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Burn or let them bury? The net social cost of producing district heating from imported waste

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ABSTRACT

In this study, a net social cost framework is applied to provide insights on policy issues relating to the cross-border trade in waste fuel. We estimate the net social cost of using imported waste fuel in a highly efficient combined heat and power plant (CHP) in a cold climate by considering both private costs and benefits as well as external costs related to energy production, alternative waste management and fuel transport. We conclude that using imported waste fuel is beneficial from a societal perspective compared to using biofuel, given the wide range of assumptions regarding technical, economic and environmental characteristics. The net social cost is mainly determined by fuel cost advantages and the external cost of greenhouse gas emissions. External costs associated with transports only marginally impact the net social cost of waste imports for incineration. The results are robust to variation in the excess heat utilisation rate, which implies that importing waste for incineration would also be beneficial in countries with warmer climates where district heating networks already exist.

1. Introduction

In recent years, the concept of circular economy has received a great deal of policy attention in the European Union (EU) and its member states. A cornerstone of the EU Action Plan for a Circular Economy is its waste hierarchy (European Commission, 2015), which establishes a clear social preference order for different residue treatments. It notes that residues should not be landfilled, but instead prevented, reused, recycled or used as fuel (European Parliament, 2008). In the spirit of the waste hierarchy, several member states have either banned or taxed landfilling, which has led to an increase in the cross-border trade in combustible solid waste. This trade is explained by the fact that, in the EU as a whole, there is insufficient incineration capacity to incinerate all of the waste that is currently treated at lower levels of the waste hierarchy, while the incineration capacity that exists is unevenly distributed geographically (Persson and Münster, 2016; Saveyn et al., 2016).

The increasing cross-border trade in waste has been questioned on environmental and moral grounds and is argued to conflict with the proximity principle formulated in EU's Waste Framework Directive (European Parliament, 2008). According to the proximity principle, each member state should establish a network of installations for waste disposal to enable treatment close to where waste is generated. However, the directive does not oppose the trade in waste fuel between member states since it aims to achieve self-sufficiency in the EU as a whole (European Parliament, 2008, Article 16).

A narrow focus on waste incineration as a form of waste treatment ignores the energy system perspective and the fact that combustible mixed waste is attractive to district heating producers in colder countries. In this sense, waste fuel only differs conceptually from other commercial resources (e.g., biofuels, coal, gas, oil etc.) with respect to its low—or even negative—price (Massarutto, 2015). For most commercial fuels export and import is seen as a regular process. In this study, we apply a net social cost approach to determine whether it makes sense from a societal perspective to import waste fuel to countries with a relatively cold climate and a pre-existing district heating network. A cold climate and high demand for district heating make it possible to recover more of the energy contained in waste compared to the more common case in which waste incineration only generates electricity (e.g., Foster et al., 2021; Dijkgraaf and Vollebergh, 2004). Therefore, if using imported waste fuel for energy production is not a good option in cold climates, it is certainly not a good option for countries with a warmer climate, ceteris paribus. In our analysis, we analyse the case of

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Sweden, as an example of a waste importing country in a cold climate. In Sweden, 50% of the residential heat demand is satisfied with district heating (Euroheat and Power, 2021), where solid waste constitutes a large share of the total input fuel. The large use of domestic and imported waste fuel has resulted in several investigations and research projects that discuss the merits of waste incineration from different perspectives, and provide information that is fundamental to calculations of the net social cost of using imported waste fuel.

Previous studies related to the pros and cons of waste incineration largely apply methods that limit their analysis to environmental impacts (life cycle analysis, LCA) and/or the cost minimisation of an energy system (energy system analysis, ESA). In this study, we aim to investigate the welfare effects of incinerating imported waste. From an economic perspective, the fundamental question is whether market failures exist in the form of uninternalised externalities that outweigh the benefits and comparative advantages of utilising cheap input fuels. This raises the following research question: Is the net social cost of waste incineration still negative when all externalities have been considered? In this paper, we compare Swedish district heating based on the incineration of either Swedish biofuel or imported waste from Norway or the UK. In 2019, the UK and Norway combined exported 1.35 million tonnes of non-hazardous waste to Sweden for incineration, which corresponds to 22% of the non-hazardous waste incinerated in Sweden each year (Swedish Waste Management Association, 2019a). In our empirical analysis, we consider environmental impacts, the recovery of energy and metals, as well as private costs and benefits. We also include the environmental impacts of fuel transport and waste treatment outside of Sweden, if these impacts are relevant to Swedish policymakers. To make the results more generalizable, also to other countries, we perform a sensitivity analysis in which we vary a large set of assumptions concerning the technological, economic and environmental factors that determine the net social cost of waste incineration.

The remainder of this paper is structured as follows. Section 2 describes the features of the Swedish district heating and waste management sectors, while Section 3 reviews the previous literature and highlights our contribution to it. In Section 4, we provide a conceptual framework for our empirical analysis, while Section 5 presents the technical and economic features of the options we compared in our net social cost calculations. Section 6 contains the results and a sensitivity analysis of the key assumptions made in the calculations. Our results are discussed in Section 7, while their policy implications are presented in the conclusion provided in Section 8.

2. Swedish district heating and waste management

In Sweden, district heating expanded on a large scale in urban areas during the 1970s and 1980s due to it being a cost-efficient and practical alternative to individual heating solutions for separate buildings. During the 1980–2019 period, the heat output of the district heating sector increased from 30 to 50 TWh. District heating systems currently exist in 283 of Sweden's 290 municipalities and most have a municipal owner (Magnusson, 2016). Over time, there has been a shift in the input fuel mix away from fossil fuels. As a result, the fossil CO_2e[1] intensity of the Swedish district heating sector has fallen drastically. In 2019, the share of fossil fuels was approximately 3%, while the main inputs included solid biofuels (41%), solid waste (22%) and flue gas condensation and industrial waste heat (20%). The remaining 16% of the energy came from peat, bio-oil, electricity and geothermal heat (Energiföretagen, 2020).

In 2019, 43 installations in Sweden burned unprocessed waste, refuse-derived fuel (RDF)[2] or a mix of plastics, paper and wood (Swedish Waste Management Association, 2019b), of which 35 were allowed to treat unprocessed municipal solid waste (Government Office of Sweden, 2017). Facilities compete in an open market for waste fuel supplied by municipalities, industries and foreign actors. Gate fees vary geographically, and in 2019, 90% of Swedish municipalities paid prices in the range of SEK 360–660/t.[3]

The import of waste for incineration in the district heating sector began to increase in 2010, which was partly the result of a deregulated European waste market. This enabled Swedish facilities to access additional waste volumes by providing other countries with a low-cost waste disposal option. The majority of Swedish waste imports originate from Norway and the UK and are transported to Sweden by trucks or freight ships. Although the quality of waste varies, waste from the UK typically has a *heating value*[4] that is higher compared to Swedish and Norwegian mixed waste due to its pre-sorting and lower shares of inert material and metals (Bisaillon et al., 2013).

Waste incineration is regulated by several policy instruments (Government Office of Sweden, 2017; Swedish Environmental Protection Agency, 2021). First, in accordance with the Environmental code (Swedish Parliament, 1998), and regulations following the EUs industrial emissions directive (European Parliament, 2010), incineration plants have to comply with technology and process standards reflecting site-specific circumstances and best available technology (BAT). Second, the facilities are part of the EU emission trading scheme (EU ETS).[5] Until recently, the cost incurred by the EU ETS has played a marginal role in this sector; however, its role has increased substantially due to a sharp increase in the EU ETS price since 2018. Third, the facilities are subject to a NO_X fee. The facilities pay SEK 50/kg NO_X and are then refunded each year based on their production of useful energy (Sterner and Höglund-Isaksson, 2006). Fourth, facilities must pay a landfill tax of SEK 540/t for landfilled incineration residue. In addition to the aforementioned environmental policies, a tax on waste incineration was introduced in 2020 at the rate of SEK 125/t.[6] In summary, the waste incineration sector in Sweden is heavily regulated.

3. Previous literature

A large body of literature has compared the benefits and costs of the use of various fuels in district heating production as well as different waste treatment options. The methods used can broadly be categorised as energy system analysis (ESA), lifecycle analysis (LCA) and welfare analysis (cost-benefit analysis, CBA). The vast majority of existing studies used ESA or LCA and thus focused on either energy system costs or environmental impacts without considering the trade-offs between benefits and costs from an overall societal perspective, which is the objective of CBA (Massarutto, 2015). That said, ESA and LCA often perform site-specific analyses that result in detailed consequences throughout entire energy and product systems, which may be of high relevance to policymakers. Such results can also serve as important

[1] CO_2e = Carbon dioxide equivalents.

[2] RDF is waste that has undergone some form of processing to increase the *heating value*.

[3] Based on our own collection of public percurrent contracts. SEK 10 = € 0.99 = US $ 1.2.

[4] The heating value of fuels can be expressed as either a lower or higher heating value. In this paper, *heating value* always refers to the lower heating value.

[5] Currently, Denmark, Sweden and Lithuania are the only countries that include waste incineration in the European Union Emission Trading Scheme (EU ETS) (European Commission, 2020b; Norden, 2015)

[6] In 2020 and 2021, tax rates are SEK 70/t and SEK 100/t; thereafter, the full tax rate is applied. In practice, the tax is lower since the taxpayers are refunded for material and residues that leave their facilities, such as ash (that will be taxed when landfilled) or recyclables (Government Office of Sweden, 2019b).

inputs in an all-inclusive CBA that aims to estimate the net social cost of energy or waste management production (Kinnaman, 2016; Massarutto, 2015).[7]

The objective of ESAs is to minimize the total system costs to satisfy production of heat, electricity and fuels, and changes in energy production may subsequently be used to study environmental impacts (Pizarro-Alonso et al., 2018b; Levihn and Nuur, 2016; Eriksson and Bisaillon, 2011; Fruergaard et al., 2010; Fahlén and Ahlgren, 2010). A Danish study used integrated system analysis to study the conditions under which imports of combustible waste lead to cost savings in the Danish energy and waste management systems (Pizarro-Alonso et al., 2018a). The analysis included detailed model simulations of waste generation, district heating and electricity production. In the analysis, the private production costs for producing energy and proper waste management treatments were minimised in the short and long run (2035). The results of this study indicate that imports of combustible waste will continue to be beneficial, especially during the winter. From a 2035 perspective, it would make economic sense to expand imports up to 3 million tonnes (i.e., 10 times more than the imports to Denmark in 2014). Notably, this analysis did not include externalities (e.g., environmental impacts). Therefore, it is not possible to draw conclusions regarding the social value of importing combustible waste.

Previous LCA studies that investigate the environmental impacts of waste incineration show that the results are greatly influenced by assumptions defining the energy system and alternative waste treatment. Moreover, two Swedish studies compared waste fuel to biofuels and showed that biofuels are associated with minor negative environmental impacts in all studied scenarios, whereas the performance of waste incineration largely depends on the alternative waste treatment (Eriksson and Finnveden, 2017; Eriksson et al., 2007). If incineration replaces landfilling, the environmental impact is positive; however, the climate impact is negative if incineration replaces recycling. It has also been shown that the type of landfill is important in terms of the overall impact on greenhouse gas (GHG) emissions (Pizarro-Alonso et al., 2018b). Also, several LCAs have shown that results are sensitive to the assumptions made regarding avoided electricity production (e.g., Pizarro-Alonso et al., 2018b; Eriksson and Finnveden, 2017; Fahlén and Ahlgren, 2010; Eriksson et al., 2007) and that these assumptions can be overly simplified (Eriksson and Bisaillon, 2011).

In a few cases, the LCA framework has been extended and used in an economic framework to weight impacts in order to determine an overall environmental impact (e.g., Fahlén and Ahlgren, 2010; Rabl et al., 2008; Eriksson et al., 2007). A study comparing the externality cost of French landfilling and incineration showed that the externality cost for incineration varied in the range of $€_{2006}$ 4–21/t_{waste}, while that for landfilling varied in the range of $€_{2006}$ 10–13/t_{waste} (Rabl et al., 2008). Moreover, this study also found that externality costs for transporting waste up to 100 km[8] were negligible per tonne of waste, which was also observed for landfill leachates.[9]

Two Swedish studies involve a rare but promising combination of LCA, ESA and an economic framework to weight impacts. Miranda and Hale (2001) estimated the social cost of energy production in Sweden for different technologies (heat-only plant, electricity-only plant, combined heat and electricity plant) in combinations with different input fuels (biofuels, coal, oil and natural gas). The results showed that biofuel technologies, which from a private production cost perspective were more expensive than fossil fuel-based technologies, turned competitive when environmental effects were accounted for. However, the study did not include waste fuel. Fahlén and Ahlgren (2010) used a similar approach to analyse the net social cost of producing district heating from different plants in the district heating system of Gothenburg, Sweden, which use waste among other fuels. The results showed that external costs make the social merit order of the plants different from a merit order based on production costs alone. However, this study did not include capital costs or cover issues relating to imports and their consequences in the waste management sector.

In the economics literature, the CBA framework has been applied to evaluate different waste treatment options. For example, studies have used this framework to compare incineration and landfilling (Dijkgraaf and Vollebergh, 2004), incineration and the recycling of plastic waste (Gradus et al., 2017) and different levels of recycling (Kinnaman et al., 2014). Dijkgraaf and Vollebergh (2004) evaluated the net social costs for landfilling and incineration in the Netherlands while accounting for the external and private cost savings arising through each treatment option's recovery of energy and metals. The comparison of net social costs builds on the logic that cost minimisation is a requirement for welfare maximisation. The authors acknowledged that their analysis likely identified a local optimum and that even better treatments may be possible.[10] The results showed that the net environmental cost was $€_{2004}$ 18/t for incineration and $€_{2004}$ 22/t for landfilling.[11] Thus, from an environmental perspective, the incineration of waste is preferable to landfilling. However, the net private cost of incineration is estimated to be $€_{2004}$ 79/t compared to only $€_{2004}$ 36/t for landfilling. Summing the environmental and private cost for each treatment gives a net social cost of $€_{2004}$ 97/t for incineration and $€_{2004}$ 58/t for landfilling. This study concluded that the EU waste hierarchy (European Parliament, 2008) causes allocation inefficiencies since it recommends incineration as a superior waste treatment technology to landfilling. Notably, similar conclusions have been reached by other CBA studies (see review in Massarutto, 2015).

In this paper, we investigate the consequences of incinerating imported waste in the context of a cold climate by using Sweden as a case study. Since a cold climate makes it possible to maximise the energy content of waste, the net benefits might be higher than in countries that have a warmer climate. In this setting, the analysis must also cover the effects of transport while also considering how to value the effects that arise in different legislative areas. Previous ESA and LCA studies typically lack such economic perspective and do not monetise effects to compare different pros and cons to reach meaningful economic conclusions.

We contribute to the existing multi-disciplinary literature on waste incineration by studying the costs and benefits associated with waste treatment in the current paradigm, where waste is traded internationally to serve as input fuel for the production of district heating in colder countries.

4. Conceptual framework – Net social cost

The ultimate goal of net social cost analyses is to compare different states of the world to determine which one is associated with the highest overall welfare level. To accomplish this in practice, one must trace the

[7] A review of 222 published LCAs on solid waste management concluded that LCAs are often case studies that builds on local conditions (e.g., regarding waste composition and energy systems) and that context dependency prevents the meaningful generalization of results (Laurent et al., 2014). This is also true for ESA (Eriksson and Bisaillon, 2011).

[8] 100 km is assumed to be one-way distance. Externalities from return transport are credited to returning goods.

[9] Concerning the approach to weighting environmental impacts, the LCA guideline provided in ISO (14040,2006) suggests that LCAs do not typically include an economic assessment and that it should be acknowledged that no agreed scientific basis exists for weighting environmental impacts to an overall environmental score (Cleary, 2009).

[10] A generic feature of CBA is that a discrete number of alternatives are investigated. The results only identify the optimal solution if it is included in the set of alternatives.

[11] In Dijkgraaf and Vollebergh (2004), the term *net* refers to both the benefits and costs associated with a specific treatment option being considered.

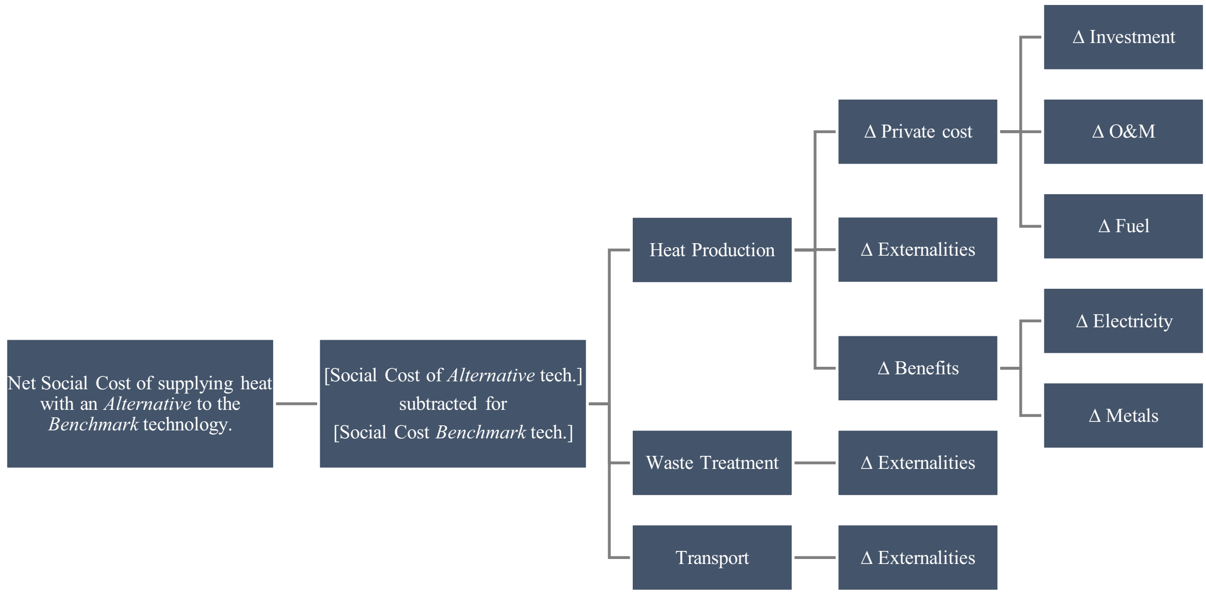

Fig. 1. Analytical framework for comparing the net social costs of incineration when using an alternative technology compared to the benchmark technology. The delta symbol refers to a difference in cost between the alternative and the benchmark.

changes of producer and consumer surpluses associated with changing from one state to another. The benefits are related to the willingness to pay (WTP) among individuals and firms for having the new state, while the costs are related to the opportunity costs of the resources required to make the change. While market prices are assumed to be informative on private benefits and costs, they may not reflect the welfare effects of externalities. Therefore, a set of shadow prices are typically used to value environmental and health effects. As will be further discussed later, externalities may be internalised in private costs by employing economic policy instruments (e.g., emissions trading or environmental taxes). In the case of full internalisation, no adjustment using shadow prices is necessary.

In our analysis, we compare district heating options using waste fuel as input, to a benchmark option that uses biofuel. All options are assumed to satisfy the same fixed heat demand (i.e., an equal amount of thermal heat is produced in all cases). In essence, we study the societal cost of satisfying heat demand using imported waste fuel *instead* of domestic biofuel. The incremental cost of supplying heat with waste fuel *instead* of biofuel is referred to as the *net social cost* of the waste fuel-fired district heating technology. Notably, the net social cost can be either positive or negative. It is important to consider that district heating plants may produce several outputs and differ in several characteristics—even if they have the same revenues from selling thermal heat. We note three key sources of differences: (1) the co-production of heat and electricity; (2) alternative waste treatment; (3) the distance that fuels must be transported. Therefore, the consequences of importing combustible waste can be categorised as being related to heat, electricity, waste management or transport. Furthermore, district heating technologies that generate outputs in addition to thermal heat are credited for doing so.

Fig. 1 illustrates that the net social cost of a waste fuel option (referred to as *alternative* technology) will be the sum of differences from the benchmark technology when considering private costs or cost savings (benefits) in district heating production, the external costs of heat production, transport and alternative waste management. Notably, we only give standing to stakeholders in the importing country (i.e., Sweden for this study). Hence, private costs for transport or waste management will not affect the analysis since the waste fuel is assumed to be imported and the transportation cost is paid by agents in the exporting country. External costs outside Sweden are also not accounted for, with an exemption for GHG emissions since the Swedish government has a clear ambition to mitigate climate change and fulfil the aims of the Paris Agreement (Government Office of Sweden, 2019a).

From a societal perspective, all available treatment options are associated with different types of costs. First, there are costs for using capital, labour and other resources, which are labelled investment, operational and fuel costs. Since these costs are paid by someone in monetary terms, they are considered private costs. Second, regardless of the fuel used, energy production causes negative externalities in the form of air and/or water pollution, smell, noise and negative visual effects on the landscape. These external costs are most often not paid by someone in monetary terms. Instead, they are realised in terms of the decreased well-being of households or decreased profits from business activities. Based on this reasoning, it follows that the net social cost of using imported waste fuel is different from the private cost. For the incineration of imported waste to be beneficial for society, the net social cost must be negative. If more than one option is compared to the benchmark, the option with the lowest net social cost is preferable if the net social cost is negative.

5. Inputs to the net social cost calculations

5.1. The district heating plants

In this section, we describe the different options that we compare in our empirical analysis. The analysis is based on an existing district heating network, where an old plant must either be replaced to satisfy existing heating demand or a new plant must be installed to satisfy a greater demand for district heating.[12] In the analysis, waste treatment services, metal recovery and electricity generation are by-products that decrease the cost of producing heat. For each option, we describe the technical, economic and environmental characteristics of the production plants as well as characteristics associated with alternative waste management and fuel transport (see Tables 1–3). Production costs presented do not include regular taxes, environmental taxes or levies.

[12] In both cases, costs associated with a district heating network will be the same and cancel out in a comparison. Therefore, such costs are not included in the analysis.

Table 1
Technical and economic characteristics of the district heating plants compared in the net social cost analysis. Economic characteristics are exclusive of taxes. All monetary values are in SEK_{2019}.

	Unit	Benchmark HOP Bio	Option A CHP Waste	Option B CHP RDF
Technology	Type	Grate	Grate	Fluidised bed
Economic lifetime	Years	25	25	25
Total input	kTonne/year	223	203	182
Heating value	GJ/t	9.36	12.00	14.04
Operating hours	Full load hours/year	6500	6500	6500
Efficiency	GWh out/GWh in	100	105	104
Heat	GWh	580	580	580
Electricity	GWh	–	130	158
Power-to-heat ratio	% share	–	0.22	0.27
Iron recovery	kg/MWh_{th}	–	6.3	0
Aluminium recovery	kg/MWh_{th}	–	0.7	0
Investment cost	MSEK/year	35	141	120
Investment cost	SEK/MWh_{th}	60	242	207
Operating cost	SEK/MWh_{th}	39	165	157
Fuel cost	SEK/MWh_{fuel}	200	−165	0
Fuel cost	SEK/MWh_{th}	200	−192	0
Electricity crediting[a]	SEK/MWh_{th}	–	−90	−109
Metals crediting[b]	SEK/MWh_{th}	–	−8	–
Private cost (sum)	SEK/MWh_{th}	299	118	255

[a] The electricity price is assumed to be SEK 400/MWh.
[b] The market price is SEK 1/kg for iron and SEK 4/kg for aluminium.

Table 2
Emissions from the district heating plants compared in the net social cost analysis.

Emission factor	Unit	Benchmark HOP Bio	Option A CHP Waste	Option B CHP RDF
Heat production				
CO_2e (fossil)	kg/GJ_{fuel}	–	38.45	22
NO_X	kg/GJ_{fuel}	0.060	0.050	0.050
SO_2	kg/GJ_{fuel}	0.010	0.002	0.002
Dioxins (air)	$\mu g/GJ_{fuel}$	0.030	0.030	0.030
Alternative waste treatment				
Landfilling GWP	kg/t	–	330	330

To determine the configuration of the district heating plants, we rely on technical and financial data from a commonly cited expert study from Sweden (Nohlgren et al., 2014).[13] This study focused on electricity production from a private investment perspective, which is reflected in the studied scenarios and the use of high discount rates (6 and 10%). Moreover, the plants are not assumed to satisfy the same heating demand and thus differ in the number of operating hours. In our analysis, we use data from Nohlgren et al. (2014) but make new calculations that are aligned with a societal perspective and better fit the scenarios under study. We use a social discount rate of 3.5%, which is in line with common practice in OECD countries (OECD, 2018). To facilitate the comparison of the different options, we present all values as per MWh of thermal heat (MWh_{th}) (deliverance of useful heat). Our normalisation differs from the one used in the previous literature, which present values as per tonne of waste treated or per MWh of electricity produced. We motivate this by the fact that heat production is the main revenue from waste incineration in the Swedish context (Nohlgren et al., 2014). All economic data in our study are presented at the 2019 price level.[14]

[13] The data are cross-checked against a public report from Denmark (Danish Energy Agency, 2020) to verify their accuracy. The outcome is that the 2014 study still contains state-of-the-art technologies.

[14] The Swedish Consumer Price Index has been used for price conversions (https://www.scb.se/en/finding-statistics/statistics-by-subject-area/prices-and-consumption/consumer-price-index/consumer-price-index-cpi/pong/tables-and-graphs/consumer-price-index-cpi/cpi-1949100/).

Table 3
Emissions from the transportation of fuel.

Emission factor	Unit	Truck[a]	Ship[b]
CO_2e	g/t-km	22.3	21
NO_X	g/t-km	0.07	0.43
PM^{10}	mg/t-km	6.67	0.01
$PM^{2.5}$	mg/t-km	1.01	0
SO_2	g/t-km	–	0.01

[a] Source: ASEK 7.0 (2020).
[b] Source: CE Delft (2016).

The benchmark against which other technologies are evaluated should be the district heating technology that would serve as the alternative to district heating production using imported waste fuels. Two biofuel-based technologies could constitute this alternative: a biofuel-fired heat-only plant (HOP) or a biofuel-fired combined heat and power (CHP) plant. Although both plant types are common in Sweden, the attractiveness of biofuel-fired CHP plants has declined with the currently low electricity price and nearly absent subsidies for producing electricity from renewable sources.[15,16] Comparison of these two technologies suggests that biofuel-fired HOP plants represent the better benchmark since they are less costly in terms of both private and externality costs.[17] Hereafter this technology is referred to as *HOP Bio*.

In the benchmark scenario, 580 GWh of district heating is produced using wood chips at SEK 200/MWh_{fuel}.[18] The plant is assumed to be at

[15] For biofuel-fired CHP to be less costly compared to biofuel-fired HOP, the electricity price would need to increase to SEK 500/MWh from the current price of approximately SEK 300/MWh.

[16] Since 2003, a renewable obligation quota system has been the main instrument for subsidizing the production of renewable electricity. The price on obligation certificates was approximately SEK 400/MWh in 2008 but is currently close to zero (http://skm.se/priceinfo/).

[17] Table 1 only presents the characteristics of a biofuel-fired HOP, which is the assumed benchmark. The characteristics of biofuel-fired CHP plants are presented in Appendix, Table A1.

[18] Cross-checked against the Swedish Energy Agency (2020).

baseload capacity as such it operates whenever there is a heat demand.[19] In our main calculation, we assume an electricity price of SEK 400/MWh, which is higher than the average price over the last decade (SEK 320/MWh). We argue that the current trends of electrification, increased transmission capacity to continental Europe and a higher price in the EU ETS will place upward pressure on the electricity price.[20]

A CHP plant is more expensive to build and operate than a HOP, and more advanced flue gas cleaning equipment is required since higher temperatures are generated by a CHP plant. By default, advanced and expensive cleaning systems are required for waste incineration, which makes it less expensive to install power generating capacity compared to the biofuel case. Therefore, we do not consider a waste-fired HOP. In our comparison to the benchmark, we include two types of CHP plants that use waste fuels:

1. The typical case with a grate plant that is allowed to burn unprocessed non-hazardous solid mixed waste and non-hazardous industrial waste (hereafter, these waste types are referred to as *mixed waste*). The gate fee is assumed to be SEK 550/t of mixed waste, which is considered a competitive market price for waste treatment with energy recovery (and waste fuel). The assumed gate fee is based on our collection of procurement data and was cross-checked against an expert report used by stakeholders in the district heating sector (Profu, 2020). Hereafter, this technology is referred to as *CHP Waste*.
2. A more futuristic case with a fluidised bed plant that burns RDF in parts smaller than 100 mm. This means that mixed waste has been processed before it is sent to incineration. We assume that the RDF is of higher quality than mixed waste, i.e., it has been separated such that the RDF contains a minimal amount of inert materials, metals and organic material. We also assume that some plastic waste has been removed to make the fuel better from a climate perspective. The higher fuel quality means that RDF in comparison to unprocessed mixed waste has a higher *heating value*[21] and causes less GHG-emissions. We assume that the gate fee for the RDF fuel is SEK 0/t, which includes the cost of waste processing and transport.[22] Hereafter, this technology is referred to as *CHP RDF*.

In all cases, the treatment of bottom ash is assumed to serve useful purposes (e.g., as construction material in landfills). Moreover, some metal scrap can be recovered from the bottom ash and sold to recyclers. However, fly ash from waste incineration is classified as hazardous waste that must be properly managed.[23] The costs associated with handling bottom and fly ash are included in the operational cost.

In summation, the facilities we compare differ with respect to their technical and economic characteristics. Although the waste incinerator is expensive to construct and operate, it has a zero or negative fuel cost. The difference in fuel price between bio and waste fuels (mixed waste and RDF) amounts to SEK 365/MWh_{fuel} and SEK 200/MWh_{fuel}, respectively, under the advantage of mixed waste and RDF.

The information on environmental impacts originates from the Swedish Environmental Protection Agency, which annually updates emission factors concerning the use of different fuels in various sectors (including the district heating sector) to reflect new legislation and trends in emission abatement technologies. In the analysis, we consider the emissions of 16 substances. Existing literature on Swedish district heating production suggests that the list of substances can be shortened to three, with negligible impacts on the environmental cost estimates (Fahlén and Ahlgren, 2012). In Table 2, we present the emission factors for these three emissions plus dioxins, whose potential harm has frequently been a focus of debates concerning waste incineration.[24]

In Sweden, public debate centres on the global warming potential (GWP) of waste incineration. CO_2e emissions are fundamentally dependent on the share of fossil material (i.e., plastics and rubber) in the waste being incinerated. This share has increased over time as more organic waste is sent to anaerobic digestion or composting (Swedish Waste Management Association, 2019b).[25] As shown in Table 2, the difference in emission factors is largest for CO_2e emissions, while there is no difference regarding dioxin emissions. Although there may be no difference with respect to emissions per GJ_{fuel}, a difference may occur per MWh_{th} due to differences in conversion efficiency, the *heating values* and the power-to-heat ratio.

5.2. Replaced waste treatment

As mentioned in Section 5.1, when decisions are made regarding the inputs used for district heating production, decisions are also implicitly made concerning waste treatment. In our case, a decision to use waste fuel will affect waste treatment in other countries because Swedish waste incineration capacity exceeds the levels of domestic waste generation. As a consequence, we assume that the new facility will only use imported waste fuel that would need to be treated elsewhere if not used as fuel in Sweden.[26]

Landfilling with energy recovery is assumed to be the alternative waste treatment for waste imported from both the UK and Norway, although only a small fraction of combustible waste in Norway is landfilled in practice (Miljødirektoratet, 2021). From a system perspective, it is reasonable to assume that landfilling is the alternative treatment because Europe as a whole currently lacks the capacity to satisfy the demand for waste incineration capacity with energy recovery (Sahlin et al., 2015). Norwegian imports of waste fuel from the UK may be a sign of this dynamic (Becidan et al., 2015).[27] Landfilling is associated with substantial emissions of CO_2 and CH_4. CO_2e emissions from

[19] Compared to Nohlgren et al. (2014), the technical and economic data have been adjusted such that the capacity utilisation is higher for a biofuel-fired CHP plant and lower for a CHP plant that burn mixed waste or RDF. Our adjustments increase the annualized investment cost per MWh_{th} for the mixed waste/RDF CHP plants, ceteris paribus. In the original setup, 4800 full load operating hours are assumed for the biofuel CHP plant and 7125 are assumed for the mixed waste/RDF CHP plants, reflecting the merit-order curve of district heating systems that consist of several plants. After the adjustment, all plants have the same amount of heat capacity installed and operated 6500 full load hours. We motivate the adjustment with the assumption that all plants satisfy the same demand for district heat and that no other production unit is affected by the new plant.

[20] SEK 400/MWh is in line with a long-term prognosis by the Swedish Energy Agency (2019).

[21] Less inert and organic materials and metals results in a higher *heating value* whereas less plastics means a lower *heating value*. Here we assume that the net effect of better source-separation is a higher *heating value*.

[22] It should be noted that Swedish municipalities that have RDF plants typical receives mixed waste and process it in facilities close to the incinerator. Here we focus on the incineration stage. Therefore, we assume that the RDF production is done and priced in a prior stage.

[23] It is assumed that the fly ash is sent for storage in a disused limestone quarry in Norway (Nohlgren et al., 2014).

[24] The full list of emission factors is presented in the Appendix, Table A2.

[25] In terms of the fossil coal content in waste, the source separation of organic waste has been more important than the increase in recycling rates to date (Swedish Waste Management Association, 2019b).

[26] A more likely scenario is that the new facility will compete in the waste fuel market and use both domestic and imported fuel, thereby indirectly forcing other facilities to increase their use of imported fuel. From a national perspective, the total effect is approximately comparable to a facility that only uses imported fuel.

[27] Becidan et al. (2015) explains the simultaneous export and import of waste fuel in Norway with a domestic lack of incineration capacity in combination with hard competition from Swedish pants that benefit from economies of scale and favorable energy prices. This means that some plants in Norway cannot satisfy their fuel demand with domestic waste but instead have to import waste fuel from the UK.

landfilling depend on the amount of biogas recovered or flared (instead of vented) to the atmosphere. In our main calculation, we use 330 kg CO_2e/t, which is the average of the range of emission factors analysed in Jeswani and Azapagic (2016). In the sensitivity analysis, we discuss how results change if the alternative treatment is changed from landfilling to incineration.

5.3. Fuel transport

The waste imported to Sweden mainly comes from the UK via freight ships and from Norway by trucks (Fråne et al., 2016). Here, we assume that waste imported from the UK travels 2150 km, while waste from Norway travels 400 km.[28] For biofuel, we assume that wood chip fuel is transported 50 km within Sweden and that transportation costs are included in the fuel cost.[29] The emission factors concerning transport by truck are collected from the public CBA guidelines for the transport sector in Sweden (ASEK 7.0, 2020), whereas the emission factors for maritime transport come from CE Delft (2016). The emission factors for road transport assume a heavy truck with a trailer that has a total freight weight of 30 t.[30]

5.4. Shadow prices

The shadow prices (Table 4) used in our calculations concerning environmental impacts are mainly collected from two different sources. First, to estimate the emissions from heat production, we rely on *Environmental Prices – Handbook EU28 version* (CE Delft, 2018). Second, to estimate emissions from transportation, we rely on the Swedish guidelines for CBA in the transport sector (ASEK 7.0, 2020). For other environmental impacts, various sources have been used.

The shadow prices for emissions to air from district heating plants correspond to averages for the EU28 and concern emissions from a 100-metre stack (CE Delft, 2018). Shadow prices for maritime transport reflect the local environment of the North Sea (Jiang et al., 2014), while the shadow prices used for road transport reflect emissions to air (NO_X and PM) and accidents (ASEK 7.0, 2020).

Since a wide range of values is possible for the shadow price of CO_2, it requires a thorough discussion. The global research community has suggested a range of shadow prices based on damage cost estimations (social cost of carbon) and marginal abatement costs associated with the fulfilment of the two-degree target or the Paris Agreement (CE Delft, 2018). Although the academic literature on social cost of carbon estimates reports a wide range of estimates, the most common estimates lie in the range of US$ 28–220/t CO_2e (Tol, 2018). Recent commissions have suggested that the social cost of carbon (NASEM, 2017) and the required shadow price of CO_2e to fulfil the aims of the Paris Agreement (CPLC, 2017), increase over time and lie in the range of US$ 40–125/t CO_2e. In our main calculation, we use the mean value of US$ 82.5, corresponding to SEK 690.

Besides the Paris Agreement, national climate policies could be used as the basis for a shadow price on CO_2. For example, the EU ETS—of which the Swedish waste incineration sector is part—implicitly values

Table 4
Shadow prices used in the main analysis. Values before adjustment to the 2019 price level.

Emission factor	Unit	Shadow price	Source
Heat production			
CO_2e	SEK/kg	0.69	CPLC (2017); NASEM (2017)
NO_X	SEK/kg	56	CE Delft (2018)
SO_2	SEK/kg	52	CE Delft (2018)
Dioxins (air)	SEK/mg	722	CE Delft (2018)
Transport by ship			
CO_2e	SEK/kg	0.69	CPLC (2017); NASEM (2017)
NO_X	SEK/kg	67	Jiang et al. (2014)
$PM_{2.5}$	SEK/kg	370	Jiang et al. (2014)
SO_2	SEK/kg	91	Jiang et al. (2014)
Transport by truck			
CO_2e	SEK/kg	0.69	CPLC (2017); NASEM (2017)
NO_X + PM	SEK/km	0.14	ASEK 7.0 (2020)
Accidents	SEK/km	1.2	ASEK 7.0 (2020)

CO_2 at € 50/t CO_2.[31] Emissions in Sweden from activities not covered under the EU ETS are subject to a carbon tax of SEK 1200/t. National GHG emission reduction targets in the domestic transport sector demand an even higher price on CO_2 emissions when compared to the carbon tax (Konjunkturinstitutet, 2015). To reach the target of a 70% decrease in emissions during the 2010–2030 period, Swedish guidelines for CBA in the transport sector suggests a shadow price of SEK 7000/t (ASEK 7.0, 2020). In the sensitivity analysis, we discuss the effects of these alternative shadow prices.

In presence of environmental policies, external costs may be internalised to varying degrees.[32] If internalisation is ignored, the climate cost will be counted twice. GHG emissions from waste incineration in Sweden are in practice internalised by the carbon price dictated by the EU ETS. However, our private cost estimates are exclusive taxes and levies, and thus the full shadow price applies. For carbon emissions within the Swedish transport sector, the general carbon tax functions as an internalisation instrument and hence the shadow price is adjusted accordingly. Upon considering GHG emissions from landfilling and incineration in the exporting country, no internalisation is present[33] and the full shadow price applies. In our calculations, the same holds for carbon emissions related to international shipping and road transport from the waste exporting country.

6. Main results and sensitivity analysis

6.1. Main results

Based on the information provided in Section 5, Table 5 presents our calculations for the social cost per MWh_{th} for the benchmark and alternative district heating technologies. In Table 6, we present our main results in terms of net social cost per MWh_{th}. If the net social cost is negative, it implies that the option is beneficial from a societal perspective when compared to the social cost of the benchmark technology.

[28] The distances are based on the locations of two Swedish CHP plants that import waste. Notably, 2150 km is the distance from Hull (on the east coast of Britain) to the Stockholm area (on the east coast of Sweden). This is double the distance compared to if we had assumed that the plant was located in the Gothenburg area (on the west coast of Sweden). Moreover, 400 km is the approximate distance from the Norwegian border to the east coast of Sweden. It should also be considered that all waste treatment options require the transportation of waste and that the distances we use in this study are approximations of the incremental distances for the cross-border trade in waste fuel.

[29] Costs for transportation via trucks include the carbon tax (in the case of biofuel transport) but not the tax for waste transport from outside Sweden.

[30] Assumption based on personal dialogue with waste importing firms.

[31] Price measured in May 2021.

[32] As noted it Section 2, district heating facilities are regulated by technology and performance standards. As a consequence, costly abatement measures have to be implemented that reduces emissions (not GHG) to air and water given any production level. This means that some of the externality cost may be reflected in the private cost. Because we base our conclusions on social costs (summed cost categories), internalised externalities is a minor issue in our analysis. It should be noted that the social cost, and how it is divided between private costs and externality costs, can differ between countries because of differences in regulations.

[33] Landfilling and waste incineration are not subject to carbon tax or the EU ETS in the UK or Norway.

Table 5
The social cost of benchmark and alternative production technologies. All values are presented as SEK_{2019}/MWh_{th}.

	Benchmark HOP Bio	Option A CHP Waste	Option B CHP RDF
Private cost effects			
Levelised private costs	299	215	363
Electricity	–	−90	−109
Materials	–	−8	–
Private cost (sum)	299	118	255
Externality effects			
Combustion	31	137	92
Transport			
Domestic biofuel	1	–	–
UK waste	–	37	33
Norwegian waste	–	8	7
Landfill	–	−83	−74
Externality cost (sum), biofuel	32	–	–
Externality cost (sum), UK waste	–	91	51
Externality cost (sum), Norwegian waste	–	62	26
Social cost, biofuel	333	–	–
Social cost, UK waste	–	210	306
Social cost, Norwegian waste	–	181	280

The results presented in Table 5 reflect private and external costs associated with producing district heating. For the benchmark, where biofuels are used in a HOP, the social cost amounts to SEK 333/MWh_{th} and mainly consists of private costs. External costs are categorised as either related to combustion, transport or alternative waste treatment (i.e., landfilling). The combustion-related external cost is approximately one-tenth of the private cost, while the external costs associated with biofuel transport are negligible.

Regarding private costs, we find that both alternatives with imported waste fuels are preferable to the benchmark HOP Bio. This is expected since it explains why the cross-border trade in combustible waste occurs. Furthermore, we find that external costs related to combustion are substantially larger for both waste fuel options when compared to the benchmark. An even greater difference is observed regarding externalities due to transport, especially when waste fuels are imported from the UK. However, external costs related to transport are still minor in relation to other costs.

As indicated in Section 3, the decision to use waste fuel instead of biofuel will also have secondary effects in the waste management sector. Given that landfilling is replaced, there is a climate benefit in the range of SEK 74–83/MWh_{th}. After crediting for avoided landfilling, waste fuel-fired district heating still yields higher external costs in all scenarios except for the case when RDF is imported from Norway. When private and external costs are summed, we find that waste fuel-fired district heating is clearly preferable in all scenarios, although by a smaller margin when waste is transported from the UK.[34] The comparison between both waste fuel options suggests that the externalities associated with *CHP Waste* are more than twice as large as the externalities associated with *CHP RDF*. Nevertheless, the high private costs of *CHP RDF* (mainly driven by the difference in gate fee) results in a social cost of SEK 280–306/MWh_{th}, which exceeds the social cost of *CHP Waste*.

Table 6 shows that the net social cost is most negative for *CHP Waste* capacity, implying that it is preferable over *CHP RDF*. This corresponds to a social cost for *CHP Waste* that is approximately 40% lower compared to the benchmark alternative. Moreover, the net social cost is lowest for Norwegian waste because the impact of fuel transport is lower when compared to the case of UK waste. Framing the social cost comparison in terms of net social costs is practical for the sensitivity analysis, which is presented in the following section.

Table 6
Results for the net social cost of waste fuel-fired CHP plants in comparison to the benchmark option of a biofuel-fired HOP. All values are presented as SEK_{2019}/MWh_{th}.

	Option A CHP Waste	Option B CHP RDF
Net private effects		
Levelised private cost	−83	64
Electricity	−90	−109
Materials	−8	–
Net private cost	−181	−44
Net externality effects		
Combustion	107	62
Transport		
UK waste	36	32
Norwegian waste	8	7
Landfill	−83	−74
Net externality cost, UK waste	60	20
Net externality cost, Norwegian waste	32	−6
Net social cost, UK waste	−121	−24
Net social cost, Norwegian waste	−149	−50

6.2. Sensitivity analysis

The net social cost estimates presented in Table 6 are uncertain since they are based on a set of assumptions that can be altered to better reflect site-specific circumstances. To account for this, we perform a sensitivity analysis to highlight how the results change if the most influential assumptions are altered. In Table 7, we present impact values (ΔNSC) and switching values (S) for key factors in each step of the net social cost calculations for each option. The impact values give the change of the net social cost following a 10% increase in a single parameter, while the switching value gives the threshold for a single parameter where the net social cost is zero. To facilitate the generalization of our results, we also present the switching values as the percentage change of the specific parameter values required to make the net social cost equal to zero.

The results presented in Table 7 indicate that several parameters have a negligible influence on the net social cost and switching values that are far from the assumed values. This is especially true for *CHP Waste* and when waste fuel is imported from Norway. The least influential parameters on the net social cost of *CHP Waste* include fuel

[34] GHGs contribute to 84% (75%) of the external costs associated with the combustion in *CHP Waste* (*CHP RDF*). The external costs associated with waste fuel transport are dominated by NO_X emissions (59%) in the case of maritime transport and accidents (66%) in case of road transport.

Table 7
Sensitivity analyses with respect to variation in key parameters. *ΔNSC* is the impact on the net social cost (SEK/MWh_{th}) due to a 10% increase in key parameters. *S* is the parameter switching value where the sign of the net social cost changes. $S_{\%}$ is the parameter switching value expressed as percentage change from the assumed value in the main analysis.

–	Option A CHP Waste			Option B CHP RDF			
	ΔNSC	*S*	$S_{\%}$	*ΔNSC*	*S*	$S_{\%}$	*Waste origin*
Operating hours[a]	−12	2270	−65	-8	4605	−29	*UK*
(full load hours/year)	−12	1860	−71	-8	2790	-57	*NO*
Investment cost[b] (MSEK/year)	24	210	49	21	134	11	*UK*
	24	227	61	21	149	23	*NO*
Electricity price (SEK/$MWh_{electricity}$)	−9	−137	-446	−11	311	677	*UK*
	−9	−265	−761	−11	217	444	*NO*
Gate fee	−19	205	−63	−17[c]	-79	[c]	*UK*
(SEK/t_{waste})	−19	124	−77	−17[c]	−158	[c]	*NO*
Biofuel price[d]	−20	79	−60	-20	176	-12	*UK*
(SEK/MWh_{fuel})	−20	51	−74	-20	150	-25	*NO*
Heating value (GJ/t)	10	24	100	−5	33	135	*UK*
	12	26	117	−2	56	299	*NO*
Fossil CO_2e intensity (kg/GJ_{fuel})	34	55	43	11	28	27	*UK*
	37	58	51	13	32	45	*NO*
GWP landfill	−8	-152	-146	−7	222	-33	*UK*
(kg CO_2e/t_{waste})	−8	−265	−180	−7	109	−67	*NO*
Distance	4	9202	328	3	3736	74	*UK*
(km)	1	7431	1758	1	3017	654	*NO*
Shadow price of CO_2e (SEK/kg)	5	2	234	1	3	339	*UK*
	4	3	368	−0.2	−17	−2524	*NO*

[a] Operating hours are adjusted simultaneously for benchmark technology and alternative technology.
[b] Investment cost is only varied for Options A–B and not for the benchmark technology (i.e., biofuel-fired HOP).
[c] A 10% increase is not possible since we assume a gate fee of SEK 0/t for RDF waste. Instead, the ΔNSC-value represents the change in net social cost when the gate fee is changed from SEK 0 to 55/t RDF (change corresponds to a 10% increase in the gate fee for mixed waste). It follows that neither $S_{\%}$ can be calculated.
[d] Price on biofuel affects all options since the benchmark technology is biofuel-fired.

transport distance, electricity price, the shadow price of CO_2e and the GWP of landfilling. The low influence of these parameters is reflected in the fact that they have to more than double to make the net social cost positive.

The sensitivity analysis concerning the *heating value* and CO_2e intensity of waste is relatively complex since it affects the total quantity of waste incinerated. In the case the *heating value* is higher, more useful heat can be extracted from 1 t of waste, and therefore less waste is required to satisfy a given heat demand. As a consequence, the revenues from gate fees (only relevant for *CHP Waste*), variable private costs and the external costs from transport will all decrease, with the decrease in revenue outweighing the decrease in cost. In the RDF case—where we assumed a gate fee of zero—a higher *heating value* reduces the net social cost. Although interesting, these effects are not decisive for our conclusion.

If the alternative waste treatment for waste imported to Sweden is not landfilling, this affects the quantity of avoided CO_2e emissions in the exporting country. In a scenario where incineration with electricity production constitutes the alternative treatment, the avoided CO_2e emissions would be 496 kg CO_2e/t (Jeswani and Azapagic, 2016), compared to 330 kg CO_2e/t under the landfill scenario. In this regard, the net social cost of both *CHP Waste* and *CHP RDF* decreases if the alternative is not landfilling.

The most influential factor for the *CHP Waste* case is the fossil CO_2e intensity of waste: a 10% increase in this intensity increases the net social cost by SEK 34–37/MWh_{th}. Since it is reasonable to assume that CO_2e intensity and *heating value* change proportionally, the effect goes through two channels.[35] In addition to the effects of an increase in the *heating value*, a higher CO_2e intensity implies that the external costs associated with combustion increase substantially due to the relatively high shadow price on CO_2e (SEK 0.69/kg). Therefore, even if modest changes in the shadow price on CO_2e have little impact on the net social cost, that price is an important determinant of the scale of the net social cost in our main calculation.

Other influential factors relate to net private costs. For example, the switching point is reached if the investment costs are increased by 49–61% in the *CHP Waste* case and 11–23% in the *CHP RDF* case.[36] Since the investment cost is directly affected by the discount rate, the latter also influences the net social cost estimates.[37] As seen in Table 7, the net social cost of *CHP Waste* is slightly more sensitive to changes in the price of biofuels than to changes in the gate fee. This is because fuel costs constitute a higher share of the social costs for the benchmark HOP when compared to *CHP Waste*. For *CHP Waste* to become unfavourable, the biofuel price would need to be 60–74% lower than we assume, while the gate fee would need to be 63–77% lower.

Interestingly, we find that net social costs are fairly robust to variation in the number of operating hours for the studied incineration options. Notably, the results indicate that all options benefit from extended operating hours. In the case of *CHP Waste*, the number of operating hours would have to be 65–71% lower for the net social cost to be zero.[38] Achieving a zero net social cost for the *CHP RDF* case would require a reduction of 29–57%. The heat production associated with such a low number of operating hours would correspond to the climate conditions

[35] 1. More CO_2e is emitted per tonne of waste. 2. Less waste is needed to supply a fixed heat demand if the *heating value* is increased.

[36] Investment cost is only varied for Options A and B (and not for the benchmark technology).

[37] In the main analysis, we assume a discount rate of 3.5%. If we instead assume a 2% discount rate the annualized investment cost for *CHP Waste* (*CHP RDF*) is reduced by 21% (16%), and ΔNSC for *CHP Waste* (*CHP RDF*) is SEK -38/MWh_{th} (SEK -30/MWh_{th}). A 5% discount rate increases the annualized investment cost of *CHP Waste* (*CHP RDF*) by 23% (17%), and ΔNSC for *CHP Waste* (*CHP RDF*) is SEK 41/MWh_{th} (SEK 33/MWh_{th}).

[38] Note that, all investigated district heating options may be unprofitable at low levels of heat demand.

of central European regions.[39] In other words, waste incineration is also likely to be socially beneficial in warmer regions if district heating networks already exists.[40]

The sensitivity analysis regarding the external cost from GHG emissions is linear in the sense that all emissions are valued equally. As mentioned in Section 5.4, Sweden currently has several prices on CO_2e emissions and the highest (SEK 7000/t CO_2e) relates to emissions from domestic transport, while the lowest relates to emissions from activities covered by the EU ETS (currently approximately SEK 500/t CO_2e). To account for contractionary shadow prices, we set up three alternative shadow price scenarios[41] and compare the net social cost of *CHP Waste* and *CHP RDF* to the main analysis results. Overall, the results show that our finding of a negative net social cost of *CHP Waste* is robust to the use of alternative shadow prices on carbon (Appendix, Table A4). Notably, the net social cost of *CHP RDF* turns positive under two of the alternative shadow price scenarios (Appendix, Table A5).

In general, we show that our main conclusion—that waste fuel fired CHP is beneficial over biofuel fired HOP—is robust for a wide range of assumptions. Variables have to change heavily, especially for *CHP Waste*, before biofuels become the preferred option instead of waste import.

7. Discussion

We apply a net social cost approach to evaluate the trade-offs between private profitability and external costs. Our results indicate that a marginal increase in the incineration of imported waste in Sweden is preferable from a societal perspective when compared to production in a biofuel-fired plant.

Our calculations of net social costs are based on expert assumptions regarding the technological, economic and environmental characteristics of production plants. Even if these assumptions are reasonable, they do not fit perfectly with a real-world investment situation and can be altered in many ways to better reflect site-specific circumstances. Through a sensitivity analysis, we demonstrated that the main conclusion of waste imports yielding net social benefits is robust to substantial changes in key parameters. This suggests that the conclusion is likely valid for many site-specific cases. By varying the capacity utilisation, we find that our results are not only relevant to the specific case of Sweden, but also to countries with somewhat warmer climates that already have district heating networks.

Importantly, we only give standing to Swedish stakeholders in our analysis. Therefore, private costs and external effects outside of Sweden are not included in our calculations, with an exemption for GHG emissions since those are global in character. If local air emissions are included in our calculations, the picture for imported waste becomes even better. However, both the previous literature and our results suggest that the external costs from these emissions are of relatively minor importance (Fahlén and Ahlgren, 2010; Dijkgraaf and Vollebergh, 2004).

Our study specifically concerns waste that is imported from countries that differ in their share of waste treated in landfills and available domestic district heating capacity. For instance, we assume that the alternative treatment of waste is landfilling. For the UK this is an obvious assumption (Fråne et al., 2016), but for Norway which has several CHP facilities in combination with a ban on landfilling of combustible waste it may seem unreasonable. From a system perspective, we argue that Sweden and Norway cannot be regarded in isolation and that all competitive incineration capacity will be utilised because Europe as a whole lacks the capacity to burn all of the combustible waste that is currently treated in the lower steps of the waste hierarchy. Therefore, the origin of the waste fuel does not warrant a different assumption regarding alternative waste treatment. However, even if one would assume that the alternative waste treatment in the exporting country is incineration, more CO_2e would be emitted than if the waste would instead be placed in a modern landfill with energy recovery. Therefore, the net social cost of incinerating foreign waste in Sweden is lower if the alternative treatment is incineration instead of landfilling. In this sense, our results are not fully aligned with the conclusions of LCA studies that only focused on the net emission of CO_2e and found that waste fuelled district heating is only preferable if it replaces landfilling (e.g., Eriksson and Finnveden, 2017). It can also be argued that incinerating waste in Sweden may be preferable from a climate perspective since Swedish waste incineration plants are part of the EU ETS, which is not the case for those in the UK and Norway. Thus, emissions that occur in Sweden will not increase global GHG emissions.

The EU ETS also influences how the electricity sector is treated in our analysis. In the LCA literature, it is common to include indirect climate effects from electricity produced in alternative waste treatment technologies. The magnitude of such indirect effects is then determined by what fuel is used in the marginal electricity production (e.g., if the marginal production is based on coal, gas or hydro power) (Eriksson and Bisaillon, 2011; Eriksson and Finnveden, 2017). However, these studies often neglect the fact that power plants in EU countries are included in the EU ETS by default and that emissions from the EU ETS will be the same regardless of whether or not electricity is co-produced in waste treatment facilities. In our analysis, the CO_2e intensity of the marginal production facilities (price-setting technology) is reflected by the pass-through of the EU ETS price to the electricity price. How electricity prices will change due to changes in the EU ETS price ultimately depends on how the CO_2e intensity of the marginal production units will change (Fell, 2010). If the permit price and CO_2e intensity change proportionally but in opposite directions, the electricity price will remain unchanged, ceteris paribus.

The main caveat with our evaluation approach is that it only compares a subset of all possible heat production technologies. For instance, we do not evaluate the net social cost of district heating compared to decentralised heating alternatives. If electricity prices will be low in the future, this would benefit heat pumps installed at the building level, which is already a common solution for single-dwelling buildings in Sweden (Swedish Energy Agency, 2017). As such, we identify a local optimum but cannot claim with certainty that this solution leads to the maximisation of overall welfare. This is also the case for previous studies on the topic (e.g., Dijkgraaf and Vollebergh, 2004; Gradus et al., 2017; Fahlén and Ahlgren, 2012; Rabl et al., 2008). It should also be noted that we study a marginal change in heat production using waste fuel and assume that the change does not affect market prices (e.g., the gate fees for incinerating or landfilling, or the price of electricity). If waste incineration capacity is expanded greatly in Sweden, it would surely place downward pressure on gate fees in Sweden and Europe. Still, our sensitivity analysis shows that gate fees must be substantially reduced for the net social cost to become positive. Gate fees would also decrease if there is a fall in supply of waste fuels. For instance, the UK supply of waste fuels is likely to decrease in the coming years due to far-reaching plans for expansion of the UK waste incineration capacity (Tolvik, 2021). However, for Europe as a whole waste supply is expected to increase due to EU's ambitions to divert waste treatment from landfilling

[39] Average annual heating degree days (HDDs) for the Stockholm area (NUTS 2 code: *SE12*) during the 2010–2020 period were 3917 (Eurostat, 2020). A reduction of 29–57% yields annual HDDs in the range of 1684–2781. This corresponds to the number of HDDs in countries such as Belgium, France, Hungary, Spain, Croatia, Bulgaria and Slovenia.

[40] District heating systems exist in most central European countries but are generally more common in northern continental Europe, the Baltic region and the Nordic countries. A list of countries' shares of residential sector heat demand (%) supplied through district heating in 2017: Belarus 70, Denmark 65, Lithuania 56, $Estonia_{2015}$ 52, Sweden 50, Poland 42, Czech Republic 40, Finland 38, Russia 25, Germany 14, Austria 14, Slovenia 9, France 5, Switzerland 4, $Netherlands_{2013}$ 4, Norway 4 (Euroheat and Power, 2021), United Kingdom 0.1% of all dwellings (IEA, 2017).

[41] Review the Appendix for a description of the scenarios.

(Persson and Münster, 2016). Thus, even if there may be geographical shifts in the market, this should not alter our findings that waste incineration is preferable over biofuel-based district heating.

From a methodological perspective, our analysis highlights the difficulty in choosing an appropriate shadow value on CO_2e emissions. Climate targets at the national and regional levels mean that emissions creating the same global damage is valued differently. Consequently, one may conclude that it is preferable to burn the waste in a neighbouring country just because a lower shadow price is assigned to emissions occurring outside a given jurisdiction. From an economics perspective, having several shadow prices on CO_2e is unfortunate because deep reductions in GHG emissions at the global or national level will not be accomplished at the lowest possible cost.

Finally, the results of net social cost analyses are typically sensitive to the social discount rates used in calculations. We also observe this sensitivity in our results when we vary the annualized investment cost, which is a direct function of the discount rate. The incineration of mixed waste (*CHP Waste*) also remains beneficial at a social discount rate of 5% (3.5% is assumed in our main analysis), irrespective of whether the waste is transported from Norway or the UK. The incineration of RDF (*CHP RDF*) only remains beneficial if the waste comes from Norway. From a private investor perspective, it is reasonable to adopt a higher discount rate to account for the high risks involved in capital-intensive investments. However, from a societal perspective, the risks are lower and few European countries adopt social discount rates above 4% (OECD, 2018).

8. Conclusion

In the previous literature, the question whether waste incineration is socially preferable or not, has often been analysed from a perspective of optimal waste management (e.g., Dijkgraaf and Vollebergh, 2004; Rabl et al., 2008; Gradus et al., 2017). In this paper we adopt an alternative perspective, based on the fact that waste is not only a necessary bad, but also a demanded input fuel in energy production in countries with a colder climate. In this perspective, waste incineration is an energy production technology as much as a waste disposal technology. Therefore, it is relevant to evaluate waste incineration against other energy production options.

The objective of this paper is to provide insights on policy issues relating to the cross-border trade in waste fuel as well as efficient thermal waste treatment services. More specifically, we apply a net social cost approach to evaluate the trade-offs between private profitability and external costs. Based on our results we conclude that a strong emphasis on the proximity principle—as seen in public debate—stands in sharp contrast to the impact that external costs from the transport of waste fuel have on the net social cost and net emissions associated with the incineration of imported fuel. This is in line with previous findings (e.g., Rabl et al., 2008; Pizarro-Alonso et al., 2018b). From our perspective, it is difficult to see any reason why transporting waste fuel is costlier from an environmental perspective than transporting other fuels or even intermediaries or goods (e.g., clothes or food products). Our main conclusion is that using imported waste fuel in highly efficient district heating facilities in Sweden is beneficial from a societal (Swedish) perspective compared to using biofuel. This conclusion holds under a wide range of assumptions regarding technical, economic and environmental characteristics.

The results also suggest that the waste incineration sector has an economic incentive to reduce the fossil coal content in waste fuel. A lower fossil share is associated with a decreased *heating value*, which has a positive impact on revenues from waste treatment since more waste can be treated, ceteris paribus. The economic incentive for using less CO_2e-intensive waste fuel is further strengthened if the CO_2e emissions are internalised via economic policy instruments (e.g., emissions trading). Therefore, the EU should create a level playing field for waste incineration by making participation in the EU ETS mandatory for waste incinerators.[42] This type of regulation should be more effective in internalising CO_2e emissions than targeting waste volumes (in practice, weights) through waste incineration taxes.

Our results should not be interpreted as an argument for subsidizing waste incineration because the environmental benefits occurring abroad are largely internalised in the gate fees, which are made possible by landfill regulations. Other countries export waste to reduce the cost of proper waste management while considering external costs. Therefore, the negative net social cost is reflected in the private profitability underlying the expansion of Swedish waste incineration capacity observed over the past decade. Our results suggest that policymakers should embrace this development instead of attempting to hold it back.

Our analysis is based on a set of discrete assumptions that are primarily informed by the existing literature. Future research should integrate ESA, LCA and net social cost analyses by considering relevant system boundaries and existing policy measures to make better site-specific predictions. Ultimately, we aim to encourage researchers and policymakers to pay more attention to industrial waste. To date, the strong focus on municipal solid waste in policy design and research has been disproportionate to its share of the total volume of generated waste.

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Appendix A

Table A1
Technical and economic characteristics of the district heating plants evaluated in the net social cost analysis, including biofuel-fired CHP (CHP Bio) as a potential benchmark option to biofuel-fired HOP (HOP Bio) Economic characteristics are exclusive of taxes. All monetary values in SEK_{2019}.

	Unit	Benchmark HOP Bio	Option A CHP Waste	Option B CHP RDF	CHP Bio
Technology	Type	Grate	Grate	Fluidised bed	Fluidised bed
Economic lifetime	Years	25	25	25	25
Total input	kTonne/year	223	203	182	291
Heating value	GJ/t	9.36	12.00	14.04	9.36
Operating hours	Full load hours/year	6500	6500	6500	6500

(*continued on next page*)

[42] To date, only Denmark, Sweden and Lithuania have included their waste incineration sectors in the EU ETS. In October 2020, the EU Commission announced its initial reform plan for the EU ETS, where it signaled that it will no longer be optional for member states to choose whether or not to include their waste incineration sector. The European Parliament and the European Council are expected to take a stance on this proposal during 2021 (European Commission, 2020a).

Table A1 (*continued*)

	Unit	Benchmark HOP Bio	Option A CHP Waste	Option B CHP RDF	CHP Bio
Efficiency	GWh out/GWh in	100	105	104	105
Heat	GWh	580	580	580	580
Electricity	GWh	–	130	158	214
Power-to-heat ratio	%-share	–	0.22	0.27	0.37
Iron recovery	kg/MWh_{th}	–	6.3	0	–
Aluminium recovery	kg/MWh_{th}	–	0.7	0	–
Investment cost	MSEK/year	35	141	120	86
Investment cost	SEK/MWh_{th}	60	242	207	149
Operating cost	SEK/MWh_{th}	39	165	157	66
Fuel cost	SEK/MWh_{fuel}	200	−165	0	200
Fuel cost	SEK/MWh_{th}	200	−192	0	261[a]
Electricity crediting[b]	SEK/MWh_{th}	–	−90	−109	−148
Metals crediting[c]	SEK/MWh_{th}	–	−8	–	–
Private cost (sum)	SEK/MWh_{th}	299	118	255	327[d]

[a] Under CHP technologies the input fuel is used for joint production of electricity and heat. As a result, to generate one MWh of thermal heat it is required more input fuel compared to the case for HOP Bio. This explains why the fuel cost in terms of SEK/MWh_{th} is higher for CHP Bio compared to HOP Bio although the efficiency of CHP Bio is higher.
[b] The electricity price is assumed to be SEK 400/$MWh_{electricit}$.
[c] The market price is SEK 1/kg for iron and SEK 4/kg for aluminium.
[d] Since private cost (sum) is lower for HOP Bio compared to CHP Bio, it follows that the former constitutes a better benchmark.

Table A2
Emission factors concerning district heating using biofuels or waste fuel. Source: Official emission factors according to the Swedish Environmental Protection Agency.

Emissions factor	Unit	Benchmark HOP Bio	Option A CHP Waste	Option B CHP RDF	CHP Bio
CO_2	kg/GJ_{fuel}	–	38.45	38.45	–
CO_4	kg/GJ_{fuel}	0.011	0.005	0.005	0.011
NO_X	kg/GJ_{fuel}	0.006	0.004	0.004	0.006
SO_2	kg/GJ_{fuel}	0.010	0.002	0.002	0.010
Dioxins (air)	mg/GJ_{fuel}	0.03	0.03	0.03	0.03
Hg	mg/GJ_{fuel}	300	1400	1400	300
PM10	kg/GJ_{fuel}	0.009	0.0002	0.0002	0.009
PM25	kg/GJ_{fuel}	0.0063	0.0002	0.0002	0.0063
CO	kg/GJ_{fuel}	0.030	0.020	0.020	0.030
As	mg/GJ_{fuel}	0.4	2	2	0.4
Cd	mg/GJ_{fuel}	1	0.5	0.5	1
Cr	mg/GJ_{fuel}	3	2.5	2.5	3
Ni	mg/GJ_{fuel}	7	2	2	7
Pb	mg/GJ_{fuel}	16	7	7	16
N_20	kg/GJ_{fuel}	0.003	0.004	0.004	0.003
NH_3	kg/GJ_{fuel}	0.001	0.002	0.002	0.001

Table A3
Full list of shadow prices concerning emissions from heat production. Values before adjustment to 2019 price level.

Emissions factor	Unit	Shadow price	Source
Heat production			
CO_2e	SEK/kg	0.69	CPLC (2017); NAS NASEM (2017)
NO_X	SEK/kg	56	CE Delft (2018)
SO_2	SEK/kg	52	CE Delft (2018)
Dioxins (air)	SEK/mg	722	CE Delft (2018)
Hg	SEK/mg	0.37	CE Delft (2018)
PM10	SEK/kg	144.01	CE Delft (2018)
PM25	SEK/kg	189.88	CE Delft (2018)
CO	SEK/kg	0.56	CE Delft (2018)
As	SEK/mg	0.01	CE Delft (2018)
Cd	SEK/mg	0.01	CE Delft (2018)
Cr	SEK/mg	0.00	CE Delft (2018)
Ni	SEK/mg	0.00	CE Delft (2018)
Pb	SEK/mg	0.06	CE Delft (2018)
NH_3	SEK/kg	64.86	CE Delft (2018)

Table A4
Net social cost of *CHP Waste* (Option A), in terms of SEK/MWh_{th}, where the shadow price of CO_2e (SEK/t) differs between emissions from different sources. All values in terms of SEK_{2019}.

	Main analysis	Current price[a]	EU target[b]	Swedish policy[c]
Shadow price				
Emissions from domestic transports	690	1200	1200	7000
Other emission within EU	690	500	1200	1200
Other emission outside EU	690	690	690	690
Net private cost	−181	−181	−181	−181
Net externality effects				
Combustion	107	71	185	185
Transport				
UK waste	36	36	36	33
Norwegian waste	8	9	9	25
Landfill	−83	−83	−83	−83
Net externality cost, UK waste	60	25	139	136
Net externality cost, Norwegian waste	32	−2	112	128
Net social cost, UK waste	−121	−156	−42	−45
Net social cost, Norwegian waste	−149	−183	−69	−53

[a] Current price: Actual carbon pricing is assumed where applicable. Emission outside of the EU is priced as the social cost value used in the main analysis.
[b] EU target: CO_2e is priced at the level required to meet EU climate targets. Emission outside of the EU is priced as the social cost value used in the main analysis.
[c] Swedish policy: CO_2e is priced to meet national targets in the domestic transport sector as well as EU climate targets. Emission outside of the EU is priced as the social cost value used in the main analysis.

Table A5
Net social cost of *CHP RDF* (Option B), in terms of SEK/MWh_{th}, where the shadow price of CO_2e (SEK/t) differs between emissions from different sources. All values in terms of SEK_{2019}.

	Main analysis	Current price[a]	EU target[b]	Swedish policy[c]
Shadow price				
Emissions from domestic transports	690	1200	1200	7000
Other emission within EU	690	500	1200	1200
Other emission outside EU	690	690	690	690
Net private effects	−44	−44	−44	−44
Net externality effects				
Combustion	62	40	109	109
Transport				
UK waste	32	32	32	30
Norwegian waste	7	8	8	22
Landfill	−74	−74	−74	−74
Net externality cost, UK waste	20	−2	67	65
Net externality cost, Norwegian waste	−6	−26	43	57
Net social cost, UK waste	−24	−46	23	21
Net social cost, Norwegian waste	−50	−70	−1	13

[a] Current price: see footnote (a) in Table A4.
[b] EU target: see footnote (b) in Table A4.
[c] Swedish policy: see footnote (c) in Table A4.

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