Norwegian Waste-to-Energy: climate change, circular economy and carbon capture and storage

Carine Lausselet[1], Francesco Cherubini [1], Gabriel David Oreggioni[1], Gonzalo del Alamo Serrano[2], Michael Becidan[2], Hu, Xiangping[1], Per Kr. Rørstad[3], Anders Hammer Strømman[1]

[1] Industrial Ecology Program, Norwegian University of Science and Technology, NTNU, Trondheim, Norway

[2] SINTEF Energy Research, Trondheim, Norway

[3]Norwegian University of Life Sciences, NMBU, Ås, Norway

*carine.lausselet@ntnu.no, +47 46 78 92 64

Abstract

Recently, the European Commission has adopted a Circular Economy package. In addition, climate change is regarded as a major global challenge, and the de-carbonization of the energy sector requires a massive transformation that involves an increase of renewable shares in the energy mix and the incorporation of carbon capture and storage (CCS) processes.

Given all this strong new momentum, what will the Norwegian waste-to-energy (WtE) look like in a decade? What threats and opportunities are foreseen? In an attempt to answer these questions, this study combines process-based life-cycle assessment with analysis of the overall energy and material balances, mathematical optimization and cost assessment in four scenarios: (1) the current situation of the Norwegian WtE sector, (2) the implications of the circular economy, (3) the addition of CCS on the current WtE system and (4) a landfill scenario.

Except for climate change, the CCS scenario performs worse than the WtE scenario. The energy recovering scenarios perform better than the recycling scenario for (1) freshwater eutrophication and human toxicity potentials due to secondary waste streams and (2) ozone depletion potential due to the additional fossil fuel used in the recycling processes. The inclusion of the near-term climate forcers decreases the climate change impacts by 1% to 13% due to a net cooling mainly induced by NO_x.

Circular economy may actually give the WtE system the opportunity to strengthen and expand its role towards new or little developed value chains such as secondary raw materials production and valorization of new waste streams occurring in material recycling.

Keywords

1. Waste-to-Energy (WtE)
2. Life-cycle assessment (LCA)
3. Carbon capture and storage (CCS)
4. Circular economy
5. Climate change
6. Near-term climate forcers

1 Introduction

The European Union's approach to waste management is based on the waste hierarchy, which sets the following priority order: prevention, reuse, recycling, energy recovery and, as the least preferred option, disposal (European Union Council 1999). The waste hierarchy's practical consequence is to divert waste from landfills to material and energy recovery. As a result, the number of Waste-to-Energy (WtE) plants has increased during the last decade in Europe (IEA Bionergy 2013). Recently, the European Commission has revised legislative proposals on waste and adopted a Circular Economy package – an economic system that leaves no waste to be landfilled and that keeps all material flows in the economy loop through reuse, redesign, material recovery or energy recovery. The European Circular Economy Package encompasses two main elements related to municipal solid waste (MSW): (1) Landfill ban/cap on specific waste fractions and (2) Recycling targets (European Commission 2015). As an EEA/EFTA country member, Norway implements all European directives and thus has a similar waste and WtE regulatory framework, e.g. Waste Hierarchy, landfill ban on biodegradable waste, Landfill Directive, Waste Framework Directive and the upcoming 2030 Energy Strategy and WtE and circular economy-related legislation and strategies.

In Norway, the latest trends in the waste management sector can be summarized as (Becidan et al. 2015): (1) strong increase in the total WtE capacity (from about 1.25 Mt/y in 2010 compared to 1.70 today) – with an average throughput of about 90 % of their nominal capacity; (2) landfill ban for organic waste (2009) followed by a reduction in the number of landfills; (3) significant MSW export to Sweden (several hundred thousand t/y); (4) a significant fraction of the energy (heat) produced is not delivered to any customer, especially during the summer; (5) the capital city Oslo has newly implemented source sorting of food waste (in addition to paper, plastic, glass and metal) and is working on the implementation of carbon capture and storage (CCS).

Almost all of the MSW (and waste in general) exported from Norway goes to Sweden and almost exclusively to WtE plants (mainly delivering district heat). Detailed statistics are difficult to obtain but it is estimated that 1.6 million tonne of MSW per year were exported over the last five years. The topic is complex, and lower gate fees in Sweden (which has a WtE overcapacity) are pointed to as being the main reason for the MSW exports. On the other hand, Norway has imported around 400’000 tonnes waste per year in the last years. For the WtE plants in particular, mainly refuse-derived fuel (RDF) from the UK has been used as fuel (Norwegian Environment Agency 2017).

Not all the materials can be recycled, and resource consumption, emissions, losses and contamination – as well as additional new waste streams – occur while material recycling (Bartl 2014). To estimate the overall environmental performance of a system and to avoid potential problem shifting when changing models – in this case from a linear to a circular economy – life-cycle assessment (LCA) is a frequently applied methodology. LCA results give an overview of how various types of environmental impacts accumulate over the different life-cycle phases, providing a basis for identifying environmental bottlenecks of specific technologies and for comparing a set of alternative scenarios with respect to environmental impacts (Finnveden 1999, Hellweg and Canals 2014).

LCA has been used extensively within the last decade to evaluate the environmental performance of waste treatment systems (Arena et al. 2003a, Bergsdal et al. 2005, Cherubini et al. 2008, 2009, Rigamonti et al. 2009, Consonni et al. 2011, Giugliano et al. 2011, Ning et al. 2013, Passarini et al. 2014, Lausselet et al. 2016). For WtE systems that combine incineration with energy recovery, or WtE value chains, the life-cycle burdens are sensitive to the energy recovery rate (Riber et al. 2008, Gentil et al. 2010, Fruergaard and Astrup 2011), the conventional fuel displaced for heat or electricity generation (Riber et al. 2008, Passarini et al. 2014, Burnley et al. 2015), the reuse of the bottom ash (Birgisdóttir et al. 2006, Birgisdóttir et al. 2007, Allegrini et al. 2014, Allegrini et al. 2015b), the leaching of key chemical elements from bottom and fly ashes (Doka and Hischier 2005, Astrup et al. 2006, Hauschild et al. 2008, Allegrini et al. 2015a, Yang et al. 2015) and the recovery of the metal or aggregate from the bottom ash (Morf et al. 2013, Burnley et al. 2015). WtE plants have been found to be a robust technology and a competitive alternative to fossil fuel based energy systems (Turconi et al. 2011, Brunner and Rechberger 2015).

LCAs available in the literature provide a variety of insights on WtE systems that combine anaerobic digestion with energy recovery, or biogas value chains. In general, biogas energy systems have lower greenhouse gas (GHG) emissions than fossil energy systems, especially when biogas is used as fuel in transportation (Liu et al. 2013, Niu et al. 2013, Lozanovski et al. 2014, Lyng et al. 2015). The results are sensitive to the management of the digestate; open storage leads to uncontrolled emissions of GHG like CH_4 and nitrous oxide (N_2O) (Blengini et al. 2011, De Meester et al. 2012, Boulamanti et al. 2013) and the use of digestate in agriculture increases the risk for human toxicity, acidification and eutrophication potentials due to the heavy metals (Patterson et al. 2011) and the high nutrient level it contains (Lozanovski et al. 2014). A recent study of Iordan et al. (2016) highlights the sensitivity of biogas systems to the choice of climate metrics and the influence of the near-term climate forcers (NO_x, SO_x, particulate matters, black carbon and organic carbon).

The different plastic recovery routes, as well as their challenges and opportunities, are explored broadly (Arena et al. 2003b, Perugini et al. 2005, Shonfield 2008, Al-Salem et al. 2009, Astrup et al. 2009a, Eriksson and Finnveden 2009, Hopewell et al. 2009, Kunwar et al. 2016, Lupo et al. 2016). A review on plastic waste management conducted by Lazarevic et al. (2010) shows: (1) the majority of the LCA study to exhibit a preference for recycling rather than for WtE, (2) the conclusions sensitive to the level of contamination and to the replacement of virgin plastic ratio, (3) landfills as the least preferred option, except for climate change. The selection of the appropriate avoided primary production of materials is also a crucial parameter in LCA studies on material recycling systems (Brogaard et al. 2014, Rigamonti et al. 2014, Turner et al. 2015). Recycling material often, but not always, reduces climate change impact (Björklund and Finnveden 2005). As an example, for paper recycling, Merrild et al. (2008) show through an LCA that recycling is clearly better than landfilling, but equal or better than WtE only if the recycling technology is at a high environmental performance level. Merrild et al. (2012) find environmental benefits when recycling the material fractions paper, glass, steel and aluminum instead of incinerating them. On the other hand, they find incineration to be a potentially better option than recycling for cardboard and plastic in some situations.

Waste treatment systems are by definition complex (Laurent et al. 2014a, Laurent et al. 2014b); they are embedded with uncertainty (Scipioni et al. 2009, Clavreul et al. 2012), and waste composition varies over time and region, influencing the results (Slagstad and Brattebø 2013, Astrup et al. 2015). In addition to treating waste and producing energy, WtE plants are becoming increasingly recognized as a means to recover materials of high importance for the economy (Morf et al. 2013, Boesch et al. 2014, Brunner and Kral 2014). Also, WtE technologies enable energy production with the advantage of not competing for land occupation as woody biomass does. Thus, in contrast to long rotation woody biomass (Cherubini et al. 2012, Guest et al. 2013a, Guest et al. 2013b), waste can be considered a carbon-neutral fuel.

Climate change is regarded as a major global challenge (IPCC 2007) that has motivated the international community to implement mitigation strategies aiming at limiting the average increase of global temperature (Riahi et al. 2007, Luderer et al. 2013). A reduction in global emissions of CO_2 can slow down the rate of warming, but a stabilization of global temperature can only occur if CO_2 emissions approach zero (Myhre et al. 2013). Energy industries have contributed to approximately 32% of global CO_2 emissions over the last 20 years (Janssens-Maenhout et al. 2012), and the de-carbonization of the energy sector requires a massive transformation that involves an increase of renewable shares in the energy mix, improvements in power plant efficiency and the incorporation of CCS processes in fossil and biomass-fuelled energy plants (Azar et al. 2013, Myhre et al. 2013, IEA 2015).

Several works analyzing the incorporation of absorptive CO_2 capture technologies in bio-refineries for liquid fuel production via gasification of woody biomass can be found in the literature (Haro et al. 2013, Heyne and Harvey 2014). Other papers study the design of pre- and post-combustion CO_2

capture technologies and the associated environmental impacts for large-scale woody biomass power plants (Corti and Lombardi 2004, Carpentieri et al. 2005, NETL 2012b, a, Schakel et al. 2014). Fewer works present techno-economic and environmental assessment of medium (1-100 MWth) fossil-fuelled CHP plants with a wide range of CO_2 capture processes (IEA 2007, Soukup et al. 2009, Singh et al. 2011). A recent series of articles analyzes the techno-environmental performance of absorptive and adsorptive pre- and post-combustion technologies in small scale woody biomass CHP (Oreggioni et al. 2015, Luberti et al. 2016, Oreggioni G D et al. 2016)

A wide range of LCA studies have been conducted on energy systems, including WtE, biogas and CCS. Yet, to our knowledge, few studies have focused on scaling up WtE technologies to a national level (e.g Gentil et al. (2009b)). A gap also exists in the knowledge base for process design and LCA studies for WtE plants with CO_2 capture technologies. In this study, we conduct an LCA and a cost assessment on the current situation of the Norwegian WtE sector, the implications of the circular economy and the introduction of CCS. The specific objectives are to assess: (1) the current situation of WtE in Norway, (2) the influence of implementing the circular economy package on the Norwegian WtE sector, (3) the addition of CCS on the current WtE plants, (4) and benchmark (1), (2) and (3) with a landfill scenario in order to check the waste hierarchy.

2 Methodology

This study combines LCA methodology with mathematical optimization, analysis of the overall energy and material balances and cost assessment.

2.1 System description

The Norwegian WtE sector currently accounts for 17 plants, spread all across Norway. Their total capacity is 1.7 million tonnes, the average throughput is at 90% of capacity, and the production is around 13300 TJ heat for district heating networks, in addition to some electricity (1200 TJ). Energy recovered from waste is the main energy source for district heating with a share of almost 50% (Statistics Norway 2014), and 50% of the energy from the WtE sector is accounted for as renewable in Norwegian national statistics. An exhaustive list of the plants is presented in Table S1 in the supplementary material.

2.2 Scenarios

This study consists of four scenarios: WtE, Circular Economy, CCS and Landfill. The scenarios are presented in Figure 1, and further explained below. Each box represents a scenario, and the outputs are given in red.

<Figure 1>

WtE - Describes the situation in 2015.

Circular economy - An increased share of plastic and paper is sent to material recycling while an increased share of organic waste is sent to anaerobic digestion with energy recovery. The recycling rates are increased from today's practice to the best practice including central waste separation in 2030; from 79% today to 93% for paper, from 23% to 63% for plastic and from 42% to 70% for organic waste (Syversen et al. 2015). The anaerobic digestion process is based on a plant located in Lindum, Norway as described in Iordan et al. (2016). The paper recycling process is from Ecoinvent 3.2, and the electricity mix used for the recycling process is switch from average European mix to NORDEL electricity mix. The recycling process uses 8 g sodium hydroxide (Arena et al. 2003b), 0.2 kWh electricity (average European mix) and 2 liters diesel fuel in onsite vehicles and the recycling process (Astrup et al. 2009a) per kg of treated waste plastic. A material replacement rate of 90% is assumed for both recycling processes.

CCS -CCS with monoethanolamine (MEA) technology is added to the 17 existing WtE plants. Despite its high energy consumption, the MEA post-combustion process was selected as the technology due to its maturity. The energy penalty caused by the additional fuel is 34%. Details on the CCS module are given in the supplementary material.

Landfill- Although not a realistic scenario since disposal of biodegradable wastes in landfills has been banned in Norway since 2009, a landfill scenario was added as a check on the waste hierarchy.

2.3 Life-cycle assessment (LCA)

Process-based LCA with system expansion is applied. Primary data (waste input, air emissions, consumables, auxiliary fuel, thermal and electrical efficiencies, transport distances) represent the majority of the input data. Arda, a Matlab routines based program developed at NTNU (Majeau-Bettez and Strømman 2016) is used. The inventory for background processes relies on Ecoinvent v3.2 (Ecoinvent Centre 2010).

2.3.1 Life-cycle inventory

An annual average mix of household (60%) and industry waste (40%) is combusted. In addition, some plants have special permits to co-combust with special waste types, such as clinical waste, hazardous waste and impregnated wood waste. The overall waste composition is provided on a waste type level in Table S4, and broken down into its chemical composition in Table S5.

The transport distances are based on expert judgments and own assumptions. For MSW, the transport distances are first 14 km for municipal waste collection, and then 100 km by truck. For RDF, the transport distances are 14 km for municipal collection, 200 km by truck (100 km in England and another 100 km in Norway) and 1000 km by ship. For organic waste, a distance of 100 km by truck is assumed. For paper to material recycling, 300 km by truck and 500 km by train (to Sweden) are assumed. For plastic to recycling, a distance of 300 km by truck and 1000 km by train (to Germany) and an additional distance of 7000 km by ship (to China) for 30% of the waste are assumed.

The direct emissions occurring at the WtE plant are either waste- or process-specific. Waste-specific emissions are calculated by multiplying each element of the waste input vector $\mathrm{WSV}(\mathrm{s})$ by its mass transfer coefficient contained in $\mathrm{TCMM}(\mathrm{s,c})$.

$$\mathrm{WSV}(\mathrm{s}) \times \mathrm{TCMM}(\mathrm{s,c}) = \mathrm{FCMM}(\mathrm{s,c}) \tag{1}$$

$\mathrm{FCMM}(\mathrm{s,c})$ is the final compartment matrix, with *s* corresponding to the 41 chemical elements constituting the waste input, and *c* to the four final environmental compartments: air, water, bottom ash and fly ash. $\mathrm{WSV}(\mathrm{s})$ and $\mathrm{TCMM}(\mathrm{s,c})$ are given in Table S5 and Tables S7-S9. For the emissions occurring at the landfill sites, $\mathrm{FCMM}(\mathrm{s,bottom\ ash})$ and $\mathrm{FCMM}(\mathrm{s,fly\ ash})$ are multiplied by leaching coefficients vectors, specific for the landfill type as described by Doka (2007) and used by e.g. Lausselet et al. (2016).

Process emissions (SO_2, particulate matters, CO, HCl, HF, NO_x and dioxin) are measured emissions at the plant site and from Doka (2007) for NH_3, NMVOC, CH_4, benzene, benzopyrene, hexachlorobenzene, pentachlorobenzene, pentachlorophenol and toluene. An exhaustive list is given in Table S10 and Table S11.

2.3.2 Functional unit and allocation

The functional unit is defined as: "To treat 1560 ktonne MSW, produce 13'309 TJ heat to feed the district heating network, 1304 TJ electricity, deliver 99 ktonne of plastic, 135 ktonne of paper and 205 tonne of fertilizer."

True system expansion is the chosen allocation approach and the system is thus expanded in order to keep the functional unit constant and deliver the same services throughout the scenarios. Primary production of plastic, paper and fertilizer are assumed to deliver the same amount of materials. To deliver the same amount of energy, electricity from hydropower and heat from oil are used for the Landfill scenario, while electricity and heat from RDF are used for the Circular economy scenario since the WtE installed capacity is in deficit, due to the diversion of plastic, paper and organic waste to material recycling and anaerobic digestion. The energy and material balances of each scenario with system expansion are presented in Table 1 below.

Table 1: Energy and material balances of each scenario with system expansion

				WtE	Circular economy	CCS	Landfill
Input	Incineration with energy recovery	MSW_{2015}	ktonn	1560		1560	
		$MSW_{circular}$	ktonn		1110		
		RDF			636*	940[5]	
	Anaerobic digestion	Organic waste	ktonn		190		
	Material recovery	Plastic	ktonn		110		
		Paper	ktonn		150		
	Landfill	MSW	ktonn				1560
	Auxiliary fuels	Heat, from fossil fuel (Diesel)	TJ	378	269	378	
		Electricity, NORDEL mix	TJ	533	379	533	
Output	District heating	Heat, from MSW_{2015}	TJ	13309		13309	
		Heat, from $MSW_{circular}$	TJ		8460[1]		
		Heat, from RDF	TJ		4849[2,*]		
		Heat, from oil	TJ				13309[3]
	Electricity	From WtE, MSW_{2015}	TJ	1163[4]		1163[4]	
		From WtE, $MSW_{circular}$			755[4]		
		From WtE, RDF	TJ		432[4,*]		
		From anaerobic digestion, organic waste	TJ		117		
		From NORDEL	TJ	141*		141*	1304*
	Material	Plastic, recycled	ktonn		99		
		Plastic, primary	ktonn	99*		99*	99*
		Paper, recycled	ktonn		135		
		Paper, primary	ktonn	135*		135*	135*
		Nitrogen fertilizer, recycled	tonn		193		
		Nitrogen fertilizer, synthetic	tonn	193*		193*	193*
		Phosphorus fertilizer, recycled	tonn		12		
		Phosphorus fertilizer, synthetic	tonn	12*		12*	12*

*For system expansion, [1]Calculated with the thermal efficiencies given in Table S1 and an average LHV of 10.3 MJ/kg, [2]Calculated with the thermal efficiencies given in Table S1 and an average LHV of 13.6 MJ/kg, [3]From Ecoinvent 3.2, [4]Calculated with the electrical efficiency given in Table S1, and LHVs of 11.5 MJ/kg for MSW_{2015}, 10.3 MJ/kg for $MSW_{circular}$ and 13.6 MJ/kg for RDF, [5]used as auxiliary fuel in the add-on boiler of the CO_2 capture process

2.4 Impact assessment

ReciPe v1.12 (hierarchist perspective) (Goedkoop et al. 2009) is the chosen impact assessment method for the following four midpoint categories: climate change impact potential (CCIP), Freshwater Eutrophication Potential (FEP), Human Toxicity Potential (HTP), and Ozone Depletion Potential (ODP). ReciPe is the chosen methodology because of the wide range of potential impacts it covers. For HTP, the missing characterization factors for methylamine, diethylamine, nitrosamine and morpholine are estimated by using USEtox (Rosenbaum et al. 2008) and potential for MEA degradation based on Veltman et al. (2010). The results are also presented according to four key single stressors: NO_x, particulate matter (PM)- PM smaller than 2.5 μm and PM smaller than 10 μm, SO_2 and carbon monoxide (CO). NO_x, PM, SO_x and CO are assessed individually because they are subject to emission limits (Norwegian Ministry of Climate and Environment 2016). In opposition to long rotation woody biomass (Cherubini et al. 2012, Guest et al. 2013a, Guest et al. 2013b), waste can be considered a carbon neutral fuel, and carbon biogenic is thus set to 0 in this study.

In addition, the latest recommendations for CCIP made by the UNEP SETAC task force on climate change (Cherubini et al. 2016, Levasseur et al. 2016) are applied to assess the current WtE system (scenario WtE, without system expansion), and a sensitivity analysis is conducted to assess the potential impact of the NTCFs. Organic and black carbons are not explicitly provided in LCA inventories. We thus derive them from the total PM emitted in the system following the methodology developed by Bond et al. (2004). Coke in blast furnace is used as a proxy for the total background emissions, and to "solid waste, all technologies" for the total foreground emissions.

2.5 Cost assessment

Standard economic metrics are applied to evaluate the economic outcome of the scenarios. Levelized cost of energy (LCOE) is widely used in the energy economics literature (see e.g. Branker et al. (2011), Chum et al. (2011), Darling et al. (2011) and (Moomaw et al. 2011)), and is used as the point of departure. The social costs are further estimated by use of the official guidelines (NOU 2012, Ministry of Finance 2014) in the assessments. The general principle is that goods and services should be valued by the best alternative use. In most cases, this means that social cost coincides with the cost that firms incur. The main difference is the discount rate. For projects with a lifespan of less than 40 years, the guidelines state that 4% pro anno should be used as the discount rate. This is clearly lower than what would be used in private firms.

Primary data from the plants is combined with data from the literature when assessing the flows and processes in Figure 1. The primary data indicates a representative investment cost for waste incineration boilers of 44.7–50.0 MNOK/tonne/hour capacity. Assuming a constant energy content of 9 MJ/kg waste, this corresponds to 14.9–16.7 kNOK/kW. The estimated average size of Norwegian waste incineration plants is about 50 MW. Norges vassdrags- og energidirektorat (2011) reports investment costs for plant sizes 10 and 30 MW at 18.9 and 15.8 kNOK/kW, respectively. These figures reflect good correspondence between the estimates. With the data available, it is not possible to estimate the investment cost for other plant sizes without making additional assumptions. We assume that the power law holds (Bruckner et al. 2011) with a scaling factor of 0.8. Operation and maintenance costs are estimated based on primary data. The estimated functions are applied to the 17 plants, and yearly capital costs and operation and maintenance costs are estimated and given in Table S14 and Table S15.

To our knowledge, no full-scale WtE CCS plant has been built so far, and cost data is thus scarce. In this study we have used data from Chandel et al. (2012). They estimate that CCS increases the investment cost of the plant by 17%. This is slightly lower than estimates for coal power plants with a representative value of about 22% according to (Rubin et al. 2015). The operation and management cost is assumed to be 2.4% of the investment cost (Chandel et al. 2012).

The Circular economy scenario requires both a sorting facility – for sorting household waste – and a biogas plant. The cost estimates for the sorting plant are based on costs for such a plant located at Haraldrud in Oslo, Norway, which handles 100 ktonne/year. Cost estimates for the biogas plant are based on Nedland (2011) and Norwegian Environment Agency (2013). For the fertilizer outputs, current market prices are used. The (positive) value of recycling paper and plastic is not included due to limited market data.

Landfilling has been banned in Norway since 2009. Thus, no current information about the cost of landfilling exists. This cost is therefore not included. The cost of producing heat from oil is based on (Water Resources and Energy Directorate 2011), using the current market price for heating oil.

3 Results

In this section, the life cycle environmental results for the environmental mid-point impacts CCIP, FEP, HTP and ODP and for the selected single stressors NO_x, SO_x, PM and CO are presented. The results are first presented without system expansion in Figure 2, and then with system expansion in Figure 3. Finally, the sensitivity analysis of the WtE scenario without system expansion is presented for GTP100, GWP100 and GWP20 with the NTCF. The cost results are integrated in Figure 3, and the absolute results for each scenario are presented in Table 2.

Table 2: Absolute results, without and with system expansion

		WtE		CCS		Circular Economy		Landfill	
		Without system expansion	With system expansion	Without system expansion	With system expansion	Without system expansion	With system expansion	Without system expansion	Wit sys expa
CCIP	kg CO_2 eq	8.0E+08	1.2E+09	3.1E+08	6.8E+08	5.9E+08	7.5E+08	8.2E+08	2.5
FEP	kg P eq	2.3E+04	1.1E+05	3.5E+04	1.2E+05	8.9E+04	1.0E+05	2.4E+04	1.
HTTP	kg 1,4-DB eq	3.2E+08	4.3E+08	4.3E+08	5.3E+08	5.7E+08	6.8E+08	6.1E+08	8.
ODP	kg CFC-11 eq	1.2E+01	3.1E+01	2.1E+01	3.9E+01	3.3E+01	4.0E+01	1.2E+01	2.

Nox	kg Nox	1.5E+06	2.2E+06	2.1E+06	2.7E+06	1.6E+06	2.1E+06	5.1E+05	2.
PM	kg PM	1.0E+05	3.7E+05	1.7E+05	4.3E+05	2.1E+05	2.7E+05	6.1E+04	5.
SO_2	kg SO_2	2.2E+05	9.7E+05	3.1E+05	1.1E+06	4.1E+05	5.2E+05	5.2E+05	3.
CO	kg CO	3.9E+05	1.1E+06	5.5E+05	1.2E+06	6.5E+05	8.2E+05	1.8E+05	1.

3.1 Results without system expansion

The results of the scenarios without system expansion cannot be compared with each other since they do neither fulfill the same functional unit nor deliver the same final service. Yet, to depict each scenario without expansion is useful to first depict the environmental bottlenecks of the different waste treatment processes, and to depict the environmental bottlenecks within each value chain or scenario.

<Figure 2>

3.1.1 Climate change impact potential (CCIP)

It is estimated that approximately 89% of the life-cycle impact for the WtE scenario is a consequence of the fossil CO_2 from the WtE plant and the remainder is caused by transport (7%), consumables used for the flue gas cleaning processes (3%) and production of material for the plant (1%). Thus, the introduction of CCS technologies into the original WtE system leads to lower CCIP, due to the direct reduction of fossil CO_2 emissions. The use of RDF as a secondary fuel in the add-on boiler does not lead to a large increase of CO_2 emissions either, since the flue gas also goes through the CCS unit. For the recycling scenario, the total impacts are caused by the incineration with energy recovery of MSW (62%), the recycling of paper (18%), the recycling of plastic (6%) and the anaerobic digestion of organic waste (1%). The burdens of the landfill scenario are caused by the biogenic CH_4 emitted at the landfill site after decomposition of biodegradable material such as paper and wood. Overall, transport is identified as a minor contributor to the total CCIP impacts for the WtE, Recycling and Landfill scenarios (7%, 12% and 5% contributions, respectively). Yet, transport's relative contribution to the CCS scenario is higher (26%), due to the reduction of CO_2 fossil emissions at the WtE plant. On a per kg basis, as shown in Table S13, with the incineration of 1 kg of MSW with the reference waste mix (MSW_{2015}) as the starting point, the total impacts are increased by 66% and 3% when sending the paper fraction to paper recycling or the same MSW mix to landfill. On the other hand, the total burdens are reduced by 4%, 22% and 78% when sending the plastic fraction to plastic recycling, changing the MSW waste mix in compliance with the circular economy and adding a carbon capture unit.

3.1.2 Freshwater eutrophication potential (FEP)

For the WtE and CCS scenarios, the impacts of FEP are driven by the phosphorus (P) content of the waste, in the ashes resulting after combustion and from leaching at the landfill site. The impacts of CCS are higher than WtE due to the use of additional fuel (RDF) in the CO_2 capture unit. The results of Landfill are higher than the results of WtE, despite the same amount of incoming P. The P in the two scenarios do not enter the landfill site in the same form; for WtE, P is in the form of ash and for Landfill, it is in the form of solid waste. The leaching of the elements contained in the waste – P in the case of FEP – is dependent on the form (MSW or ashes) of the waste entering the landfill, and the results are thus different, despite the same amount of entering P. The disposal of the consumables used in the flue gas processes is responsible for 12% of the total impacts of the WtE scenario. The impact of transport is marginal, and is caused by the combustion of fossil fuel while driving the lorries that transport the waste. For the Circular Economy scenario, the impacts are mainly caused by the leaching of P on agricultural land; P is either contained in the ash mixture resulting from the recycling paper process or in the digestate from anaerobic digestion.

3.1.3 Human toxicity (HTP) and ozone depletion potentials (ODP)

The same elements of the value chains are causing the burdens of FEP and HTP, with the difference that it is not the leaching of P, but of the heavy metals that causes the environmental harm. For ODP, transport is the main contributor for the Landfill, CCS and WtE scenarios with respective shares of 56%, 62% and 74%. The second ODP contributor is the use, and thus combustion, of fossil fuels while producing the different consumables used in the recycling and flue gas cleaning processes and while running and building the waste treatment sites. For the Circular Economy scenario, the total impact of transport is comparable to the other scenarios. However, its contribution to the total impacts is lower (33%) due to the larger use of fossil fuel in the recycling processes.

3.1.4 Single stressors

For NO_x, the direct emissions occurring at the waste treatment plant constitute the majority of the emissions for all the scenarios but Landfill, with sources being the combustion of the waste (MSW and RDF to feed the add-on boiler for CCS) and the use of fossil fuels in the recycling process. The NO_x emissions at the landfill site come from the combustion of fossil fuels to run the landfill. Transport contributes with a share of around 30% for the three first scenarios, and is the major contributor for the Landfill scenario (60%).

The major sources of PM are the combustion of fossil fuel in transport and background processes while producing the materials to build the different waste treatment plant, and the auxiliary materials in the different value chains. The direct emissions of PM at the WtE plant contribute only

11% of the total burdens of the WtE scenario. Thus, despite the co-capture efficiency of 50% in the CCS process and the potential reduction of PM in the system, the small share of direct emissions is offset by the impact of fossil fuel use in the background process, while producing the auxiliary materials (NaOH, MEA and activated carbon) as well as the CCS infrastructure.

For SO_2, in opposition to PM, direct emissions occurring at the WtE plant are the main contributor for the WtE scenario with a contribution of 75% to the total impacts. In addition to the high co-capture of SO_2 efficiencies of the CCS process (99.5%), the impact of fossil fuel combustion in the background value chain is almost totally offset, and the CCS subgroup ends up with a marginal net share.

For CO, as with PM, the combustion of fossil fuels in background processes and in transport is the main source of emissions.

Overall, we can see: (1) the major influence of fossil CO_2, NO_x and SO_2 at the WtE plant, (2) the contribution of transport, mainly for ODP, PM and CO, for all the scenarios, (3) the impact of the production of the auxiliary materials used in the flue gas cleaning, recycling and CCS processes, (4) the impact of secondary waste streams for FEP and HTP, and (5) the introduction of CCS technologies in the original WtE system leading to lower CCIP but increased life-cycle values for all the other mid-point impacts and stressors.

3.2 Results with system expansion

The results of Figure 2 are now presented with system expansion in Figure 3. The total results of Figure 2 are grouped under the sub-group Waste treatment, and the results for system expansion are presented separately for each material and energy to be provided.

<Figure 3>

All the scenarios increase their total impact for each single stressor and impact category. Landfill is the scenario that increases its impact the most with an increase from 32% for HTP to 2027% for ODP. The Circular economy scenario is the least altered, with an increase ranging from 11% for FEP to 32% for NOx. On a material basis, the direct emissions caused by the burning of oil impacts mainly the burdens for CCIP, ODP, and PM, SO_2 and CO. The same is valid for the production of plastic where the combustion of fossil fuels in the production chain induces the same increases. The primary production of paper affects the same single stressors and impacts categories as the production of plastic. In addition, it also affects FEP, due to the phosphorus contained in the waste generated on the production site. On the other hand, due to its small quantity, the production of synthetic fertilizer does not affect the results. Nor does electricity impact the results, due to its renewable source.

The estimated costs – with the limitations mentioned in the methodology – are the lowest for the WtE scenario (1383 MNOK). The costs of the CCS and Circular Economy scenario are similar. The increase for the former is caused by the use of auxiliary fuel (RDF) and the CCS process. The increase for the latter is caused by the additional recycling facilities to be built (material recycling and anaerobic digestion plants). The landfill scenario entails the largest costs (200% increase) even without including the direct costs of the landfill itself. The reason is that heat produced from heating oil is roughly three times more expensive than WtE due to the high heating oil price. Even if we exclude the heating oil tax and the CO_2 tax, in total about 25% of the price, this scenario will still be the most expensive. The value of material recovery (99 ktonne recycled plastic and 135 ktonne recycled paper) is not included in the net cost estimates. Given the large amounts, it is likely that the net total cost is lower for Circular economy than CCS when the income effect (saved costs) of recycling is taken into account. This will happen if the average price is above about NOK 340/tonne. It is, however, highly likely that the total cost of WtE will still be the lowest even when all costs and incomes are taken into account.

The Landfill scenario ranks worst for all the impact categories and single stressors assessed, since landfill is the only waste treatment option that does not recover either materials or energy. Except for CCIP, the CCS scenario performs worse than the WtE scenario. The order of recycling and energy recovery of the waste hierarchy is not respected for: (1) HTP, due to secondary waste streams and (2) ODP, due to addition fossil fuel used in the recycling processes. The waste hierarchy is respected for FEP, PM, SO_2, PM and NO_x.

3.2.1 Sensitivity analysis on the climate metrics

The results for the WtE scenario without system expansion are presented in Figure 4 with a sensitivity analysis on the climate metrics.

<Figure 4>

Since CO_2 contributes mainly to CCIP, and because of the long atmospheric lifetime of CO_2, assessing the CCIP by using GWP100 or GTP100 does not change the overall CCIP results much. When including the NTCF, the overall results may decrease by a maximum of 13% in the best case for very-short term climate change impacts (GWP20). In total, the CCIP results vary with a decrease ranging from 1% to 13%, caused by net cooling mainly due to NO_x.

4 Discussion

For the WtE scenario, we find a total contribution to CCIP of 507 g CO_2 eq/kg of waste. This result is in line with previous studies on similar systems; Lausselet et al. (2016) find a contribution of 265 to

637 g CO_2 eq/kg of waste, Astrup et al. (2009b) 347–371 g CO2 eq/kg of waste for the direct emissions occurring at the WtE plant, and Turconi et al. (2011) find fossil CO_2 emissions of 280–450 g/kg of waste. Furthermore, these studies, as well as the present study, show the importance of using quicklime as a consumable. The measured air emissions at WtE plants, as reported by Norwegian Environment Agency (2016) and used here, are in line with the air emissions reported by Damgaard et al. (2010) for similar air pollution control technologies (APC 5 and 6).

Direct emissions of fossil CO_2 occurring at the WtE plants are the main driver for CCIP. CO_2 emissions are unavoidable and cannot be reduced by conventional flue gas treatment, but only by the use of CCS. On the other hand, NO_X, SO_2, HCl, PM, dioxins and heavy metals are relatively low since the plants are equipped with efficient flue gas treatment technologies, in accordance with (Turner et al. 2011, Polettini 2012, Turner et al. 2015, Lausselet et al. 2016).

Accounting of GHG emissions is a major focus within waste management (Gentil et al. 2009a), and climate change is affected by a variety of forcing agents. In addition to the well-known well-mixed GHG (WMGHGs), human activities disturb the climate system through emissions of pollutants such as NO_x, CO, volatile organic compounds (VOCs), black carbon (BC), organic carbon (OC), and sulphur oxides (SO_x). The net climate impacts of NTCFs are the result of many complex opposing effects with different temporal evolutions at play; NO_x, CO, VOCs are tropospheric ozone formation precursors, BC and OC are primary aerosols, while NO_x, SO_x, NH_3 are precursors to secondary aerosols. Quantifying them is subject to uncertainties that are larger than for WMGHGs. The few LCA studies that take into account the NTCFs, all argue for their routine inclusion in environmental system analysis (Peters et al. 2011, Tsao et al. 2012, Cherubini et al. 2016, Iordan et al. 2016, Levasseur et al. 2016).

In this work, we analyse the incorporation of the post-combustion MEA CO_2 capture process in the WtE plants. In order to produce the required steam for the operation of the stripper, we assume an additional boiler is installed and fed by RDF, leading to an important increase in auxiliary fuel. Novel solvents are currently the subject of research aiming to reduce the energy penalty of the carbon capture processes (Artanto et al. 2014, Sanchez Fernandez et al. 2014, Manzolini et al. 2015), and other gas separation technologies, such as pressure swing adsorption or membranes, could enable lower energy penalties (Merkel et al. 2010). However these processes require electricity instead of heat.

The leaching of some elements of the bottom ash, fly ash and filter cake are pointed out as significant contributors to FEP and HTP. This finding coincides with the conclusion drawn by Cherubini et al. (2008), Cherubini et al. (2009), Christensen et al. (2007), Allegrini et al. (2015a) and

Burnley et al. (2015). In addition to the leaching of P and heavy metals from the bottom and fly ash, the leaching of the same elements from the new waste streams emerging from the recycling processes have also been shown to influence HTP and FEP in this study. The recovery of the bottom ash, fly ash and new waste streams or the use of other treatments could drastically reduce the FEP and HTP impacts, as highlighted. As an alternative, bottom ash could for instance be recovered as road construction material (Birgisdóttir et al. 2006, Birgisdóttir et al. 2007).

Waste treatment systems are by definition complex and embedded with uncertainties (Clavreul et al. 2012, Laurent et al. 2014a), and this study is no exception. The uncertainty in the incoming waste is somehow mitigated, since waste composition varies throughout the year. The waste mix assumed here can thus be seen as a realistic estimation of the annual average waste mix. The uncertainty in the measurement data can be assumed to be low for the air emissions at the WtE plant site. The opposite is true for the measurements of the chemical elements contained in the bottom ash from Heie et al. (2015). Uncertainties are also embedded in the choice of background processes, in the low availability of raw recycling process data and in the severe lack of data for some recycled materials in LCA databases (Brogaard et al. 2014). This uncertainty also applies to the chemical composition of the waste input.

While performing mathematical optimization to find the calibrated mass transfer coefficient matrices for each individual plant, the condition described in equation (8) could not always be fulfilled. If the total of the row was higher than 1, each element of the row in question had to be divided by the total, in order for it to be equal to 1. This was sometimes true for elements such as chlorine and fluorine, and especially true for heavy metals such as arsenic, barium, cadmium, manganese, antimony and zinc. As a result, one might suspect the level of these particular elements to be higher in the incoming waste than assumed in this analysis.

The gradual implementation of circular economy (EU package) will lead to a diversion of MSW from landfills to both material and energy recovery (minimally relevant for Norway with a landfill ban on biodegradable waste since 2009) and increase material recycling for specific fractions (before energy recovery of the residual fractions). These movements will clearly affect both the quality and quantity of the fuel mix going to energy recovery, with potentially large consequences on logistics and operation. What these changes will be remains to be seen and is difficult to predict today. Another aspect is the development of new technologies and treatment routes for utilizing or upgrading residues (fly ash, bottom ash).

WtE is a well-established, knowledgeable sector with a large network of operating sites. Its role today is mainly twofold: waste disposal (volume and weight reduction) in a safe way (destruction of contaminated materials) and energy production. A circular economy may actually give WtE the opportunity to strengthen and expand its role with new or little developed value chains, such as secondary raw materials production (metals and minerals from ash, building materials from ash or RDF production) and a stronger involvement in material recycling with more on-site sorting. This expansion might also give a push towards new, advanced concepts, such as carbon capture use and storage (CCUS), energy storage and flexibility and new- or multi-products systems (e.g. waste refineries, biofuel production, biogas + WtE). In other words, the WtE sector activities will both broaden and "go up" the waste hierarchy.

The opportunities for the WtE sector to play an "extended role" in waste management and move up the waste hierarchy and towards new products are not without their hurdles. The challenges are techno-economic (the costly development of new technologies and investment in new machinery), political/regulatory (WtE actors need a stable framework to evolve and invest in the nascent circular economy) and operational (the changing quantities and properties of the MSW fractions to be energy recovered). One should not underestimate job creation in an extended WtE sector (also central in the EU Energy Union strategy), especially when it is connected to a reduction in the carbon footprint/GHG emissions from waste management.

Several Norwegian WtE plants are currently suffering from low profitability. The main reason is overcapacity in Scandinavia, where Swedish WtE sets the gate fees (Becidan et al. 2015). They can offer lower rates because of higher revenues from energy delivered in well-developed district heating (DH) systems. Another challenge is the lack of new projects that can secure long-term, strong revenue streams from energy. The major cities in Norway already have well-developed district heating infrastructures, so the remaining district heating market is limited to small-scale applications.

Yet, this study illustrates new potential for WtE plants; the focus for WtE systems has traditionally been on the energy recovery aspect, often neglecting the potential for recovery of materials that end up and accumulate in incineration residues. As stressed by Morf et al. (2013) and Boesch et al. (2014), waste incineration has great potential for recovering metal resources. Incineration plays an important role as an element of industrial ecology, providing waste disposal services and helping to close material and energetic cycles.

5 Conclusion

In this paper, LCA is combined with analysis of the overall energy and material balances, mathematical optimization and cost assessment in order to assess the current Norwegian WtE system and the implication of the circular economy package and the addition of CCS. Also, a landfill scenario is added as a check on the waste hierarchy.

The Landfill scenario ranks worst for all the impact categories and single stressors assessed. Except for climate change, the CCS scenario performs worse than the WtE scenario. The order of recycling and energy recovery in the waste hierarchy is not respected for: (1) HTP, due to secondary waste streams and (2) ODP, due to additional fossil fuel used in the recycling processes. The waste hierarchy is respected for PM, SO_2, PM and NO_x. The inclusion of near-term climate forcers decreases the climate change impacts 1% to 13% due to a net cooling mainly due to NO_x.

A circular economy may actually give WtE systems the opportunity to strengthen and expand their role in growing new or little developed value chains, such as secondary raw materials production (e.g. recovery of metals and minerals, building materials and fertilizers) and valorization of new waste streams occurring during material recycling. Additional costs will also be incurred in order to build the new required infrastructure. However, some of this cost could potentially be decreased by reusing the secondary waste streams that are generated.

Acknowledgements

This publication has been funded by CenBio – Bioenergy Innovation Centre. CenBio is co-funded by the Research Council of Norway (193817/E20) under the FME scheme and the research and industry partners.

6 References

Al-Salem, S. M., P. Lettieri, and J. Baeyens. 2009. Recycling and recovery routes of plastic solid waste (PSW): A review. Waste Management **29**:2625-2643.

Allegrini, E., S. Butera, D. S. Kosson, A. Van Zomeren, H. A. Van der Sloot, and T. F. Astrup. 2015a. Life cycle assessment and residue leaching: The importance of parameter, scenario and leaching data selection. Waste Management **38**:474-485.

Allegrini, E., A. Maresca, M. E. Olsson, M. S. Holtze, A. Boldrin, and T. F. Astrup. 2014. Quantification of the resource recovery potential of municipal solid waste incineration bottom ashes. Waste Management **34**:1627-1636.

Allegrini, E., C. Vadenbo, A. Boldrin, and T. F. Astrup. 2015b. Life cycle assessment of resource recovery from municipal solid waste incineration bottom ash. Journal of Environmental Management **151**:132-143.

Arena, U., M. L. Mastellone, and F. Perugini. 2003a. The environmental performance of alternative solid waste management options: A life cycle assessment study. Chemical Engineering Journal **96**:207-222.

Arena, U., M. L. Mastellone, and F. Perugini. 2003b. Life cycle assessment of a plastic packaging recycling system. International Journal of Life Cycle Assessment **8**:92-98.

Artanto, Y., J. Jansen, P. Pearson, G. Puxty, A. Cottrell, E. Meuleman, and P. Feron. 2014. Pilot-scale evaluation of AMP/PZ to capture CO2 from flue gas of an Australian brown coal-fired power station. International Journal of Greenhouse Gas Control **20**:189-195.

Astrup, T., T. Fruergaard, and T. H. Christensen. 2009a. Recycling of plastic: Accounting of greenhouse gases and global warming contributions. Waste Management and Research **27**:763-772.

Astrup, T., H. Mosbæk, and T. H. Christensen. 2006. Assessment of long-term leaching from waste incineration air-pollution-control residues. Waste Management **26**:803-814.

Astrup, T., J. Møller, and T. Fruergaard. 2009b. Incineration and co-combustion of waste: Accounting of greenhouse gases and global warming contributions. Waste Management and Research **27**:789-799.

Astrup, T. F., D. Tonini, R. Turconi, and A. Boldrin. 2015. Life cycle assessment of thermal Waste-to-Energy technologies: Review and recommendations. Waste Management **37**:104-115.

Azar, C., D. J. A. Johansson, and N. Mattsson. 2013. Meeting global temperature targets - The role of bioenergy with carbon capture and storage. Environmental Research Letters **8**.

Bartl, A. 2014. Moving from recycling to waste prevention: A review of barriers and enables. Waste Management and Research **32**:3-18.

Becidan, M., L. Wang, M. Fossum, H.-O. Midtbust, J. Stuen, J. I. Bakken, and E. Evensen. 2015. Norwegian Waste-to-Energy (WtE) in 2030: Challenges and Opportunities. Chemical Enginnering Transactions **vol. 43**.

Bergsdal, H., A. H. Strømman, and E. G. Hertwich. 2005. Environmental assessment of two waste incineration strategies for Central Norway. International Journal of Life Cycle Assessment **10**:263-272.

Birgisdóttir, H., G. Bhander, M. Z. Hauschild, and T. H. Christensen. 2007. Life cycle assessment of disposal of residues from municipal solid waste incineration: Recycling of bottom ash in road construction or landfilling in Denmark evaluated in the ROAD-RES model. Waste Management **27**:S75-S84.

Birgisdóttir, H., K. A. Pihl, G. Bhander, M. Z. Hauschild, and T. H. Christensen. 2006. Environmental assessment of roads constructed with and without bottom ash from municipal solid waste incineration. Transportation Research Part D: Transport and Environment **11**:358-368.

Björklund, A., and G. Finnveden. 2005. Recycling revisited - Life cycle comparisons of global warming impact and total energy use of waste management strategies. Resources, Conservation and Recycling **44**:309-317.

Blengini, G. A., E. Brizio, M. Cibrario, and G. Genon. 2011. LCA of bioenergy chains in Piedmont (Italy): A case study to support public decision makers towards sustainability. Resources, Conservation and Recycling **57**:36-47.

Boesch, M. E., C. Vadenbo, D. Saner, C. Huter, and S. Hellweg. 2014. An LCA model for waste incineration enhanced with new technologies for metal recovery and application to the case of Switzerland. Waste Management **34**:378-389.

Bond, T. C., D. G. Streets, K. F. Yarber, S. M. Nelson, J. H. Woo, and Z. Klimont. 2004. A technology-based global inventory of black and organic carbon emissions from combustion. Journal of Geophysical Research D: Atmospheres **109**.

Boulamanti, A. K., S. Donida Maglio, J. Giuntoli, and A. Agostini. 2013. Influence of different practices on biogas sustainability. Biomass and Bioenergy **53**:149-161.

Branker, K., M. J. M. Pathak, and J. M. Pearce. 2011. A review of solar photovoltaic levelized cost of electricity. Renewable and Sustainable Energy Reviews **15**:4470-4482.

Brogaard, L. K., A. Damgaard, M. B. Jensen, M. Barlaz, and T. H. Christensen. 2014. Evaluation of life cycle inventory data for recycling systems. Resources, Conservation and Recycling **87**:30-45.

Bruckner, T., H. Chum, A. Jäger-Waldau, Å. Killingtveit, L. Gutiérrez-Negrín, J. Nyboer, W. Musial, A. Verbruggen, and R. Wiser. 2011. Annex III: Recent Renewable Energy Cost and Performance Parameters in O. Edenhofer, R. Pichs-Madruga, Y. Sokona, K. Seyboth, P. Matschoss, S. Kadner, T. Zwickel, P. Eickemeier, G. Hansen, S. Schlömer, and C. von Stechow. Cambridge, United Kingdom and New York, NY, USA.

Brunner, P. H., and U. Kral. 2014. Final sinks as key elements for building a sustainable recycling society. Sustainable Environment Research **24**:443-448.

Brunner, P. H., and H. Rechberger. 2015. Waste to energy – key element for sustainable waste management. Waste Management **37**:3-12.
Burnley, S., T. Coleman, and A. Peirce. 2015. Factors influencing the life cycle burdens of the recovery of energy from residual municipal waste. Waste Management **39**:295-304.
Carpentieri, M., A. Corti, and L. Lombardi. 2005. Life cycle assessment (LCA) of an integrated biomass gasification combined cycle (IBGCC) with CO2 removal. Energy Conversion and Management **46**:1790-1808.
Chandel, M. K., G. Kwok, R. B. Jackson, and L. F. Pratson. 2012. The potential of waste-to-energy in reducing GHG emissions. Carbon Management **3**:133-144.
Cherubini, F., S. Bargigli, and S. Ulgiati. 2008. Life cycle assessment of urban waste management: Energy performances and environmental impacts. The case of Rome, Italy. Waste Management **28**:2552-2564.
Cherubini, F., S. Bargigli, and S. Ulgiati. 2009. Life cycle assessment (LCA) of waste management strategies: Landfilling, sorting plant and incineration. Energy **34**:2116-2123.
Cherubini, F., R. M. Bright, and A. H. Strømman. 2012. Site-specific global warming potentials of biogenic CO2 for bioenergy: Contributions from carbon fluxes and albedo dynamics. Environmental Research Letters **7**.
Cherubini, F., J. Fuglestvedt, T. Gasser, A. Reisinger, O. Cavalett, M. A. J. Huijbregts, D. J. A. Johansson, S. V. Jørgensen, M. Raugei, G. Schivley, A. H. Strømman, K. Tanaka, and A. Levasseur. 2016. Bridging the gap between impact assessment methods and climate science. Environmental Science and Policy **64**:129-140.
Christensen, T. H., G. Bhander, H. Lindvall, A. W. Larsen, T. Fruergaard, A. Damgaard, S. Manfredi, A. Boldrin, C. Riber, and M. Hauschild. 2007. Experience with the use of LCA-modelling (EASEWASTE) in waste management. Waste Management and Research **25**:257-262.
Chum, H., A. Faaij, J. Moreira, G. Berndes, P. Dhamija, H. Dong, G. Gabrielle, A. Goss Eng, W. Lucht, M. Mapako, O. Masera, T. Cerutti, T. McIntyre, K. Minowa, and K. Pingoud. 2011. Bioenergy.in O. Edenhofer, R. Pichs-Madruga, Y. Sokona, K. Seyboth, P. Matschoss, S. Kadner, T. Zwickel, Eickemeier, P., G. Hansen, S. Schlömer, and C. von Stechow-IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation. Cambridge, United Kingdom and New York, NY, USA.
Clavreul, J., D. Guyonnet, and T. H. Christensen. 2012. Quantifying uncertainty in LCA-modelling of waste management systems. Waste Management **32**:2482-2495.
Consonni, S., M. Giugliano, A. Massarutto, M. Ragazzi, and C. Saccani. 2011. Material and energy recovery in integrated waste management systems: Project overview and main results. Waste Management **31**:2057-2065.
Corti, A., and L. Lombardi. 2004. Biomass integrated gasification combined cycle with reduced CO2 emissions: Performance analysis and life cycle assessment (LCA). Energy **29**:2109-2124.
Darling, S. B., F. You, T. Veselka, and A. Velosa. 2011. Assumptions and the levelized cost of energy for photovoltaics. Energy and Environmental Science **4**:3133-3139.
De Meester, S., J. Demeyer, F. Velghe, A. Peene, H. Van Langenhove, and J. Dewulf. 2012. The environmental sustainability of anaerobic digestion as a biomass valorization technology. Bioresource Technology **121**:396-403.
Doka, G. 2007. Life Cycle Inventories of Waste Treatment Services. Swiss Centre for Life Cycle Inventories, Dübendorf.
Doka, G., and R. Hischier. 2005. Waste treatment and assessment of long-term emissions. International Journal of Life Cycle Assessment **10**:77-84.
Ecoinvent Centre. 2010. Ecoinvent data and reports v2.2. Swiss Centre for Life Cycle Inventories, Dübendorf, Switzerland.
Eriksson, O., and G. Finnveden. 2009. Plastic waste as a fuel - CO2-neutral or not? Energy & Environmental Science **2**:907-914.

European Commission. 2015. Communication from the commission to the European parliament, the council, the European economic and social committee and the committee of the regions - Closing the loop - An EU action plan for the Circular Economy.
European Union Council. 1999. Council Directive 1999/31/EC on the landfill of waste. OJ L 182, Brussels.
Finnveden, G. 1999. Methodological aspects of life cycle assessment of integrated solid waste management systems. Resources, Conservation and Recycling **26**:173-187.
Fruergaard, T., and T. Astrup. 2011. Optimal utilization of waste-to-energy in an LCA perspective. Waste Management **31**:572-582.
Gentil, E., T. H. Christensen, and E. Aoustin. 2009a. Greenhouse gas accounting and waste management. Waste Management and Research **27**:696-706.
Gentil, E., J. Clavreul, and T. H. Christensen. 2009b. Global warming factor of municipal solid waste management in Europe. Waste Management and Research **27**:850-860.
Gentil, E. C., A. Damgaard, M. Hauschild, G. Finnveden, O. Eriksson, S. Thorneloe, P. O. Kaplan, M. Barlaz, O. Muller, Y. Matsui, R. Ii, and T. H. Christensen. 2010. Models for waste life cycle assessment: Review of technical assumptions. Waste Management **30**:2636-2648.
Giugliano, M., S. Cernuschi, M. Grosso, and L. Rigamonti. 2011. Material and energy recovery in integrated waste management systems. An evaluation based on life cycle assessment. Waste Management **31**:2092-2101.
Goedkoop, M., R. Heijungs, M. A. J. Huijbregts, A. D. Schryver, J. Struijs, and R. V. Zelm. 2009. ReCiPe 2008. Dutch Ministery of the Environment, The Hague, Netherlands.
Guest, G., R. M. Bright, F. Cherubini, and A. H. Strømman. 2013a. Consistent quantification of climate impacts due to biogenic carbon storage across a range of bio-product systems. Environmental Impact Assessment Review **43**:21-30.
Guest, G., F. Cherubini, and A. H. Strømman. 2013b. The role of forest residues in the accounting for the global warming potential of bioenergy. GCB Bioenergy **5**:459-466.
Haro, P., F. Trippe, R. Stahl, and E. Henrich. 2013. Bio-syngas to gasoline and olefins via DME - A comprehensive techno-economic assessment. Applied Energy **108**:54-65.
Hauschild, M., S. I. Olsen, E. Hansen, and A. Schmidt. 2008. Gone...but not away - Addressing the problem of long-term impacts from landfills in LCA. International Journal of Life Cycle Assessment **13**:547-554.
Heie, A., H. Rau Heien, K. Sandberg, M. Zabrodina, and T. Metlie Hagen. 2015. Characterization of the bottom ash from incineration plants in Norway (in Norwegian). Avfall Norge (Norwegian Waste Management Association).
Hellweg, S., and L. M. I. Canals. 2014. Emerging approaches, challenges and opportunities in life cycle assessment. Science **344**:1109-1113.
Heyne, S., and S. Harvey. 2014. Impact of choice of CO2 separation technology on thermo-economic performance of Bio-SNG production processes. International Journal of Energy Research **38**:299-318.
Hopewell, J., R. Dvorak, and E. Kosior. 2009. Plastics recycling: Challenges and opportunities. Philosophical Transactions of the Royal Society B: Biological Sciences **364**:2115-2126.
IEA. 2007. CO2 capture from medium scale combustion installations.
IEA. 2015. World Energy Outlook.
IEA Bionergy. 2013. Waste to Energy - Summary and Conclusions from the IEA Bioenergy ExCo71 Workshop.
Iordan, C., C. Lausselet, and F. Cherubini. 2016. Life-cycle assessment of a biogas power plant with application of different climate metrics and inclusion of near-term climate forcers. Journal of Environmental Management **184**:517-527.
IPCC. 2007. Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor and H.L. Miller (eds.)]. Cambridge, United Kingdom and New York, NY, USA.

Janssens-Maenhout, G., F. Dentener, J. Van Aardenne, S. Monni, V. Pagliari, L. Orlandini, Z. Klimont, J. Kurokawa, H. Akimoto, T. Ohara, R. Wankmueller, B. Battye, D. Grano, A. Zuber, and T. Keating. 2012. EDGAR-HTAP: a harmonized gridded air pollution emission dataset based on national inventories. European Commissio- Joint Resarch Center (JRC).

Kunwar, B., H. N. Cheng, S. R. Chandrashekaran, and B. K. Sharma. 2016. Plastics to fuel: a review. Renewable and Sustainable Energy Reviews **54**:421-428.

Laurent, A., I. Bakas, J. Clavreul, A. Bernstad, M. Niero, E. Gentil, M. Z. Hauschild, and T. H. Christensen. 2014a. Review of LCA studies of solid waste management systems – Part I: Lessons learned and perspectives. Waste Management **34**:573-588.

Laurent, A., J. Clavreul, A. Bernstad, I. Bakas, M. Niero, E. Gentil, T. H. Christensen, and M. Z. Hauschild. 2014b. Review of LCA studies of solid waste management systems – Part II: Methodological guidance for a better practice. Waste Management **34**:589-606.

Lausselet, C., F. Cherubini, G. del Alamo Serrano, M. Becidan, and A. H. Strømman. 2016. Life-cycle assessment of a Waste-to-Energy plant in central Norway: Current situation and effects of changes in waste fraction composition. Waste Management **58**:191-201.

Lazarevic, D., E. Aoustin, N. Buclet, and N. Brandt. 2010. Plastic waste management in the context of a European recycling society: Comparing results and uncertainties in a life cycle perspective. Resources, Conservation and Recycling **55**:246-259.

Levasseur, A., O. Cavalett, J. S. Fuglestvedt, T. Gasser, D. J. A. Johansson, S. V. Jørgensen, M. Raugei, A. Reisinger, G. Schivley, A. Strømman, K. Tanaka, and F. Cherubini. 2016. Enhancing life cycle impact assessment from climate science: Review of recent findings and recommendations for application to LCA. Ecological Indicators **71**:163-174.

Liu, B., Q. Wei, B. Zhang, and J. Bi. 2013. Life cycle GHG emissions of sewage sludge treatment and disposal options in Tai Lake Watershed, China. Science of the Total Environment **447**:361-369.

Lozanovski, A., J. P. Lindner, and U. Bos. 2014. Environmental evaluation and comparison of selected industrial scale biomethane production facilities across Europe. The International Journal of Life Cycle Assessment **19**:1823-1832.

Luberti, M., S. Natarajan, Y. Zaccheus, and A. Giovanoglou. 2016. Depressurization of CO2 rich mixtures: Challenges for the safe process design of CCS facilities and CO2 EOR systems. Pages 361-366 Chemical Engineering Transactions.

Luderer, G., R. C. Pietzcker, C. Bertram, E. Kriegler, M. Meinshausen, and O. Edenhofer. 2013. Economic mitigation challenges: How further delay closes the door for achieving climate targets. Environmental Research Letters **8**.

Lupo, E., M. Moroni, F. La Marca, S. Fulco, and V. Pinzi. 2016. Investigation on an innovative technology for wet separation of plastic wastes. Waste Management **51**:3-12.

Lyng, K. A., I. S. Modahl, H. Møller, J. Morken, T. Briseid, and O. J. Hanssen. 2015. The BioValueChain model: a Norwegian model for calculating environmental impacts of biogas value chains. International Journal of Life Cycle Assessment **20**:490-502.

Majeau-Bettez, G., and A. H. Strømman. 2016. Documentation for Arda Calculator, version 1.8.3. 2016.

Manzolini, G., E. Sanchez Fernandez, S. Rezvani, E. Macchi, E. L. V. Goetheer, and T. J. H. Vlugt. 2015. Economic assessment of novel amine based CO<inf>2</inf> capture technologies integrated in power plants based on European Benchmarking Task Force methodology. Applied Energy **138**:546-558.

Merkel, T. C., H. Lin, X. Wei, and R. Baker. 2010. Power plant post-combustion carbon dioxide capture: An opportunity for membranes. Journal of Membrane Science **359**:126-139.

Merrild, H., A. Damgaard, and T. H. Christensen. 2008. Life cycle assessment of waste paper management: The importance of technology data and system boundaries in assessing recycling and incineration. Resources, Conservation and Recycling **52**:1391-1398.

Merrild, H., A. W. Larsen, and T. H. Christensen. 2012. Assessing recycling versus incineration of key materials in municipal waste: The importance of efficient energy recovery and transport distances. Waste Management **32**:1009-1018.

Ministry of Finance. 2014. Principles and requirements for social cost-benefit analyses (in Norwegian). Oslo, Norway.

Moomaw, W., P. Burgherr, G. Heath, M. Lenzen, J. Nyboer, and A. Verbruggen. 2011. Annex II: Methodology.in O. Edenhofer, R. Pichs-Madruga, Y. Sokona, K. Seyboth, P. Matschoss, S. Kadner, T. Zwickel, Eickemeier, P., G. Hansen, S. Schlömer, and C. von Stechow, IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation. Cambridge, United Kingdom and New York, NY, USA.

Morf, L. S., R. Gloor, O. Haag, M. Haupt, S. Skutan, F. D. Lorenzo, and D. Böni. 2013. Precious metals and rare earth elements in municipal solid waste - Sources and fate in a Swiss incineration plant. Waste Management **33**:634-644.

Myhre, G., D. Shindell, F.-M. Bréon, W. Collins, J. Fuglestvedt, J. Huang, D. Koch, J.-F. Lamarque, D. Lee, B. Mendoza, T. Nakajima, A. Robock, G. Stephens, T. Takemura, and H. Zhang. 2013. Anthropogenic and Natural Radiative Forcing. In: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.)]. Cambridge, United Kingdom and New York, NY, USA.

Nedland, K. 2011. Development of biogas in Norway - Cost of biogasproduction in Norway, Sweden and Danemark (in Norwegian). Avfall Norge/Asplan Viak AS.

NETL. 2012a. Greenhouse gas reductions in the power industry using domestic coal and biomass, vol.2 National Energy Technology Labotory, USA.

NETL. 2012b. Greenhouse Gas reductions in the power industry using domestic coal and biomass, vol. 1. National Energy Technology Labotory, USA.

Ning, S.-K., N.-B. Chang, and M.-C. Hung. 2013. Comparative streamlined life cycle assessment for two types of municipal solid waste incinerator. Journal of Cleaner Production **53**:56-66.

Niu, D.-j., H. Huang, X.-h. Dai, and Y.-c. Zhao. 2013. Greenhouse gases emissions accounting for typical sewage sludge digestion with energy utilization and residue land application in China. Waste Management **33**:123-128.

Norges vassdrags- og energidirektorat. 2011. Kostnader ved produksjon av kraft og varme (Costs of production of electricity and heat, in Norwegian). Norwegian Water Resources and Energy Directorate, Oslo.

Norwegian Environment Agency. 2013. Background materials for intersectoral biogas strategy. Climate and Pollution directorate, Oslo, Norway.

Norwegian Environment Agency. 2016. Total emissions to air in Norway.

Norwegian Environment Agency. 2017. Import and export of waste (in Norwegian). Norwegian Environment Agency,, Oslo, Norway.

Norwegian Ministry of Climate and Environment. 2016. Forskrift om gjenvinning og behandling av avfall (avfallsforskriften) (Regulation on waste recovery and treatment). Lovdata, Oslo, Norway.

NOU. 2012. 16. Cost-Benefit Analysis (in Norwegian). Ministry of Finance, Oslo, Norway.

Oreggioni G D, Singh B, F. Cherubini, G. Guest, C. Lausselet, M. Luberti, H. Ahn, and A. Strømman. 2016. Environmental trade off associated to carbon capture technologies in biomass fuelled plants: The case of absorptive and adsorptive carbon capture units in biomass gasification CHP plants. Accepted in International Journal of Greenhouse Gas Control.

Oreggioni, G. D., S. Brandani, M. Luberti, Y. Baykan, D. Friedrich, and H. Ahn. 2015. CO<inf>2</inf> capture from syngas by an adsorption process at a biomass gasification CHP plant: Its comparison with amine-based CO<inf>2</inf> capture. International Journal of Greenhouse Gas Control **35**:71-81.

Passarini, F., M. Nicoletti, L. Ciacci, I. Vassura, and L. Morselli. 2014. Environmental impact assessment of a WtE plant after structural upgrade measures. Waste Management **34**:753-762.

Patterson, T., S. Esteves, R. Dinsdale, and A. Guwy. 2011. Life cycle assessment of biogas infrastructure options on a regional scale. Bioresource Technology **102**:7313-7323.

Perugini, F., M. L. Mastellone, and U. Arena. 2005. A life cycle assessment of mechanical and feedstock recycling options for management of plastic packaging wastes. Environmental Progress **24**:137-154.

Peters, G. P., B. Aamaas, M. T. Lund, C. Solli, and J. S. Fuglestvedt. 2011. Alternative "global warming" metrics in life cycle assessment: A case study with existing transportation data. Environmental Science and Technology **45**:8633-8641.

Polettini, A. 2012. Waste and climate change: Can appropriate management strategies contribute to mitigation? Waste Management **32**:1501-1502.

Riahi, K., A. Grübler, and N. Nakicenovic. 2007. Scenarios of long-term socio-economic and environmental development under climate stabilization. Technological Forecasting and Social Change **74**:887-935.

Riber, C., G. S. Bhander, and T. H. Christensen. 2008. Environmental assessment of waste incineration in a life-cycle-perspective (EASEWASTE). Waste Management and Research **26**:96-103.

Rigamonti, L., M. Grosso, J. Møller, V. Martinez Sanchez, S. Magnani, and T. H. Christensen. 2014. Environmental evaluation of plastic waste management scenarios. Resources, Conservation and Recycling **85**:42-53.

Rigamonti, L., M. Grosso, and M. C. Sunseri. 2009. Influence of assumptions about selection and recycling efficiencies on the LCA of integrated waste management systems. International Journal of Life Cycle Assessment **14**:411-419.

Rosenbaum, R. K., T. M. Bachmann, L. S. Gold, M. A. J. Huijbregts, O. Jolliet, R. Juraske, A. Koehler, H. F. Larsen, M. MacLeod, M. Margni, T. E. McKone, J. Payet, M. Schuhmacher, D. Van De Meent, and M. Z. Hauschild. 2008. USEtox - The UNEP-SETAC toxicity model: Recommended characterisation factors for human toxicity and freshwater ecotoxicity in life cycle impact assessment. International Journal of Life Cycle Assessment **13**:532-546.

Rubin, E. S., J. E. Davison, and H. J. Herzog. 2015. The cost of CO<inf>2</inf> capture and storage. International Journal of Greenhouse Gas Control **40**:378-400.

Sanchez Fernandez, E., E. L. V. Goetheer, G. Manzolini, E. Macchi, S. Rezvani, and T. J. H. Vlugt. 2014. Thermodynamic assessment of amine based CO2 capture technologies in power plants based on European Benchmarking Task Force methodology. Fuel **129**:318-329.

Schakel, W., H. Meerman, A. Talaei, A. Ramírez, and A. Faaij. 2014. Comparative life cycle assessment of biomass co-firing plants with carbon capture and storage. Applied Energy **131**:441-467.

Scipioni, A., A. Mazzi, M. Niero, and T. Boatto. 2009. LCA to choose among alternative design solutions: The case study of a new Italian incineration line. Waste Management **29**:2462-2474.

Shonfield, P. 2008. LCA of management options for mixed waste plastic. Waste Resource Action Programme (WRAP), London.

Singh, B., A. H. Strømman, and E. G. Hertwich. 2011. Comparative life cycle environmental assessment of CCS technologies. International Journal of Greenhouse Gas Control **5**:911-921.

Slagstad, H., and H. Brattebø. 2013. Influence of assumptions about household waste composition in waste management LCAs. Waste Management **33**:212-219.

Soukup, G., C. Pfeifer, A. Kreuzeder, and H. Hofbauer. 2009. In situ CO2 capture in a dual fluidized bed biomass steam gasifier - Bed material and fuel variation. Chemical Engineering and Technology **32**:348-354.

Statistics Norway. 2014. Fjernvarme, 2013 (District heating, 2013).

Syversen, F., H. Bratland, and O. Skogesal. 2015. Scenarios for waste quantity and treatment capacity up to 2030. The Norwegian Waste Management and Recycling Association.

Tsao, C. C., J. E. Campbell, M. Mena-Carrasco, S. N. Spak, G. R. Carmichael, and Y. Chen. 2012. Increased estimates of air-pollution emissions from Brazilian sugar-cane ethanol. Nature Climate Change **2**:53-57.

Turconi, R., S. Butera, A. Boldrin, M. Grosso, L. Rigamonti, and T. Astrup. 2011. Life cycle assessment of waste incineration in Denmark and Italy using two LCA models. Waste Management and Research **29**:78-90.

Turner, D. A., S. Kemp, and I. Williams. 2011. Carbon footprinting in the UK waste management sector. Carbon Management **2**:677-690.

Turner, D. A., I. D. Williams, and S. Kemp. 2015. Greenhouse gas emission factors for recycling of source-segregated waste materials. Resources, Conservation and Recycling **105**:186-197.

Veltman, K., B. Singh, and E. G. Hertwich. 2010. Human and environmental impact assessment of postcombustion CO2 capture focusing on emissions from amine-based scrubbing solvents to air. Environmental Science and Technology **44**:1496-1502.

Water Resources and Energy Directorate. 2011. Costs of production of electricity and heat (in Norwegian). Oslo, Norway.

Yang, N., A. Damgaard, P. Kjeldsen, L. M. Shao, and P. J. He. 2015. Quantification of regional leachate variance from municipal solid waste landfills in China. Waste Management **46**:362-372.

List of figures