# DECARBONISATION OPTIONS FOR THE DUTCH WASTE INCINERATION INDUSTRY

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## MIDDEN

**Manufacturing Industry Decarbonisation Data Exchange Network**

## Colophon

**Decarbonisation options for the Dutch Waste Incineration Industry**

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# Summary

This report focuses on the decarbonisation of the current waste combustion processes in the Netherlands. We define decarbonisation as the reduction of the net greenhouse gas (GHG) emissions resulting from changes in the management of the following waste streams that are incinerated at present: municipal solid waste, contaminated soil and sewage sludge. Apart from end-of-pipe carbon capture technologies, a number of recycling technologies have been discussed in this report. This is especially true for plastic containing waste streams, since these contain the bulk of the fossil carbon in municipal solid waste. It is important to make the distinction between the goal of this research (net fossil GHG emission reduction stemming from application of alternatives or modifications to the current incineration processes), and the much broader objective of attaining a circular economy. While we have analysed technologies available to alternatively process the waste streams (partly) containing fossil-based carbon, this report excludes topics such waste prevention. Recycling methods that do not result in net GHG emission reduction have not been addressed.

Waste management in the Netherlands includes numerous activities. These include collection, separation, recycling and various ways of waste disposal such as incineration and landfilling. About 23% of the Dutch greenhouse gas emissions from the waste management sector are emissions from landfilling sites. Of the remaining emissions, which are reported to The Netherlands' Pollutant Release and Transfer Register, 99% originates from combustion processes by 20 companies in three sectors: waste incineration, sludge incineration and thermal soil remediation (Emissieregistratie, 2018).

The Netherlands is among the EU member states with lowest landfilling rates, and at the same time has a residual capacity of waste incinerators which is filled with waste imports, mostly from the UK. Waste imports peaked in 2016 and have been declining since then. They are expected to decline further, since the scope of the tax on combustion of waste has been extended in 2020 to include imported waste streams. In 2018, 7.5 million tonnes (Mt) of municipal solid waste was incinerated, causing 7.8 Mt $CO_2$-eq emissions. 63% of these emissions were of biogenic origin, while the remaining 37% were fossil $CO_2$ emissions. In 2018, Dutch waste incinerators generated 12.1 PJ of electricity and 14.9 PJ of useful heat with a net cogeneration efficiency of 36%. Sludge incineration installations processed 303 kt of dry mass sludge in 2018, leading to 334 kilotons (kt) of $CO_2$-eq emissions. Additionally, sewage treatment plants emitted 220 kt $CO_2$-eq in the form of methane. Thermal soil remediation plants emitted 323 kt of $CO_2$ according to the Netherlands' Pollutant Release and Transfer Register for 2018. Finally, landfills release GHG emissions mainly in the form of methane, amounting to 2.3 Mt of $CO_2$-eq emissions in 2020. These emissions are experiencing a natural decline, and are expected to decrease to 0.9 Mt in 2040.

Decarbonisation options covered in this report are carbon capture and utilisation (CCU), carbon capture and storage (CCS), municipal solid waste gasification, plastic pyrolysis, depolymerisation of polyethylene terephthalate (PET) and solvolysis of expanded polystyrene (EPS) (see Table 1). Decarbonisation options for sludge incinerators and thermal soil remediators are qualitatively discussed. Findings include possible decarbonisation due to recycling options, with a combined emissions reduction potential of ~1.2 Mt $CO_2$ per year, largely due to the storage of $CO_2$ combined with gasification. If the $CO_2$ from gasification is not stored, emissions reduction is estimated at ~150 kt $CO_2$ annually. Pyrolysis was found to be only applicable to polyethylene (PE) and polypropylene

(PP) waste, which is available in relatively limited quantities (47 kt in 2018). Potentially, however, more material may become available for recycling than estimated in this research. Such additional feedstock may result from increased efforts in source separation of recyclable waste, as well as increased post-collection separation. The quantities above are limited to packaging plastic waste, which constitutes ~30% of the total plastic waste. Non-packaging plastics, which make up the remaining 70%, are not widely covered in literature and have been excluded from calculations in this research. A larger focus on this non-packaging plastic waste stream in research and policy making is an important step in decarbonising the waste incineration sector.

EPS solvolysis and PET depolymerisation result in significant GHG emission reduction per unit of feedstock input, but again, available feedstock quantities are small at 6.5 and 30 kt per year respectively. All plastic recycling options were found to have negative $CO_2$ reduction costs due to substantial revenues from the recycled products. Deep decarbonisation (even resulting in net negative emissions) can be realised with CCU/S options.

A summary of the incinerated municipal solid waste and decarbonisation options discussed in this report is shown in Table S.1.

**Table S.1**
Summary of incinerated municipal solid waste and applicable decarbonisation options (2018). Based on average subdivisions of MSW (Corsten et al., 2013; Lu et al., 2017; Makarichi et al., 2018). Plastics subdivision and carbon content based on Brouwer et al. (2018) and Larsen and Astrup (2011).

| Category | Estimated amount incinerated (Mt, 2018) | Estimated amount, percentage of total | Emissions (Mt $CO_2$/y) (fossil by default) | Emissions, percentage of total (only fossil) | Decarbonisation options[b] (see Section 5) |
|---|---|---|---|---|---|
| **Total municipal solid waste** | 7.5 | 100% | 2.9 (fossil) | 100% | See subdivision in this table |
| **Organic** | 2.1-2.6 | 28-35% | PM | 0% | CCUS, gasification |
| **Paper** | 1.5-2.3 | 20-30% | <0.1 | <1% | CCUS, gasification |
| **Plastics** | 0.9 | 12% | 2.3-2.6 | 80-95% | See subdivision in this table |
| *Plastic packaging* | 0.26 | ~3% | 0.6-0.8 | 23-25% | See subdivision in this table |
| • PE/PP | 0.07 | ~1% | 0.2 | 7% | CCUS, gasification, pyrolysis |
| • PET | 0.04 | <1% | 0.1 | 3% | CCUS, gasification, PET depolymerization |
| • EPS | 0.01 | <1% | 0.03 | 1% | CCUS, gasification, EPS solvolysis |
| • others | 0.1-0.15 | 1-2% | ~0.3 | 10-14% | CCUS, gasification |
| *Plastic non-packaging* | ~0.6 | ~9% | 1.7-1.9 | 57-65% | CCUS, gasification, possibly others |
| **Glass, textile, metals** | ~0.9 | ~12% | 0.1 | 4% | CCUS, gasification |
| **Others[a]** | 0.6-1.6 | 8-22% | 0.2-0.4 | 5-14% | CCUS, gasification |

  [a] Contains plastics in unknown amounts.

  [b] If the option is applied to only a fraction of the mixed solid waste stream, a separation process has to be applied (either before or after collection).

# 1    Introduction

This report describes the current situation for waste incineration in the Netherlands and the options and preconditions for its decarbonisation. The study is part of the MIDDEN project (Manufacturing Industry Decarbonisation Data Exchange Network). The MIDDEN project aims to support industry, policymakers, analysts, and the energy sector in their common efforts to achieve deep decarbonisation. The MIDDEN project will update and elaborate further on options in the future, in close connection with the industry.

## Scope

This study contributes to the MIDDEN project by analysing the Dutch waste incineration industry. Specifically, the research is limited to companies in this industry that emit more than 10 kt of $CO_2$-equivalent emissions per year. This research has opted to take 2018 as the year of reference due to the availability of public data for this year. Whenever relevant developments have taken place after 2018, they are included in the report.

Over 99% of Dutch waste management GHG emissions registered in The Netherlands' Pollutant Release and Transfer Register originate from 20 companies operating in three sectors: Waste Incineration Plants (WIPs), Thermal Soil Remediation Plants (TSRPs) and Sludge Incineration Plants (SIPs). Together, these 20 companies emitted 8.2 Mt of biogenic and fossil $CO_2$-eq emissions in 2018 (Emissieregistratie, 2018). WIPs represent 95% of these emissions and are thus the main point of focus for this report. WIP emissions originate from the incineration of municipal solid waste (MSW), of which 7.5 Mt was incinerated in 2018 (Rijkswaterstaat, 2020). A significant part of this waste is of biogenic origin. Fossil $CO_2$-equivalent emissions, including 0.1 Mton $CO_2$-eq in the form of $N_2O$, were 3.0 Mt (RIVM, 2021). TSRPs remediate contaminated soil and currently rely on the combustion of natural gas to do so. The 4 companies involved in this industry processed 0.6 Mt of soil and emitted 0.3 Mt of fossil $CO_2$-eq emissions in 2018. Lastly, SIPs process dried municipal sewage sludge, emitting 0.4 Mt $CO_2$-eq in 2018, of which 0.10-0.14 Mt were fossil emissions.

## Reading guide

Section 1 introduces the Dutch waste management industry and the European context. Section 2 characterises the waste streams and discusses GHG emissions related to these streams. Section 3 describes the relevant processes to manage the major waste streams. Options for decarbonisation are quantified and evaluated in terms of costs and net emission reduction potential in Section 4. A discussion of techno-economic analyses and policy is presented in Section 5.

# 2 Waste management in the Netherlands

This section discusses the historic developments in the Dutch waste management industry. Next, details are provided on the companies within the scope of the research.

## 2.1 Developments in waste management, EU and the Netherlands

In the early 1990's, waste management policy in the Netherlands and other EU countries was centred around disposal. Recycling was not a primary focus, and the result was large amounts of waste that were either incinerated or, to a larger extent, landfilled. The landfilling rate in the Netherlands peaked at this time. After the early 1990's, policy started to shift towards reducing the amount of disposed waste by increasing recycling efforts. Where recycling was not an option, incineration was a favourable alternative. Examples of policy efforts to aid the goal of minimising disposal are the Dutch landfill tax, introduced in 1996, and the ''Kaderrichtlijn Afvalstoffen'' , first issued in 2006 (European Parliament and Council, 2006). Figure 2.1 shows the development of the amount of waste processed for each processing category. It must be noted that not all waste generated in the Netherlands is processed within its borders. Exports are not shown in Figure 2.1. The large amount of dredge material processed in 2018 was due to the application of stored material in two large infrastructural projects, Houtribdijk in Lelystad and the Marker Wadden.

**Figure 2.1**
Development of Dutch waste management by processing category.

*Comparison based on total mass per category, data for 1992-2019 (Rijkswaterstaat, 2021a). The origin of landfilled waste is discussed in Section 2.3.2. For treated soil and processed dredge material were no figures available prior to 2009.*
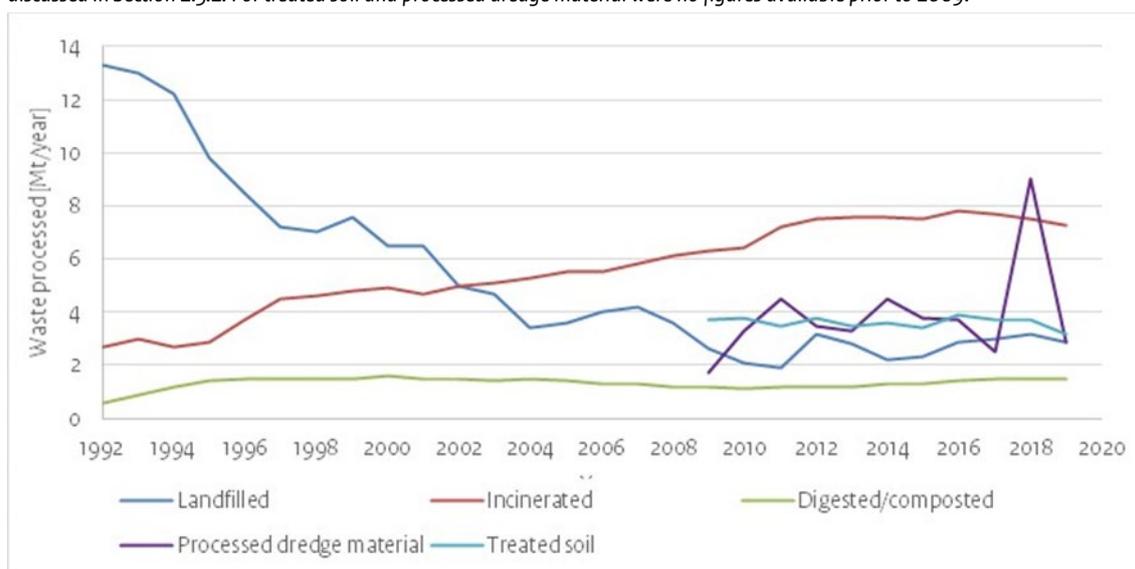


More recently, the EU has been increasing its focus on the prevention of waste disposal. The European Commission (EC) has established the waste hierarchy, which is shown in Figure 2.2. The

waste hierarchy is accompanied by a number of policies, such as a minimum recycling rate for all member states that increases every five years. The minimum recycling rate goals for municipal waste are 55% in 2025, 60% in 2030 and 65% in 2035. There are also specific directives dealing with packaging waste or electric and electronic appliances (EEA, 2018). The Netherlands reported a recycling rate of 80% of all waste produced and treated domestically in 2020, which is above the European average of around 61% (PBL, 2021). It must be noted that a major part of this recycled fraction is labelled low-grade recycling, meaning the resulting product of recycling is of low quality. 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).

**Figure 2.2**
European Commission waste hierarchy.



Despite these efforts, some fraction of the waste generated must be disposed. Due to methane emissions associated with landfilling and groundwater protection, incineration is preferred. The incineration and landfilling rates for disposed waste in all EU countries and the UK are shown in Figure 2.3. Out of all waste that is disposed by either landfilling, incineration, the Netherlands has one of the highest incineration rate for waste disposal at close to 90%. Figure 2.3 only includes domestic waste disposal, thus including imported waste but excluding export (Eurostat, 2021). The figure also details initial disposal, thus excluding double counting of mass such as the landfilling of waste incinerator bottom ash.

**Figure 2.3**
Landfilling and incineration rates of the EU for 2018.
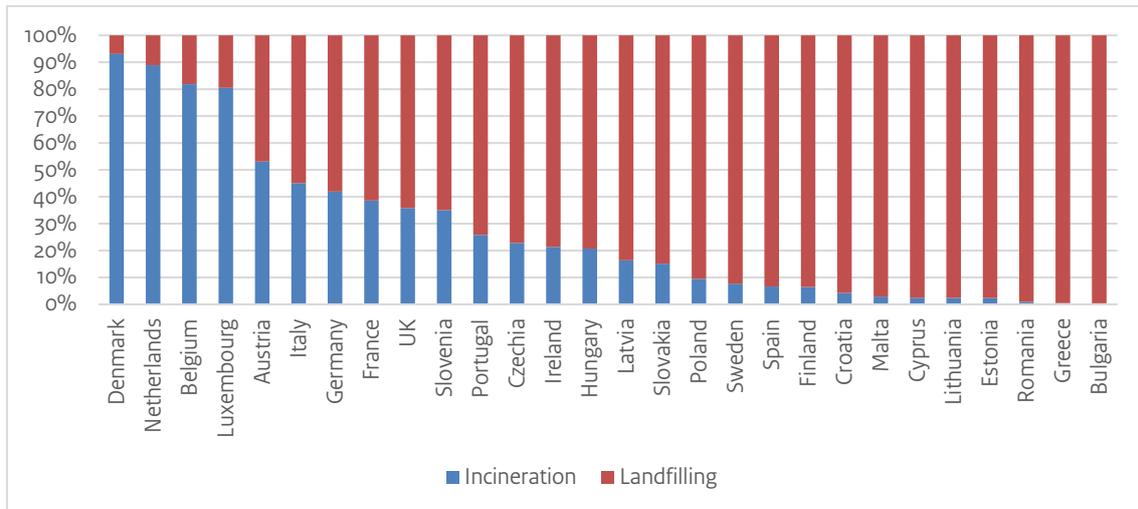


*Rate of incineration or landfilling for all 28 member states, domestic disposed waste only, 2018 data (Eurostat, 2021).*

## 2.2    Overview of waste types

In this section, waste is discussed by dividing it in three types: municipal solid waste, waste water (both sewage and industrial) and contaminated soil (including asphalt waste). Each of these types is discussed based on their relevance to decarbonisation efforts in the waste management industry.

### 2.2.1    Municipal solid waste

Although it is not the biggest fraction of waste by mass, municipal solid waste (MSW) is an important stream due to its size and composition. In this report, MSW includes (bulky) household waste, solid waste from businesses, non-hazardous industrial waste and other residual streams. MSW composition differs per region and throughout the year. Composition also depends on the method of collection that is applied. MSW can be recycled, depending on the type of waste. Bijleveld et al. (2021) lists 13 types of waste and the most common disposal and processing methods, as well as some recycling rates. Table 2.1 shows their main findings. Residual waste is not included in this table, as this is not a discrete flow with one single recycling method or rate.

**Table 2.1**
Individual types found in municipal waste, and typical processing methods (Bijleveld et al., 2021).

| Waste stream | Typical processing method |
|---|---|
| Paper/cardboard | 85% pre-sorted and recycled up to 7 times. Remainder is incinerated with energy recovery. |
| Plastics | PET bottles 99% pre-sorted. Part of HDPE, LDPE and non-bottle PET is pre-sorted in PMD, remainder 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 a waste 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 ends up in the bottom ash, where it may be reclaimed and recycled. |
| Stone/rock | Pre-sorted at municipal recycling centre, or sent to a waste incinerator. In the latter case, it ends up in the bottom ash, where it may be reclaimed and recycled. |
| 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. |

## *MSW collection in the Netherlands*

The streams associated with waste in the Netherlands are the product of the collection and separation techniques that are in place, and to which extent these techniques are applied by all the relevant parties. Collection of Dutch municipal solid waste 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 2.4 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).

**Figure 2.4**
MSW collection in the Netherlands, 1995-2020.
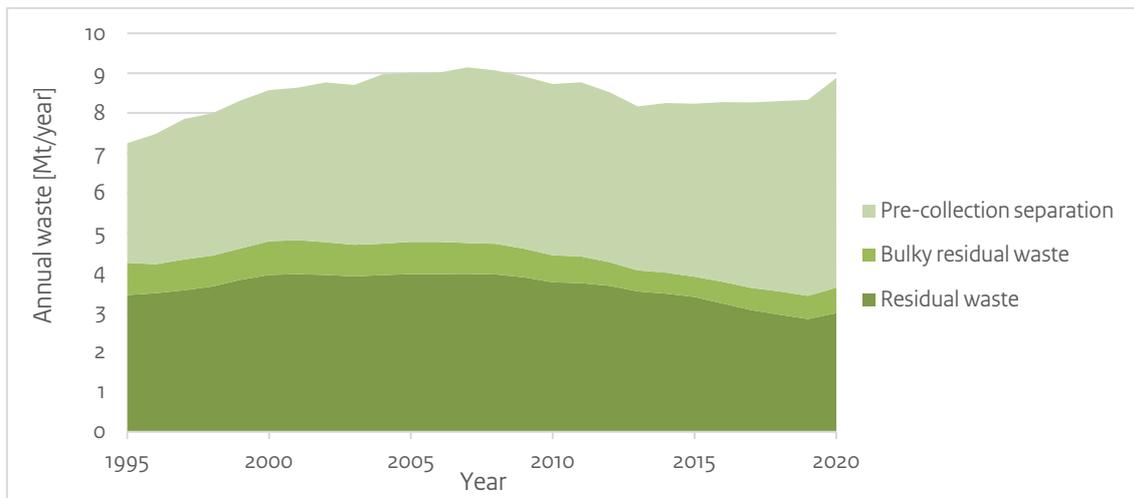


*Figure excludes pre-collection separation commissioned by other parties than municipalities as it constitutes less than 1% of total collected waste from 1995-2020. Data from CLO (2021). Increase in all types of waste in 2020 coincides with the COVID-19 pandemic.*

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 including, but not limited to, Amsterdam, Rotterdam, Leiden 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 occasionally be highly contaminated with other waste, dirt, food residue and moisture (Gradus, 2020). When the contamination exceeds a certain level, often set at 10-15%wt, freights are rejected.

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 2.2 shows the typical composition for solid municipal waste that enters an incinerator. It shows the fractions of materials as weight percentages. This composition is known to vary, but is shown as a indication of the composition of incinerated waste.

**Table 2.2**
Composition of incinerated waste in the EU.

| 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 |

*Moisture content is not separately addressed in these figures, but is part of the fractions presented.*

Figure 2.5 shows the origin of incinerated waste in the Netherlands in 2018. The sum of industrial waste (part of "other" in the figure) and business waste is roughly 18%, since most of these wastes are recycled in some way. The fraction ''separation residues'' in Figure 2.5 is composed of residues from business waste, industrial waste and municipal waste. The subdivision among these origins is not known.

**Figure 2.5**
Origin of incinerated waste in the Netherlands in 2018.
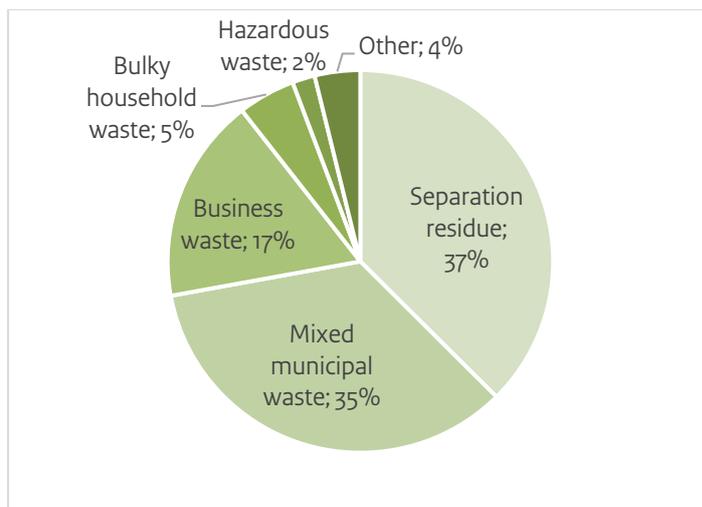


*''Business waste'' is from the trade, service and government sectors. ''Other'' includes composting residues, industrial waste, hospital waste and residue from municipal waste services. Data from Rijkswaterstaat (2020).*

### Plastic waste in the Netherlands

Plastic waste is a primary focus of this research, as it is the main source of fossil $CO_2$ emissions upon incineration. Other fossil $CO_2$ emissions from incineration originate from fossil-based textiles, paint, ink and other chemicals. Worldwide, it is estimated that 80-90% of fossil $CO_2$ emissions from waste incineration originate from plastic incineration (Larsen & Astrup, 2011; Yang et al., 2012). 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.

Packaging waste was incinerated at a rate of 52%, while 48% was recycled in some way. Table 2.3 shows the breakdown 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. The amount of recycled material in plastics was around 20% in 2018 (Krebbekx et al., 2018). The vast majority of this plastic is recycled mechanically. The extent to which currently incinerated plastics can be made available for enhanced recycling efforts is discussed in Sections 4.4.1, 4.4.4 and 4.4.5.

**Table 2.3**
Plastic packaging waste processed in the Netherlands in 2017 (in kt plastic).

| 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 |

*Data from 2017, values in kt (Snijder & Nusselder, 2019). Note that in Section 4.4.6, 320 kt of mechanically recycled waste is identified. This is larger than the total figure identified here, as it includes non-packaging plastic waste.*

## 2.2.2   Wastewater

Wastewater constitutes any aqueous stream that is of no use to its disposer. Wastewater can be divided into two main categories; sewage wastewater and industrial wastewater. Sewage wastewater can be lightly contaminated (greywater) or highly contaminated (blackwater). Greywater originates from household water applications such as washing machines, sinks and showers, and typically contains soap residue, fats and oils. Blackwater is contaminated with faecal matter. Blackwater and greywater are mixed upon entering the sewage system and are transported to sewage treatment plants. This stream contains many different organic and inorganic compounds and is difficult to fully characterise. Instead, the level of contamination is often approximated with the biochemical oxygen demand, or BOD: the amount of oxygen consumed by microorganisms to break down the contamination. Note that this is only a measure of the biological decomposition of organic matter. The BOD of sewage wastewater is typically around 100-300 mg per litre, whereas the threshold for disposing the wastewater into surface water bodies is 7 mg per litre. The sewage treatment plant ensures that the BOD and other indicators of contamination are brought below the threshold of 7 mg per litre by various processes. In these processes, a sludge is generated. Roughly half of this sludge is incinerated in dedicated incinerators, while the remainder is incinerated at electricity plants or cement kilns (CBS, 2020).

The second type of wastewater comes from industry. The Netherlands has 182 industrial wastewater treatment plants, of which 15 are of substantial size (van Wezel & van den Hurk, 2018). The goal of industrial wastewater treatment is to decrease contamination to a level where discharge to surface water is not hazardous. The method of achieving this depends on the type and level of contamination. This varies greatly upon the industry producing the wastewater. Major emitters of industrial wastewater and the associated contaminants are listed in Table 2.4.

**Table 2.4**
Major emitters of industrial wastewater and common examples of associated emissions (Micronics, n.d.).

| Industry type | Emissions | Examples |
|---|---|---|
| **Battery manufacturing** | Heavy metals | Cadmium, chromium, cobalt, copper, cyanide, iron, lead, manganese, mercury, nickel, oil & grease, silver and zinc |
| **Organic chemical manufacturing** | Organic chemicals | Benzene, chloroform, naphthalene, phenols, toluene and vinyl chloride |
| **Inorganic chemical manufacturing** | Various | Aluminium compounds, potassium compounds, heavy metals etc. |
| **Fossil fuel power plants** | Metals | Lead, mercury, cadmium and chromium |
| **Food industry** | Various | Suspended solids, nitrogen, phosphorus, high BOD |
| **Iron/steel industry** | Gasification products | Benzene, naphthalene, anthracene, cyanide, ammonia, phenols, cresols, polycyclic aromatic hydrocarbons |

## 2.2.3  Contaminated soil

Soil can be contaminated with a variety of compounds that severely limits its uncontained use in the environment. Common types of contaminants are mercury, oil residues, dioxins and cyanide (AJansen Website, n.d.). In 2019, roughly 3.2 Mt was treated (Rijkswaterstaat, 2021a). Soil treatment methods in the Netherlands include thermal soil remediation, biological remediation, wet remediation and immobilisation. In the context of this report, thermal remediation is of interest since it consumes large amounts of fossil fuels and is applied widely. In 2019, 238 kt of contaminated soil was thermally remediated to allow other applications of the soil (Rijkswaterstaat, 2021a). Tar-containing asphalt residues (TAG) are also considered contaminated, as they cannot freely be used in new roads in the Netherlands since legislation passed in 2017, limiting the tolerable concentration of polycyclic aromatic hydrocarbons. The estimated annual TAG stream is roughly 1.4 Mt per year (LAP3, 2014). This quantity may increase due to TAG imports.

# 2.3  Dutch waste management GHG emissions

The largest GHG emitting waste management processes are sewage sludge incineration (0.4 Mt fossil $CO_2$-eq), municipal solid waste (MSW) incineration (3.0 Mt fossil $CO_2$-eq emissions), landfilling (2.3 Mt fossil $CO_2$-eq), and thermal soil remediation (0.3 Mt fossil $CO_2$-eq emissions). These activities also include a biogenic emission fraction which varies depending on the stream that is processed, as well as the kind of processes employed. Each of the major emitters in Dutch waste management is discussed in this section in terms of their fossil and biogenic GHG emissions and industry and company data.

## 2.3.1   Waste incineration GHG emissions

There are 12 waste incineration plants in the Netherlands with energy recovery, which together processed 7.5 Mt of waste in 2018. Table 2.5 shows the waste incineration plants (WIPs) in the Netherlands and associated data. The amount of incinerated waste peaked in 2016 at 7.8 Mt and has been stagnant since then (Rijkswaterstaat, 2020). Since 2010, substantial quantities of waste have been imported, mostly from the UK. Imports of waste peaked in 2016 and have been decreasing since then. In 2018, waste imports amounted to 1.7 Mt, roughly 23% of total incinerated waste. The waste incineration industry employed 3,365 FTE in 2018 according to annual reports from the WIPs (this number does not include employees outside the incineration sector, such as waste collection). The total turnover for all WIPs was EUR 1,153 mln in 2018.

GHG emissions from waste incineration in 2018 totalled 7.8 Mt $CO_2$-eq when adding biogenic and fossil emissions together (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). 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. The fossil $CO_2$ emissions of the waste incineration sector were 2.9 Mt in 2018, as reported in the National Inventory Report to the UNFCCC (RIVM, 2021). Previous studies have shown that 80-90% of fossil emissions in waste incineration originate from the incineration of plastic waste (Larsen & Astrup, 2011; Yang et al., 2012). Clearly, the precise fossil emissions per tonne of waste depends on the composition of the waste stream. The remainder of fossil emissions originate from additives in the paper fraction and the unidentified residual fraction.

Some of the emissions from WIPs are captured and utilised at present. AVR Duiven installed a large $CO_2$ capturing installation in 2019 with a capacity of up to 60 kt $CO_2$ per year. In 2020, it managed to capture and transport 30 kt of $CO_2$ to greenhouses (AVR, 2021). Twence currently captures $CO_2$ for use as a feedstock for sodium bicarbonate (baking soda) production. This is however limited to 2 kt per year (Rijkswaterstaat, 2020). In addition, Twence also started capturing and liquifying $CO_2$ for transport to greenhouses. In 2020, it delivered 3.6 kt and it strives to increase this capacity to 100 kt per year (Twence, 2020). More information about $CO_2$ utilisation in greenhouses can be found in Section 4.1.2.

**Figure 2.6**
Locations of the 12 Dutch waste incineration plants.

**Table 2.5**
Waste incineration plants in the Netherlands, emissions, employment and incineration data for 2018.

| | Location | Estimated GHG emissions[a] [kt CO₂—eq] | Estimated fossil GHG emissions[b] [kt CO₂.eq | Employees in 2018[c] [FTE] | Permitted capacity [kt waste] | Incinerated waste in 2018[d] [kt waste] | Installed thermal capacity [MW$_{th}$] | Installed electric capacity [MW$_e$] |
|---|---|---|---|---|---|---|---|---|
| **AEB Amsterdam** | Amsterdam | 1549 | 592 | 400 | 1350 | 1487 | 495 | 154 |
| **ARN B.V.** | Weurt | 243 | 93 | 106 | 310 | 233 | Unknown | Unknown |
| **AEC Moerdijk** | Moerdijk | 924 | 353 | 800 | 1200 | 887 | 339 | 16.2 |
| **Attero Noord B.V. GAVI Wijster** | Wijster | 676 | 259 | 800 | 719 | 649 | 180 | 54 |
| **AVR Afvalverwerking B.V.** | Duiven | 410 | 157 | 433 | 400 | 394 | 120 | 31.4 |
| **AVR Afvalverwerking Rijnmond** | Rotterdam | 1378 | 527 | 433 | 1300 | 1323 | 394 | 140 |
| **EEW Delfzijl B.V.** | Delfzijl | 398 | 152 | 58 | 576 | 382 | 180 | 36 |
| **HVCafvalcentrale, Alkmaar** | Alkmaar | 669 | 256 | 1025 | 675 | 642 | 243 | 71.2 |
| **HVCafvalcentrale, Dordrecht** | Dordrecht | 292 | 112 | 1025 | 396 | 280 | 112 | 32.5 |
| **REC Harlingen** | Harlingen | 226 | 86 | 226 | 280 | 217 | 106 | 17 |
| **PreZero ReEnergy** | Roosendaal | 381 | 146 | 50 | 336 | 366 | 124 | 39 |
| **Twence Afval en energie** | Hengelo | 633 | 242 | 235 | 650 | 608 | 220 | 56 |
| **Total** | | 7780 | 2976 | 3333 | 8192 | 7468 | 2513 | 647 |

a) Estimate based on a national average emission factor of 1.05 kg CO₂-eq/kg waste calculated from NIR total emissions (RIVM, 2021).

b) Based on national average fossil CO₂ emission factor including N₂O emissions of 0.05 kg N₂O/tonne waste (RIVM, 2021).

c) Employee data for Attero, AVR and HVC is aggregated for all plants of the respective company.

d) Total incinerated waste can exceed permitted capacity, since the permit is sometimes based on the LHV of the waste. A lower LHV allows for more waste to be incinerated. The permit may also be based on an estimation of the total operational hours.

## 2.3.2    Landfilling GHG emissions

Landfilling has decreased significantly since 1990's peak levels. However, landfills are still responsible for considerable amounts of GHG emissions. These emissions are in the form of methane, which is formed during the process of biological decomposition of organic waste. Landfills in the Netherlands emitted 2.3 Mt $CO_2$-eq of methane in 2020, down from 13.7 Mt in 1990 (CBS, 2021b) (IPCC AR5, 100-year global warming potential equivalence used). The Climate and Energy Outlook by PBL (2020) projects that with current policies, these emissions will decrease to 0.9 Mt in 2040. The projected drop in landfill methane emissions is in part due to the diminishing ''legacy emissions'', as emission patterns lag behind landfilling rates by roughly 15-20 years (Kumar et al., 2004). Figure 2.7 shows the types of waste that are landfilled in the Netherlands. (Bulky) household waste is not present, since there is a ban on landfilling this type of waste. Figure 2.8 shows the annual $CO_2$-equivalent emissions from landfills, as well as the forecasted emissions until 2040. The Netherlands also extracts part of the produced methane (not part of the emissions discussed above). In 2018, roughly 0.3 PJ of energy was produced from this methane, mostly being used for electricity generation (CBS, 2021a).

**Figure 2.7**
Types of waste landfilled in 2019.
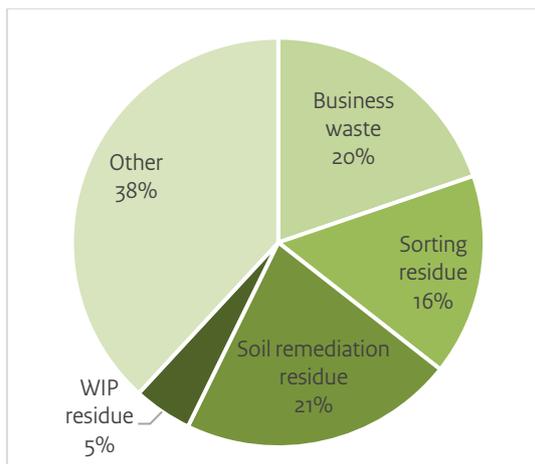


*"Other" waste includes soil, construction-, shredder-, sludge- and uncategorised waste.*
*Adapted from (Rijkswaterstaat, 2021a).*

**Figure 2.8**
Methane emissions from landfills in the Netherlands, 1990-2040.
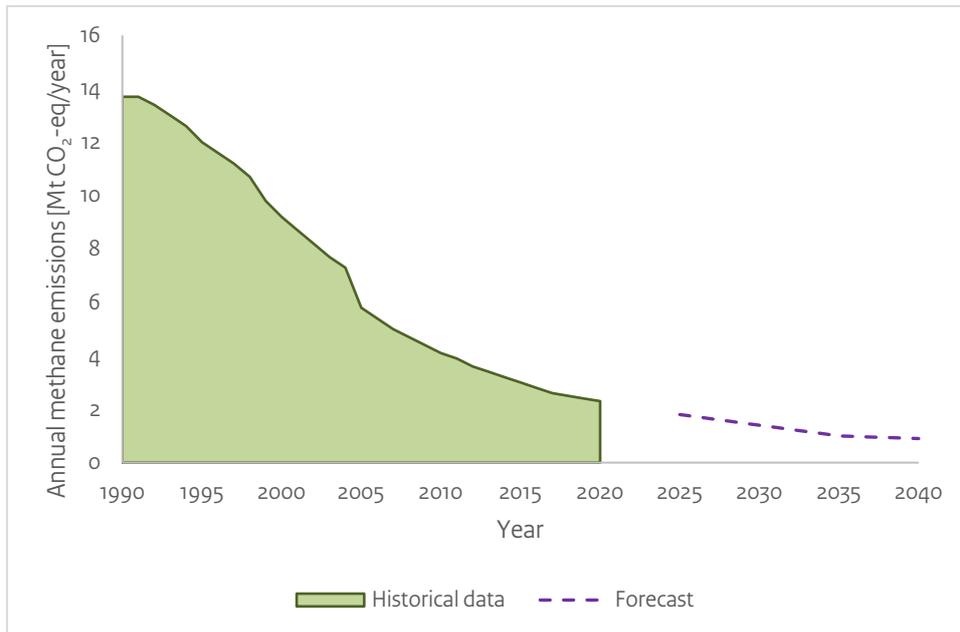


*1990-2020 data from CBS (2021b). 2025-2040 forecast is a linearisation of 5-year interval data from PBL (2020).*

## 2.3.3 Sewage water processing and sludge incineration GHG emissions

After sewage wastewater is transported to municipal sewage treatment plants, a number of processes are carried out to ensure the removal of toxic contaminants. In these processes, an estimated 0.22 Mt of $CO_2$-equivalent emissions were released in 2019 in the form of methane (PBL, 2020). The methane is formed during decomposition of various organic compounds present in the wastewater. After the wastewater is decontaminated, it is released into surface water bodies or directly used by industry. The remaining sludge is further processed.

In total, the Netherlands processed 1,250 kt of wet sewage sludge in 2018, amounting to 303 kt of dry mass (CBS, 2020). The main process applied to the sludge is incineration, as it has been for many years. Figure 2.9 shows that apart from incineration in a dedicated plant, sewage sludge is also landfilled or used as a fuel for cement kilns or co-fired in electricity production plants. There are 2 dedicated sludge incinerators in the Netherlands. These are HVC and SNB (Haskoning DHV, 2018). AEB Amsterdam incinerates sludge together with its municipal waste, with a capacity of 160 kt wet mass per year (Rijkswaterstaat, 2020). The dedicated sludge incinerators employed 67 FTE in 2018 and had a total turnover of MEUR 55. See Table 2.6 for details on the dedicated sludge incinerators.

**Figure 2.9**
Dry sludge processing in the Netherlands (CBS, 2020).
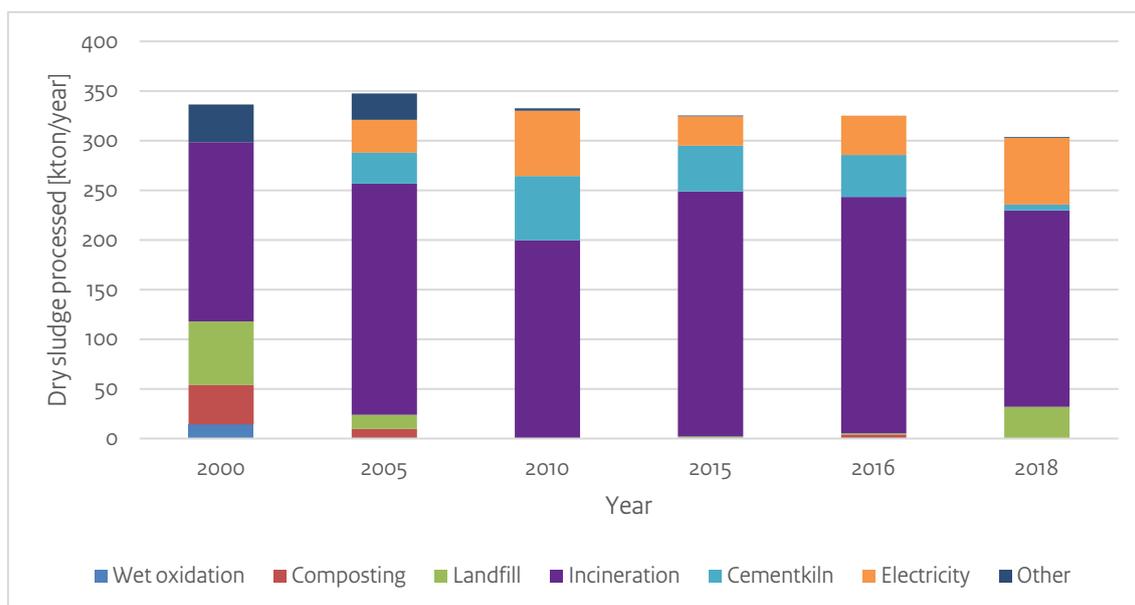


The two dedicated sewage sludge incinerators produced 334.2 kt of $CO_2$-equivalent emissions in 2018 (Emissieregistratie, 2018). The majority of these emissions are biogenic in origin, although studies have indicated that 25-35% of emissions are from fossil carbon in the sludge (Griffith et al., 2009; Kang et al., 2018). This amounts to 84-117 kt of fossil $CO_2$-eq emissions in 2018.

**Table 2.6**
Dedicated sewage sludge incinerators in the Netherlands, data from 2018.

| Plant name | Location | Emissions (ktonne $CO_2$-eq) | Employment (FTE) | Turnover (mln EUR) | Capacity (ktonne wet mass) | Processed sludge (ktonne) |
|---|---|---|---|---|---|---|
| **HVC Slibverwerking** | Dordrecht | 164.2 | 20 | 30.8 | 380.0 | 371.8 |
| **Slibverwerking Noord-Brabant** | Moerdijk | 170.0 | 47 | 24.0 | 420.0 | 419.2 |
| **Total** | | 334.2 | 67 | 54.8 | 800.0 | 791.0 |

## 2.3.4    Thermal soil remediation GHG emissions

In the Netherlands, there are four plants that perform thermal soil remediation. These are ATM (Moerdijk) , REKO (Rotterdam), Theo Pouw (Eemshaven) and Vink (Barneveld). Together they employ 800 FTE and have an annual turnover of EUR 200 mln. In 2018, these four companies emitted 323 kt of $CO_2$-eq, where 212 kt came from ATM alone (Emissieregistratie, 2018). These emissions originate from all processes carried out on the premises of the aforementioned companies. The fraction of emissions that emerge from thermal soil and TAG remediation are estimated here.

The emissions originate mostly from the combustion of fossil fuels used to heat up the soil, while there are also some emissions associated with the pyrolysis of the organic fraction in the soil (Amponsah et al., 2018). TSRPs in the Netherlands use fuel oil, natural gas or LPG as a fuel source.

On average, thermal remediation of 1 tonne of soil requires the equivalent of 40-50 litres of diesel oil of energy (Bodemrichtlijn, n.d.). This translates to an energy use of 1.5-1.8 GJ per tonne soil. If diesel oil is used, the resulting $CO_2$ emissions are in the range of 110-130 kg $CO_2$ per tonne soil remediated.

Tar-containing asphalt granulate (TAG) is also incinerated by the TSRPs. It is often mixed with the contaminated soil and incinerated together (Transport-Online, 2018). Upon incineration of the TAG, the bitumen in the asphalt combusts and releases $CO_2$. We estimate that, assuming a bitumen content of 5-7.5% in asphalt, the emissions from TAG incineration are 160-240 kg $CO_2$/tonne TAG.

# 3 Waste management processes

## 3.1 Waste incineration process

This section describes the current processes applied in the Dutch waste incineration industry. Some process steps are plant-specific, and are indicated as such whenever relevant. Data on processes is sourced from public information supplied by the WIPs on websites, annual reports and newspaper articles. Figure 3.1 serves as an overview of the points discussed in this section.

**Figure 3.1**
Process flowchart of waste incineration.
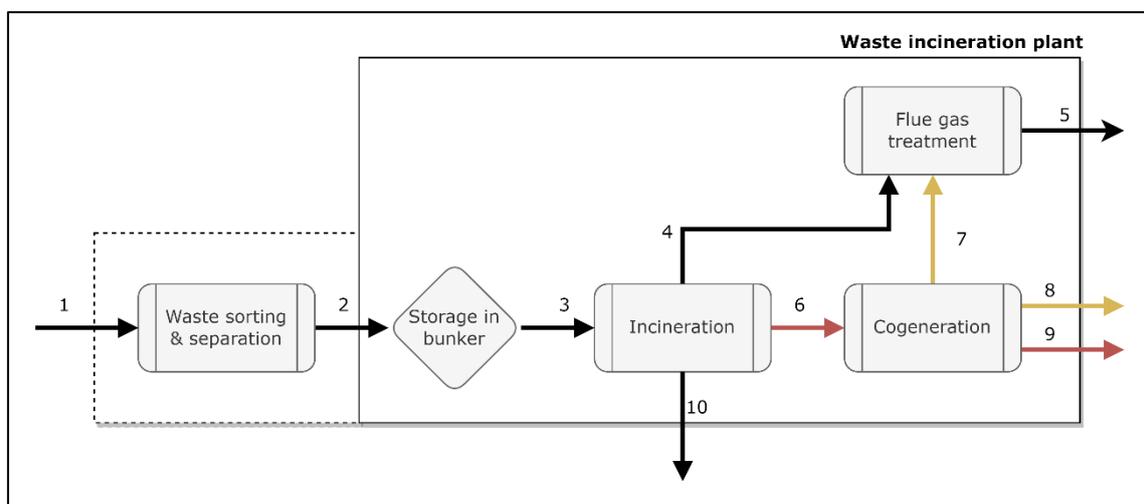


*Numbers indicate different streams. Black, yellow and red arrows indicate mass, electricity and heat flows respectively.*

| 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 | Heat | Heat flow from incineration of waste |
| 7 | Electricity | Self-consumed electricity for on-site processes |
| 8 | Heat | Useful heat export, may be steam of condensate depending on application |
| 9 | Electricity | Net electricity export after self-consumption |
| 10 | Mass | Bottom ash |

### 3.1.1 Input waste

Figure 3.1 shows a process flowchart of the general steps in waste incineration. The waste incineration process in the Netherlands starts at the final sorting and separation phase of the waste stream. The composition of waste streams indicated by 1, 2 and 3 differs per plant. The typical

composition of stream 2 is shown in Table 2.2. Some WIPs include a final waste separation stage on the same site as the incineration process. These are AEB Amsterdam, Attero Noord BV and AVR Rijnmond (Rijkswaterstaat, 2020).

Due to the variability of final waste composition, the Lower Heating Value (LHV) of the waste also differs between plants and from time to time. The Netherlands reports the annual average LHV of the waste in the National Inventory Report (NIR) for the UNFCCC. From 2000-2019, the average LHV was between 9.8-10.6 GJ/tonne. The 2018 average LHV was 10.0 GJ/tonne (RIVM, 2021). Although this country average value is reported annually, plant-level LHVs are not published. For this report, plant-level LHV values for input waste were calculated from the so-called R1-coefficient, which is published annually for each WIP except (Rijkswaterstaat, 2021b). The R1-coefficient is a performance measure for the useful energy recovery from waste incineration plants. The R1-coefficient is calculated using the following equations (EC, 2008).

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

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

Here, $\eta_{R1}$ is the R1-value, $E_p$ 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. The R1-value $\eta_{R1}$ and the CCF are calculated annually to ensure that the R1-certification is still relevant (Rijkswaterstaat, 2021b). This research assumes that $E_f$ and $E_i$ are 0. This assumption means that plants do not import energy for the process of waste incineration. This simplification is necessary due to a lack of plant-specific data on energy imports. 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 3.1. 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.

**Table 3.1**
Estimated plant level LHV for input waste in 2018, own calculation.

| 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 |
| **PreZero ReEnergy Roosendaal** | 10.0 |
| **Twence BV** | 11.3 |
| **Total calculated weighted average** | 10.0 |

## 3.1.2 Waste incineration

Waste is incinerated in a continuously operated furnace in all Dutch plants, 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 3.2 shows a schematic of this kind of furnace. Waste is inserted in the funnel on the left (stream 3 in Figure 3.1), 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). In the Netherlands, WIPs most often use gas burners to increase the temperature in the furnace and aid the incineration process (RWS, personal communication, 2022).

**Figure 3.2**
Schematic of moving grate furnace for waste incineration.
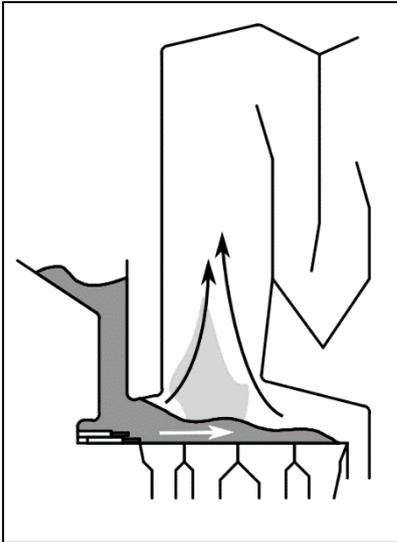


*Waste enters the moving-grate furnace at the left (funnel with gray area) and is transported into the furnace by the moving grate (white arrow).*
*Image from Van Blijderveen (2012).*

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 3.3 (Rijkswaterstaat, 2020).

Figure 3.3 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 3.3 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 (Lu et al., 2017; Makarichi et al., 2018). It must be noted that Dutch WIPs were initially designed for low-calorific value waste, and are often unable to process very high waste streams with a very high LHV (Personal communication, 2022).

**Figure 3.3**
Furnace diagram for a hypothetical waste incinerator.
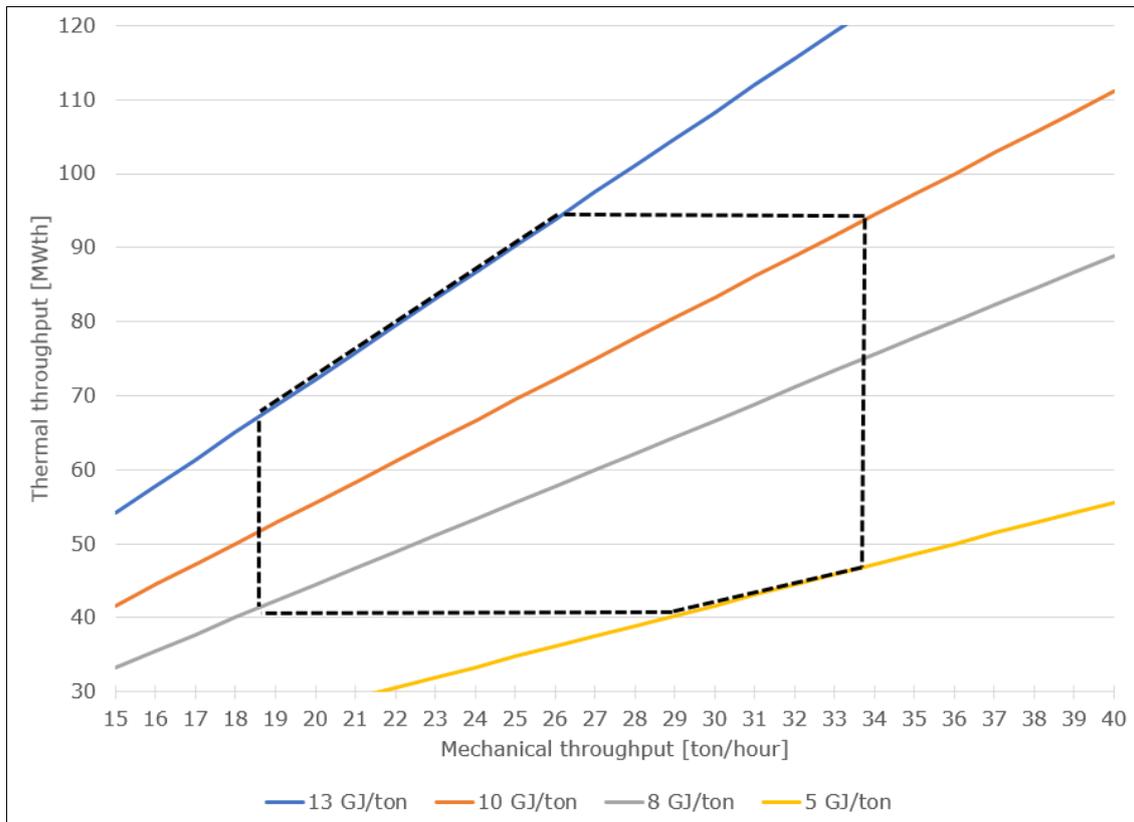


*Example of a hypothetical furnace with a mechanical capacity of 18-34 tonne/hour and a thermal capacity of 40-95 MWth input.*

### 3.1.3 Flue gas treatment

The flue gas from the waste incineration process is indicated with streams 4 and 5 in Figure 3.1. 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 maximum emissions of Dutch waste incinerators are derived from the European Waste Incineration Directive, which includes limiting values for all of the aforementioned pollutants (EC, 2000). Dutch emission limitations are generally more stringent than the limitations in the EU directive. To limit the emission of 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). Some plants also use DeNO$_x$ treatment systems. 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 flue gas treatment and ancillary processes on the waste incinerator site (Rijkswaterstaat, 2020). This energy demand is shown in stream 7 in Figure 3.1.

## 3.1.4 Cogeneration, efficiencies & energy balance

All WIPs have the R1-status indicating that these WIPs produce enough energy to be classified as plants with a useful application. In the Netherlands, this status means the WIP is allowed to import waste from other countries for the purpose of incineration (Rijkswaterstaat, 2020). In order to attain the R1-status, the WIP must have an R1-coefficient of 0.60 (for plants built before 2009) or 0.65 (for plants built in 2009 or later). The net thermodynamic efficiency of plant (i), $\eta_i$ in this report does not contain the correction factors used by the EU described in Section 3.1.1 and is instead calculated as follows.

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

$E_{el}$ 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 3.1. The efficiencies of the WIPs are shown in Table 3.2. Note that, in part due to the assumption that there are no energy imports, efficiencies presented may deviate from actual efficiencies. Data is presented to accentuate the high degree of differentiation between WIPs in terms of cogeneration efficiency. 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 3.2. This is simply the ratio of useful heat exported to gross electricity production. Figure 3.4 shows a correlation between HP ratio and efficiency.

**Table 3.2**
Estimated net efficiencies and HP ratio for WIPs in the Netherlands, 2018 data.

| Plant name | Estimated 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** | 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 |
| **PreZero 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 calculation, see equation above.
[b] (Rijkswaterstaat, 2020)
[c] Own calculation, ratio of gross electricity production and net heat delivered.

**Figure 3.4**
Heat/power ratio and net efficiency of Dutch waste incinerators in 2018.
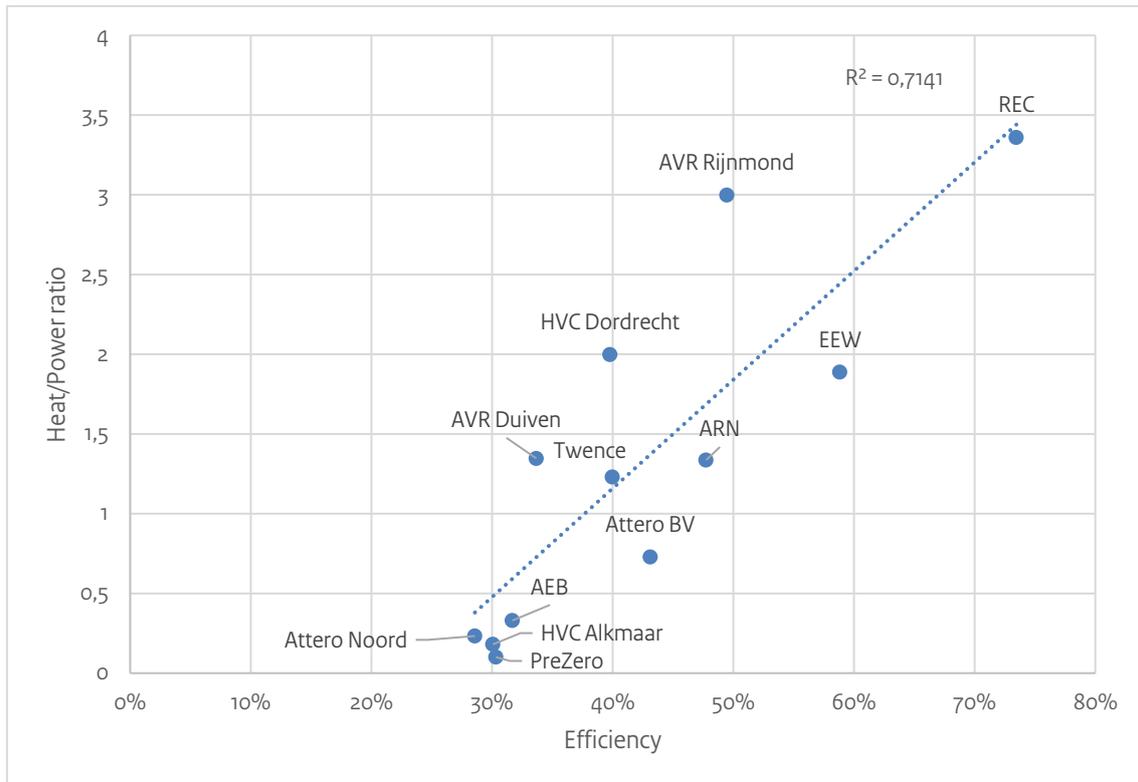


*The heat/power ratio differs for each waste incinerator. Generally, a higher heat/power ratio correlates with a higher net efficiency.*

Rijkswaterstaat states that 20% of the gross electricity production from WIPs is used for on-site processes such as flue gas scrubbing. This report follows that assumption and calculates the plant-specific self-consumption of electricity based on the weight of the incinerated waste per plant, with the following equation. Here, $E_{el,i}^{SC}$ is the amount of electricity that is self-consumed by plant $i$ in TJ, $M_w$ is the total mass of incinerated waste by all plants in kt, $M_{w,i}$ is the waste incinerated by plant $i$ in kt, $E_{el}$ is the total gross electricity generation by all plants in TJ and *SCRatio* is the average rate of self-consumption (20%). The resulting self-consumption of electricity is thus constant per tonne waste and is found to be roughly 0.4 GJ/t.

$$E_{el,i}^{SC} = \frac{M_{w,i}}{M_w} * E_{el} * \text{SCRatio}$$

The rest is sent to the electricity grid (Rijkswaterstaat, 2020). The useful heat production is mostly exported in the form of steam for industrial uses or steam/hot water for district heat networks. The type of external party for each WIP is shown in Table 3.3. Figure 3.5 shows an example of an energy balance per tonne input waste for the average Dutch WIP with a net efficiency of 36%.

**Table 3.3**
Heat delivery type for Dutch waste incinerators.

| Plant name | Industrial heat | District heat network |
|---|---|---|
| AEB Amsterdam | | x |
| ARN B.V. | x | |
| AEC Moerdijk | x | |
| Attero Noord B.V. GAVI Wijster | x | |
| AVR Afvalverwerking B.V. | | x |
| AVR Afvalverwerking Rijnmond | x | x |
| EEW Energy From Waste Delfzijl | x | |
| HVCafvalcentrale, Alkmaar | | x |
| HVCafvalcentrale, Dordrecht | x | |
| REC Harlingen | x | |
| PreZero ReEnergy | x | x |
| Twence Afval en energie | x | x |

**Figure 3.5**
Energy balance for average Dutch waste incineration.
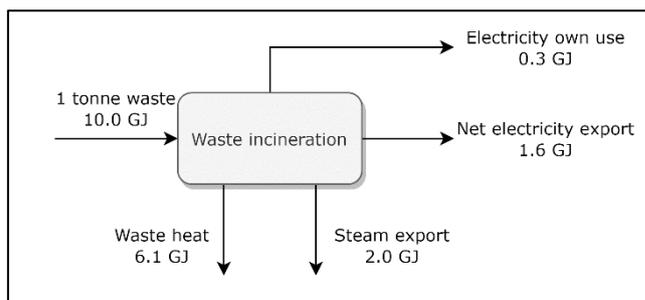


*Functional unit of 1 tonne waste with a LHV of 10.0 GJ per tonne. Net cogeneration efficiency is 36%.*

## 3.1.5    Residual products & mass balance

The non-energetic solid products from the incineration process are indicated as stream 10 in Figure 3.1. 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). Bottom ash was used as an IBC construction material until 2020. IBC construction material is applied with special precautions to prevent contamination of soil or groundwater (Ministerie van Infrastructuur en Waterstaat, 2021). Since 2020, other applications of the bottom ash have been found, such as filler material in foundations and concrete structures (AVR, personal communication, 2022). 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 stabilising material in German salt mines (Twence, 2019).

Figure 3.6 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 2.2 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 3.6**
Mass balance for the incineration of 1 tonne of waste.
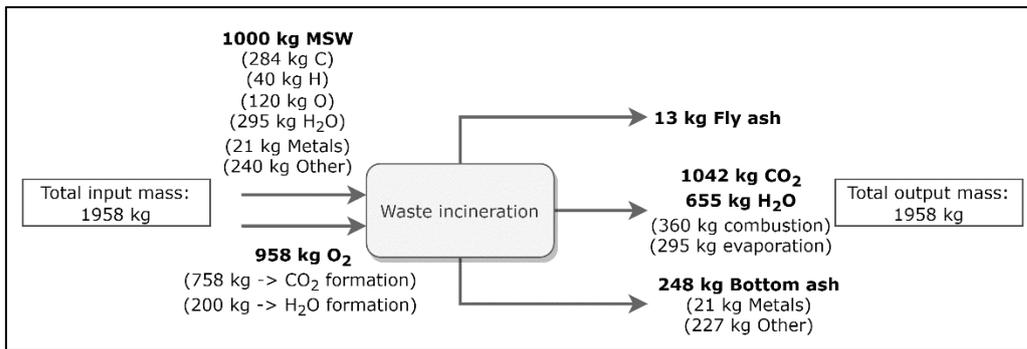


## 3.2 Sludge incineration process

In the Netherlands an worldwide, the dominant technology for sewage sludge incineration is the fluidized bed incinerator (Schnell et al., 2020). The advantage of this technology is that, due to constant and intense mixing of the combustion air and the sludge, the heat transfer is thorough, enhancing full combustion of the sludge. The incineration process is similar to waste incineration, although the high moisture content and energy intensive processes related to by-products of sewage sludge limit the capabilities for heat and electricity exports. Therefore, most Dutch sludge incinerators are either net importers of energy, or are close to being energy neutral (RegioInBedrijf, 2019; SNB, n.d.).

In addition to the baghouse- and E-filters present in municipal waste incinerators, sewage sludge incineration plants may also include an acid scrubber and a neutral scrubber. Most plants add ammonia prior to incineration to reduce nitrogen oxides to elemental nitrogen (Schnell et al., 2020; Te Marvelde et al., 1994). This ammonia is reclaimed using some of the heat from the cogeneration boiler. Figure 3.7 shows the typical process schematically.

**Figure 3.7**
Process flowchart of sewage sludge incineration.
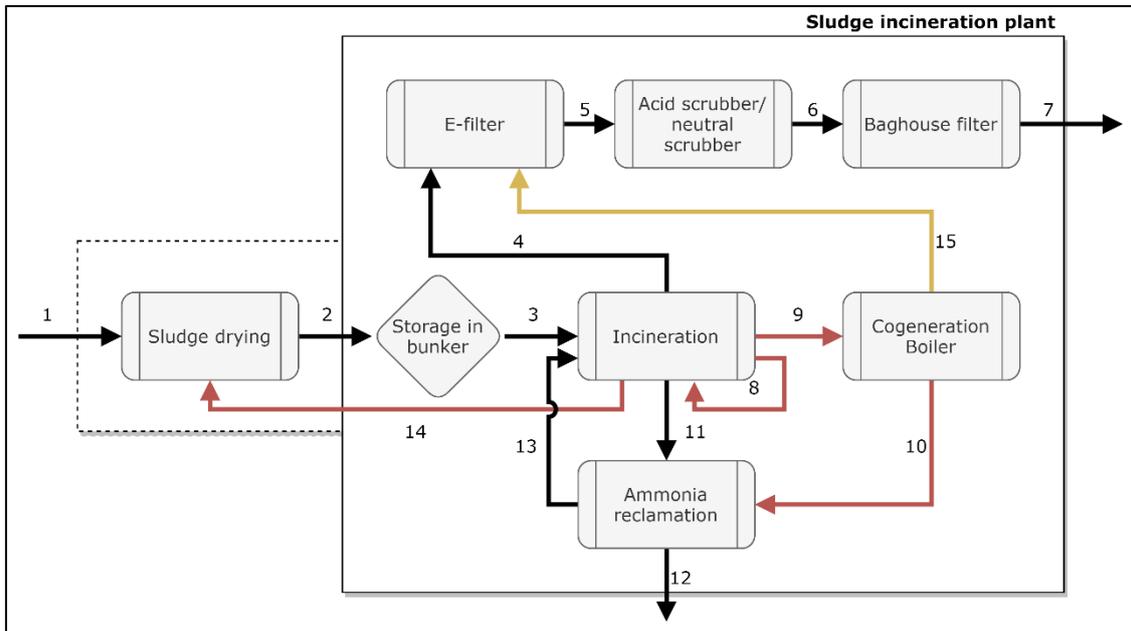


*Numbers indicate different streams. Black, yellow and red arrows indicate mass, electricity and heat flows respectively.*

| Flow number | Flow type | Description | Flow number | Flow type | Description |
|---|---|---|---|---|---|
| 1 | Mass | Wet sludge | 9 | Heat | Heat from incineration of sludge |
| 2 | Mass | Dry sludge | 10 | Heat | Self-consumed heat |
| 3 | Mass | Dry sludge to incinerator | 11 | Mass | Ammonia-rich condensate |
| 4 | Mass | Unfiltered flue gas | 12 | Mass | Waste water |
| 5 | Mass | Filtered flue gas | 13 | Mass | Reclaimed ammonia |
| 6 | Mass | Scrubbed flue gas | 14 | Heat | Self-consumed heat for drying |
| 7 | Mass | Vented flue gas | 15 | Electricity | Self-consumed electricity |
| 8 | Heat | Self-consumed heat | | | |

## 3.3 Thermal soil remediation process

Thermal soil remediation is a technique to remove contaminants from soil, and has been applied by several companies in the Netherlands for a long time. Soil remediation can be done in several ways, such as biological or chemical remediation, but thermal remediation is preferred when fast and reliable removal of contaminants is necessary (O'Brien et al., 2018). One of the downsides of this method of remediation is that the soil is rendered virtually sterile, making it unable to sustaining vegetation. Additionally, the method is highly energy intensive, requiring large amounts of natural gas (Bodemrichtlijn, n.d.). In 2018, 2.4 Mt of soil was remediated, 0.6 Mt of which was through thermal soil remediation (Rijkswaterstaat, 2020).

In the Netherlands, thermal soil remediation is primarily used for heavily contaminated soil containing organic compounds and heavy metals such as mercury, and also for the removal of cyanide contamination (Bodemrichtlijn, n.d.). The Dutch guidelines on thermal soil processing

describe the process as threefold: soil dehydration and pyrolysis, afterburning of contaminated flue gasses and flue gas scrubbing. The soil is first brought to a rotating metal drum where it is heated to circa 300 °C. During this period, the water in the soil completely evaporates and organic compounds decompose into large hydrocarbons. Further temperature increase to 450-600 °C breaks down hydrocarbons and promotes the release of mercury and cyanide contamination. During afterburning, the flue gasses are subject to higher temperatures of up to 1100 °C in order to fully break down hydrocarbons into $CO_2$ and water. This flue gas mixture does contain a high degree of contamination and must be scrubbed afterwards (Bodemrichtlijn, n.d.).

# 4    Options for decarbonisation

This section lists the most viable options for decarbonisation of the currently incinerated waste stream. The section will first discuss the possibility of Carbon Capture and Storage (CCS) and Carbon Capture and Utilisation (CCU). Hereafter, $CO_2$ emission mitigation through the use of alternative (recycling) processes will be investigated. These alternative processes are MSW gasification and plastic pyrolysis, gasification, solvolysis and depolymerisation. These technologies differ in the way their products re-enter the production chain. This is shown graphically in Figure 4.1. It also features a qualitative assessment of the decarbonisation options for thermal soil remediation and sludge incineration. The decarbonisation options are assessed based on their carbon mitigation potential, technological readiness, feasibility and associated costs. The technological readiness level (TRL) can be assessed through the TRL framework as defined by U.S. Department of Energy (2010). This framework is shown in Table 4.1.

**Figure 4.1**
Overview of recycling and carbon capture technologies covered in this research, as well as points on the plastic production chain where recycled materials re-enter this chain. Carbon capture is included.
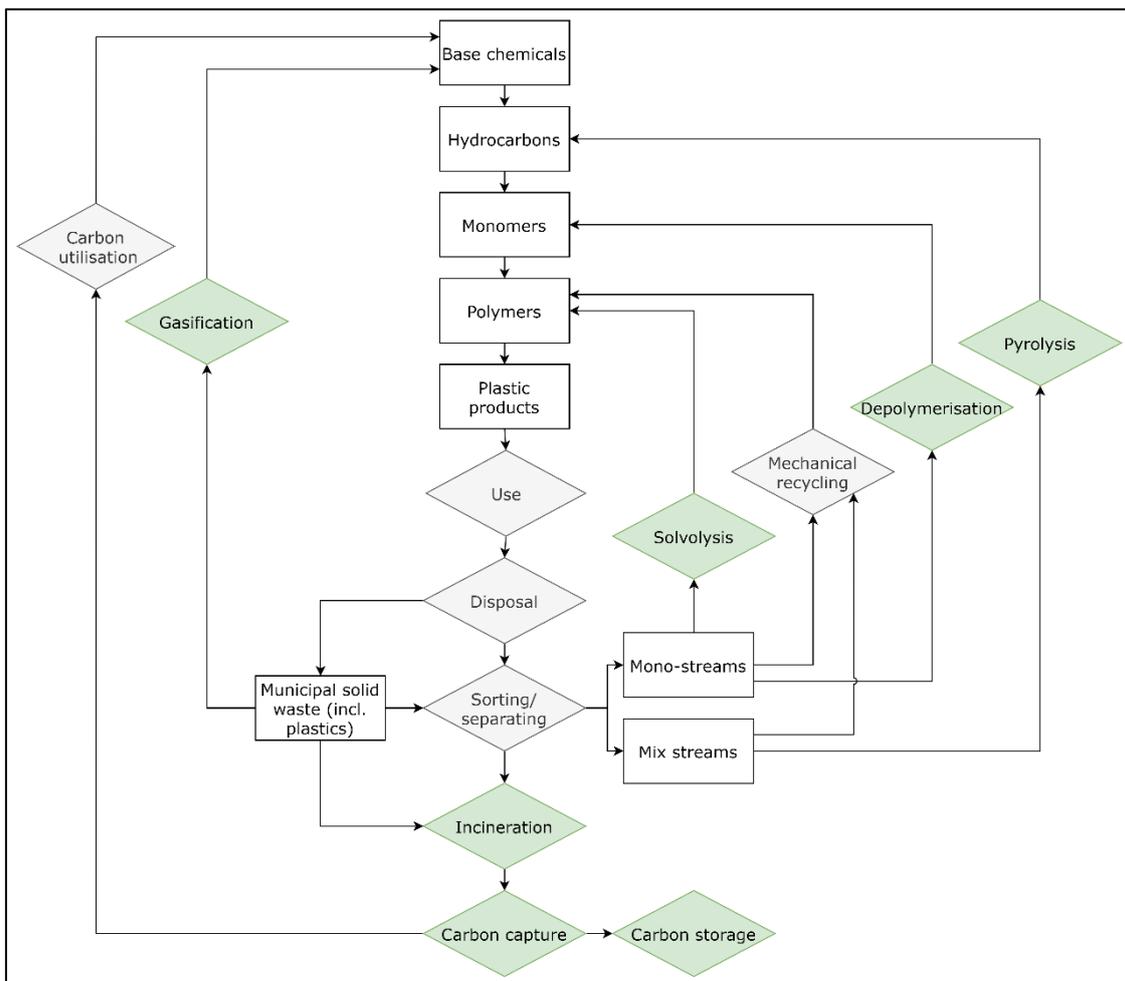
**Table 4.1**
Technology readiness level (TRL) simplified descriptions (IEA, 2014).

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

# 4.1    Carbon Capture for waste incinerators

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 4.2. 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% (Personal communication, 2021). 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 4.2**
Process flowchart of chemical based absorption carbon capture.
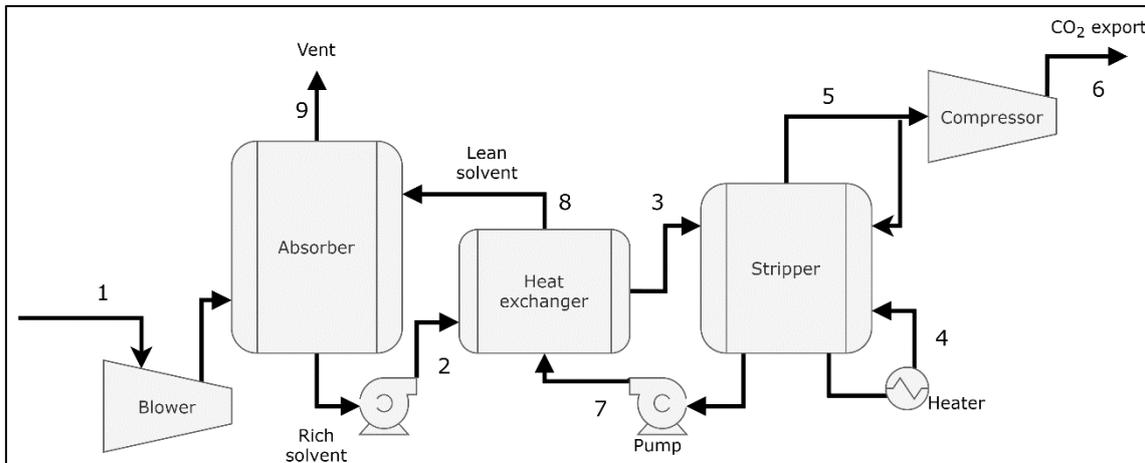


*Adapted from Tang and You (2018) and Mumford et al. (2015).*

| 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 4.2. 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 4.2, after which 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).

### 4.1.1　Carbon capture and storage

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 are not analysed further.

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 (at present emission volumes). In this sense, CCS should be viewed as a transition technology.

### 4.1.2　Carbon Capture and 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

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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).

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 (Lensink & Schoots, 2020).

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 production, in combination with a lower heat demand (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, e.g. via the Porthos infrastructure in Rotterdam.

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:

$$NaOH(aq) + CO_2(g) \rightarrow NaHCO_3(aq)$$
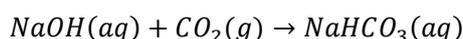
Twence uses the sodium bicarbonate for flue gas scrubbing (Twence, 2020), but the product is versatile in its application and has a global market size of 1.6 billion USD and an expected annual growth rate of 0.8% (More, 2021). If used as a $CO_2$ emissions reduction technology, this type of CCU requires that the product replaces 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. According to representatives from waste incinerators, food-grade application is an unsuitable option for waste incineration CCU plants, despite adequate $CO_2$ purity standards. 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 $CO_2$ (Personal communication, 2021). In the long-term future, it is expected that $CO_2$ can act as a feedstock for many organic compounds, but this technology requires further development.

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 on the short- to medium-term. 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 CCU/CCS summary

Table 4.2 summarises the three identified CCU and CCS variants that are applicable to the decarbonisation of WIPs. 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. Annualised investment cost (AIC) is calculated using the following equation. Here, $r$ is the discount rate, set at 6% for this case, and $n$ is the number of payments. Since the lifetime of the carbon capture unit is assumed to be 15 years, $n$ is equal to 15.

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

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 (Lensink & Schoots, 2020). Revenue from the sale of $CO_2$ to greenhouses is estimated at € 60/t $CO_2$ in consultation with industry (Personal communication, 2021). Any benefits gained in the form of reduced carbon taxes for CCS options are not taken into account in these calculations.

The avoidance rate of $CO_2$, defined as the ratio of $CO_2$ avoided and the $CO_2$ captured, is calculated by accounting for emissions associated with the generation of electricity and heat needed for the processes of carbon capture, compression and liquefaction. The emission factors for electricity and heat are estimated at 0.40 and 0.23 kg/kWh respectively, see Box 4.1. The reduction in fossil $CO_2$ emissions depends on the fraction of fossil emissions in the flue gas of WIPs, which is 36.6%. The resulting cost of fossil $CO_2$ avoided is shown at the bottom of Table 4.2.

**Box 4.1: Emission factors for electricity & heat.**
Emission factors electricity are calculated based on the emission factor for electricity in the Climate and Energy Outlook (KEV) 2021 with the Reference Park Method, which 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. (2014). 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 useful heat, the reference emission factor is a gas boiler with 90% conversion efficiency, making the emission factor 0.23 kg CO2/kWh. It should be noted that, as the Dutch electricity grid moves to net-zero emissions in 2050, the emissions factor of electricity would decrease. Decarbonisation of the heat supply will also cause gas boilers to be a less suitable reference case. These changes to the energy mix beyond 2030 are not taken into account in these estimates, but will substantially impact the net GHG emissions related to processes discussed.

**Table 4.2**
Assumed technical and economic parameters of a reference CCU/CCS plant.

| Reference plant parameters | CCU only (greenhouses) | CCS only | CCUS hybrid (greenhouses) |
|---|---|---|---|
| Operational hours (hours/year) | 4000 | 8000 | 8000 |
| Peak $CO_2$ capture rate (t $CO_2$ captured/hour) | 13.75 | 13.75 | 13.75 |
| Captured $CO_2$ for utilisation (kt $CO_2$ captured/year) | 55 | 0 | 55 |
| Captured $CO_2$ for storage (kt $CO_2$ 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 $CO_2$ sales (M€/year) | 3.30 | 0 | 3.30 |
| Profit (M€/year) | -6.62 | -17.63 | -11.06 |
| Cost per tonne $CO_2$ captured (€/t $CO_2$) | 120 | 176 | 111 |
| $CO_2$ avoidance rate (t $CO_2$ avoided/t $CO_2$ captured) | 56% | 76% | 65% |
| Cost per tonne $CO_2$ avoided (€/t $CO_2$) | 214 | 232 | 170 |
| Fossil fraction of emissions (%) | 0.366 | 0.366 | 0.366 |
| Cost per tonne fossil $CO_2$ avoided (€/t $CO_2$) | 586 | 633 | 464 |

*Data for post-combustion carbon capture using amines, including liquid $CO_2$ transport.*
*Based on SDE++ 2021 data (Lensink & Schoots, 2020).*
*Other CCU than in greenhouses is possible, but no figures were available for this option.*

## 4.2 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. It should be noted that landfilling of these residuals may have other negative environmental impacts besides GHG emissions. Especially the char and tar compounds are likely to contain high levels of toxic chemicals (Arena, 2012).

Before assessing the net emissions from the gasoline produced, the system boundary must be set. Figure 4.3 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 4.3**
Cradle-to-Grave scope for gasification pathway for producing gasoline.
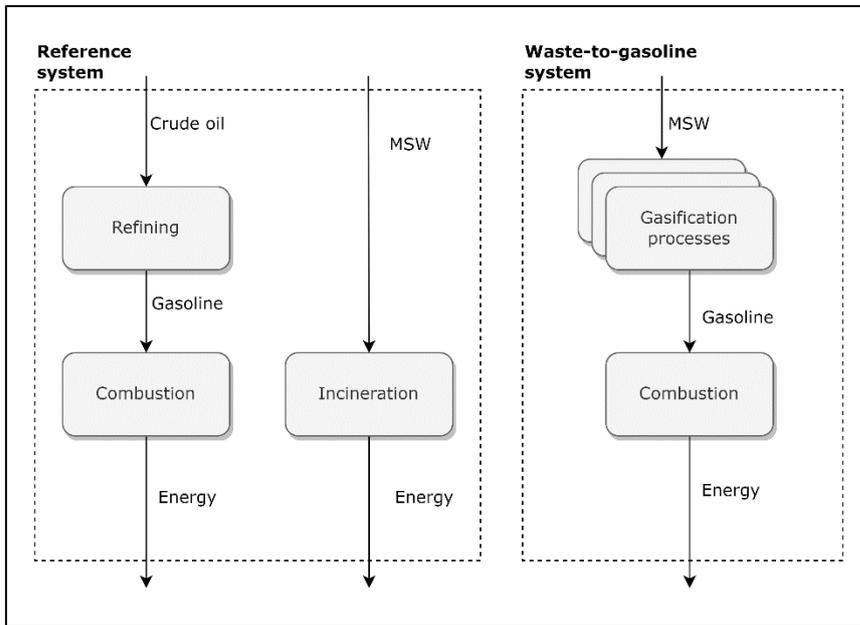


*Dotted line indicates the system boundary. Left is the reference pathway.*

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 4.4. 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. Avoided emissions from electricity and heat are based on the assumption that residual streams are incinerated in an average Dutch waste incinerator with a net cogeneration efficiency of 36%, see details in Box 4.1. AlMohamadi (2021) mentions the processes used are currently at laboratory scale, making the TRL 4-5. Figure 4.4 shows that net $CO_2$ emissions for the gasification and gasoline substitution pathway are higher than net emissions incurred from the reference system when counting both biogenic and fossil emissions. If only fossil $CO_2$ emissions are included, the gasification pathway results in 0.29 kg $CO_2$/kg MSW and the reference pathway has 0.22 kg $CO_2$/kg MSW, leading to the same conclusion of increased net emissions for the gasoline pathway. For this reason, this pathway is not analysed further.

**Figure 4.4**
Net emissions for waste-to-gasoline pathway.



**Net emissions [kg CO₂/kg MSW]**

Legend:
- Process emissions (biogenic)
- Process emissions (fossil)
- Avoided emissions electricity
- Avoided emissions heat
- Virgin gasoline production
- Gasoline substitute combustion (biogenic)
- Gasoline substitute combustion (fossil)
- Virgin gasoline combustion
- Total
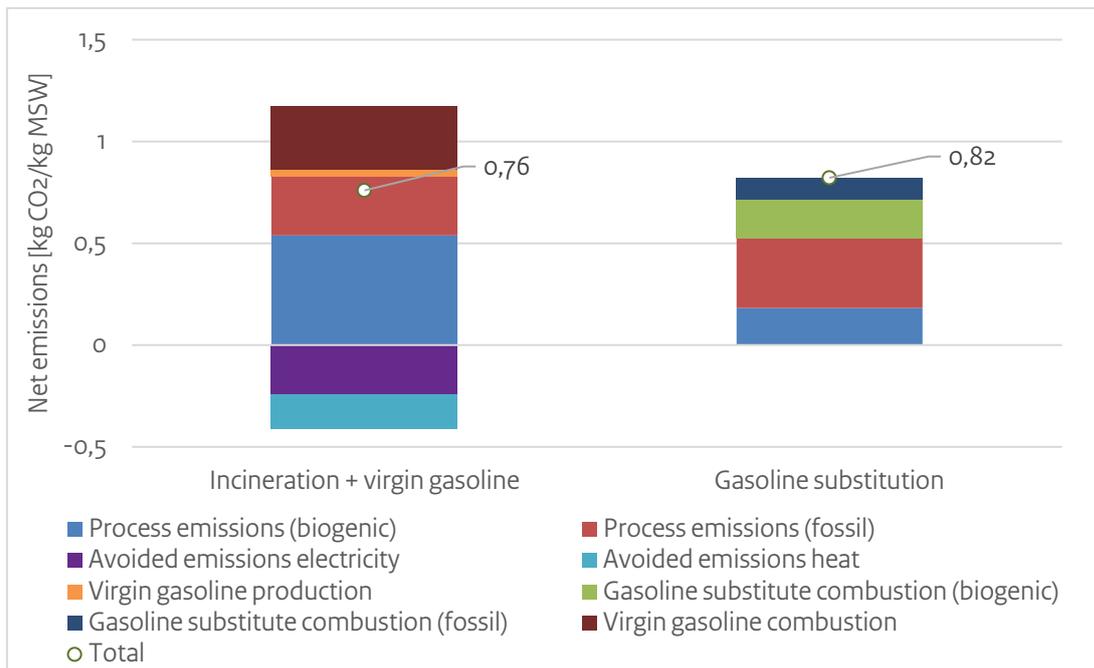
## *Methanol production*

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 4.5. The input for the process is typically not MSW but a derivative called RDF. The assumptions behind the conversion from MSW to RDF are detailed in Box 4.2.

**Box 4.2: MSW-to-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).

The total domestic potential for RDF production in the Netherlands is estimated at 1,700 kt, based on 5,850 kt of domestic MSW and the ratio described above (Rebel, 2021). Since most of the imported waste is already classified as RDF, the total volume of imported waste is available for gasification. This quantity was stable at around 1,700 kt per year before the waste import tax was implemented (Ligthart, 2020). However, since the import tax on waste was implemented, this stream is expected to decline rapidly or disappear in the near future (Haskoning DHV, 2020).

**Figure 4.5**
Process flowchart of Enerkem MSW-to-Methanol process.
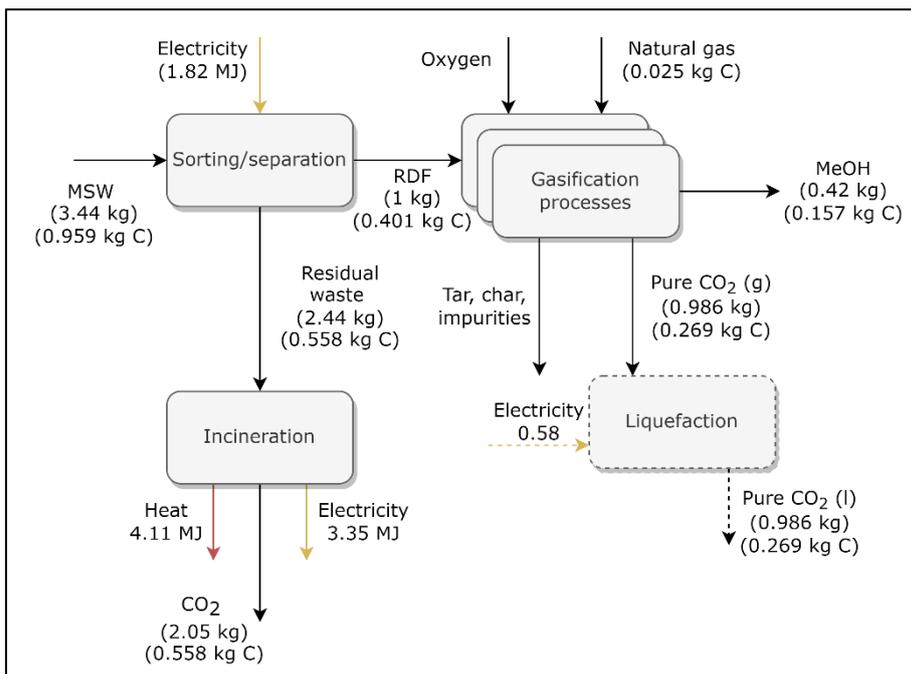


*Process requires a further 0.08 MJ for the air separation unit to supply the oxygen for the system. Dotted lines indicate possibility of CO₂ storage. The system has a 39% carbon efficiency from RDF to methanol. If MSW is included, the carbon efficiency is 16%. Adapted from (Iaquaniello et al., 2017).*

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 of 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 4.3.

**Table 4.3**
Assumed key parameters for the MSW-to-Methanol process for a 100 kt RDF per year plant.

|  | Parameter | Unit | Figure |
|---|---|---|---|
| **Input** | MSW | kt/year | 344 |
| **Input** | RDF | kt/year | 100 |
| **Input** | Electricity | TJ/year | 190 |
| **Input** | Natural gas | kt/year | 2.87 |
| **Output from gasification** | Methanol | kt/year | 42 |
| **Output from gasification** | Pure $CO_2$[a, b] | kt/year | 99 |
| **Output from gasification** | Residual waste | kt/year | 244 |
| **Output from incineration** | $CO_2$ emissions[a] | kt/year | 205 |
| **Output from incineration** | Electricity[c] | TJ/year | 335 |
| **Output from incineration** | 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 4.4. 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 (see Box 4.3 for details on scaling).

**Table 4.4**
Assumed economic parameters for 10, 100 and 1000 kt methanol per year plant for MSW-to-MeOH system using the Enerkem process.

| Parameter | Unit | 10 kt plant | 100 kt plant | 1000 kt plant |
|---|---|---|---|---|
| **CAPEX excl. liquefaction** | 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 4.6. 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 $CO_2$ can be stored via the Porthos infrastructure. Investment and operational costs of running the carbon capture facility are included in the cost figures presented in Table 4.4. Additionally, energy consumption from Table 4.3 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 4.6**
Cradle-to-gate scope for net $CO_2$-eq emission calculations for the Waste-to-methanol system and the reference system.
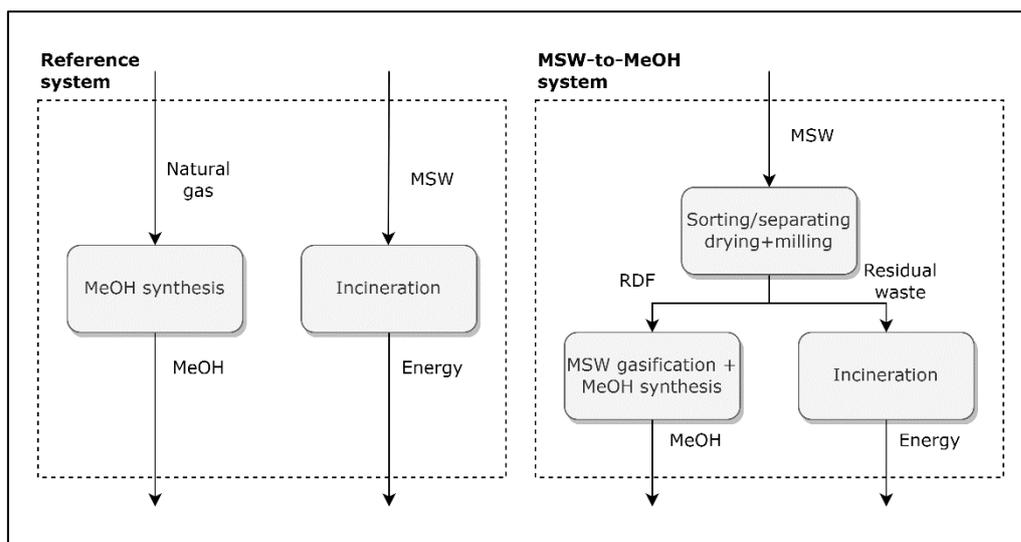


## Net $CO_2$ emissions of MSW-to-MeOH system

The net $CO_2$-eq emissions for both systems are shown in Figure 4.7. 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, see Box 4.1 for details. Avoided emissions from heat based on emission factor of a 90% efficient natural gas boiler, 0.23 kg $CO_2$/kWh. The figure includes a net emission estimation for a gasification configuration including and excluding storage of the pure $CO_2$ stream, as is described above.

**Figure 4.7**
Net CO₂-eq emissions associated with the reference system and the MSW-to-Methanol route, either with or without storage of the CO₂ from the gasification process.
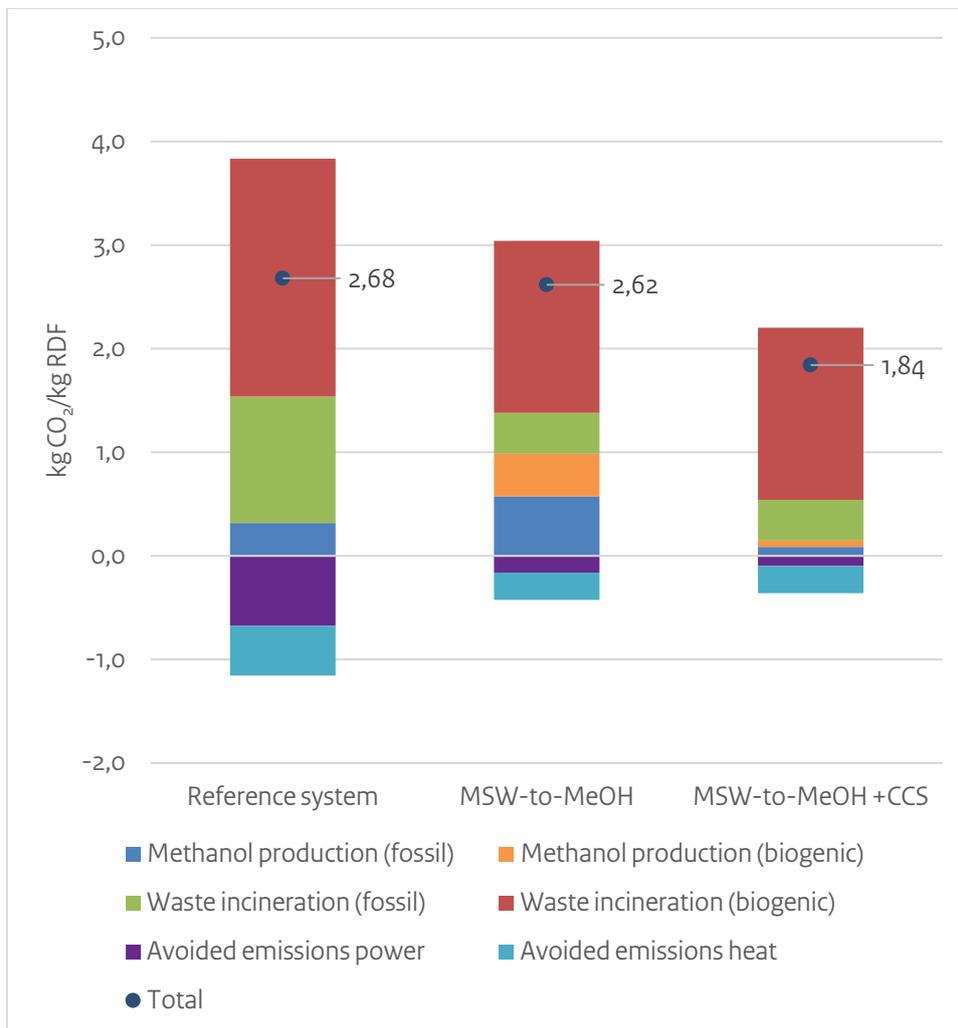


*Functional unit is 1kg of RDF (meaning in 3.44 kg of MSW total).*

The CO₂ emissions related to methanol production and waste incineration shown in Figure 4.7 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 CO₂ 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 4.8. 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 CO₂, 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 total potential feed for gasification, depends partly, as described in Box 4.2, on the availability of imported waste. If only the domestic waste (5,850 kt) is used for RDF production (1,700 kt) and eventual gasification, the total potential fossil $CO_2$ emission reduction is estimated at 1,037 kt, following IPCC GHG accounting guidelines. Importing waste at 2018 levels (another 1,700 kt) could add a similar net emission reduction to the total potential.

# 4.3    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 ⁰C) 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.

## 4.4    Enhanced recycling of plastic waste

This section details potential methods for the decarbonisation of waste management through increased and enhanced plastic recycling. The main reason for the focus on this stream is the high fossil carbon content of plastic waste. An estimated 80-90% of fossil carbon emissions from waste incineration originate from the incineration of plastics (Larsen & Astrup, 2011; Yang et al., 2012) Additionally, plastic waste has the highest energy content of all waste fractions, making it a likely candidate for fuel production. Dutch politics and industry have identified the positive environmental potential of enhanced plastic recycling, as well as the potential for a viable business case. As such, many pilot-scale and full-scale plastic recycling plants are being constructed or are planned for construction in the near future. A list of planned and constructed plants in the Netherlands and Germany is shown in Table 4.5.

**Table 4.5**
Existing or planned (plastic) waste chemical recycling plants. Missing information is indicated by minus sign.

| Technology | Owner (project) | Site | Status | In use | Input | Output | Input (kt) | Output (kt) |
|---|---|---|---|---|---|---|---|---|
| Torrefaction + pyrolysis | RWE (Furec) | Chemelot | FID planned in 2023 | - | Mixed waste | Hydrogen | 350-400 | 40-60 |
| Gasification/Fischer-Tropsch | Enerkem & Shell | Botlek | Permitting procedures | 2026 | RDF | SAF (75%), chemical feedst. (25%) | 360 | 220 |
| Gasification (HT Winkler) | GIDARA Energy (AMA) | Port of Amsterdam | - | 2023 | Biomass & RDF | Methanol and $CO_2$ (for CCU) | 180 | 90 |
| Gasification (HT Winkler) | GIDARA Energy (AMR) | Botlek | - | 2025 | Biomass & RDF | Methanol and $CO_2$ (for CCU) | 180 | 90 |
| Thermochemical liquefaction | Neste & Ravago | Vlissingen | - | - | Mixed plastics | - | 55 | - |
| 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 | Pryme | Rotterdam | Under construction | 2022 | Mixed plastics | Pyrolysis oil | 60 | - |
| Pyrolysis | Pryme | Rotterdam | - | 2024 | Mixed plastics | Pyrolysis oil | - | 350 |
| Solvolysis | PolystyreneLoop | Terneuzen | Operations stopped | 2019 | EPS & XPS | PS polymer & Bromine | 3 | - |
| Pyrolysis | Multiple parties (Pyrolyseproeftuin) | Moerdijk | Constructed | - | - | - | - | - |
| Pyrolysis | Itero | Chemelot | - | 2024 | Mixed plastics | - | 27 | - |
| Upgrading pyrolysis oil | DOW | Terneuzen | - | - | Pyrolysis oil | Pyrolysis oil | 10 | - |

### 4.4.1 Mixed plastic waste recycling feedstock

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 2.3 shows that there are two streams that come into consideration for incineration reduction. These are the pre-collection sorted consumer plastics and the sortable plastic fraction in residual waste. 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 made visual in Figure 4.9. Source separation of recyclable materials is, especially for waste originating from the trade, services and governmental sectors, a promising method of preventing incineration of waste (Haskoning DHV, 2020).

Note that in Figure 4.9, the non-packaging plastic fraction and its component fractions are rough estimates. These estimated are based on the total annual plastic mass processed in the Netherlands (1,650 kt), the total amount of plastic packaging waste processed (496 kt) and the percentage plastic material in incinerated MSW (12%, ~950 kt). These estimates serve to address the fact that a focus on packaging material draws attention from this larger fraction. In further analyses, the non-packaging material is not taken into consideration as a potential feedstock however. The main reason for this is the unknown composition of this material. Due to the size of this fraction, there exists a large underutilised potential for decarbonisation. We therefore stress that a larger focus on this non-packaging plastic waste stream in research and policy making is an important step in decarbonising the waste incineration industry.

**Figure 4.9**
Estimated potential plastic packaging material for alternative processing methods.



*Circle size scaled to total annual mass of the stream (see legend). Green circles indicate plastic waste suitable and obtainable for alternative processing relative to the current situation.*

The utilisation of the separable fraction of the plastic packaging waste in the residual waste stream does necessitate the construction of additional separation facilities, if the total processed waste exceeds the 1.7 Mt installed capacity (Kerstens & Blanksma, 2019). The resulting plastic waste composition is assumed to be similar to Dutch mixed plastic waste (DKR 350) as modelled by Brouwer et al. (2018). DKR 350 is a mixture of plastics and residues that remains after the currently useful fractions have been removed from plastic waste. The composition of DKR 350 is variable, but a typical composition is shown in Table 4.6. Included are the estimated LHV, total carbon content and fossil carbon ratio. These are calculated by Larsen and Astrup (2011). The carbon content for PE and PP is estimated at 85wt% as is done in Fivga and Dimitriou (2018), whose paper was used for a subsequent techno-economic analyses in Section 4.3.

**Table 4.6**
Composition of Dutch mixed plastic waste.

| 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 | |

*The composition of mixed plastic waste is assumed to be identical to DKR 350 as defined by Brouwer et al. (2018). Fossil carbon content from Larsen and Astrup (2011) is assumed to be 100% for plastics and 0% for organic material. Unknown fractions are assumed to be similar to "other" waste in carbon content.*

## 4.4.2 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.

## 4.4.3 Plastic waste 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 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 Ludwigshafen (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. This naphtha substitute can be used to produce base chemicals, which can be used for 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. 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 4.10 and Figure 4.11 respectively.

The 1kg of plastic waste is assumed to be similar to the standard DKR 350 mixed plastic waste stream, shown in Table 4.6. 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.

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 4.10 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 3.6. 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 4.10**
Mass balance based on process flows for waste plastic pyrolysis with hydrocracking.
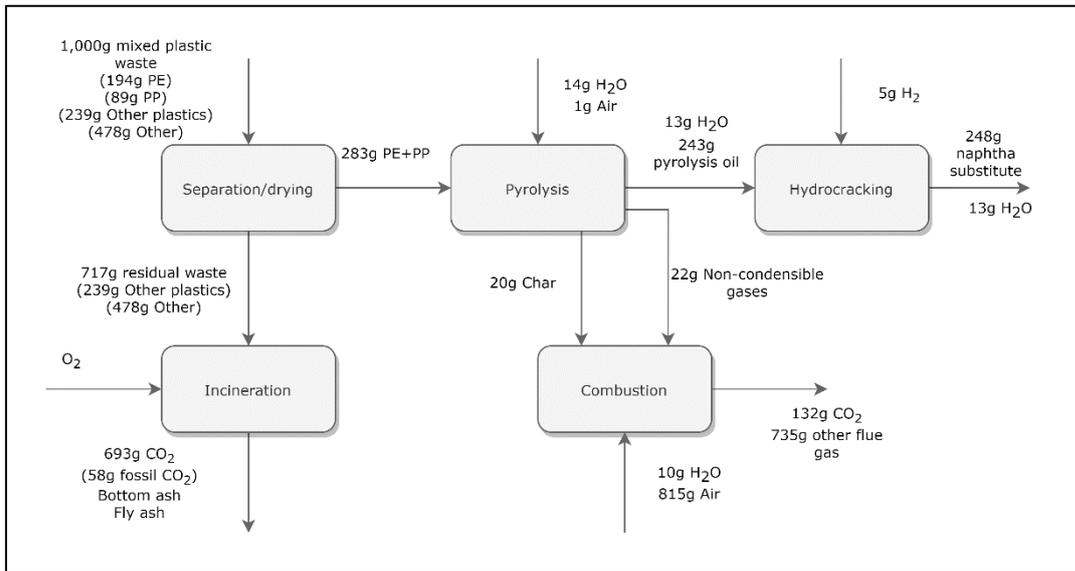


*Adapted from (Fivga & Dimitriou, 2018).*

The energy balance in Figure 4.11 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% net efficiency respectively. The conversion efficiencies are calculated from a weighted average of all Dutch WIPs (Rijkswaterstaat, 2020).

**Figure 4.11**
Energy balance for waste plastic pyrolysis for chemical feedstock substitution.
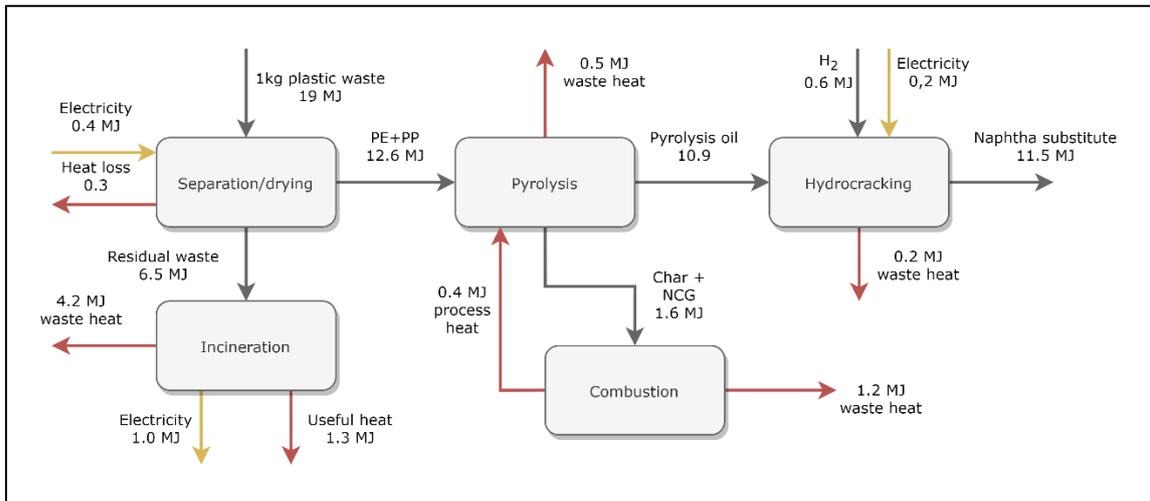


*Energy flows are based on the LHV. The energy efficiency from PE+PP input to naphtha output on LHV basis is 86% accounting for energy imports from electricity and hydrogen. The total energy efficiency from input to useful outputs (including incineration and its benefits) is 69%. Adapted from (Fivga & Dimitriou, 2018).*

Table 4.7 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 4.10. 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 4.7**
Assumed technical parameters for mixed plastic waste pyrolysis for chemical feedstock substitution.

|  | Parameter | Unit | Figure |
|---|---|---|---|
| **Input** | Mixed plastic waste | kt/year | 353 |
| **Input** | Sorted PE+PP input | kt/year | 100 |
| **Input** | Electricity demand | TJ/year | 198 |
| **Input** | Hydrogen | kt/year | 1.83 |
| **Output** | Pyrolysis oil | kt/year | 86 |
| **Output** | Naphtha substitute | kt/year | 88 |
| **Output** | Direct $CO_2$ emissions | kt/year | 47 |
| **Output** | Leftover waste | kt/year | 253 |

Table 4.8 shows the financial data of a mixed plastic pyrolysis plant. CAPEX and OPEX figures are adopted from TNO (2018) and scaled up using a scaling factor of 0.7, see Box 4.3 for details. It is likely that SABIC and Shell could make use of existing infrastructure for hydrocracking (TNO

personal communication, 2021). 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) predicts 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.

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**Box 4.3: Cost-to-capacity scaling method for cost estimation of various plant sizes.**

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 (Reilly, 2015). It follows the general formula below. $C_1$ 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).

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

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**Table 4.8**
Assumed financial parameters for the mixed plastic waste pyrolysis.

| 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 4.12 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 4.12**
Cradle-to-Gate scope for comparing naphtha substitution to the reference case.
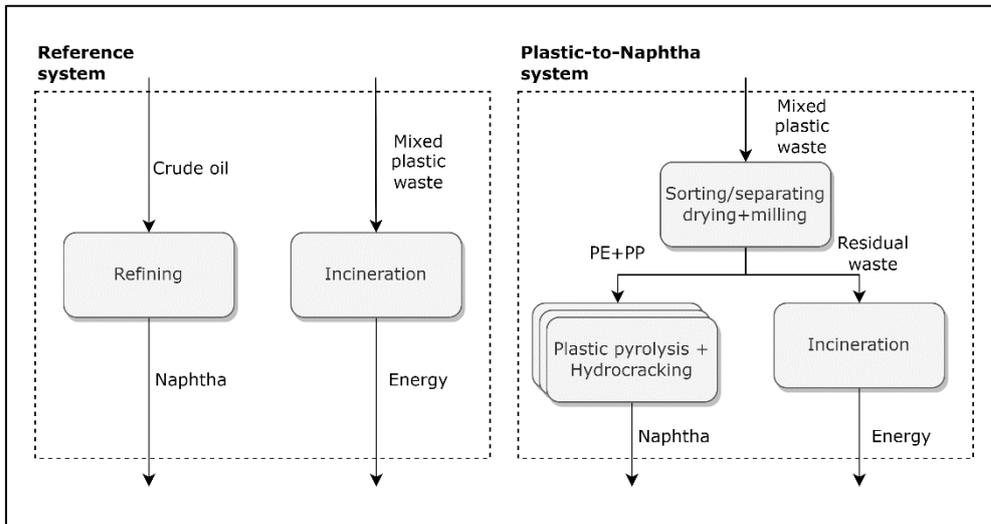


Figure 4.13 shows the results of the net $CO_2$-eq emissions calculation for the processing of 1 kg of DKR 350. The DKR 350 fraction that is not used for pyrolysis is incinerated. It is assumed that all incineration takes place in a Dutch WIP with average useful heat and electricity conversion efficiencies of 20% and 16% respectively. The emissions and energy production from incineration are calculated with the emission factors and LHVs from Table 4.6. 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. See Box 4.1 for details on electricity and heat emission factors. These emission factors are multiplied with the LHV and mass of the input plastic waste. For DKR 350 incineration, the LHV is estimated at 19 MJ/kg (see Table 4.6 for details).

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 4.13 shows that the emissions from incineration of waste are more than halved, as is shown by the height of the red section decreasing from 1.58 to 0.69 kg $CO_2$/kg DKR 350. 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 2.2.1, the total potential for additional plastic recycling is estimated at 166 kt per year. Thus, the total emission reduction potential of this system is 55 kt $CO_2$.

**Figure 4.13**
Net $CO_2$-eq emissions for the reference system and the plastic-to-naphtha system for the processing of 1 kg DKR 350.
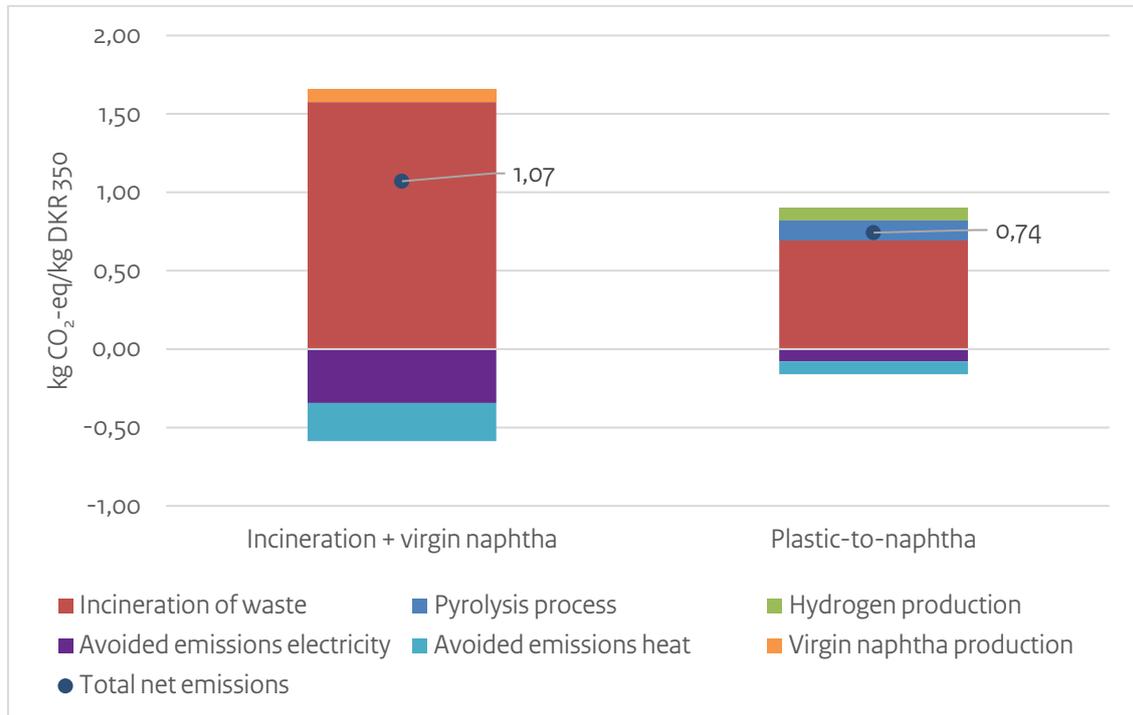


### Fuel substitute production

Fivga and Dimitriou (2018) mention that the pyrolysis oil produced may also be used as a substitute for heavy fuel oil[2] (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 4.10 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 4.9. Financial parameters from Table 4.8 excluding the hydrocracker apply in this case.

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[2] According to the LAP3 (National waste management plan), using waste as a fuel constitutes as a 'useful application' and is only allowed when earlier steps in the waste hierarchy (prevention, reuse and recycling) are not possible. Since this instance only concerns plastic waste that is otherwise incinerated, there is likely no conflict with LAP3. However, when prevention, reuse and recycling are possible in the future, they are preferred.

**Table 4.9**
Assumed technical parameters for mixed plastic waste pyrolysis for fuel substitution.

|  | Parameter | Unit | Figure |
|---|---|---|---|
| **Input** | Mixed plastic waste | kt/year | 353 |
| **Input** | Sorted plastic input | kt/year | 100 |
| **Input** | Electricity demand | TJ/year | 130 |
| **Output** | Pyrolysis oil (HFO substitute) | kt/year | 86 |
| **Output** | Direct $CO_2$ emissions | kt/year | 47 |
| **Output** | Leftover waste | kt/year | 253 |

*Plant is modelled to process up to 100 kt per year of PE and PP.*

The Cradle-to-Grave scope for the HFO substitution pathway is shown in Figure 4.14, along with the reference pathway. In this case, the system boundary includes the combustion of the HFO. This is chosen to more accurately depict the emissions resulting from both pathways.

**Figure 4.14**
Cradle-to-Grave scope for the substitution of heavy fuel oil. Dotted line indicated the system boundary.
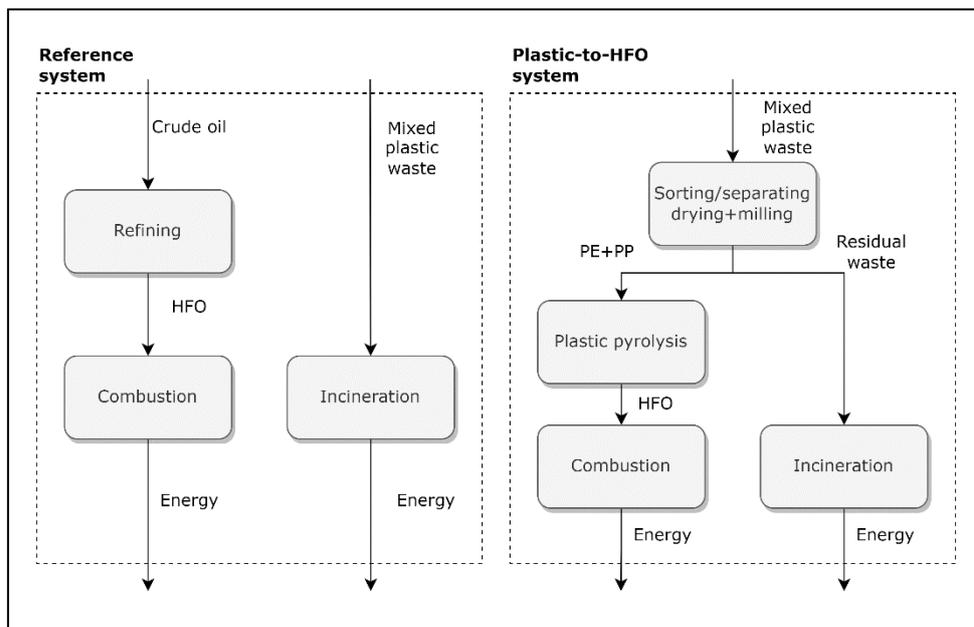


Net $CO_2$-eq emissions are calculated similarly to the calculation for naphtha substitution, with the addition of the combustion step. The emission factor and energy content for crude oil based HFO are sourced from RVO (2020). Emissions from virgin HFO production are based on carbon intensity of Dutch 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 4.15**
Net CO$_2$-eq emissions for the reference system and the plastic-to-HFO system for the processing of 1 kg DKR 350.
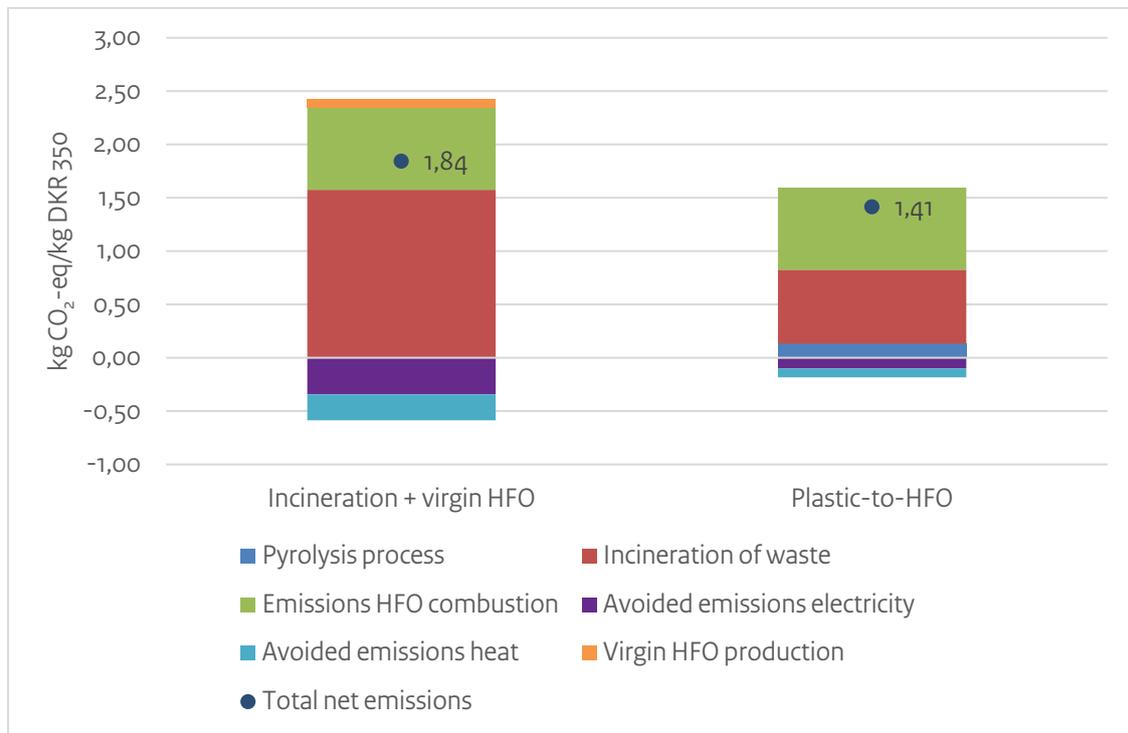


## 4.4.4 EPS Solvolysis

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 (hexabromocyclododecane), 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 0.46 kg CO$_2$-eq per kg EPS input for pre-treatment and other processes respectively. Process emissions are related to the incineration of residual material

that enters the process together with the EPS, such as dirt, cement and other solid impurities (Demacsek et al., 2019).

## 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 4.16.

**Figure 4.16**
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.
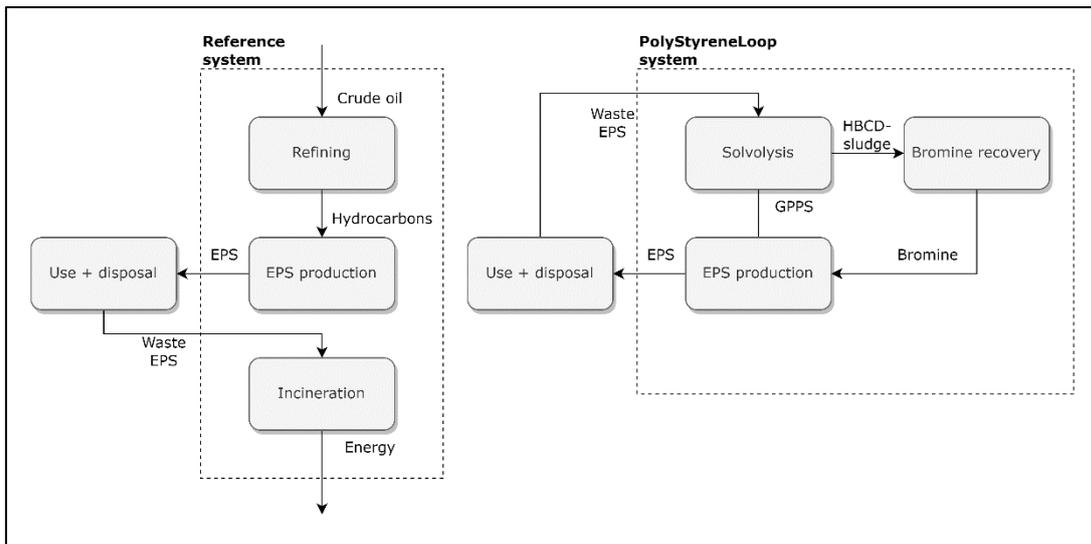


Figure 4.17 shows the estimated net emissions of the systems described above. Negative emissions are applied to the reference scenario to account for the electricity and useful heat that are generated in the WIP. Process emissions for the reference system are the result of oxidation of carbon in the 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 4.17 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 4.17**
Net $CO_2$-eq emissions for the reference and solvolysis pathways for processing 1kg of EPS waste.
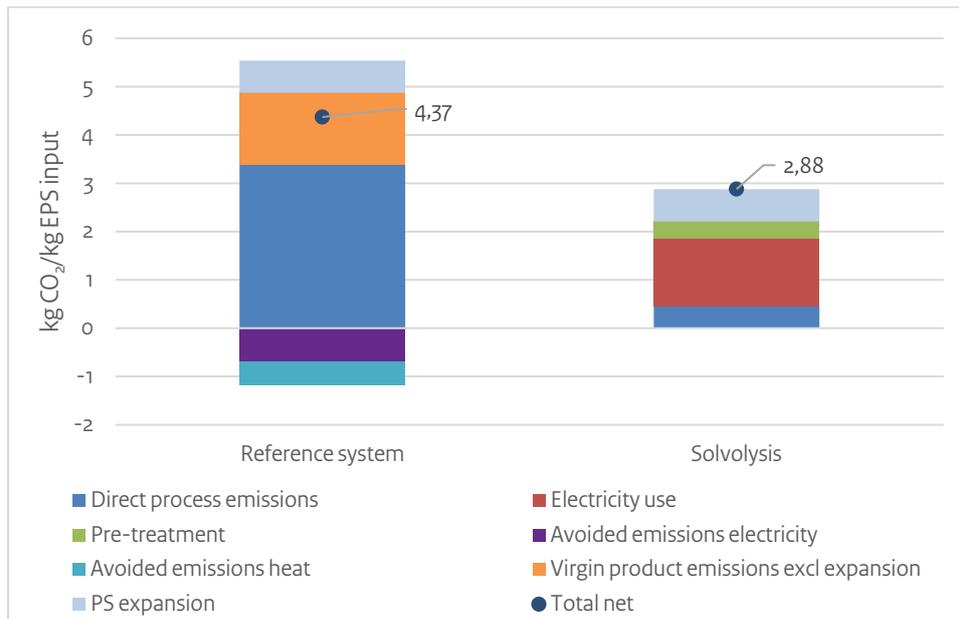


The largest factor in annual expenses for the EPS plant is in the OPEX, which are dominated by energy use for distillation and drying processes. Although process costs are fairly high (~800 €/tonne EPS waste feed), the high value of the product (~1700 €/tonne GPPS) allows the process to likely be profitable (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 4.10.

**Table 4.10**
Assumed 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 | 20 | 50 |
| **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 | 12.10 | 30.25 |

## 4.4.5   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). The process is estimated to have a 95% material efficiency (KIDV, 2018). It is not known where the loss of materials takes place, but inability to deal with certain contaminations or incomplete reclamation of the material after depolymerisation are possible reasons.

### 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 a net electrical and useful thermal efficiency of 16% and 20% respectively. Figure 4.18 shows the scope used for comparing the reference system with the depolymerisation pathway.

**Figure 4.18**
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.
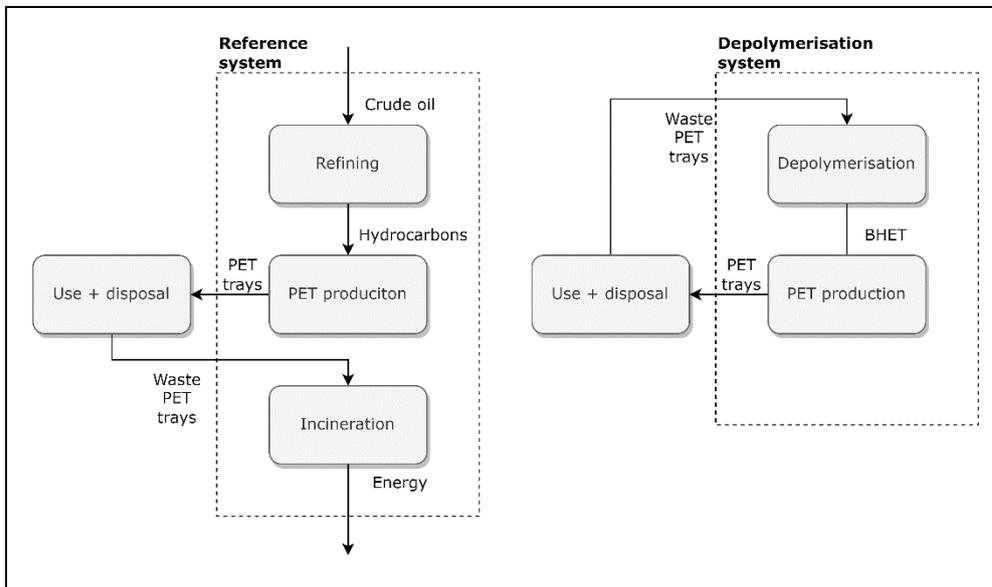


The net $CO_2$-eq emissions for the two pathways for PET tray disposal are shown in Figure 4.19. Avoided emissions from electricity and heat are elaborated above. Incineration emissions are estimated at 2.3 kg $CO_2$-eq per kg of PET incinerated following Chilton et al. (2010). Virgin product emissions 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 4.19, 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. These emission factors are described in more detail in Box 4.1.

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 73-95 kt $CO_2$-eq per year. At 100% effective separation of PET trays from residual waste, the total waste stream of 60 kt would lead to 176-229 kt $CO_2$-eq emissions reduction.

**Figure 4.19**
Net $CO_2$-eq emissions from the reference and depolymerisation pathways for processing 1kg of waste PET trays.
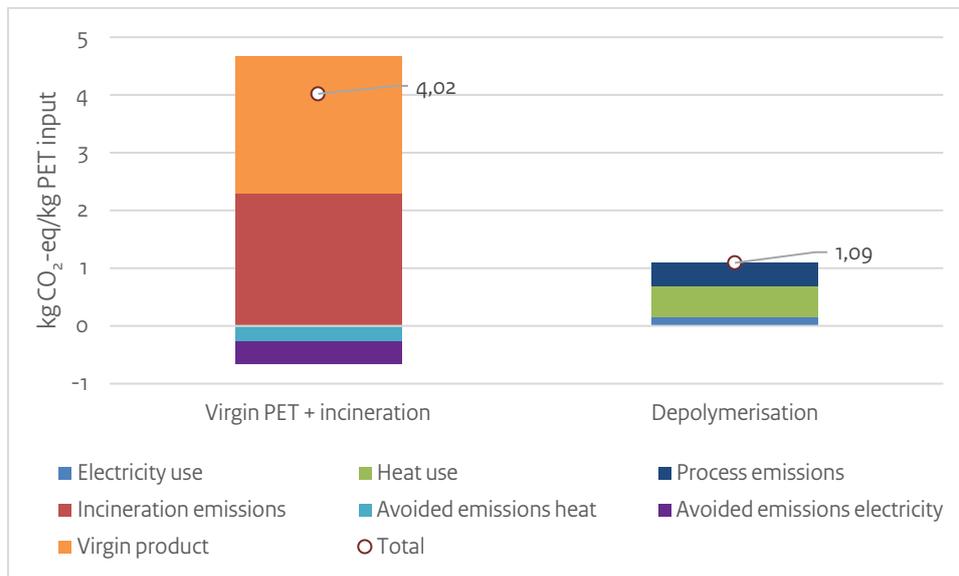


An analysis of the economic performance of PET depolymerisation was done by TNO (2018). The base scale for the plant is 20 kt per year output of recycled PET. For the economic parameters shown in Table 4.11, scaling to different sizes was done using a scaling factor of 0.7 and the methodology described in Box 4.3. TNO further assume that the recycled PET can be sold for ~€100 per tonne.

**Table 4.11**
Assumed 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 PET/year | 5 | 20 | 50 |
| 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 |

## 4.4.6   Mechanical recycling of plastics

Mechanical recycling is the process of breaking down the structure of plastic products without breaking down the chemical bonds. The product is a secondary raw material which can be reshaped into a new products. Mechanical recycling of plastics is, wherever possible, the preferential method of recycling in terms of process complexity, emissions and materials use. The prime example of mechanical polymer recycling is PET recycling. Around 99% of PET bottles collected through the Dutch deposit system are mechanically recycled (Snijder & Nusselder, 2019). Additionally, plastic waste from cars, electric and electronic equipment and pre- or post-collection separated MSW plastic can often be mechanically recycled (Vereniging Afvalbedrijven, personal communication, 2022).  However, in many cases, mechanical recycling is not possible. This may be due to contamination of the feed, layering of multiple materials or the use of certain additives.

In 2020, ~320 kt of plastics were mechanically recycled. KIDV (2021) predicts that in 2030, 340 kt of plastics will be mechanically recycled. This is a 6% increase relative to 2020, and it is mostly attributed to increased production volumes of virgin material. Due to the maturity of the Dutch mechanical recycling industry, the impact of additional policies or technologies is not expected to have a large effect on the recycled volume. Therefore mechanical recycling is not investigated further as a potential decarbonisation technology.

# 4.5 Thermal soil remediation decarbonisation

Although some $CO_2$ emissions from thermal soil remediation arise from the decomposition of organic compounds in the soil itself, it is assumed in this research that this amount is negligible compared to the emissions resulting from the combustion of fossil fuels in the process. These emissions amount to 110-130 kg $CO_2$/tonne soil (or TAG) processed.

### Fuel substitution

Fuel substitution is possible since fossil fuels could be substituted with alternative fuels to supply the reaction heat. The process goes through 3 heating stages. First, primary heating and moisture evaporation is performed (300 ⁰C). Next, pyrolysis of organic compounds and release of heavy metals and cyanide compounds occurs (450-600 ⁰C) and lastly, afterburning of the flue gases (1100 ⁰C). Thus, any fuel that can bring the oven to the desired temperatures qualifies for fuel substitution. Hydrogen and biogas are both viable candidates for fuel substitution.

Biogas has net-zero GHG emissions from combustion since the carbon is 100% of biogenic origin (Chan et al., 2019). Biogas can be sourced from organic waste, sewage sludge, energy crops and more. To replace the energy requirement of 40-50 litres of diesel oil for thermal remediation of 1 tonne of soil, 67-84 $Nm^3$ of biogas would be needed (at a LHV of 21.8 MJ/kg (RVO, 2020)). In case of a gas-fired furnace, minimal adjustments to the plant would be necessary, keeping investment costs low.

In the case of hydrogen, there is a strong dependence of related GHG emissions on the production method for hydrogen. These methods include but are not limited to 1) green hydrogen from electrolysis of water, 2) blue hydrogen from natural gas with carbon capture, 3) grey hydrogen from natural gas without carbon capture or 4) hydrogen from biomass gasification. These types of hydrogen have different emission factors. When using green hydrogen, the emissions can be brought down to zero since no fossil fuels are needed. The TRL for this technology is 9.

### Electrification

Electrification of the thermal soil remediation plants may also be possible. In China, ex-situ thermal soil remediation for mercury removal is done with electric furnaces consuming roughly 237 kWh or 0.85 GJ of electricity per tonne of soil treated (Hou et al., 2016). Based on an emission factor for electricity of 0.40 kg $CO_2$-eq/kWh, full electrification of the thermal soil remediation process would result in 94.8 kg $CO_2$-eq per tonne soil treated. This is 14-27% lower than the current 110-130 kg $CO_2$-eq per tonne soil. Additionally, as the future emission factor of the Dutch electricity grid is expected to decrease, the emissions related to electric thermal soil remediation will decrease accordingly. Investment costs for electrification are expected to be high, since an overhaul of the entire process scheme is necessary. We have not attempted to quantify the costs associated with electrification of thermal soil remediation plants. The existence of full-scale operational electric plants makes the TRL of electric furnace thermal soil remediation 9.

*Utilisation of residual heat*

Dutch thermal soil remediators are reported to have high rates of residual energy utilisation (Bodemrichtlijn, n.d.). However, additional utilisation of heat may be an option to decarbonise the industry further. Residual heat utilisation can be done for processes within the plant, or at neighbouring facilities or district heat networks. In some cases, heat pumps may be necessary to elevate the temperature of the residual heat flow. Residual heat may also be used to produce cooling using an absorption refrigerator (Kampman & Nieuwenhuijse, 2019). Goovaerts et al. (2007) note that, since most residual heat from thermal soil remediation is low-temperature, this heat is suitable for the built environment in the form of district heat networks.

*Carbon capture*

CCU/CCS is also considered a viable option for thermal soil remediators. It is expected that the flue gases from waste incineration and thermal soil remediation are generally the same in terms of composition, especially after flue gas scrubbing. This is expected because both operations involve combustion processes in air. It is therefore assumed that the data from Table 4.2 apply to thermal soil remediators as well.

# 4.6   Sludge incineration decarbonisation

This research focuses on the most concentrated source of GHG emissions from sludge processing, which is incineration. Sludge contains a variety of organic and inorganic compounds. The organic compounds can be sources from fossil carbon since soap and oil residues are present in the sludge. Upon incineration, the resulting $CO_2$ emissions are therefore not 100% of biogenic origin. Tseng et al. (2016) notes that the fossil fraction of the carbon present in sewage sludge generally varies between 25-35%.

The situation for sludge incinerators is analogous to waste incinerators, in that common decarbonisation techniques such as fuel/feedstock substitution or electrification are not an option. The waste sludge needs to be processed one way or another. In the past, utilisation of sludge as fertiliser for agriculture was possible, but this practice has been phased out due to resulting methane emissions. The sludge may be used for further biogas generation. Enhanced biogas extraction or the production of syngas from sewage sludge are possibilities, but are not covered in this research.

As a means of decarbonisation, CCU/CCS may be applicable to sludge incinerators. Both WIPs and SIPs carry out incineration processes in air, so flue gases are expected to have similar compositions, temperatures and pressures after flue gas scrubbing and filtration. The technologic and economical aspects of CCU/CCS for SIPs are therefore assumed to be analogous to those for WIPs, meaning data from Table 4.2 apply. Poblete et al. (2022) modelled such a system, but instead of direct sludge incineration, only the biogas was incinerated. The researchers expressed emission reduction per kg of biochemical oxygen demand (BOD) removed from the sludge. The research found GHG emissions reduction of 6 kg $CO_2$/m³ wastewater processed relative to the reference process without CCS. Currently, SNB already applies a form of CCU, where flue gases are transported via pipeline to a neighbouring facility from Omya (a chalk manufacturing company), where the $CO_2$ is used in the manufacturing process (SNB, n.d.). The quantity of $CO_2$ that is sequestered by Omya is not known to the authors.

## 4.7 Financial analysis for waste incineration decarbonisation

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 the following formula, adopted from Roussanaly (2019):

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

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

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 in Box 4.3. The plant capacities, feedstock prices and product sales prices are shown in Table 4.12.

**Table 4.12**
Assumed parameters for $CO_2$ avoidance cost calculation (TNO, 2018).

| Parameters [unit] | Plant size [kt output/year} | Feedstock cost [€/tonne] | Product sales price [€/tonne] |
|---|---|---|---|
| **Gasification: MSW-to-MeOH** | 100 | 0 | 250 |
| **Pyrolysis: plastic-to-naphtha** | 30 | -50 | 500 |
| **Pyrolysis: plastic-to-HFO** | 30 | -50 | 550 |
| **PET Depolymerisation** | 20 | 100 | 960 |
| **EPS solvolysis** | 20 | 50 | 1720 |

Figure 4.20 shows the results of the $CO_2$ avoidance cost analysis. 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. Gasification without CCS has very high costs and revenues per tonne of $CO_2$ avoided, since the net emission reduction is very small (0.04 kg $CO_2$/kg RDF processed). Thus, to achieve 1 tonne of $CO_2$ avoidance, more activity is needed than for other technologies. Gasification is also the only chemical recycling system with a positive $CO_2$ avoidance cost, largely caused by the high AIC

and OPEX, as well as the relatively low sales price of methanol. Due to the large net emission reduction of the gasification configuration with CCS, the $CO_2$ avoidance cost observed is relatively low. $CO_2$ avoidance costs for WIP CCS and CCU have previously been discussed in Section 4.1. Table 4.13 specifies different components of the $CO_2$ avoidance cost calculation in terms of their contribution to the $CO_2$ avoidance cost shown in Figure 4.20.

**Figure 4.20**
$CO_2$ avoidance cost for the chemical recycling options as well as the configurations for carbon capture.
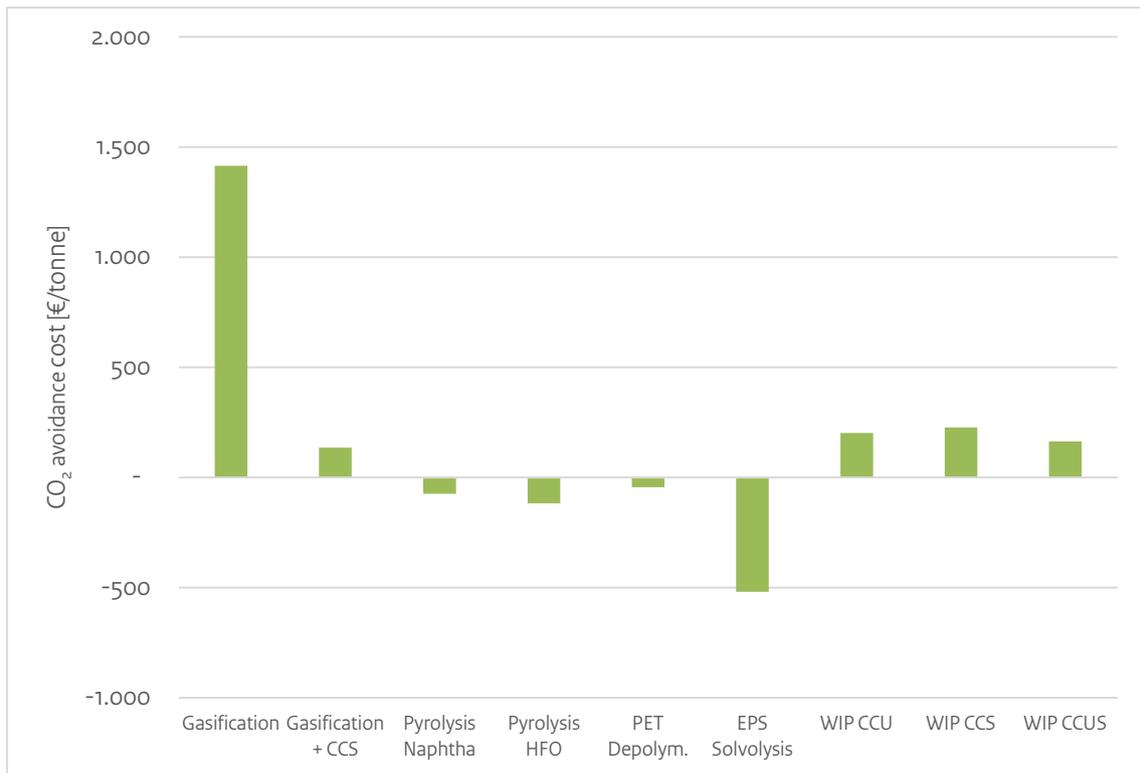
**Table 4.13**
Estimated parameters for the calculation of $CO_2$ avoidance cost for each technology. All values are shows as €/tonne $CO_2$ avoided.

| | AIC | Feedstock cost | OPEX | Revenues | LOI | $CO_2$ avoidance cost |
|---|---|---|---|---|---|---|
| **Gasification: MSW-to-MeOH** | 1,215 | - | 1,194 | -1,668 | 674 | 1,415 |
| **Gasification + CCS** | 105 | - | 102 | -125 | 53 | 135 |
| **Pyrolysis: plastic-to-naphtha** | 100 | -153 | 240 | -378 | 116 | -74 |
| **Pyrolysis: plastic-to-HFO** | 52 | -117 | 171 | -312 | 89 | -117 |
| **PET Depolymerisation** | 29 | 34 | 182 | -312 | 22 | -44 |
| **EPS solvolysis** | 80 | 33 | 385 | -1,094 | 77 | -519 |
| **CCU** | 176 | - | 133 | -107 | - | 202 |
| **CCS** | 71 | - | 155 | - | - | 227 |
| **CCU/S hybrid** | 83 | - | 131 | -51 | - | 164 |

Following IPCC guidelines, biogenic carbon emissions that cause a net change in the biogenic carbon stock have already been accounted for in other sectors, such as indirect land-use change. To avoid double counting, the biogenic fraction of the emissions may be removed. The effect this has on the $CO_2$ avoidance cost only presents itself in the gasification system as well as the CCU and CCS systems. The reason for this is that the other systems only deal with plastic waste streams, which are assumed to contain 100% fossil carbon. The results are shown in Figure 4.21. Gasification without CCS is not shown, since it has no avoided $CO_2$ if biogenic emissions are not counted. This is detailed in Figure 4.8.

**Figure 4.21**
Fossil $CO_2$ avoidance cost for the plastic recycling options and the three configurations for carbon capture.
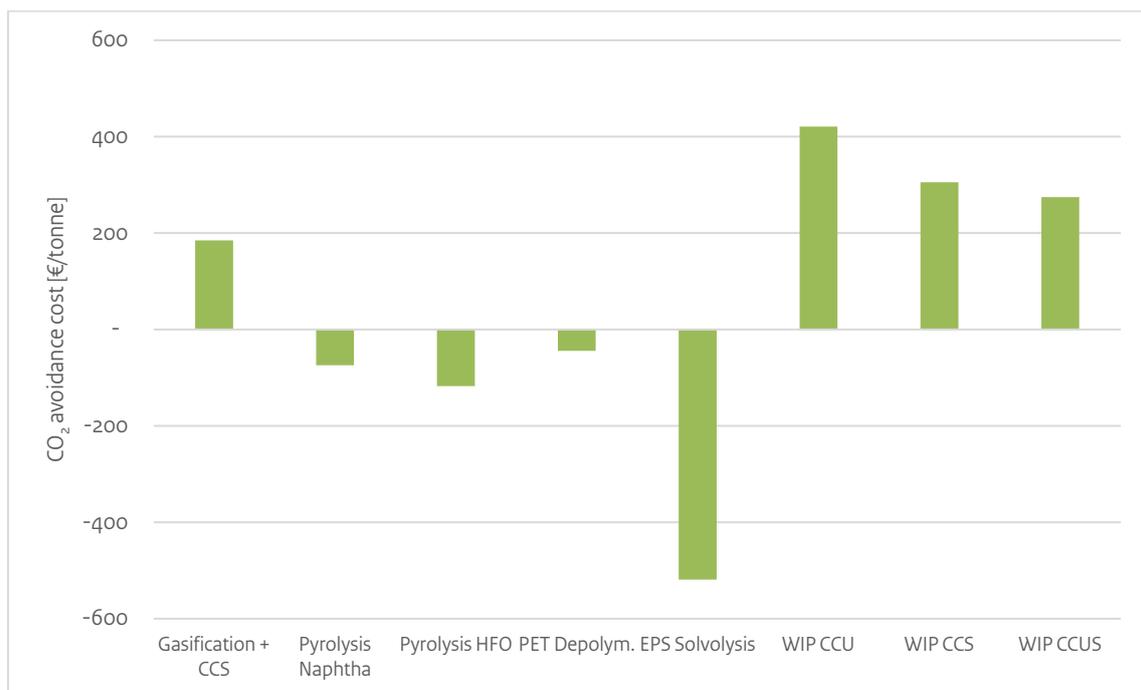
# 5 Discussion

This section provides a discussion on the quantitative results presented in Section 4, as well as a qualitative discussion regarding decarbonisation of waste incineration. Due to the more generic nature of the assessment of SIP and TSRP decarbonisation, these subjects are not covered in this discussion. Instead, more information on them can be found in other MIDDEN publications covering similar decarbonisation technologies. For instance, information about biogas production and fuel substitution to biogas and hydrogen can be found in e.g. Muller et al. (2021). CCUS could also be applicable to SIP and TSRP and is discussed below.

### Qualitative discussion on decarbonising waste incineration

Decarbonisation of the waste management industry is fundamentally different from most other industrial decarbonisation efforts. The waste management industry does not need to produce any product, 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. 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.

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 on the 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). Alternatively, the $CO_2$ may be utilised in chemical industry in processes which run throughout the entire year.

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, presently, 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 $CO_2$, as is shown in Table 4.2. 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.

It should be noted that although CCS is currently a useful method to aid in the decarbonisation of waste incineration, it is not seen as a long-term solution to the problem of fossil $CO_2$ emissions from waste. Similar to other sectors, CCS for waste incinerators is a temporary solution to bridge the gap between now and a more permanent solution. Nonetheless, CCS is projected to play an essential role in the transition during the coming decades. Reusing the carbon from waste as a feedstock for the organic chemicals production is the long-term strategy that also fits in a more circular economy. Complementary, replacing fossil carbon in the production of chemicals and plastics (e.g. by biogenic carbon) reduces fossil emissions of waste incineration too. Enhanced plastic recycling measures are a well-suited solution to this end. A wide variety of alternative plastic recycling methods exist or are being developed to increase the total rate of recycling and avoid the incineration of plastics. A distinction can be made between niche methods and omnivorous processes. Niche methods, such as EPS solvolysis and PET depolymerisation, focus on a specific and very narrowly defined waste stream. While this specificity allows for the development of very efficient processes (leading to low net emission factors), the creation of such highly specific waste streams inevitably creates a stream of "rejects" that often end up in incineration. For omnivorous processes such as mixed plastic pyrolysis, the net emission factor of the process may not be much lower than that of the reference system (incineration and virgin material production). However, the higher flexibility of the process with respect to the input streams enhances the potential feedstock, resulting in fewer "rejected" flows.

In Section 4.4.1, an estimation of the available feedstock of mixed plastic waste for chemical recycling was made. Part of this estimation is based on a post-collection separation efficiency of 50%. In theory, higher separation rates may be possible, thereby increasing the amount of feedstock available for recycling and decreasing incineration GHG emissions. Furthermore, improved source separation of plastics was not included in the estimation of the available feed for chemical recycling. Improved source separation could lead to higher amounts of available feedstock. For instance, Haskoning DHV (2020) estimate that 54-84% of residual waste from the trade, services and government sectors is recyclable. In total, residual waste from trade, services and government sector made up 22% of the incinerated waste in 2018 (Rijkswaterstaat, 2020). Haskoning DHV (2020) states that source separation is a promising method to prevent incineration of this material. To which extent this waste contains fossil carbon and how effectively the material can be separated from the unrecyclable waste is not clear. Finally, the level of decarbonisation related to the effective recycling of these materials depends on the processes applied to recycle them. Where mechanical recycling is possible, and a high material efficiency can be achieved, emissions are likely to decrease significantly when compared to incineration.

As mentioned in Box 4.1, the emission factors used for electricity and heat pertain to the average marginal emission factors for electricity and heat production between 2019 and 2030. For virgin material production, the reference method pertains to the emissions incurred from these processes as they are now, not as they might be in, say, 2050. While the Netherlands progresses in the transition to net-zero GHG emissions, the avoided emissions from heat and electricity production,

as well as emissions from virgin product production, may approach zero. In this way, technologies that are currently penalised for large energy consumption may become more attractive relative to waste incineration over time. Additionally, the net emissions for plastic recycling are heavily influenced by avoided emissions from virgin product production. As these avoided emissions decrease towards 2050, net emissions for these technologies may increase.

### Quantitative discussion GHG emission reduction waste incineration

Figure 5.1 shows the total potential emission reduction, which is the product of the net emission reduction times the potential annual feed. The potential annual feed of mixed plastics is estimated at 166 kt, for EPS 6.5 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. Although PET and EPS recycling have very high emission reduction on a per-kg basis, the total reduction of pyrolysis and HFO substitution is higher due to the larger size of its potential feed. Naturally, the two pyrolysis processes make use of the same feedstock (mixed plastic waste), meaning an estimated 71 kt $CO_2$/year can be reduced by implementing plastic pyrolysis.

The total potential feedstock for the MSW gasification process is limited to the amount of RDF that can be produced. At an RDF yield from MSW of 29% (see Box 4.2 for details), roughly 1.7 Mt are available annually. If RDF imports at 2018 levels are included, another 1.7 Mt would be potentially available for gasification. Without CCS, the gasification pathway does not offer meaningful reduction. When storing the pure $CO_2$ stream that is produced however, an estimated net emission reduction of 0.6 kg fossil $CO_2$/kg RDF can be achieved, resulting in 1,037 kt fossil $CO_2$ reduction potential annually, without RDF imports. If included, RDF imports could double this $CO_2$ emission reduction potential. The total potential GHG emissions reduction resulting from the implementation of mixed plastic pyrolysis, PET depolymerisation and EPS solvolysis is estimated at ~150 kt $CO_2$-eq per year.

**Figure 5.1**
Estimated potential $CO_2$ emission reduction per recycling technology.
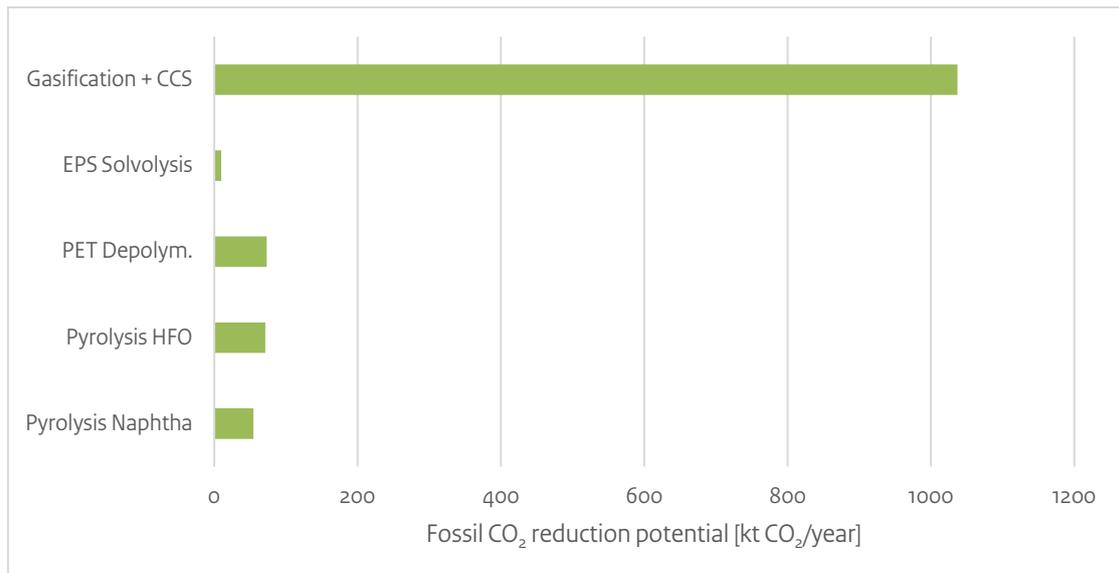
*Additional decarbonisation options waste incineration*

This report has not been focussed on the use of residual heat from waste incinerators. Making adequate use of heat from the incineration of waste is already an important component in the business of waste incinerators. However, given the energy balance for the average Dutch waste incinerator, there is still around 64% of thermal energy loss as residual heat. The differences between waste incinerators are quite large in this respect, where the occurrence of residual heat can be correlated well with a high rate of electricity production at the facility, as is seen in Figure 3.4. There may be opportunities for waste incinerators to change operations to a higher Heat/Power ratio, thereby increasing the overall thermodynamic efficiency of the plant. As the total amount of waste incinerated in the Netherlands decreases due to a reduction in imports and domestic recycling measures, it is probable that more efficient waste incinerators would be given preference due to their more competitive position in the sector. This would, in time, result in an autonomous overall efficiency gain for the entire sector.

Another opportunity which is being investigated by Dutch WIPs is the use of cascading energy flows, where steam may be exported to industrial partners, whereafter it returns as condensed hot water, and subsequently delivered to the built environment or horticulture. The sector estimates that WIPs can export an additional 10 PJ of heat relative to 2020 levels (Vereniging Afvalbedrijven, personal communication, 2022). This would increase the overall net efficiency of the WIPs from 36% to roughly 50% if electricity output remains unaffected. The applicability of such options depend on the demand for heat in the vicinity of the WIP and the economic attractiveness compared to other heating options for the sectors to which the heat could be delivered.

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