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Life-cycle assessment of a Waste-to-Energy plant in central Norway: Current situation and effects of changes in waste fraction composition

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ABSTRACT

Waste-to-Energy (WtE) plants constitute one of the most common waste management options to deal with municipal solid waste. WtE plants have the dual objective to reduce the amount of waste sent to landfills and simultaneously to produce useful energy (heat and/or power). Energy from WtE is gaining steadily increasing importance in the energy mix of several countries. Norway is no exception, as energy recovered from waste currently represents the main energy source of the Norwegian district heating system. Life-cycle assessments (LCA) of WtE systems in a Norwegian context are quasi-nonexistent, and this study assesses the environmental performance of a WtE plant located in central Norway by combining detailed LCA methodology with primary data from plant operations. Mass transfer coefficients and leaching coefficients are used to trace emissions over the various life-cycle stages from waste logistics to final disposal of the ashes. We consider different fractions of input waste (current waste mix, insertion of 10% car fluff, 5% clinical waste and 10% and 50% wood waste), and find a total contribution to Climate Change Impact Potential ranging from 265 to 637 g CO_2 eq/kg of waste and 25 to 61 g CO_2 eq/MJ of heat. The key drivers of the environmental performances of the WtE system being assessed are the carbon biogenic fraction and the lower heating value of the incoming waste, the direct emissions at the WtE plant, the leaching of the heavy metals at the landfill sites and to a lesser extent the use of consumables. We benchmark the environmental performances of our WtE systems against those of fossil energy systems, and we find better performance for the majority of environmental impact categories, including Climate Change Impact Potential, although some trade-offs exist (e.g. higher impacts on Human Toxicity Potential than natural gas, but lower than coal). Also, the insertion of challenging new waste fractions is demonstrated to be an option both to cope with the excess capacity of the Norwegian WtE sector and to reach Norway's ambitious political goals for environmentally friendly energy systems.

1. Introduction

Waste-to-Energy (WtE) technologies consist of any waste treatment process that uses a waste source to create energy in the form of electricity, heat and/or transport fuels. The most common WtE technology used worldwide is the incineration of municipal solid waste (MSW) in a moving grate combustion system with combined heat and power production (CHP) (World Energy Council, 2013; Lombardi et al., 2015). Modern WtE has dual functions (Brunner and Rechberger, 2015) as: (1) Waste treatment – acts as a sink for pollutants with its thermal treatment processes destroying organic pollutants and extracting (and concentrating) chemical pollutants via advanced flue gas cleaning systems and transferring them into landfills and (2) Energy producer – recovery of useful energy from waste streams and possible reduction in the dependency on fossil sources. Owing to European legislation that discourages disposal to landfills as the environmentally and economically worst option (European Union Council, 1999), the number of WtE plants is steadily increasing in Europe, reaching 455 plants in 2012 (IEA Bionergy, 2013).

The Norwegian WtE sector is following this trend, and it has been a growing industry for the last decade, increasing from a total capacity of 1.3 million tonnes/year in 2010 to 1.7 million tonnes today. The sector currently accounts for 17 plants, spread all across Norway. The average throughput is 90% of the capacity, and the production is around 4 TW h for district heating networks, in addition to some electricity and process steam for industries located

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near the plants (Becidan et al., 2015). 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.

Several Norwegian WtE plants are currently suffering from low profitability. The main reason is that the processing capacity exceeds the waste produced in the Scandinavian market, where the gate fee is basically set by the Swedish plants (Becidan et al., 2015). A market with excess capacity will put the gate fees under pressure, which is not financially viable in the long run. Two alternatives are either to reduce the processing capacity or to increase the demand for processing capacity. An increase in demand for processing capacity can be achieved by importing waste from markets with insufficient capacity, i.e. countries where the waste would otherwise be landfilled and/or by the insertion—and thus co-combustion—of available challenging new waste fractions such as, in a Norwegian context, car fluff, clinical waste and wood waste.

The consideration of environmental aspects is playing an increasingly important role in the development of WtE projects (World Energy Council, 2013), and life-cycle assessment (LCA) is a methodology that has been used extensively within the last decade to evaluate the environmental performance of waste treatment systems (Arena et al., 2003; Björklund and Finnveden, 2005; Finnveden et al., 2005; Moberg et al., 2005; Buttol et al., 2007; Cherubini et al., 2008, 2009; Christensen et al., 2009; Rigamonti et al., 2009; Zhao et al., 2009a; Lazarevic et al., 2010; Consonni et al., 2011; Giugliano et al., 2011; Manfredi et al., 2011; Merrild et al., 2012) and in particular WtE technology such as incineration (Hellweg et al., 2001; Riber et al., 2008; Scipioni et al., 2009; Fruergaard et al., 2010; Fruergaard and Astrup, 2011; Boesch et al., 2014; Passarini et al., 2014; Burnley et al., 2015). 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).

For WtE systems, the environmental bottlenecks are typically influenced by the energy recovery rate (Gentil et al., 2010; Giugliano et al., 2011; Tunesi, 2011; Turconi et al., 2011), the composition of the incoming waste (Astrup et al., 2011; Clavreul et al., 2014; Edjabou et al., 2015), the final disposal and leaching of the bottom ash (Doka and Hischier, 2005; Astrup et al., 2006; Hauschild et al., 2008; Allegrini et al., 2015a), the reuse of the bottom ash (Birgisdóttir et al., 2006, 2007; Allegrini et al., 2014, 2015b; Passarini et al., 2014), and the recycling of the metals (Morf et al., 2013; Boesch et al., 2014). Different technology options influence the performance of WtE plants (Tabasová et al., 2012; Arena and Di Gregorio, 2013; Ning et al., 2013; Passarini et al., 2014), and technology improvements can lead to drastic changes in their environmental profile; these changes are mostly due to the improved flue-gas cleaning achieved by stricter emission limits for species like Hg, As, heavy metals, and dioxins (Damgaard et al., 2010).

In general, WtE plants are found to be a robust technology for energy recovery from mixed waste (Astrup et al., 2009, 2011; Turconi et al., 2011; Brunner and Rechberger, 2015), and efficient WtE plants have been shown to be a competitive alternative to today's fossil fuel based energy system and complementary to a future energy system based on 100% renewable energy (Fruergaard and Astrup, 2011).

The combustion or co-combustion with energy recovery of challenging new waste fractions such as car fluff (Ciacci et al., 2010; Vermeulen et al., 2011; Passarini et al., 2012; Cossu and Lai, 2015) and clinical waste (Zhao et al., 2009b) are demonstrated to be more advantageous than landfills. Car fluff has awakened much interest in the EU in recent years, as 2–2.5 million tonnes are produced every year (Al-Salem et al., 2009), and the growing awareness of sustainability issues amongst the stakeholders is driving many industries to undertake environmentally conscious policies all along the value-chain (Subramoniam et al., 2009). Health care waste represents only a minor volume (e.g. 200–300 tonnes per year for St. Olavs Hospital, the main hospital in Central Norway), but its responsible and efficient treatment is a matter of public health (Harhay et al., 2009; Soares et al., 2013) and more research is needed given the trend in increased clinical waste production (Windfeld and Brooks, 2015). Most Norwegian healthcare facilities have closed down their own incinerators and have also experienced difficulty in finding facilities or actors to accept their waste, hence the importance of considering co-combustion with MSW in WtE plants.

The total amount of wood-based residues has been evaluated to 1 300 000 tonnes per year in Norway (Statistics Norway, 2011). However, this number includes many different fractions, and Statistics Norway does not have more details. A significant portion of wood waste is currently exported to Sweden, but local energy recovery would be preferred, in order to help Norway reach its ambitious political goals for environmentally friendly energy systems; Norway has implemented, through the EEA/EFTA agreement, the EU Renewable Directive with a national goal of 67.5% renewable energy sources by 2020 from a 2012 value of about 64.5%. For bioenergy, the aim is to double the production (including WtE) by 2020; from 14 to 28 TW h per year (Ministry of Petroleum and Energy, 2008).

Despite an increasing interest in LCA outcomes as a decision-support tool, LCAs of WtE in a Norwegian context are quasi-nonexistent (Bergsdal et al., 2005). Geographic and waste composition specificities have an impact on the results (Gentil et al., 2010; Turconi et al., 2011; Astrup et al., 2015), and the aim of this paper is to assess a Norwegian case by means of combining detailed LCA with operational data of a WtE plant located in Central Norway. Building on the state of the field, and following the recommendations provided by the recent reviews by Astrup et al. (2015), Laurent et al. (2014a,b), the specific objectives are: (1) to assess the current waste mix, (2) to assess the co-combustion of the current waste mix with car fluff, clinical waste and wood waste, (3) to provide a high resolution and geographical specificities on chemical waste composition, (4) to break down the results for any single chemical element constituting the waste, and influencing the results.

2. Methodology

2.1. System description

The WtE plant is divided into four subsystem areas (SAs), and the system description is presented in Fig. 1.

SA1 stands for the transport system. Household wastes are first collected throughout the city and then transported to the WtE facility while commercial and industrial (C&I) wastes are directly transported from their source to the WtE facility. The bottom ash is transported and handled to the municipal landfill. The fly ash (boiler ash and electronic precipitator ash) and filter cake are transported and handled to the hazardous landfill (located in Southern Norway).

For SA2, the conversion of waste to energy is largely based on the so-called lines 1 and 2 at the Heimdal WtE plant near Trondheim, which is owned and operated by Statkraft Varme AS, part of Statkraft, Europe's largest generator of renewable energy. The Heimdal plant supplies hot water to the district heating system

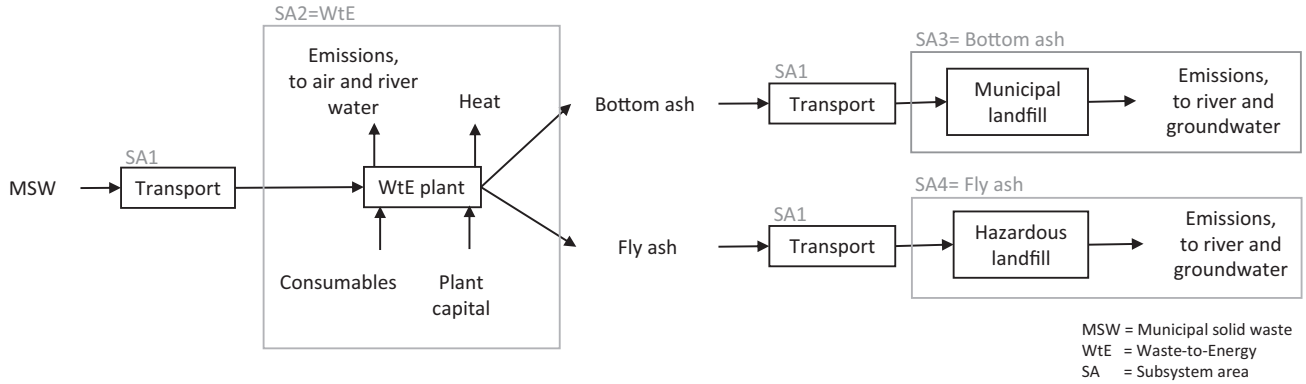

Fig. 1. System description – Foreground system boundary.

in Trondheim. The plant has a thermal capacity of 33 MW, a waste throughput of 90 k tonnes, annual heat production of 240 GW h, annual exported heat of 200 GW h and gross thermal efficiency of 83%. The present analysis is based on these figures.

For SA3, there is no metal separation before incineration, but metal is recovered from the bottom ash before it is sent to the municipal landfill. In 2014, around 9% iron and other metals were sifted out from the total mass of the bottom ash. The capital cost and the specific burdens of the landfills are included in SA3 and SA4.

2.2. Life-cycle-assessment (LCA)

Primary data from the assessed waste combustion plants represent the majority of the input data. This study uses the Arda software (Majeau-Bettez and Strømman, 2016) developed at NTNU and the EcoInvent v3.2 (Ecoinvent Centre, 2010) for background data and the missing information to conduct a process-based LCA.

2.2.1. Life-cycle impact assessment (LCIA)

ReciPe v1.12 (hierarchist perspective) (Goedkoop et al., 2009) is the chosen impact assessment method, which includes an analysis of the impacts in four midpoint categories: Climate Change Impact Potential (CCIP), Freshwater Eutrophication Potential (FEP), Human Toxicity Potential (HTP), and Ozone Depletion Potential (ODP). The results are also presented according to four key single stressors: nitrogen oxides (NO_x), particulate matter (PM)- PM smaller than 2.5 μm and PM smaller than 10 μm, sulfur dioxide (SO_2) and carbon monoxide (CO). ReciPe is the chosen methodology because of the wide range of potential impacts it covers. NO_x, PM, SO_2 and CO are assessed individually as single stressors because they are subject to emission limits (Norwegian Ministry of Climate and Environment, 2016).

2.2.2. Functional unit (FU)

A first functional unit (FU) is defined to depict the environmental impact of a change in waste composition during normal operation as (FU1) 1 kg of waste going to WtE in Central Norway and. A second FU is defined to assess the impact of the co-combustion on the energy output of the WtE plant as (FU2) 1 MJ of heat produced from a Central Norway WtE plant.

To go from the first to the second FU, the results from FU1 of each waste mix *(k)* are divided by a waste fuel dependent normalization factor (n_k). n_k is found by multiplying the resulting thermal energy efficiency of the plant $(\varepsilon_{th,k})$ in combusting the waste mix k by the lower heating value (ΔH_k^{LHV}) of the waste mix k. Because a change in waste composition leads to a change in ΔH_k^{LHV} and $\varepsilon_{th,k}$, each waste mix k is assigned a specific n_k, as shown in Eq. (1).

$$n_k = \varepsilon_{th,k} \cdot \Delta H_k^{LHV} \quad (1)$$

2.3. Waste composition

MSW composition varies over time and season (Slagstad and Brattebø, 2013; Laurent et al., 2014b), and its generation is influenced by economic development, the degree of industrialization and consumers' habits (World Energy Council, 2013). Also, waste chemical composition data are time consuming and expensive to produce, only a few detailed datasets are available (Clavreul et al., 2012) and uncertainty is thus by definition embedded in waste heterogeneity.

To best cope with uncertainty in waste composition, an integrated approach has been taken. First, the waste mix is defined on a 21 waste-type/category level *(i)*, e.g. plastics, cardboard, glass, and gypsum. Information on the share of each waste type is gathered from official Norwegian statistics (Statistics Norway, 2011) and is further refined by data from sampling analyses (Marthinsen et al., 2011). Then, each waste type i is characterized by 43 elements *(j)*, which consist of 41 chemical elements covering the main, minor, ash-forming and trace components of the waste, the water content and lower heating value (LHV). The elements' compositions are obtained from Doka (2007). All the waste types used, along with their composition of elements, are grouped in the waste fraction composition matrix *(C)* with dimension *(j,i)*. To obtain the scenario waste element matrix *(E)* with dimension *(j,k)*, C is multiplied by the case waste type matrix (M) with dimension (i,k), as shown in Eq. (2). E is given in the Supplementary material (see Table 7).

$$E = C \cdot M \quad (2)$$

2.4. Transfer coefficients

The chemical elements constituting the waste can end up in different compartments *(l)*: bottom ash, fly ash, filter cake, river water, or in the air as air emission after the flue gas cleaning. To describe the partitioning patterns of the chemical elements, mass transfer coefficients are used, and grouped in the transfer coefficient matrix (T) with dimension *(j,l)* as in Hellweg et al. (2001). To find out the final compartment matrix (F_k) with dimension *(j, l)* for each waste mix k, each column e_k of E is diagonalized, and multiplied by T. Eq. (3) is repeated for each waste mix k.

$$F_k = \widehat{e_k} \cdot T \quad (3)$$

T is defined by the current waste mix, measurement data for air emissions, water emissions, as well as bottom ash and fly ash com-

position. To fill in the gaps, recent transfer coefficients computed by Doka (2013) are used. The data used to compute *T*, and *T* are available in the Supplementary material (Table 3–5, Table 6). One *T* matrix is defined and is kept constant throughout the scenarios.

2.5. Air emissions

Two types of emissions are defined: waste-specific and process-specific. When a clear causality between the waste input and the output flow can be observed, emissions are considered waste-specific. Waste-specific emissions are calculated using *T*, and are dependent on the waste input. Process-specific emissions do not vary with the waste composition as they are assumed to be dependent on the combustion conditions (CO, dioxins, particulate matter (PM), volatile organic compounds (VOC)) or considered dependent on the flue gas cleaning technologies (NO_x, methane, benzene, toluene, pentachlorobenzene, hexachlorobenzene, pentachlorophenol, benzo(a)pyrene).

All S, Cl and F in the flue gas are assumed to be emitted to the air, as SO_2, HCl and HF respectively. S, N and P in water are similarly assumed emitted as sulfate, nitrate and phosphate.

To compute CO_2 emissions, which are waste-specific, the amount of carbon from the CO emissions, which are constant since considered process-specific, is first deduced from the total amount of carbon contained in the waste. The remaining carbon is then oxidized to CO_2 to close the mass balance. A further distinction is made between biogenic and fossil CO_2 and CO by using the carbon biogenic fraction. Also, for biogenic CO_2, a characterization factor of 0 is assumed, and biogenic CO_2 is thus considered climate neutral.

The N contained in the waste and transferred to air is assumed to be emitted as nitrogen oxide (NO_x) and ammonia (NH_3). Both NO_x and NH_3 emissions are assumed to be process-specific.

The main air emissions are given in the Supplementary material (see Table 8).

2.6. Leaching modeling

For both SA3 and SA4, the modeling of the leaching of the chemical elements contained in the ash is taken from Doka (2007). In short, the emissions are divided into two categories: (1) Short-term emissions occurring the first 100 years, and (2) Long-term emissions occurring after 100 years and up to 60 000 years. Short-term emissions end up in river water while long-term emissions end up in ground water. Short- and long-term emissions are computed using leaching transfer coefficients, which are time and type-of-landfill specific. The leaching transfer coefficients are computed with a first-order decay dynamic model. We refer the reader to the main report for more information.

2.7. Scenarios description

WtE plants receive a so-called gate fee to accept and process waste. To include unusual waste fractions can be seen as a win-win solution; the waste producer/owner gets rid of the waste in a responsible manner, while the WtE plant owner may generate extra income from a possibly higher gate fee (or remedy under-capacity issues). The design of our scenarios reflects some relevant alternative waste mixtures for future developments of the Norwegian WtE sector. Five scenarios are assessed: (1) Base case (BC), (2) Car fluff – 10% (CF-10%), (3) Clinical waste – 5% (CW-5%), (4) Wood waste – 10% (WW-10%), and (5) Wood waste – 50% (WW-50%). BC describes the current situation while scenarios CF-10%, CW-5%, WW-10% and WW-50% are prospective scenarios in which a new waste fraction is inserted into the current waste mix, which is reduced proportionally.

BC: The waste input mix consists on average of 60% household waste and 40% C&I waste. On a waste-type level (Avfall Norge, 2010; Statistics Norway, 2015), this waste input translates into: 26% organic waste, 18% wood, 16% paper, 13% plastic, 7% minerals, 6% textile, 5.4% diapers, 1.5% electronic waste, 2.0% glass, 2.0% waste water treatment sludge, 1.9% metals, 1.0% plaster (gypsum) and 0.20% hazardous waste (batteries, etc.). The category "wood" is assumed to be 90% virgin wood and 10% wood waste. The wood waste chemical composition is taken from Burman (2005). The category "paper" is further detailed into 71% mixed cardboard, 13% laminated packaging, 12% laminated materials and 4% paper.

CF-10%: Some Norwegian WtE plants have been asked about the possible co-combustion of car fluff. The composition of the car fluff is computed by first averaging out the 3 car fluff compositions (from 3 different cars) described in Passarini et al. (2012) and then by breaking down the average composition into its chemical elements, as described above in Section 2.3.

CW-5%: As of today, some Norwegian WtE plants are already co-combusting clinical waste while some others have been asked by local hospitals to co-combust clinical waste. Norwegian authorities have expressed interest in evaluating the effects (operation, logistics, emissions) of expanding and implementing such co-processing in existing WtE plants. The LHV, water content and chemical composition of the inserted clinical waste fraction is taken from Delay et al. (2001).

WW-10% and WW-50%: Wood waste cannot be combusted in wood-chip-fired district heating plants because of the impurities it contains. It has to be sent to WtE plants or dedicated wood waste heat/CHP plants, which have higher flue-gas cleaning technology and stricter emissions limits. The co-combustion of wood waste can thus be seen as a good opportunity for the WtE sector to increase its renewable energy fraction by using an available waste fuel.

2.8. Key parameters

The waste's chemical composition of the major chemical elements and key parameters of each scenario are given in Table 1, and the full overview is given in the Supplementary material (see Tables 9–11).

3. Results

The results are presented first for FU1, and then for FU2. For each FU and within each of the eight impact categories and single stressors analyzed, the overall picture between the scenarios is mixed, but all absolute values are in the same order of magnitude. Therefore, for both FUs, the results are quite comparable, but with some important share differences among the SAs. Beginning with the impact category CCIP, the analysis of the results is discussed for each impact category and for each single stressor for FU1, and the scenarios are compared for both FUs.

3.1. Functional unit 1 (FU1)

The absolute results for all the scenarios for FU1: "1 kg of waste going to WtE in Central Norway" are presented in Table 2.

Structural path analysis is then applied to identify the hot spots in the value chain, and the absolute results from Table 2 are presented by SA in Fig. 2.

The absolute results given in Table 2 are normalized to BC inside each impact category and are presented in Fig. 3. This further step is taken in order to compare BC with the other scenarios inside each impact category.

Table 1
Key parameters for the scenarios.

				BC	CF-10%	CW-5%	WW-10%	WW-50%
Chemical composition	Oxygen (without O from H2O)	O	kg/kg	2.3E−01	2.2E−01	2.3E−01	2.4E−01	2.9E−01
	Hydrogen (without H from H2O)	H	kg/kg	4.7E−02	4.9E−02	4.8E−02	4.7E−02	4.9E−02
	Carbon	C	kg/kg	3.3E−01	3.5E−01	3.3E−01	3.4E−01	3.7E−01
	Sulfur	S	kg/kg	3.2E−03	2.9E−03	3.2E−03	2.9E−03	1.7E−03
	Nitrogen	N	kg/kg	4.2E−03	5.5E−03	4.3E−03	3.9E−03	2.6E−03
	Phosphorus	P	kg/kg	6.9E−04	6.3E−04	7.5E−04	6.4E−04	4.0E−04
	Chlorine	Cl	kg/kg	6.8E−03	1.2E−02	7.7E−03	6.2E−03	3.8E−03
	Fluorine	F	kg/kg	7.1E−05	6.5E−05	6.7E−05	6.6E−05	4.6E−05
	Arsenic	As	kg/kg	7.4E−07	7.9E−07	7.0E−07	1.1E−06	2.6E−06
	Barium	Ba	kg/kg	8.9E−05	9.4E−05	1.7E−04	9.1E−05	1.0E−04
	Cadmium	Cd	kg/kg	8.2E−06	1.1E−05	3.0E−05	7.4E−06	4.2E−06
	Chromium	Cr	kg/kg	1.7E−04	1.6E−04	2.2E−04	1.5E−04	9.1E−05
	Copper	Cu	kg/kg	2.3E−03	2.9E−03	2.2E−03	2.1E−03	1.2E−03
	Mercury	Hg	kg/kg	9.2E−07	8.7E−07	2.3E−05	8.3E−07	5.0E−07
	Manganese	Mn	kg/kg	1.7E−04	1.6E−04	1.6E−04	1.6E−04	1.3E−04
	Lead	Pb	kg/kg	6.6E−04	7.1E−04	7.2E−04	6.0E−04	3.4E−04
	Antimony	Sb	kg/kg	1.6E−05	2.0E−05	1.5E−05	1.4E−05	7.8E−06
	Water content	H_2O	kg/kg	2.9E−01	2.7E−01	2.8E−01	2.8E−01	2.3E−01
	Carbon, biogenic share		%	63%	54%	62%	67%	83%
	Normalization factor	n_k	–	9.4	10.4	9.8	9.7	10.5
Ash	Bottom ash		kg/kg	1.5E−01	1.6E−01	1.5E−01	1.3E−01	8.7E−02
	Fly ash		kg/kg	2.8E−02	3.2E−02	2.9E−02	2.6E−02	1.5E−02
Energy balance	Lower heating value	LHV	MJ/kg	11.5	12.7	11.7	11.8	12.7
	Thermal input	–	MJ	11.5	12.7	11.7	11.8	12.7
	Thermal output	–	MJ	9.43	10.5	9.62	9.70	10.5
	Losses	–	MJ	2.07	2.25	2.08	2.10	2.18
	Thermal energy efficiency	$\varepsilon_{th,k}$	–	82.0%	82.3%	82.2%	82.2%	82.8%
Transport	Household waste collection[a]	–	km	14	14	14	14	14
	To WtE plant[a]	–	km	50	30	200	100	100
	Bottom ash[a]	–	km	10	10	10	10	10
	Fly ash[a]	–	km	550	550	550	550	550

[a] Average data based from operators.

Table 2
Absolute results for functional unit 1 (FU1: 1 kg of waste going to WtE in Central Norway).

	CCIP kg CO_2 eq	FEP kg P eq	HTP kg 1,4-DB eq	ODP kg CFC-11 eq	NO_x kg NO_x	CO kg CO	PM kg PM	SO_2 kg SO_2
BC	4.98E−01	1.33E−05	1.74E−01	6.05E−09	1.09E−03	3.89E−04	1.92E−04	8.82E−05
CF−10%	6.37E−01	1.25E−05	1.84E−01	6.19E−09	1.08E−03	3.63E−04	2.12E−04	8.98E−05
CW-5%	5.09E−01	1.42E−05	2.27E−01	6.37E−09	1.08E−03	3.89E−04	1.98E−04	9.26E−05
WW-10%	4.51E−01	1.26E−05	1.71E−01	5.92E−09	1.08E−03	3.58E−04	1.86E−04	8.79E−05
WW-50%	2.65E−01	9.48E−06	1.57E−01	5.51E−09	1.07E−03	2.35E−04	1.68E−04	8.43E−05

3.1.1. Climate change Impact Potential (CCIP)

The direct emissions emitted at the WtE plant account for more than 90% of SA2 and have fossil CO_2 as a key contributor. A small share of the environmental burden of SA2, around 3%, is due to various consumables, and the consumption of quicklime and ammonia accounts for 50% and 5% of the burdens associated with consumables. The variations between the scenarios are principally due to the share of biogenic carbon in the input waste. The performance of the scenarios in the impact category CCIP is thus directly connected to the share of fossil CO_2.

3.1.2. Freshwater Eutrophication Potential (FEP)

For FEP, the impacts are more distributed along the value chain. The burdens related to SA1 are occurring in background processes (i.e. processes required to supply the consumables and energy consumption of the WtE system), and are mainly due to the end-of-life of various materials. The environmental burdens occurring at SA2, SA3 and SA4 are waste-specific and directly related to the phosphorus content of the input waste. The total impact of SA4 is higher than of SA3, despite the larger amount of bottom ash than fly ash and filter cake produced per kg waste due to the higher leaching coefficient of phosphorus to hazardous landfill than to municipal landfill.

3.1.3. Human Toxicity Potential (HTP)

For HTP, the majority of the impact is caused by the leaching of the heavy metals contained in the bottom ash in the municipal landfill, followed by the impact of the leaching of the heavy metals contained in the fly ash and filter cake in the hazardous landfill. The leaching elements that affect the results the most vary slightly throughout the cases. For SA3 and for all the cases, the main drivers are manganese, lead and zinc. In addition, barium plays an important role for CW-10% and so does arsenic for WW-50%. For SA4, antimony and arsenic are the key drivers throughout all the cases except CW-10%, where mercury is the key driver.

3.1.4. Ozone Depletion Potential (ODP)

The small variations observed along the scenarios are due to SA1, and are caused by the transport of the waste to the WtE and the transport of bottom ash and fly ash and filter cake to their respective landfills. The transport distances of the waste to the WtE vary throughout the scenarios while the weight is constant. On the other hand, for the bottom ash, fly ash and filter cake transported to their respective landfill, the distances are constant while the weights vary along the scenarios. For SA2, SA3 and SA4, the use of fossil fuels is causing the environmental burdens. For SA2, the large majority (80%) of the burdens come from the combustion of

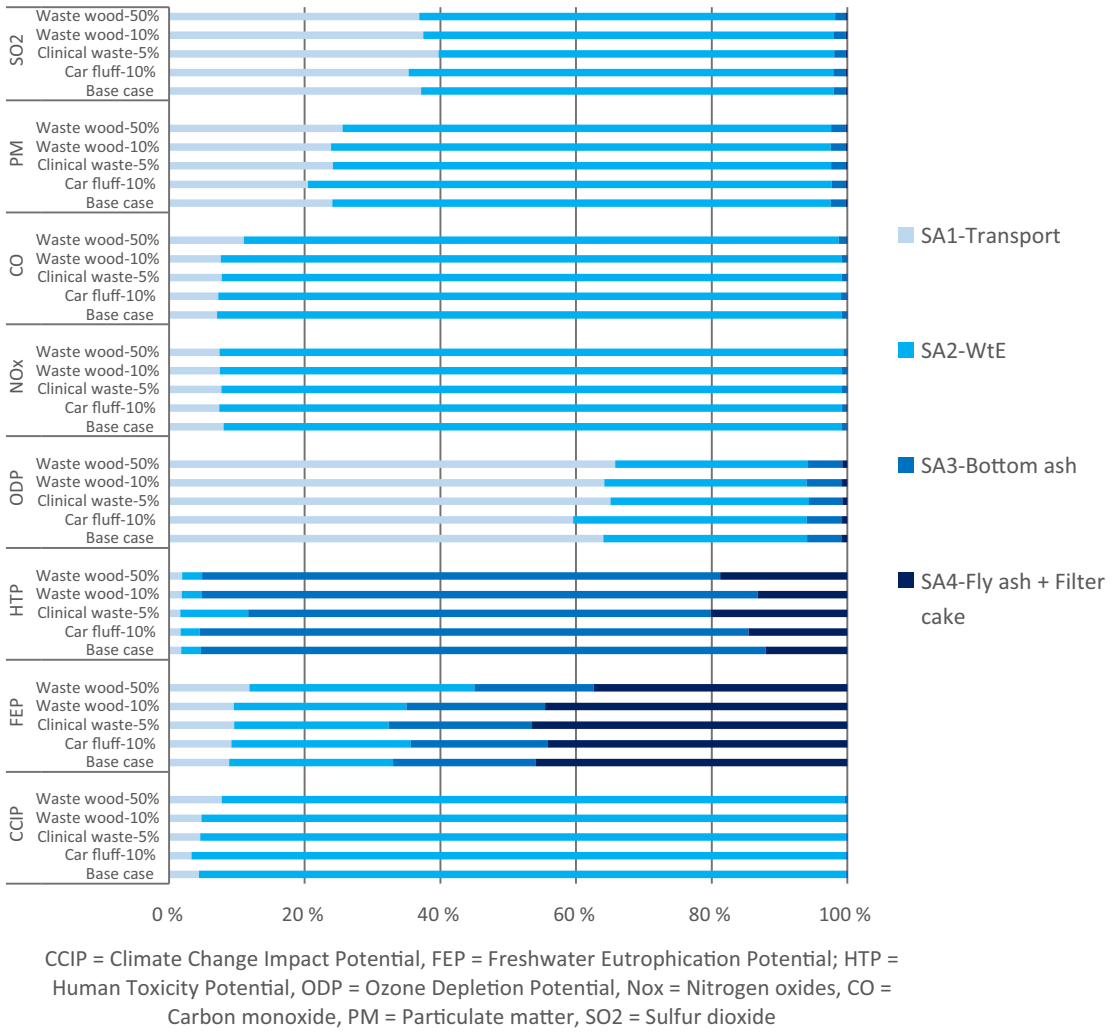

Fig. 2. Contribution analysis of the environmental impacts of each scenario.

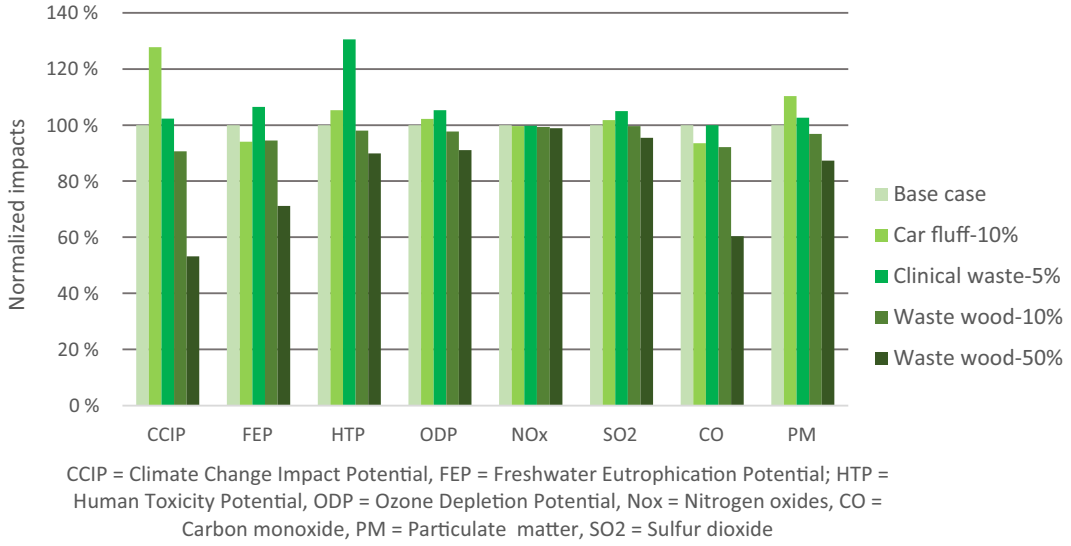

Fig. 3. Environmental impacts due to functional unit 1 (FU1: 1 kg of waste going to WtE in Central Norway), normalized to the Base case scenario (BC).

light fuel oil (mainly during start-up procedures) used as a consumable, and the rest of the burdens are allocated to the use of fossil fuel during plant construction. For SA3 and SA4, the opposite is true; the large majority of the burdens arise from activities during the construction of the plant, while the rest of the emissions are due to fossil fuels used to operate the landfill facility.

3.1.5. Nitrogen oxides (NO_x)

Over 95% of the environmental burden of SA2 is caused by direct emissions of NO_x, which are considered process-specific. A small share of SA2's burden (around 3%) is due to the production of quicklime, which is used as a consumable, and to the combustion of diesel. Around 20% of the total environmental burden is due to the fossil fuel used for transport – SA1 (direct emissions of diesel combustion and emissions of natural gas in the background processes, i.e. producing the transport engine). Less than 1% of the total burden is allocated to SA3 and SA4 (building and running both landfill sites). The total emissions of NO_x are well distributed along the scenarios, and no one scenario is clearly better than another. This is due to the fact that the main contributor for the impact category/single stressor NO_x are the direct emissions occurring at the WtE plant. These direct emissions are kept constant throughout the scenarios, leaving little room for variation amongst the scenarios. The small variations amongst the cases (2–3%) are due to the variation of the transport distances and the use of quicklime. The consumption of quicklime varies across the scenarios, and is, as explained in the Supplementary material, dependent on the chlorine and sulfur content in the waste. The more sulfur and/or chlorine in the waste, the more quicklime is consumed. CW-5% has the highest concentration of chlorine and sulfur, the longest transport distance and the second highest amount of ash, so not surprisingly, it performs the worst.

3.1.6. Carbon monoxide (CO)

40% of SA2 emissions are due to direct emissions, another 40% due to the use of consumables (85% of which allocated to quicklime production). The rest of the emissions (20%) occur in the background system, especially while producing steel used to build the WtE plant. The emissions occurring at SA3 and SA4 are due to steel production, polyethylene used in the landfill and to the diesel employed while constructing and running the plant. The resemblance of the ranking pattern of the scenarios for CCIP and CO is obvious. The relative differences between the scenarios are higher for CCIP than for CO, due to the fact that different sub-processes are the key drivers for CCIP and CO. Yet both CO_2 and CO are divided into fossil and biogenic emissions, and this parameter triggers the resemblance of the ranking for CCIP and CO.

3.1.7. Particulate matters (PM)

For PM, as is also the case for NO_x, SA2 is responsible for the majority (60%) of the environmental burdens. For SA2, the emissions pattern is distributed along the subsystems. The opposite is true for NO_x, where 90% of the emissions are direct emissions. For PM, SA2 burdens are distributed as followed: process-specific emissions with 48%, consumables with 30% and plant capital with 22%. 20% of the burdens allocated to consumables are due to quicklime production, the rest being allocated to diverse background processes. The emissions allocated to plant capital cost are due to the production of steel and to the fossil fuel used in the production chain of the background processes. When comparing with other single stressors kept constant throughout the scenarios, the differences in the ranking of the scenarios are larger than for NO_x, but smaller than for CO. The differences are larger than for NO_x because of the stronger influence of transport to PM emissions and the use of quicklime. Also, for NO_x the majority of the emissions is caused by the direct emissions occurring at the WtE, limiting the possibilities of variations amongst the scenarios. The differences between the scenarios are smaller than for CO because no further differentiation is made, as for instance fossil and biogenic fraction in the case of CO and CCIP.

3.1.8. Sulfur dioxide (SO_2)

SO_2 is also a single stressor, like NO_x, CO and PM described above. The main difference between SO_2 and NO_x, CO and PM is that SO_2 is considered waste-specific, while the three others are process-specific. SO_2 total burdens follow the same trend as NO_x; more than 90% of the burdens are allocated to SA2, whereof 93% are due to direct emissions and 5% to the use of quicklime as a consumable and 1% to the plant capital cost (steel and cement production). For SA1, SA3 and SA4, the emissions of SO_2 arise in the background processes, while burning hard coal, natural gas, and heavy and light fuel oil. Since the key driver of the total impact for SO_2 are the direct emissions occurring at the WtE plant (SA2), and since they are considered waste-specific and thus depend directly on the sulfur content of the waste, the effect of the insertion of a new waste fraction can be deduced from the sulfur content of the waste input.

3.2. Functional unit 2 (FU2)

When assessing our system as an energy producer (Fig. 4), the influence of the LHV through $n_k$is clearly shown.

We also benchmark the environmental performances of our WtE plant against those of conventional fossil-based systems producing the same amount of heat energy from coal, oil or natural gas (Fig. 5).

In general, heat from WtE has lower environmental impacts than heat from fossil systems. All the scenarios outperform heat from coal for all the impact categories but ODP. For FEP, heat from natural gas outperforms all the WtE scenarios, due to the fact that the combustion of natural gas does not produce ash, which is the main contributor to FEP. Similarly, the leaching of the heavy metals from the fly ash, filter cakes and the bottom ash makes it difficult for WtE scenarios to outperform natural gas and oil for HTP.

4. Discussion

4.1. Impacts at the Waste-to-Energy plant

We found a total contribution of our system to CCIP ranging from 265 to 637 g CO_2 eq/kg of waste for FU1 and of 25–61 g CO_2 eq/MJ of heat for FU2. The WtE plant assessed in this study shows similar patterns to the plant assessed by Astrup et al. (2009), where the direct emissions occurring at the WtE plant are assigned 347–371 g CO_2 eq/kg of waste. For the same geographical context, Bergsdal et al. (2005) report fossil CO_2 emissions of 133 g/kg of waste. Direct emissions occurring at the WtE plant also constitute the key contributor in the study conducted by Turconi et al. (2011), where fossil CO_2 emissions are 280–450 g/kg of waste. Furthermore, this latter study, as it is the case for this present study, shows the importance of the use of quicklime as a consumable. The air emissions used in our study are compatible with the air emissions reported by Damgaard et al. (2010) for similar air pollution control technologies (APC 5 and 6). For fossil CO_2, they report a value of 300 g/kg of waste. For SO_2, our range of 294–324 g/tonne of waste aligns well with their range of 109–164 g/tonne of waste. For HCl, we find emissions from 37 to 117 g/tonne of waste while they find a range of 11–27 g/tonne of waste. For NO_x, we keep constant emissions of 963 g/tonne of waste while they keep 900 g/tonne of waste. For NH_3, we use a constant emission value of 8 g/tonne of waste while they use 11 g/tonne of waste. For PM, we report a value of 23 g/tonne of waste while they report a value of 11 g/tonne of waste. For dioxins, we use a value of 0.21 μg/tonne of waste while they use a value of 0.30 μg/tonne of waste. We can thus conclude that the results calculated in this study are within the range of previous studies on similar WtE systems.

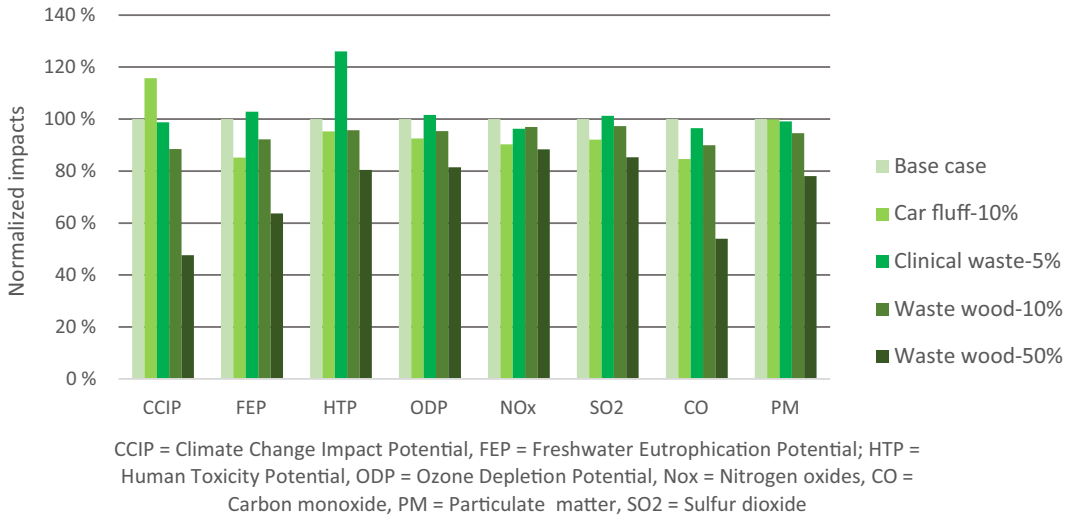

Fig. 4. Environmental impacts due to functional unit 2 (FU2: 1 MJ of heat produced from a Central Norway WtE plant), normalized to the Base case scenario (BC).

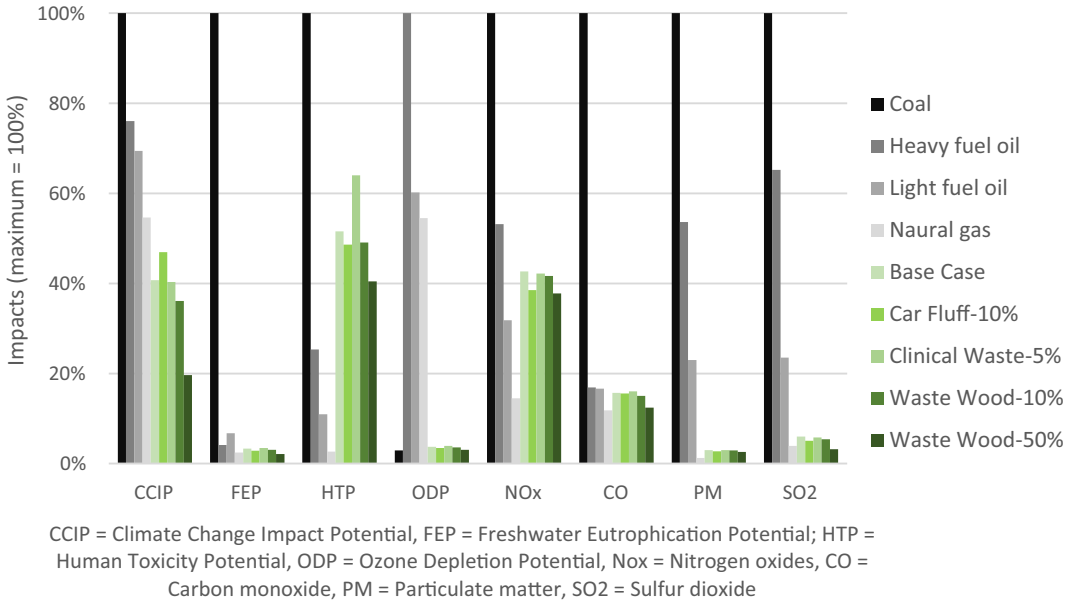

Fig. 5. Benchmarking against fossil fuels.

4.2. Functional units

Two FUs are used in this study. The co-combustion of waste fractions such as car fluff and clinical waste, which contain potentially harmful environmental substances such as heavy metals and a higher share of fossil CO_2, but which have a higher LHV, will worsen the environmental performance of the plant when assessing a waste treatment system (FU1), but may improve it when assessing the system as an energy producer (FU2). Using an input-based FU (FU1) is relevant for evaluating the performance of different WtE technologies for transferring harmful substances to the air, water effluents (if any) and solid residues. FU1 can facilitate comparison of different waste management options and evaluation of the environmental performance of value chains when, for example, aspects such as material recovery from ash streams or improvements in the leachability of landfills are considered. Using an output-based FU (FU2) is relevant when considering a WtE plant as an energy recovery facility rather than as a waste disposal system. FU2 also facilitates comparison of the environmental performance with other heat energy production systems with the same energy output and/or energy carrier, based on the energy content of the input fuel and the energy efficiency in the overall conversion.

4.3. Leaching modeling

The landfill model used allowed us to estimate the impact of pollutants transferred to the water bodies (river and ground) via leaching. Leachates can contain substantial amounts of pollutants, and the temporal dynamic of their release is difficult to predict. It is a function of the pH-value, redox potential in the landfill body as well as the chemical composition, speciation and mineralogy of the waste, in this case ash (Doka, 2007). All these parameters are site-specific, and the landfill model used in this study has been calibrated for Swiss conditions. This lack of geographical specificity introduces a higher uncertainty in the model, which could be reduced by a further calibration of the model for Norwegian, or even specific, landfill conditions. Also, other predictive models developed by e.g. Hauschild et al. (2008) or Christensen et al.

(2007) could be used. Also, given the importance of the leaching of the heavy metals for human toxicity, a sensitivity analysis using USEtox (Rosenbaum et al., 2008) - the scientific consensus model endorsed by the UNEP/SETAC Life Cycle Initiative for characterizing human and ecotoxicological impacts of chemical – could be undertaken.

4.4. Re-use of the bottom ash

The leaching of some elements of the bottom ash, fly ash and filter cake are pointed out as contributing significantly to FEP and HTP, in accordance with Cherubini et al. (2008,2009), Christensen et al. (2007), (Allegrini et al., 2015a) and Burnley et al. (2015). The phosphorus content in the waste is responsible for the burdens to FEP, and the heavy metals for the burdens to HTP. On the other hand, studies where the bottom ash, fly ash and filter cake are re-used and thus not sent to landfills, e.g. Damgaard et al. (2010) or Turconi et al. (2011), do not show a significant contribution from either bottom ash or fly ash and filter cake. Thus, the recovery of the bottom ash and fly ash and filter cake and/or the use of other treatments could drastically reduce the FEP and HTP impacts. As an alternative, bottom ash could for instance be recovered as road construction material (Birgisdóttir et al., 2006, 2007) or to produce glass frit for ceramic glaze (fritted glaze) (Barberio et al., 2010). Also, environmentally stable material could be obtained from bottom ash, fly ash and various types of air pollution control residues from WtE plants with thermal treatment methods such as sintering, vitrification, and melting (Lindberg et al., 2015). In Norway, a possible route could be to send the bottom ash to hazardous landfills where the bottom ash would act as buffering material. Practically, it would imply that the WtE plant owners would have to pay for the transport, but would be discharged from the landfill gate fee. From an LCA perspective, this would result in a cut-off of the burden related to the bottom ash.

4.5. Uncertainty assessment

The analysis performed in this study is based on average operational data from the WtE plant. In reality, both the waste composition and the process conditions in the WtE plant vary (Astrup et al., 2011; Clavreul et al., 2012), leading to variations in the energy efficiency of the plant and the transfer of elements from the waste to the air emissions, water effluents and solid residues. For example, an increase in both the nitrogen content and the calorific value of the waste can increase the NO_x emissions during combustion, leading to higher ammonia (used as a consumable) content in the flue gas to keep the same level of NO_x in the air emissions. Higher calorific value in waste can also increase the temperature conditions in the grate process, which can lead to higher transfer of sulfur and metals to the gas phase. Evaluating the variability in waste composition and process conditions requires taking into account detailed aspects of the processes involved in the thermal conversion and the flue gas cleaning, which is outside the scope of this paper.

4.6. Barriers and opportunities

The co-combustion of car fluff is not a common practice in Norway. Yet the combination of large volumes and high-energy content makes car fluff an attractive candidate for co-combustion in WtE plants, even though challenges (e.g. content of metals, organic pollutants, slagging) must be addressed, especially if the fraction combusted becomes significant. Clinical waste represents only a minor volume, and since most, and perhaps all, Norwegian healthcare facilities have closed down their own incinerators (for example, shutdown in 2007 of the Syfa plant in Fredrikstad that was burning the clinical waste of Southern Norway) and have also experienced difficulty finding facilities or actors to accept their waste, the co-combustion of clinical waste with the current waste mix could be a potential solution for its responsible and efficient treatment. Still, hospital (or health care in general) waste requires careful handling and poses specific challenges both in terms of logistics and plant operation, since it is hazardous. Also, the composition of the bottom ash might have to be monitored on a more regular basis to verify if the bottom ash still complies with the maximum limit values accepted to send it to municipal landfills. Because of the impurities it contains, wood waste cannot be combusted in wood-chip-fired district heating plants, and it has to be sent to WtE plants or dedicated wood waste heat/CHP plants, which have higher flue-gas cleaning technologies and stricter emissions limits. The co-combustion of wood waste can thus be seen as a good opportunity for the WtE sector to increase the renewable fraction of its waste by using an available fuel. This would also avoid investments in new, dedicated wood waste plants.

5. Conclusion

This paper combines detailed LCA with technology evaluation. The plant model and actual data on the input waste, air and water emissions and ash composition are used to assess the influence of a change in waste composition on the environmental performance of a WtE plant located in Central Norway.

The share of biogenic carbon, the level of heavy metals, the LHV, the phosphorus content, the leaching at the landfill sites and to a lesser extent the use of consumables are influencing the environmental performance of the assessed WtE system.

In future work, the approach taken in our study should be extended to also account for the sensitivity of the final results to uncertainties in the waste composition and process conditions.

The results of this study show: (1) heat from the WtE sector to be a climate-friendly alternative to the fossil fuel heat energy system, (2) WtE plants as not only disposal sites but also energy recovery sites (and material recovery for metals) (3) the insertion of challenging new waste fractions as an option to cope with the excess capacity of the Norwegian WtE sector (car fluff, clinical waste, waste wood) and to reach ambitious Norwegian political goals for environmentally friendly energy systems (waste wood), (4) the most common MSW WtE technology (grate firing) used in Europe combined with modern flue gas treatment as a robust and efficient solution for an array of waste fractions (whenever necessary) to advance cleaner production, (5) an environmentally sound solution for specific waste fractions across Europe, while opening new markets for the WtE sector.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.wasman.2016.09.014.

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