a patented feedstock which can be upgraded and refined to produce a naphtha substitute to be used by SABIC in the production of new polymers (Chemelot website, 2021). In addition, Shell is building a 30 kt/year pilot plant at Moerdijk with a similar process (Shell website, 2021). In line with the developments in the sector the TRL of mixed plastic waste pyrolysis for chemical feedstock production is estimated at 6.

Fivga and Dimitriou (2018) performed a techno-economic assessment by modelling a plastic pyrolysis plant in the UK. The plant includes a fluidized bed reactor for pyrolysis, a hydrocracking reactor where the pyrolysis oil is upgraded, and 4 heat exchangers within the system. Additionally, a sink-float sorting system is included to sort out suitable plastics from the waste mix. Supposing a plastic mixture of 50% polyethylene (PE), 25% polypropylene (PP) and 25% polystyrene (PS), it was found that the pyrolysis oil yield was roughly 87%wt and the non-condensable gas and char fractions were able to supply more than enough energy to sustain the pyrolysis process. The thesis by Ferjan (2020) adapted the process for a Dutch mixed plastic waste stream. The resulting mass and energy balances of this process are shown in Figure 19 and Figure 20 respectively.

The 1kg of plastic waste is assumed to be similar to the standard DKR 350 mixed plastic waste stream, shown in Table 11. It contains 28.3%wt PE+PP total. A sink-float sorting system discards the other 71.7%wt of the initial mass. The process produces 0.248kg of naphtha substitute for every 0.283kg of PE+PP mix, making the yield 87%wt. The pyrolysis of the 0.283kg of plastic produces 0.132kg of direct CO_2 emissions from the combustion of char and non-condensable gases.

Table 11. Average composition of Dutch mixed plastic waste (Brouwer et al., 2018).

Mixed plastic waste	Relative mass (wt%)	LHV (MJ/kg)	Total carbon content (g/kg wet)	Fossil carbon ratio (wt% wet)
PET	18.5%	21.3	672	100%
PE+PP	28.3%	44.5	850	100%
Not identifiable	3.2%	10.1	276	21%
Laminated packaging	1.6%	29.0	672	100%
Non-packaging plastics	3.0%	29.0	672	100%
Organics	5.2%	4.0	141	0%
Paper/board	4.4%	12.4	346	1%
Metal	0.6%	1.2	0	0%
Moisture/dirt	34.3%	0	0	0%
Fractions <0.5%	0.9%	10.1	276	21%
Total	100.0%	19.0	430	

Hydrocracking of the resulting pyrolysis oil is done to remove oxygen, nitrogen and sulphur contaminations, and to break down carbon-carbon bonds. The latter is necessary since the process by Fivga and Dimitriou (2018) produces primarily hydrocarbons of C8 or longer, and over 35% is estimated to be >C18. In order to get a hydrocarbon size closer to that of naphtha (C6-12), hydrocracking is used. This process makes use of hydrogen, high pressure and high temperatures to break down longer hydrocarbons.

Figure 19 also shows that the residual waste after separation is assumed to be incinerated in a waste incineration plant. The resulting CO_2 emissions are calculated using the carbon contents from waste from (Larsen & Astrup, 2011). The carbon is assumed to be fully oxidised to CO_2, similar to the mass balance of MSW incineration in Figure 9. The final naphtha stream has 205 out of 241 grams of carbon in the PE+PP stream, making the carbon efficiency of the process 85%. Relative to the entire plastic waste stream the carbon efficiency is 48%.
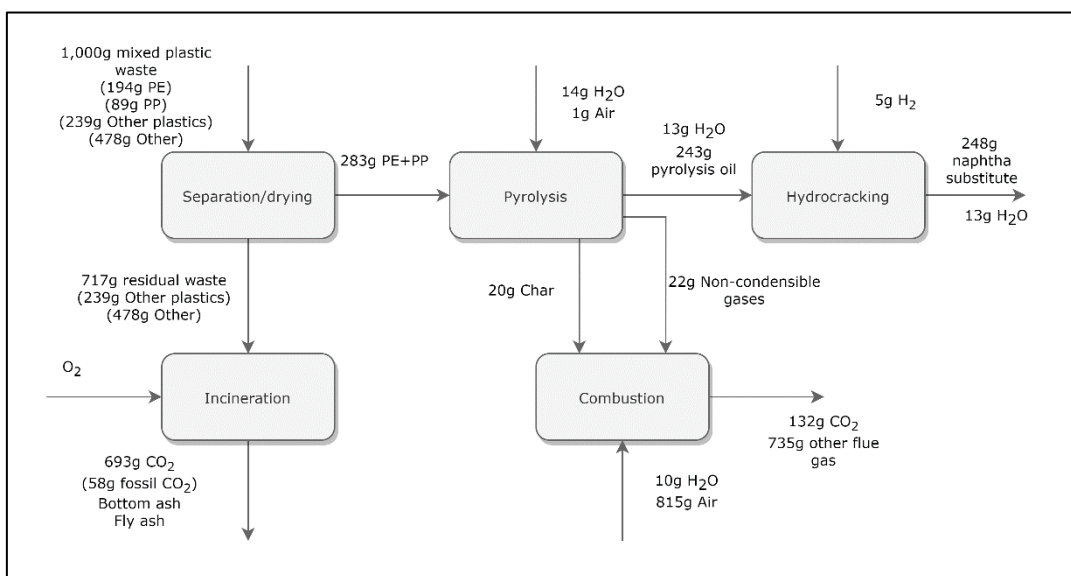
Figure 19. Mass balance based on process flows for waste plastic pyrolysis with hydrocracking (Ferjan, 2020; Fivga & Dimitriou, 2018).

Figure 20 shows the energy balance for the plastic pyrolysis system. Energy flows are based on the LHV. The energy efficiency from PE+PP input to naphtha output on LHV basis is 86% accounting for energy imports from electricity and hydrogen. The total energy efficiency from input to useful outputs (including incineration and its benefits) is 69%. The energy balance in Figure 20 shows that apart from the waste plastic input, the process needs roughly 0.1 MJ of electrical per kg plastic for drying, shredding and operating the sink-float separator. A further 0.2 MJ of electrical energy is needed per kg mixed plastic waste in the hydrocracking process. The system supplies its own heat for pyrolysis through combustion of the non-condensable gas and char. In theory the combustion of these materials has the potential for additional useful heat exports. However, due to the uncertainties in pyrolysis energy requirements highlighted by Rollinson and Oladejo (2019), the excess heat is assumed to be used as buffer energy to mediate fluctuations in the feed. The end product has an energy content of 11.5 MJ and 0.5 MJ of heat is lost in the pyrolysis processes, while 0.2 MJ is lost in the hydrocracking process, assuming that all the hydrogen is consumed. The incineration of the leftover waste stream generates useful heat and electricity with 20% and 16% efficiency respectively. The conversion efficiencies are calculated from a weighted average of all Dutch WIPs (Rijkswaterstaat, 2020).
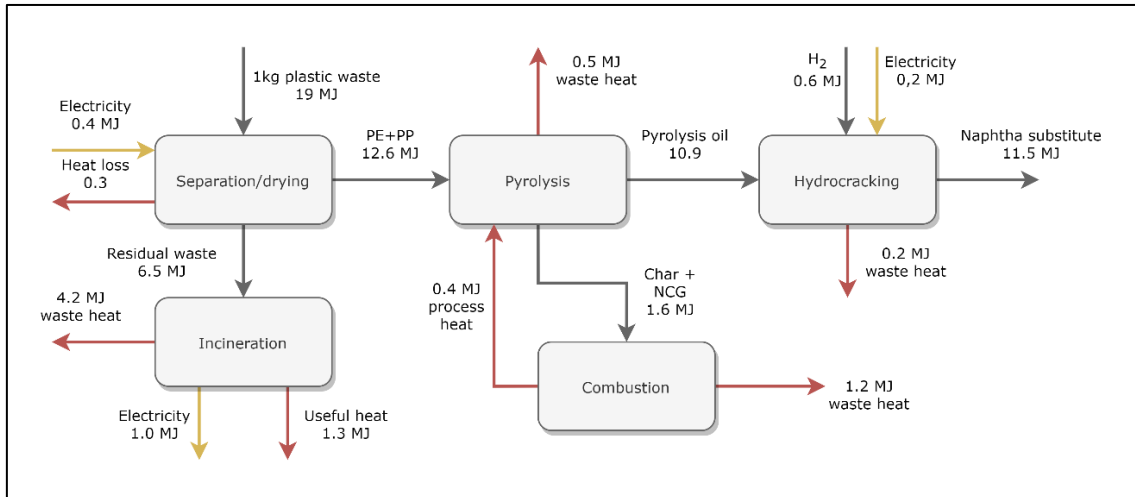
Figure 20. Energy balance for waste plastic pyrolysis for chemical feedstock substitution (Ferjan, 2020; Fivga & Dimitriou, 2018).

Table 12 shows the technical parameters for a pyrolysis system for mixed plastic waste. The standard plant size is 100 kt per year of input plastic for the pyrolysis unit. Note that for this amount of plastic feed, 3.5 times more mixed plastic waste is needed as total input, since not all plastics are suitable for the pyrolysis process, see Figure 19. Only PP and PE are sorted out. The electricity demand is based on the total plant electricity demand from Fivga and Dimitriou (2018) (0.37 MJ per kg mixed plastic feed). Here, we also include electricity for hydrocracking (0.19 MJ per kg mixed plastic feed). A plastic recycling facility in the UK called CLARITER also uses a hydrocracking process for upgrading pyrolysis oil (Akinshipe & Bird, 2016). It is assumed that a similar amount of hydrogen and electricity would be needed. Hydrogen is used on a 2.0%wt basis relative to the amount of pyrolysis oil, and is used to remove any oxygen and sulfur from this oil. The hydrogen is also used to crack the larger molecules into naphtha-like hydrocarbons.

Table 12. Technical parameters for mixed plastic waste pyrolysis for chemical feedstock substitution.

	Parameter	Unit	Figure
Input	Mixed plastic waste	kt/year	353
	Sorted PE+PP input	kt/year	100
	Electricity demand	TJ/year	198
	Hydrogen	kt/year	1.83
Output	Pyrolysis oil	kt/year	86
	Naphtha substitute	kt/year	88
	Direct CO2 emissions	kt/year	47
	Leftover waste	kt/year	253

Table 13 shows the financial data of a mixed plastic pyrolysis plant. CAPEX and OPEX figures are adopted from TNO (2018). It is likely that SABIC and Shell could make use of existing infrastructure for hydrocracking. Therefore, the associated CAPEX are shown separately. The plant CAPEX is determined to a large extent by the cost of the reactor and scrubber. Most of the annual costs are in the OPEX. TNO (2018) estimated a negative feedstock price for the mixed plastic waste of -50

€/tonne. At this price, it is deemed likely that the feedstock is heavily contaminated and additional sorting and separating is necessary. This is included in the cost calculations by TNO.

Table 13. Financial data for the mixed plastic waste pyrolysis (Fivga & Dimitriou, 2018; TNO, 2018).

Economics	Unit	10 kt/year	100 kt/year	1000 kt/year
CAPEX excl. hydrocracker	M€	13.14	65.84	329.97
CAPEX excl. hydrocracker	M€/kt naphtha/year	1.31	0.66	0.33
Fixed OPEX	M€/year	4.19	21.00	105.23
CAPEX Hydrocracker	M€	6.03	30.23	151.51

In order to calculate the net emissions for each of the pathways, the system boundaries must be defined. Figure 21 shows the Cradle-to-Gate scope for the chemical feedstock pathway for plastic pyrolysis. The dotted line indicates system boundary. Left system is the reference, right the proposed chemical feedstock pathway. On the left, crude oil is converted to naphtha through refining, whereafter it can be used as a feedstock for polymer production. The DKR 350 is assumed to be incinerated, as this is the reference process. Since it is assumed that the naphtha from the chemical feedstock pathway is an identical substitute, there are no differences between the processes conducted after naphtha production. For this reason, the system boundary ends here.
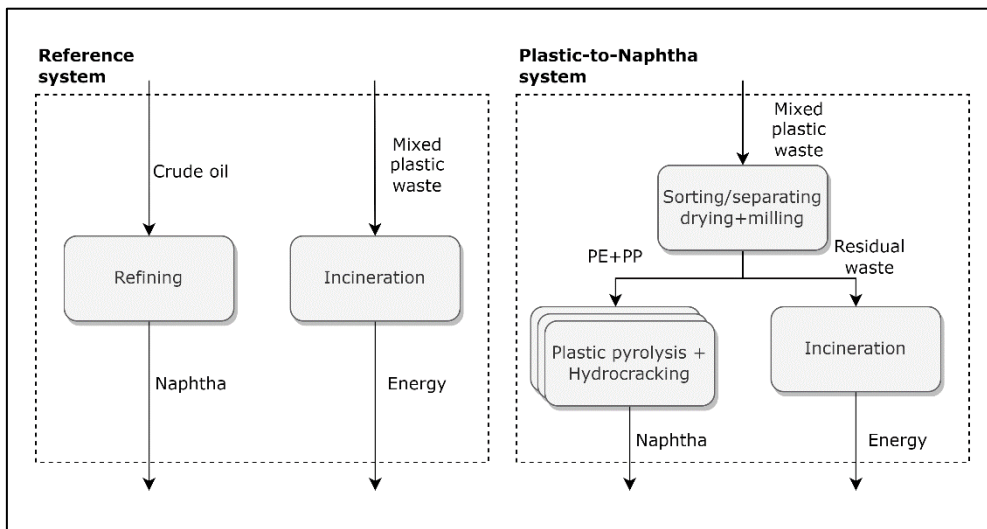
Figure 21. Cradle-to-Gate scope for comparing naphtha substitution to the reference case.

Figure 22 shows the results of the net CO_2-eq emissions calculation for the processing of 1 kg of DKR 350. The DKR 350 fraction that is not used for pyrolysis is incinerated. It is assumed that all incineration takes place in a Dutch WIP with average useful heat and electricity conversion efficiencies of 20% and 16% respectively. The emissions and energy production from incineration are calculated with the emission factors and LHVs from Table 11. Avoided emissions from electricity are 0.40 kg/kWh. For heat, the reference emission factor is a gas boiler with 90% conversion efficiency, making the emission factor 0.23 kg CO_2/kWh. Hydrogen related emissions are calculated with the average emission factor for Dutch grey hydrogen, which is 15.63 kg CO_2/kg H_2. Figure 22 shows that the emissions from incineration of waste are more than halved. This is caused by only removing 28% of the mass from the initial input waste, since this waste (PE+PP) has a relatively high carbon content. The total net emissions of the reference system are 1.07 kg CO_2/kg DKR 350 while the naphtha

production system emissions are 0.74 kg CO_2/kg DKR 350. As mentioned in Section 3.1, the total short-term potential for additional plastic recycling is estimated at 166.4 kt per year. Thus, the total emission reduction potential of this system is 54.9 kt CO_2.

Figure 22. Net CO_2-eq emissions for the reference system and the plastic-to-naphtha system for the processing of 1 kg DKR 350.

Fuel substitute production

Fivga and Dimitriou (2018) mention that the pyrolysis oil produced may also be used as a substitute for heavy fuel oil (HFO). HFO, also known as bunker fuel, is suitable mostly as an fuel for marine vessels due to its high degree of contamination. The pyrolysis oil output shown in Figure 19 can act as a direct substitute for this fuel without hydrocracking. This eliminates hydrogen demand of the process and decreases electricity demand relative to the chemical feedstock production pathway. Technical parameters of this pathway are shown in Table 14. Financial parameters from Table 13 excluding the hydrocracker apply in this case.

Table 14. Technical parameters for mixed plastic waste pyrolysis for fuel substitution (Fivga & Dimitriou, 2018).

	Parameter	Unit	Figure
Input	Mixed plastic waste	kt/year	353
	Sorted plastic input	kt/year	100
	Electricity demand	TJ/year	130
Output	Pyrolysis oil (HFO substitute)	kt/year	86
	Direct CO2 emissions	kt/year	47
	Leftover waste	kt/year	253

The Cradle-to-Grave scope for the HFO substitution pathway is shown in Figure 23, along with the reference pathway. In this case, the system boundary includes the combustion of the HFO. This is chosen because the HFO substitute (based on plastic pyrolysis) has different properties in terms of calorific value and carbon content when compared to HFO from crude oil, which can only be taken into account if HFO combustion is included in the system.
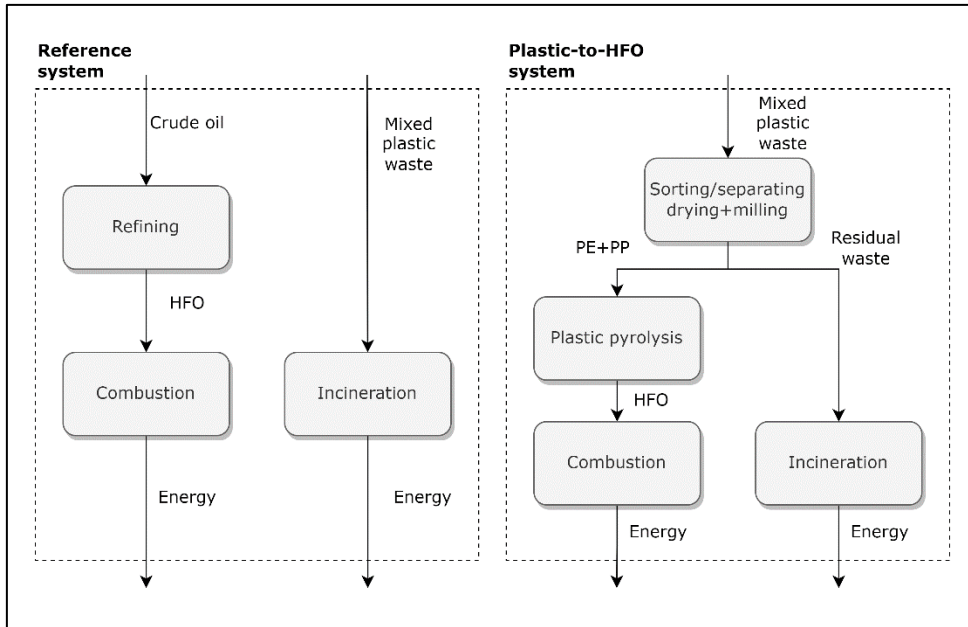
Figure 23. Cradle-to-Grave scope for the substitution of heavy fuel oil. Dotted line indicated the system boundary.

Net CO_2-eq emissions shown in Figure 24 are calculated similarly to the calculation for naphtha substitution, with the addition of the combustion step. The emission factor and energy content for crude oil based HFO are sourced from RVO (2020). Emissions from virgin HFO production are based on carbon intensity of Dutch oil refinery, which is 0.33 kg CO_2 per kg crude oil refined (Jing et al., 2020). This carbon intensity is multiplied with the amount of HFO that is substituted by the pyrolysis pathway. The total net emissions of the reference system are 1.84 kg CO_2/kg DKR 350 processed, while HFO substitution system emissions amount to 1.41 kg CO_2/kg DKR 350. At a possible feedstock of 166 kt per year of mixed plastic waste, the total reduction potential is estimated at 71 kt CO_2 per year.
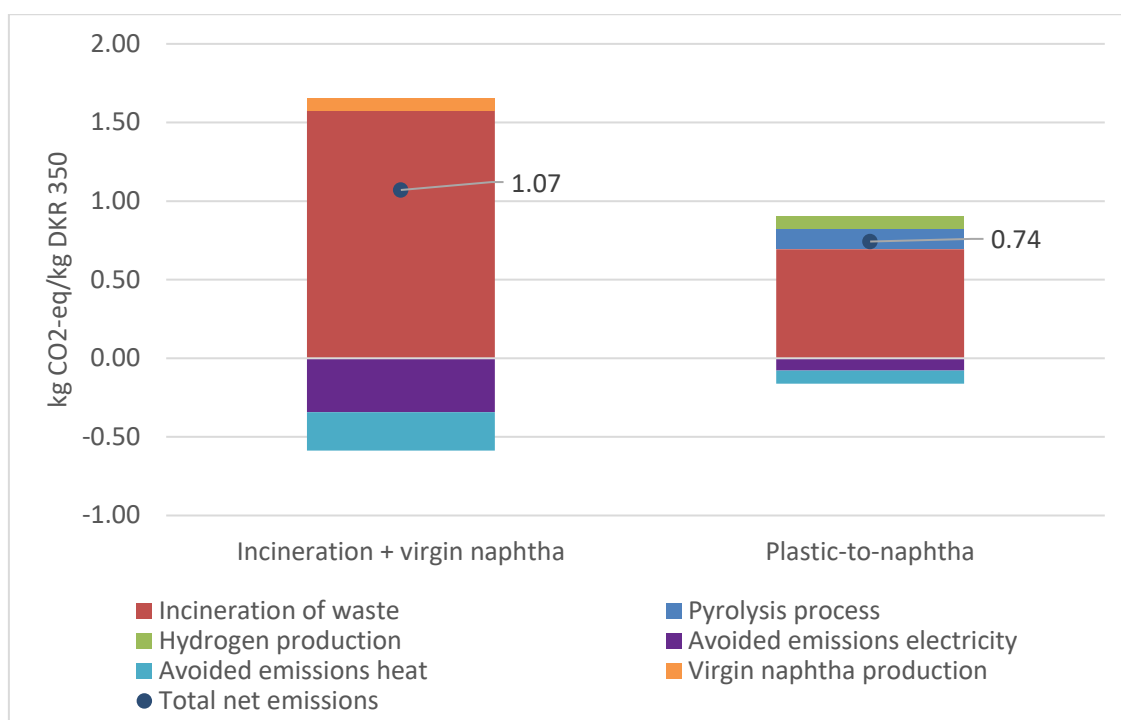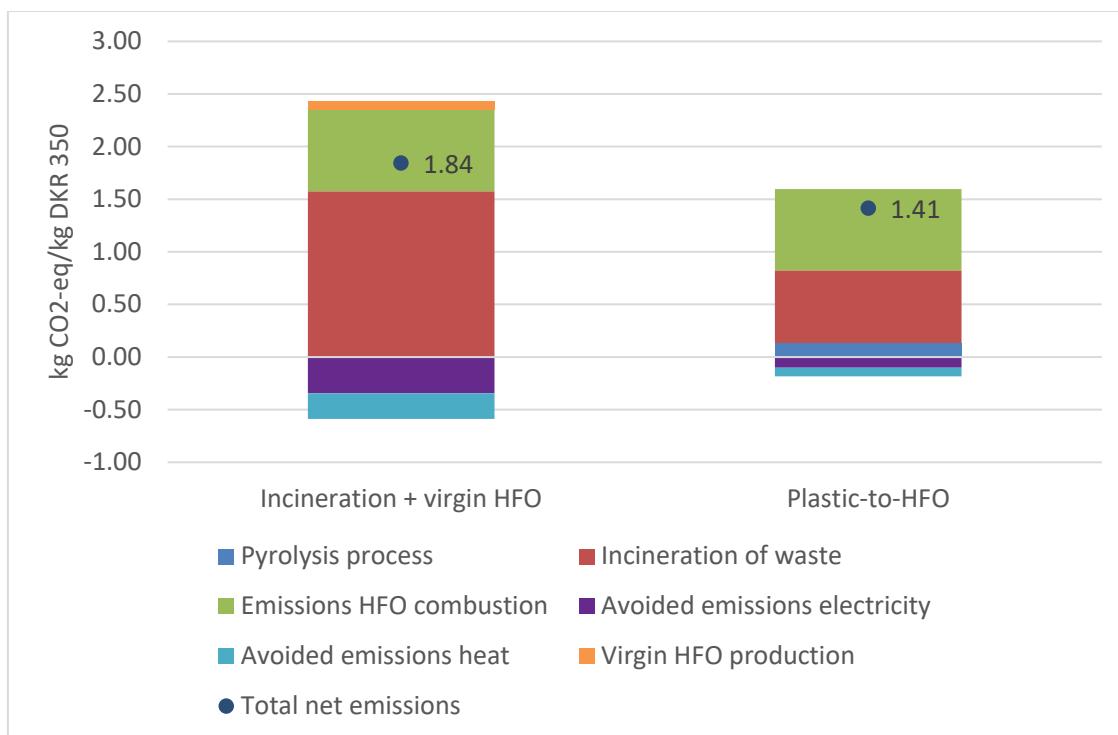
Figure 24. Net CO_2-eq emissions for the reference system and the plastic-to-HFO system for the processing of 1 kg DKR 350.

4.2.5 EPS solvolysis

EPS (expanded polystyrene) is mainly used as an insulator in construction, as well as a packaging material. It is very light weight due to the foam structure, being over 98% air in its expanded state (Demacsek et al., 2019). EPS is suitable as a construction material when it is combined with HBCD, a bromine based flame retardant. The addition of HBCD however makes the material unsuitable for mechanical recycling, which is possible for EPS without HBCD. Instead, HBCD-containing EPS can be chemically recycled by a process called solvolysis. Since 2015, HBCD is banned from use in construction. Still, it is estimated that, given the lifetime of buildings, around 6.5 kt of HBCD-containing EPS will become available annually (Broeren et al., 2019).

In the solvolysis process, the EPS is dissolved in a liquid solution specific to the EPS, so that impurities can be separated. Hereafter, a second solution is added to the reactor, turning the EPS in to a gel-like substance. The HBCD remains in the primary solution while the PS-gel is separated, dried and extruded into a product similar to virgin GPPS (general purpose polystyrene) granulate (Demacsek et al., 2019). The HBCD is separated as a sludge and can be recycled as well. This process is currently applied by PolyStyreneLoop, a Dutch company that has been operating a 3 kt feed plant since 2021. This would bring to TRL of this process to 7.

Energy demand for the process is in the form of electricity only as steam is assumed to be generated on site. Electricity demand amounts to 12.5 MJ/kg EPS input (Muller & Lensink, 2020). The process is assumed to have a material efficiency of 95%, meaning 5% of EPS material is lost per cycle on average. Process emissions were calculated by TÜV Rheinland in an LCA and were reported by Demacsek et al. (2019) to be 0.36 and o.46 kg CO_2-eq per kg EPS input for pre-treatment and other processes respectively.

Reference system

Since the general purpose polystyrene (GPPS) produced by the PolyStyreneLoop is found to have similar properties to virgin material, it is assumed that the product from EPS solvolysis replaces virgin GPPS in EPS production. This means the emissions resulting from the process of expanding GPPS to EPS must be added to the process emissions for the PolyStyreneLoop system. The current standard method is to produce EPS from polymerisation of the monomer styrene, which is in turn produced from dehydrogenation of ethylbenzene using pentane. Ethylbenzene is a product of benzene and ethylene. All of the aforementioned reactants are produced from crude oil refining. An LCA performed by PlasticsEurope (2015) estimated the process emissions of EPS synthesis at 2.37 kg CO_2-eq per kg EPS. This excludes end-of-life treatment. Since HBCD-containing EPS is currently not suitable for mechanical recycling, it is assumed that it is incinerated after the use phase. EPS has an LHV of 38 MJ/kg, and is assumed to be incinerated in an average Dutch WIP with average electrical and (useful) thermal efficiencies of 16% and 20%, respectively. The scope of the net emission calculation showing this reference system and the PolyStyreneLoop system is shown in Figure 25.
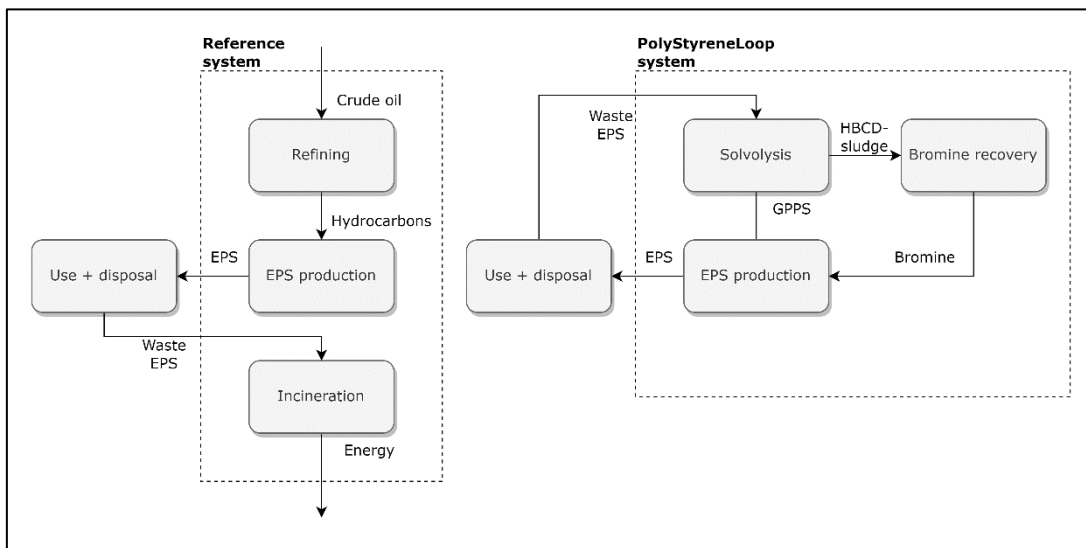
Figure 25. Cradle-to-grave scope for the reference and PolyStyreneLoop systems for calculating net emissions. PolyStyreneLoop system does not show material losses of 9%. Use and disposal phases not included in scope.

Figure 26 shows the estimated net emissions of the systems described above. Negative emissions are applied to the reference scenario to account for the electricity and useful heat that are generated in the WIP. Process emissions for the reference system are the result of oxidation of carbon in the waste EPS to carbon dioxide. Since the material efficiency of the PolyStyreneLoop process is 95%, the virgin product emissions in the reference system are 95% of the emissions associated with the production of 1 kg virgin EPS. Electricity is assumed to come from the grid, and an emission factor of 0.40 kg CO_2/kWh is used. For heat, we assume a 90% efficient natural gas boiler with an emission factor of 0.23 kg CO_2/kWh. Figure 26 also shows direct process emissions for the solvolysis process of 0.45 kg CO_2/kg EPS input. These emissions are related to the incineration of residual products that are separated from the EPS upon dissolving in the solvent (Demacsek et al., 2019). Finally, the emissions for expanding GPPS to EPS were estimated. AFPR (2009) note that 31% of virgin product emissions are related to expansion of PS. The net emissions for the solvolysis pathway are 1.49 kg CO_2-eq lower for each kg of EPS waste processed. This can also be expressed as a 34% decrease in emissions.

Following these findings, the total potential GHG emission reduction that can be achieved by full utilisation of the available EPS waste stream in the Netherlands can be estimated. At a potential HBCD-containing EPS waste stream of 6.5 kt per year, the annual net CO_2-eq savings would amount to 9.7 kt.
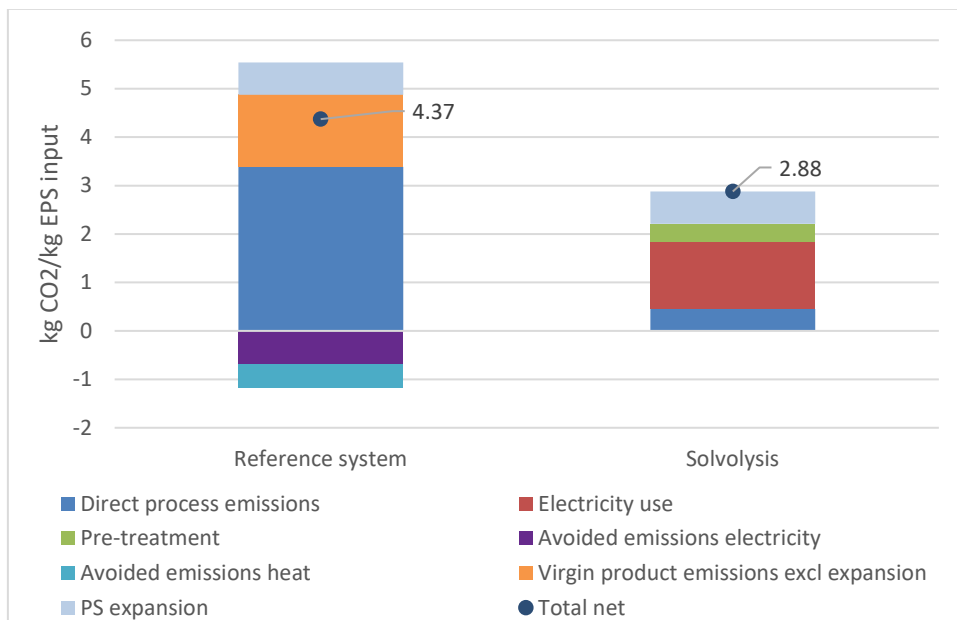
Figure 26. Net CO_2-eq emissions for the reference and solvolysis pathways for processing 1kg of EPS waste.

The largest factor in annual expenses for the EPS plant is in the OPEX, which are dominated by energy use for distillation and drying processes. Although process costs are fairly high (~800 €/tonne EPS waste feed), the high value of the product (~1700 €/tonne GPPS) allows the process to likely be profitable (TNO, 2018). Scale is also found to be an important factor in total cost per tonne GPPS. The breakdown for three different plant scales is shown in Table 14.

Table 15. Economic parameters for a EPS solvolysis plant of 5, 20 and 50 kt annual GPPS output.(TNO, 2018)

Parameter	Unit	Scale 1	Scale 2	Scale 3
Output	kt GPPS	5.00	20.00	50.00
CAPEX	M€	9.89	26.10	49.57
CAPEX/kt output/year	M€/kt/year	1.98	1.31	0.99
OPEX incl. electricity	M€/year	3.03	12.1	30.25

4.2.6 PET depolymerisation

Over 99% of Dutch PET bottles collected through the deposit system are mechanically recycled (Snijder & Nusselder, 2019). This monostream can be characterised by mostly clear, clean and non-coloured bottles of similar composition. This is not the case for all PET waste. PET trays are often used in food packaging. The trays are layered with different plastics (often LDPE films) and are more likely to be contaminated with other wastes, whether they are collected in PMD-collection systems or separated from residual waste. This makes PET trays difficult to mechanically recycle for multiple

reasons. The primary complication for mechanical PET recycling is the presence of various additives and other polymers in the trays. Another reason is the small particles that form during the grinding process, which disrupt machine operation (TNO, 2018).

The total amount of PET trays in the Netherlands is estimated at 65 kt per year. About 30 kt are separated into a monostream at recycling facilities (Broeren et al., 2019; Lensink & Schoots, 2020). Recycling of PET trays is currently limited to around 5 kt per year and is done by mixing in with PET bottle mechanical recycling, or by blending with the DKR 350 mixed plastic stream (DKR 350 has a limit to how much PET it can contain). The remaining 25 kt of the monostream are stockpiled by recycling companies and eventually incinerated in WIPs (KIDV, 2016).

An alternative to mechanical recycling or incineration is PET depolymerisation. In this method, the PET is dissolved in glycol, after which contaminations are removed and a crystalline monomer BHET is extracted. This material can be used to synthesise virgin-grade PET. The Dutch company Ioniqa Technologies has developed a PET depolymerisation process that is also capable of handling coloured PET feedstock while producing clear PET pellets of virgin quality. The company is operating its first pilot-scale plant, processing up to 10 kt of PET feed per year. Ioniqa plans to expand this to a 50 kt plant in the future (Snijder & Nusselder, 2019).

Reference system

TNO (2018) states that PET from magnetic depolymerisation has similar characteristics to virgin PET. It is therefore assumed that the PET produced by depolymerisation replaces virgin PET. Virgin PET $(C_{10}H_8O_4)_n$ is produced by of esterification of terephthalic acid and ethylene glycol, both of which are crude oil-based. The process uses energy, mostly in the form of combusted natural gas, and has a carbon footprint of around 2.5 kg CO_2-eq per kg PET (Lindgreen & Bergsma, 2018). In the reference system (the current situation), PET trays are stockpiled and later incinerated. It is assumed that stockpiles are unsustainable and all PET trays that are not mechanically recycled will at some point be incinerated. This incineration is done in an average Dutch WIP with an electrical and useful thermal efficiency of 16% and 20% respectively. Figure 27 shows the scope used for comparing the reference system with the depolymerisation pathway.
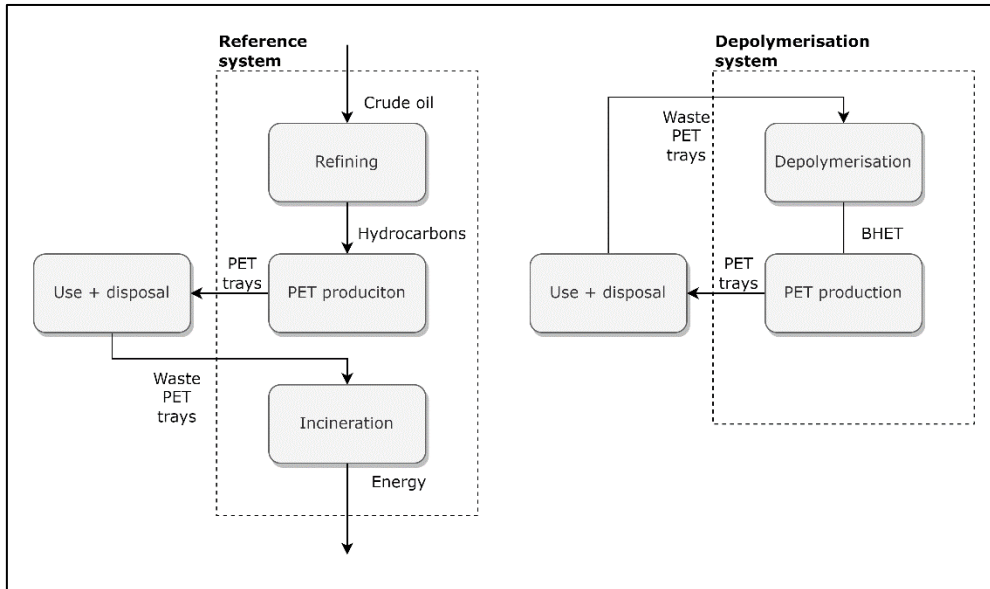
Figure 27. Cradle-to-grave scope for the reference and depolymerization pathways. Depolymerisation figure does not show the 5% material loss which occurs during the recycling process. Use and disposal phase is outside the scope.

The net CO_2-eq emissions for the two pathways for PET tray disposal are shown in Figure 28. Avoided emissions from electricity and heat are elaborated above. Incineration emissions are estimated at 2.3 kg CO_2-eq per kg of PET incinerated following Chilton et al. (2010). Virgin product emissions are multiplied with the 95% material efficiency for depolymerisation, since this amount is replaced by the alternative process. Process emissions excluding energy-related emissions are estimated by subtracting these energy-related emissions from the total emissions of the depolymerisation process found by Broeren et al. (2019). The total emissions range from 1.0-1.3 kg CO_2-eq/kg output PET depending in the size (and efficiency) of the plant. In Figure 28, the average of these two values is taken and multiplied by the material efficiency to find 1.09 kg CO_2-eq/kg input PET waste. After subtracting energy-related emissions, 0.41 kg CO_2-eq/kg input PET waste is left as process emissions. Electricity is assumed to come from the grid, and an emission factor of 0.40 kg CO_2/kWh is used. For heat, we assume a 90% efficient natural gas boiler with an emission factor of 0.23 kg CO_2/kWh.

The total potential emission reduction from PET depolymerisation depends in part on how effectively the PET trays that end up in residual waste are separated in the future. The currently available stream is limited to 25 kt/year of waste PET trays, making the total GHG emission reduction potential 73.3-95.2 kt CO_2-eq per year. At a (theoretical) 100% effective separation of PET trays from residual waste, the total waste stream of 60 kt would lead to 175.8-228.5 kt CO_2-eq emissions reduction.
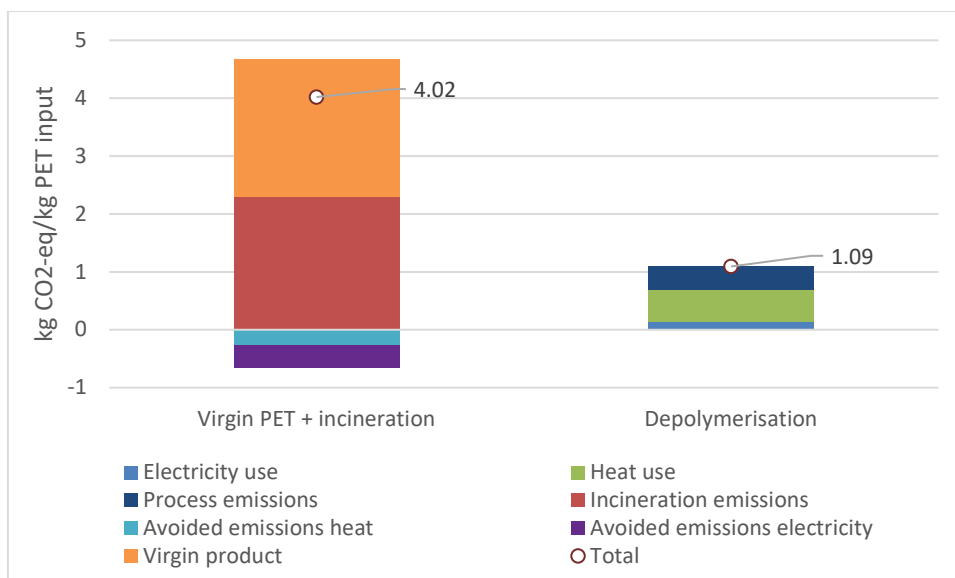
Figure 28. Net CO_2-eq emissions from the reference and depolymerisation pathways for processing 1kg of waste PET trays.

An analysis of the economic performance of PET depolymerisation was done by TNO (2018). The base scale for the plant is 20 kt per year output of recycled PET. Economic parameters are shown in Table 16. TNO further assume that the recycled PET can be sold for ~€100 per tonne.

Table 16. Economic parameters for a PET depolymerisation plant of 5, 20 and 50 kt annual recycled PET output (TNO, 2018).

Parameter	Unit	Scale 1	Scale 2	Scale 3
Output	kt	5.00	20.00	50.00
CAPEX	M€	7.09	18.70	35.51
CAPEX/kt output/year	M€/kt/year	1.42	0.94	0.71
OPEX incl. electricity	M€/year	2.80	11.20	28.00

4.3 CO_2 avoidance cost results

The analysis above concludes the comparative life cycle assessments of the technologies. Excluding low-TRL technologies and technologies that did not cause net emission reduction, the analysis of the costs continues for MSW gasification, plastic pyrolysis (HFO and naphtha), PET depolymerisation and EPS solvolysis. Using the methods described in Section 2.4, the CO_2 avoidance cost was estimated for the technologies described above. A summary of input parameters is given in Table 17. Plant sizes are chosen in accordance with the standard plant capacities used by TNO (2018).

Table 17. Parameters for CO2 avoidance cost calculation (TNO, 2018).

Parameters	Plant size	Feedstock cost	Product sales price
Unit	kt output/year	€/tonne	€/tonne
Gasification	100	0	250
Pyrolysis Naphtha	30	-50	500
Pyrolysis HFO	30	-50	550
PET Depolym.	20	100	960
EPS Solvolysis	20	50	1720

Figure 29 shows the results of the CO_2 avoidance cost analysis. The emissions for technologies that deal with feedstocks that contain biogenic and fossil carbon were accounted for using IPCC guidelines. Gasification without CCS has no avoided CO_2 if biogenic emissions are not counted, as is shown in Figure 18 in Section 4.2.1. As a result, CO_2 avoidance costs are inverted and shown as negative here, and are shown with a hatched pattern. Gasification with CCS has a slightly higher cost of CO_2 avoided due to the biogenic emissions not being included. This effect is smaller however, because biogenic CO_2 stored is counted as a negative emissions following IPCC guidelines, as is shown in Figure 18. All plastic recycling options show a negative cost of CO_2 avoided. This can be interpreted as positive business case. For PET depolymerisation and EPS solvolysis this is in accordance with the negative subsidy level calculated for the SDE++ 2021 (Lensink & Schoots, 2020). The main reason is the high sales prices of the products. CO_2 avoidance costs for CCS and CCU have previously been discussed in Section 4.1. Table 18 shows the CO_2 avoidance cost calculation in more detail.
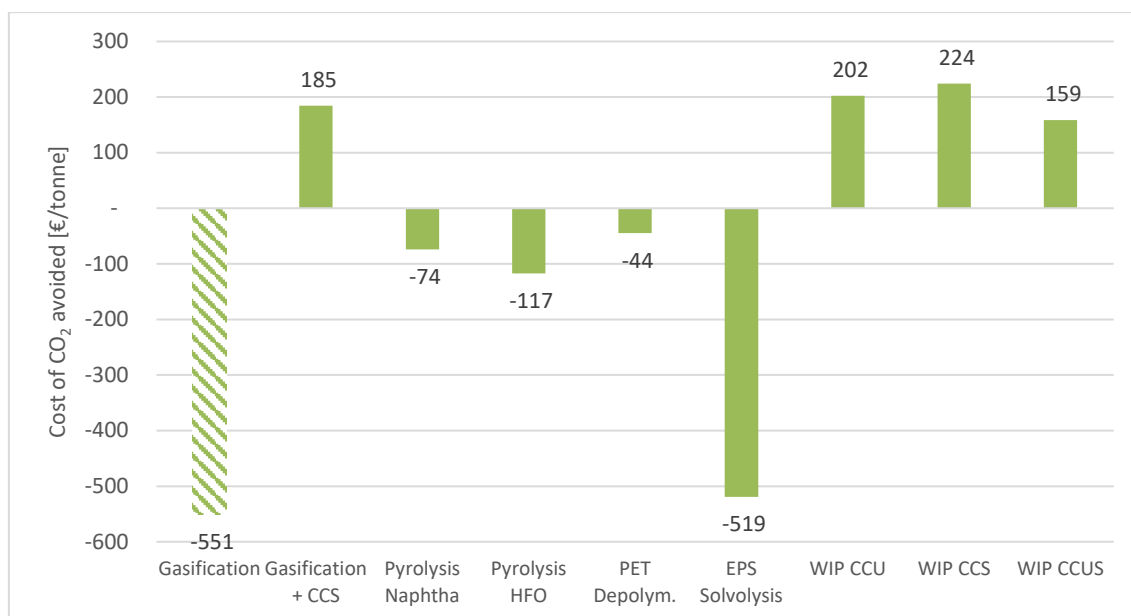

Figure 29. CO_2 avoidance cost for emission reduction technologies following IPCC guidelines. Hatched pattern indicates negative CO_2 avoidance with positive avoidance cost.

Table 18. Calculation data for cost of CO_2 avoided for each technology. All values are shows as €/tonne CO_2 avoided.

	AIC	Feedstock cost	OPEX	Revenues	LOI	CO_2 avoidance cost
Gasification	-473	-	-465	649	-262	-551
Gasification + CCS	144	-	140	-171	72	185
Pyrolysis Naphtha	100	-153	240	-378	116	-74
Pyrolysis HFO	52	-117	171	-312	89	-117
PET Depolym.	29	34	182	-312	22	-44
EPS Solvolysis	80	33	385	-1,094	77	-519
WIP CCU	481	-	364	-292	-	202
WIP CCS	195	-	425	-	-	224
WIP CCUS	228	-	359	-138	-	159

4.4 Uncertainty analysis

The uncertainty analysis was done using the Monte Carlo method described in Section 2.5. The first analysis was conducted on the net emission reductions per unit input found in the comparative life cycle assessments of the enhanced recycling technologies. Parameter, range and distribution selection was discussed in Section 2.5. The results are shown in Figure 30. Error bars indicate 90% confidence intervals and the green bars indicate the median values computed by @RISK 8.0.

The 90% confidence interval for gasification without CCS included the possibility for positive emission reduction, with range -0.3-0.1 kg CO_2 reduced per kg RDF processed. Uncertainty was found to be most sensitive to the emission factor of virgin product production (methanol synthesis). The second and third most sensitive parameters were the grid electricity emission factor and the emission factor of heat. Gasification with CCS was found to have a 90% confidence interval between 0.7-1.1 kg CO_2 reduction / kg RDF. Sensitive parameters were identical to gasification without CCS, being virgin product production, electricity emission factors.

For both pyrolysis systems, as well as the PET depolymerisation system, the most sensitive parameter was the emission factor of heat. PET depolymerisation net emission reduction was also highly sensitive to virgin product emissions. Pyrolysis net emission uncertainty was largely caused by the grid electricity emission factor. EPS solvolysis emission reduction was highly sensitive to the grid electricity emission factor as well, since the process is assumed to generate heat on-site with electric furnaces/boilers.

Uncertainty for net emission reduction of WIP CCS were not calculated separately since this parameter only depended on the variations in the CO_2 capture rate. This rate was varied with a triangular PDF from 80-90% with a preferred value of 85%.

Material efficiency distribution for all processes ranged from a lower bound of 20% below the LCA values up to the values found in the LCA. The material efficiencies presented in the LCA sections did not include material losses that were not inherent to the physical processes (such as rejects due to

contamination or process errors). For this reason, the material efficiencies found in the literature research and presented in LCA were assumed to be the maximum material efficiencies. Due to this relatively small variation, material efficiency was only a minor contributor to total net emission reduction in all technologies.
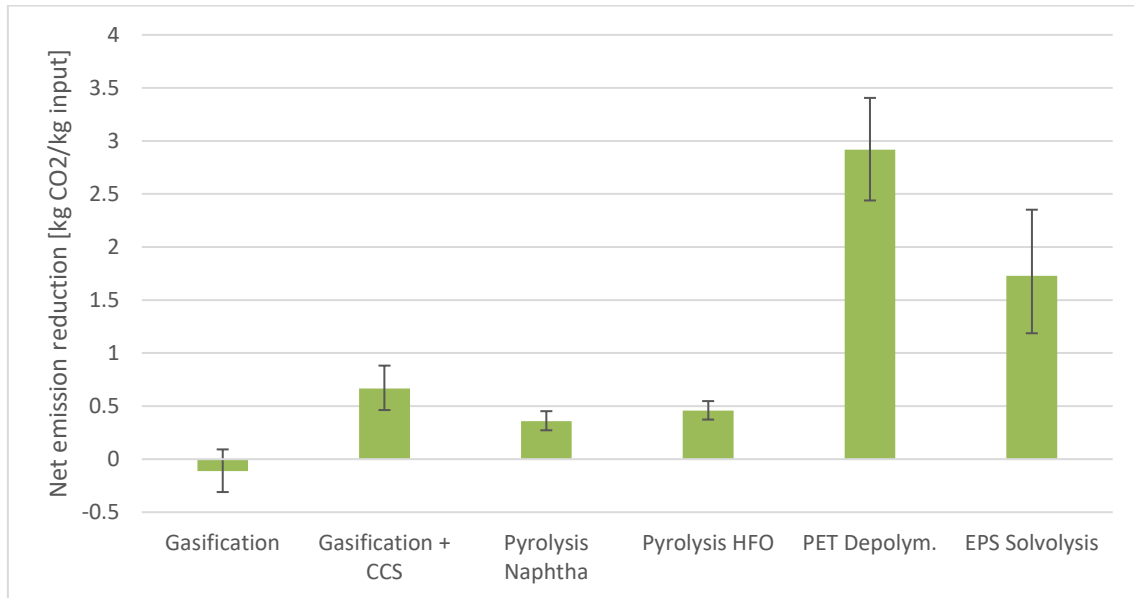
Figure 30. Monte Carlo analysis of net CO_2 emission reduction factors per kg input for enhanced recycling technologies including 90% confidence interval.

The uncertainties calculated above cascade into uncertainties for the CO_2 avoidance cost, which are presented in Figure 31. Gasification without CCS yielded negative CO_2 reduction, inverting the results. Gasification with CCS shows large upwards uncertainty, indicating a high probability of large CO_2 reduction costs. The most sensitive parameters were found to be the gasification plant size, the resulting CCS equipment capacity and the equipment lifetime.

Pyrolysis error margin indicated that the uncertainty is very large relative to this median. For PET depolymerisation and EPS solvolysis, the uncertainty in CO_2 avoidance cost is smaller than for pyrolysis. Additionally, for solvolysis and depolymerisation, the full range of the 90% confidence interval does not include a positive CO_2 avoidance cost.

In terms of the relative sensitivity of parameters, pyrolysis was most heavily affected by changes in plant size, feedstock cost and product price, the former being the most sensitive. PET depolymerisation was affected by the same parameters, although the most sensitive parameter was found to be the product price. Solvolysis sensitivity parameters were, from most to least sensitive, the feedstock cost, product price and grid electricity emission factor.

90% confidence interval for the WIP CCS options was relatively small, indicating relative certainty to the CO_2 avoidance cost found. Uncertainty in the CO_2 avoidance cost depended most heavily on plant size, the emission factor for heat and the equipment lifetime.
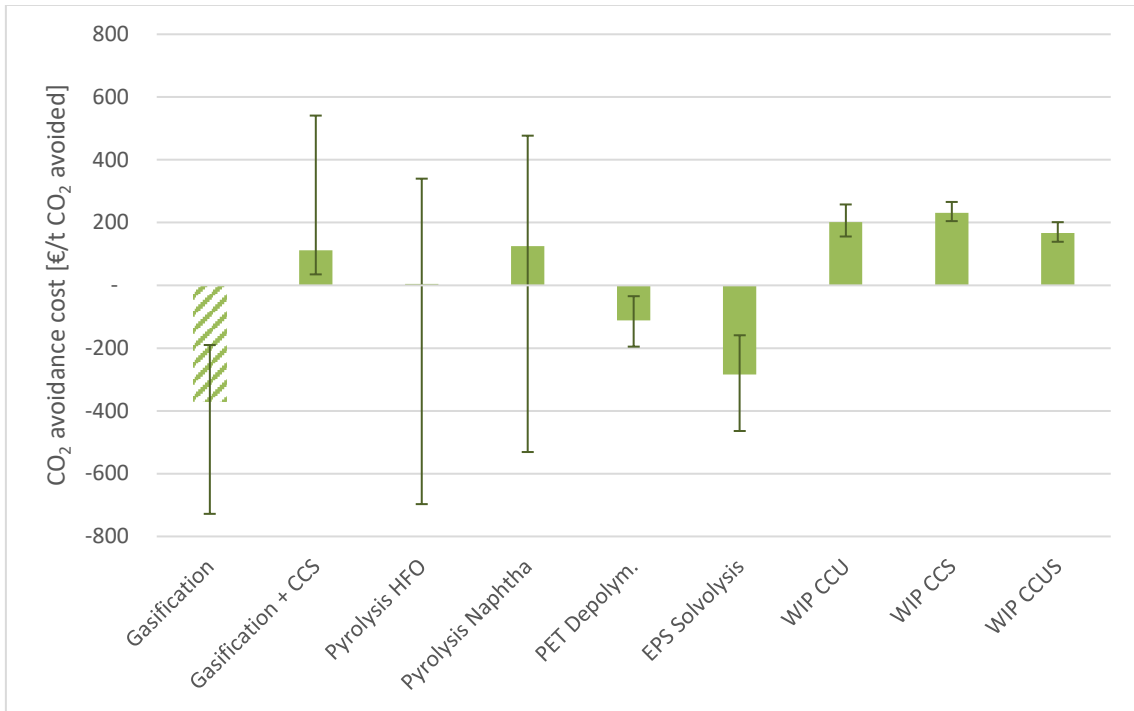
Figure 31. Monte Carlo analysis of CO_2 avoidance cost for emission reduction technologies with 90% confidence interval. Hatched pattern indicates negative CO_2 avoidance with positive avoidance cost.

5 Discussion

Section 4 outlines the selected technologies that were identified by this research relating to their possible contribution to reducing the GHG emissions from the currently incinerated waste stream. The emission reduction potential was assessed in terms of the total potential feed, the net reduction per unit input, the technology readiness level and the financial implications of application of the technology. This section will discuss the limitations imposed on the research by the methods that were applied. Next, the results are discussed, as well as qualitative aspects of the technologies that were selected. Finally, recommendations for further research are discussed.

5.1 Limitations of the research

The primary limitation of the research related to the scope that was set. Reducing the GHG emissions related to the incineration of waste is fundamentally different from many other industrial sectors. The waste management industry does not need to produce certain products, but merely offers the service of waste disposal at low cost and minimal environmental impact. The rise of "R-strategies" (reduce, reuse, repair, remanufacture, recycle, recover) in government policy have placed waste incineration (recover) at the very bottom of the hierarchy of circular economics (PBL, 2021). This report has looked at the recovery and recycling strategies, but to minimise emissions in the entire life cycle of products, efforts in the other R-strategies are also needed.

Furthermore, the calculation of net emissions was done by incorporating avoided emissions from heat and electricity production at the waste incineration plant. The height of these avoided emissions relate to the emission factors for heat and electricity over the period 2019-2030. As a result, the development of the reference emission factors after 2030 was not included in the analysis. Additionally, developments in the emission factors for the production of virgin products after 2030 were also not taken into account in the research.

To assess the effect that the change in reference emission factors may have on the net emission reduction, the contribution of these factors are removed from the calculation. The underlying assumption is that energy generation and virgin product production are net-zero processes by 2050. The effect of this assumption is shown in Figure 32. Under these assumptions, the net emission reduction compared to the reference is expected to increase in case of pyrolysis, making it more favourable compared to the reference method of incineration and virgin material production. This is related to the large amount of energy extracted from the plastic in the reference scenario for heat and power production, which no longer contributions to the emission reduction in 2050 if the alternative method of heat and power production would have zero emissions. The emission reduction for PET and EPS recycling heavily depend on the emissions from the production of the virgin product in the reference case. When the emissions of virgin production are set to zero, the relative reduction compared to the reference case is lower. For MSW gasification, a similar effect as in the case of plastic pyrolysis occurs:, when the avoided emissions from heat and electricity no longer factor in for the reference scenario, MSW gasification becomes more attractive.

For CCU at WIPs, the 2050 scenario shows net-zero emissions, since there is no longer any form of compensation from CO_2 delivery to greenhouses. This effect is also seen for CCUS for WIPs. CCS

emission reduction rises, since the penalty for the consumption of electricity and heat no longer applies.
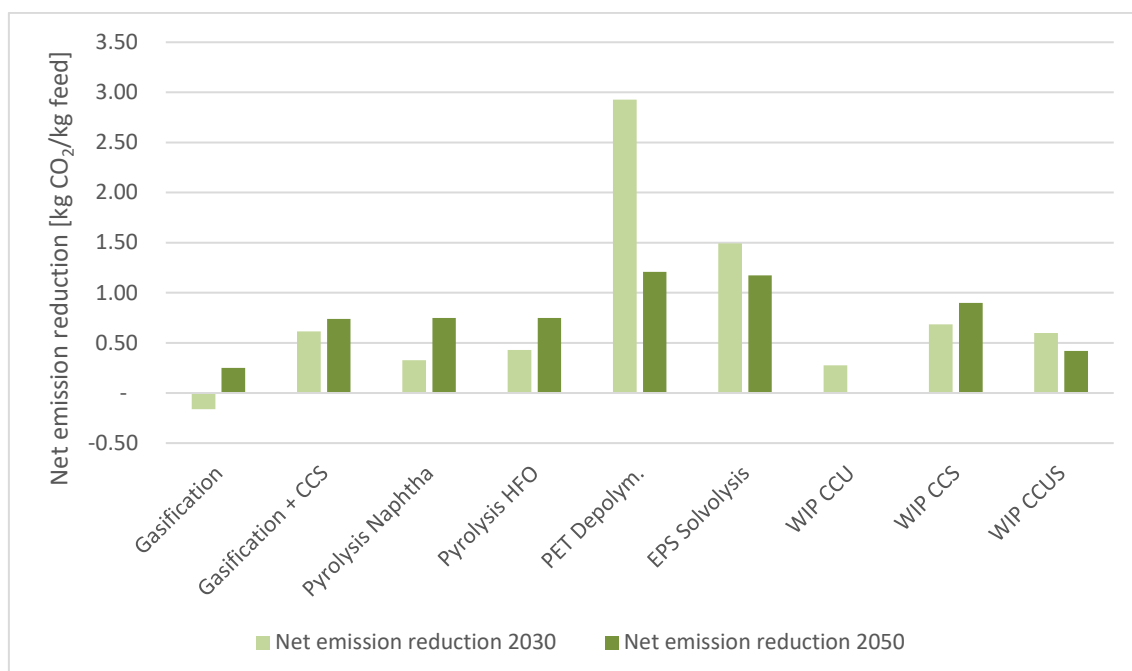
Figure 32. Net emission reduction per kg input for emission reduction options following assumptions of a 2050 scenario.

As a means of identifying sensitive parameters and quantifying the confidence interval of the results obtained by the emissions- and financial analysis, the Monte Carlo analysis was performed. This method is, relative to the practice of error propagation, seen as a robust assessment of uncertainty. However, McMurray et al. (2017) note that due to the large number of simulations conducted, uncertainty tends to converge to the median. This convergence may lead to underestimation of the actual uncertainty. It should therefore be concluded that the confidence interval presented in Section 4.4 may underestimate the true uncertainty given the input parameters. Reducing the number of simulations may alleviate this underestimation, but causes problems as the lack of convergence in the resulting PDF leads to high inaccuracies in the median and confidence interval results.

Additionally, the ranges and input parameters chosen for the PDFs in the Monte Carlo analysis may not truly reflect the uncertainty that certain parameters are subject to. A good example for this is the electricity and gas prices. In light of the recent gas-crisis in Europe, EPEX day-ahead spot prices have soared to ten times their annual average value over the last 3 years (EPEXSpot, 2022). Fluctuations of this magnitude are not included in the uncertainty range of the energy prices, and may indicate that the uncertainty range does not reflect the heavy-tailed nature of these distributions.

5.2   Quantitative discussion of results

The Monte Carlo analysis presented in Section 4.4 indicated that there was some uncertainty in the net emission reduction as a function of the input mass. For all enhanced recycling technologies with the exception of gasification, the resulting confidence interval indicated that results were robust,

and net emissions are expected to substantially decrease if these technologies are implemented. For the financial analysis however, results of the Monte Carlo indicated that uncertainty was too large to draw any conclusions from the data as to the avoidance cost of CO_2 given the implementation of these technologies. Despite the large uncertainties, some conclusions can still be drawn from the financial analysis. For EPS solvolysis and PET depolymerisation, the entire 90% confidence interval did not include a positive CO_2 avoidance cost. This implies a very small likelihood that these two technologies can be implemented with a negative overall financial impact, given the scope and parameters chosen.

GHG emission reduction potential for gasification is highly uncertain yet also close to zero when compared to the incineration case. Pyrolysis, EPS solvolysis and PET depolymerisation were found to result in the highest net CO_2 reductions per kg input material. The potential annual feed of mixed plastics is estimated at 166 kt, for EPS 6 kt is available annually and PET trays are estimated at 25 kt (Broeren et al., 2019; KIDV, 2016; Snijder & Nusselder, 2019). For PET, this does not include trays currently incinerated as part of residual waste. Th pyrolysis-to-HFO system was found to result in higher net emission reduction than the Naphtha route. When all recycling technologies and their effects on net emissions are combined, the total potential fossil CO_2 reduction given the feedstocks described above is 154 kt CO_2 per year. This is small in comparison to the total fossil CO_2 emission of 2,745 kt per year, at an estimated reduction of around 6%. The only exception is applying CCS on the gasification pathway. At a total feed of 1,700 kt of RDF, this can result in 1,043 kt CO_2 reduction annually.

In terms of the reduction potential of carbon capture, Section 4.1 showed that total reduction was highest for a 100% CCS configuration. However, due to the much higher costs, a CCUS hybrid system is more likely to be implemented on the short term. The incentives at play for waste incinerators between CCU and CCS are detailed further on, in Section 5.3. The total emission reduction potential for the technologies is found by multiplying the total available feed with the CO_2 avoidance rate of the technology. Results are displayed in Figure 33, which show the limited potential of enhanced recycling based on the research conducted when compared to carbon capture technologies.
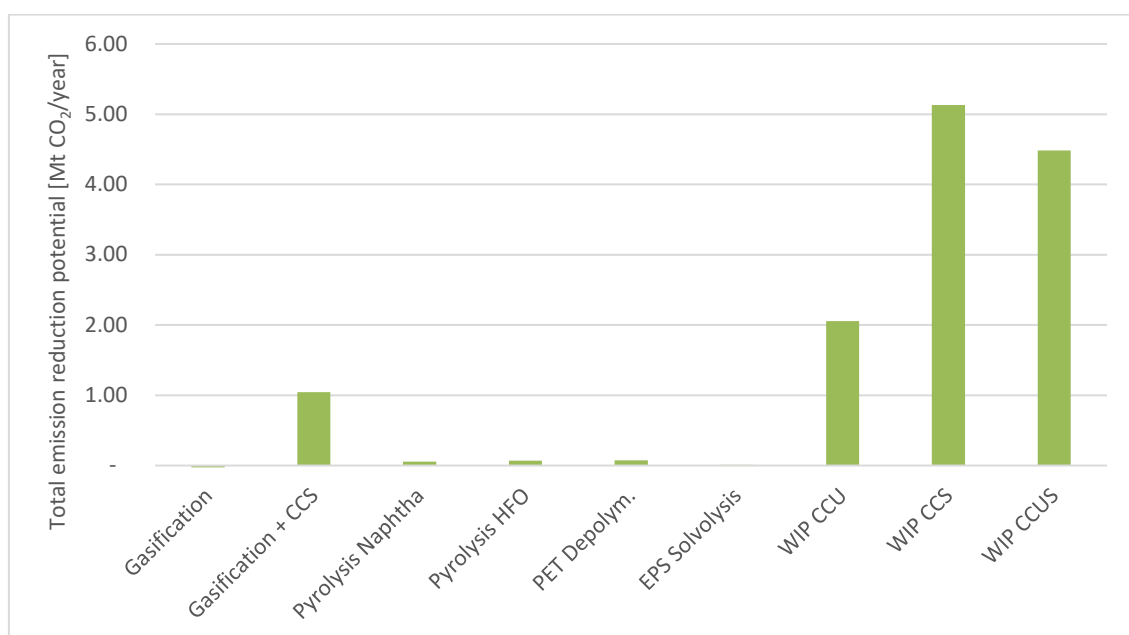

Figure 33. Total potential annual CO_2 emission reduction of emission reduction technologies.

5.3 Qualitative discussion of decarbonisation options

This section discusses some qualitative aspects related to the reduction of emissions associated with incinerating waste. The discussion is non-exhaustive and only encompasses point which are, in the authors opinion, vital to include in this research to avoid over-simplification of the issues with must be resolved in the waste management industry.

5.3.1 Waste incineration in the Dutch context

Dutch waste incinerators belong to the most efficient in the world, in terms of energy extraction from waste. This high efficiency translates to more avoided emissions in the context of the current energy system. The reason is that the electricity and heat generated from waste incineration would have been generated by other means if not generated by waste incinerators, and would have resulted in additional GHG emissions. The high efficiency causes emission reduction options that are relevant in other countries to be less relevant in the Dutch context. Every alternative technology or pathway for waste management must compete with this high efficiency of waste incineration. The result is that alternative waste management methods that make sense from a circular economy perspective, for example because they reduce the reliance on fossil resources or increase material efficiency, do not necessarily reduce the total emissions of the system. Although the net GHG emissions are the focus of this report, we want to emphasise that emission reduction is only one dimension of the transition that the waste management industry has to go through, given the various policy ambitions.

5.3.2 Carbon capture technology policy

The most direct and effective method of GHG emission reduction for waste incineration is carbon capture and subsequent storage of utilisation of the captured CO_2. Currently in the Netherlands, several WIPs have carbon capture installations that capture part of their emissions. The captured CO_2 is utilised and no CO_2 is stored long term. When these carbon capture installations are primarily used to supply CO_2 to greenhouses, which experience peak CO_2 demand during the summer, the carbon capture installations only run about 6 months per year (during the growing season).

Current policies disincentivise CCU compared to CCS for WIPs. The Dutch CO_2-levy system for industry – which also includes WIPs – prescribes that CO_2 utilisation in greenhouse horticulture is not seen as emission reduction for the CO_2 supplier, since the carbon uptake by the crops that are grown in the greenhouses is temporary and CO_2-emissions are merely delayed. Moreover, only a small fraction of the CO_2 delivered is actually taken up by the crops. With respect to emission accounting, the Dutch levy system follows international GHG accounting rules. The emission reduction stemming from the CO_2 utilisation, caused by lowering CO_2 production in the horticulture itself using gas boilers or combined heat-power installations, is accounted to the horticulture. If the captured CO_2 is permanently stored, it does count as an emission reduction for WIPs, however; application of CCS thus contributes to lowering the emissions that fall under the Dutch CO_2-levy system. As a result, the relative attractiveness of the business case for CCU compared to CCS is declining.

Additionally, it is difficult for WIPs to obtain SDE++ subsidies for CCS, while subsidies such as SDE++ are essential to make CCS economically feasible. The reason is that the options for the SDE++-subsidy are ranked according to their amount of subsidy needed per tonne avoided fossil CO_2 emission. Since the fossil carbon content of waste incinerator emissions is only 36% on average, the cost per tonne of fossil CO_2 avoided is nearly triple that of plants that emit 100% fossil CO_2, if biogenic CO_2 is not counted as negative emissions, which is currently the case for CCS in the Netherlands. Moreover, the subsidy demand for CCS at WIP's is higher compared to industrial ETS-installations with similar emission characteristics. This is because for those industrial ETS-installations, the subsidy only covers the additional costs on top of the ETS-price. This makes the attainment of SDE++ subsidies for WIP CCS retrofits unlikely if the total subsidy budget is restrictive.

5.4   Research implications & further research

This research contributes to the MIDDEN project, which aims to provide an open-source database with up-to-date information on GHG emission reduction options for all relevant stakeholders. This research has made an attempt at comparing a variety of different GHG emission reduction options, configurations and pathways and to compare them based on techno-economic analyses. The results may form the basis for future attempts to reduce the emissions related to waste incineration. Although the financial analysis was largely inconclusive due to large uncertainties, the LCA results offer a knowledgebase to aid decision-making for the Dutch government, waste incinerators and other stakeholders.

The list of technologies included in this research in not exhaustive. Technologies that were excluded may become more relevant in the future as their TRL increases with more plants being constructed, process efficiencies increase as a result of industrial scale applications, and potential feedstocks increase due to better product design or disposal methods. Future research may therefore be conducted to include these technologies and their economic and environmental implications for Dutch waste management and society.

As the enhanced recycling technologies studied enter more mature phases in their development and application, the uncertainty in many parameters will likely decrease, giving decision-makers a better understanding of the environmental and financial implications of the decisions they are faced with. It is therefore necessary for further research to reduce these uncertainties and make more accurate prediction of environmental and financial implications possible.

6   Conclusion

In the context of the MIDDEN project, this research set out to provide an assessment of the energy and material flows related to waste incineration practices and to identify the potential for GHG emission reduction by means of carbon capture or enhanced recycling technologies. The highly energy efficient Dutch waste incineration industry provides many benefits relative to other established waste disposal methods. It vastly reduces CO_2-equivalent emissions relative to landfilling, and reduces the overall waste volume by up to 90%. Nonetheless, for the Dutch government to realise its climate ambitions, significant reduction of the 3.0 Mt fossil CO_2-eq emissions annually emitted by waste incineration is necessary. This research has reiterated that carbon capture technology presents a feasible short-term solution to the problem. Feasible, as companies such as AVR Duiven and Twence have shown, in a configuration where CO_2 is utilised in greenhouse horticulture or elsewhere. However, without sequestration of the captured CO_2, emissions may remain fairly constant. Carbon capture and storage solutions were found to be out of reach of waste incinerators due to conflicting incentives in the form of taxes and subsidies, as well as high costs.

Enhanced recycling concerning certain plastic streams, was found to be a potent method of reducing CO_2 emissions per unit input. Solvolysis of EPS and depolymerisation of waste PET trays reduce the net emissions per kg input by 1.7 (±0.5) and 2.9 (±0.6) kg CO_2 respectively. These processes were also found to likely be profitable when applied in full-scale industrial settings. However, due to the limited availability of input waste for these processes, total GHG emission reduction was limited to roughly 3% of total fossil CO_2 emissions from waste incineration. Pyrolysis of mixed plastic waste was found to result in emission reductions of 0.4 (±0.1) to 0.5 (±0.1) kg CO_2 per kg of plastic waste, with the process producing the HFO substitute having the higher net emission reduction. The processes examined only process PE and PP waste, necessitating the incineration of the left-over fraction and limiting GHG emission reduction. The total potential feedstock for the process is much larger than for solvolysis or depolymerisation, but total GHG emission reduction was limited to 3% of total fossil CO_2 emissions from waste incineration. Gasification of municipal solid waste was found to potentially decrease CO_2 emissions by 0.3 (±0.2) kg CO_2 per kg of RDF when combined with storage of the captured pure CO_2 stream inherently created by the process. At a total potential feed of 1,700 kt RDF in the Netherlands, this results in an estimated potential CO_2 emission reduction of over 1.0 Mt annually.

Over the course of this research, it became clear that significant short-term GHG emission reduction is most likely to be achieved through the large-scale implementation of carbon capture technologies. In part, this is due to the highly specific waste streams necessary for the current methods of enhanced recycling, which are available in limited amounts. Additionally, the recycling processes often proposed cause significant emissions themselves. Developments in the EPS and PET tray recycling sectors are aided by the negative CO_2 avoidance cost, indicating possibility for a profitable, self-sustaining business case. Carbon capture and storage proves to be the most potent short-term solution to waste incineration GHG emissions. In the long-term, GHG emission reduction must be achieved by full separation of the plastic waste from the residual fraction and subsequent recycling of this plastic material. The methods and practices to achieve this, however, are not available at this time.

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Appendix

Table 19. Construction planning of novel (plastic) waste recycling technology pilot plants. Missing information is indicated by minus sign.

Technology	Owner	Project name	Site	Status	In use	Input	Output	Input (kt)	Output (kt)
Torrefaction	RWE	Furec	Chemelot	FID planned in 2023		Mixed waste	Hydrogen	350-400	40-60
Gasification/Fischer-Tropsch	Enerkem & Shell	-	Botlek	Permitting procedures started	2026	RDF	SAF (75%), chemical feedst. (25%)	360	220
Enzymatic conversion	DGB Bio Energy	-	Delfzijl	construction will start 2022	2022	Paper sludge	BioLNG and fertilizer	270	155
Gasification (HT Winkler)	GIDARA Energy	Advanced Methanol Amsterdam (AMA)	Port of Amsterdam	-	2023	Biomass & RDF	Methanol en CO2 (for CCU)	175	87.5
Pyrolysis	Waste4Me	-	Moerdijk	-	2023	Mixed plastics	-	35	-
Pyrolysis	Shell & Blue Alp	-	Moerdijk	-	2023	Mixed plastics	Naphtha substitute	30	-
Pyrolysis +hydrogenation	Sabic & Plastic Energy	-	Chemelot	Under construction	2022	Mixed plastics	Naphtha substitute (Tacoil)	20-25	-
Pyrolysis (LT depolym.)	Patpert Teknow Sys.	Xycle	Rotterdam	Detailed process design study	2023	-	-	20	-
Pyrolysis	Multiple parties	Pyrolyseproeftuin Zuid-Nederland	Moerdijk	Constructed	-	-	-	-	-
Upgrading pyrolysis oil	DOW	-	Terneuzen	-	-	Pyrolysis oil	Pyrolysis oil	10	-