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# Areas on which to focus when seeking to reduce the greenhouse gas emissions of commercial waste management. A case study of a hypermarket, Finland

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ABSTRACT

This study focuses on commercial waste, which has received less attention than household waste in regards to greenhouse gas emission research. First, the global warming potential (GWP) of commercial waste management was calculated. Second, the impacts of different waste fractions and the processes of waste management were recognised. Third, the key areas on which to focus when aiming to reduce the greenhouse gas emissions of commercial waste management were determined.

This study was conducted on the waste generated by a real hypermarket in South-East Finland and included eight different waste fractions. The waste treatment plants were selected based on the actual situation. Three different scenarios were employed to evaluate the environmental impact of managing mixed waste: landfilling, combustion and more accurate source separation. The GaBi software and impact assessment methodology CML 2001 were used to perform a life cycle assessment of the environmental impacts associated with the waste management.

The results indicated that the total GWP of commercial waste management could be reduced by 93% by directing the mixed waste to combustion instead of landfill. A further 5% GWP reduction could be achieved by more accurate source separation of the mixed waste. Utilisation of energy waste had the most significant influence (41–52%) on the total GWP (−880 to −860 kg$CO_2$-eq./t), followed by landfilling of mixed waste (influence 15–23% on the total GWP, 430 kg$CO_2$-eq./t), recycling polyethylene (PE) plastic (influence 18–21% on the total GWP, −1800 kg$CO_2$-eq./t) and recycling cardboard (influence 11–13% on the total GWP, 51 kg$CO_2$-eq./t). A key focus should be placed on treatment processes and substitutions, especially in terms of substitutions of energy waste and PE plastic. This study also clarified the importance of sorting PE plastic, even though the share of this waste fraction was not substantial.

The results of this paper were compared to those of previous studies. The output of this analysis indicated that the total GWP can be significantly reduced by identifying an alternative recycling or incineration location for cardboard where it is used to substitute virgin material or replace fossil fuels respectively. In conclusion, it is essential to note that waste management companies have a notable influence on the emissions of commercial waste management because they choose the places at which the waste fractions are treated and utilised.



## 1. Introduction

While municipal waste is defined as waste from households, it also includes the similar waste generated by additional sources (1999/31/EC), such as commercial enterprises, offices and public institutions (Eurostat). According to the Waste Framework Directive (2008/98/EC), waste should be recycled before it is used to generate energy or placed in a landfill. Elevating municipal solid waste (MSW) management up the waste hierarchy offers one method by which it may be possible to reduce greenhouse gas (GHG) emissions. Methane emissions from landfills have decreased considerably in the past decade. At the same time, an increase in the amount of waste that is recycled has allowed recycled materials to replace virgin materials, and this has reduced the GHG generated during primary production (European Environment, 2013). Furthermore, member states of the European Union (EU) have been encouraged to promote the waste management practices that offer the best overall environmental outcome. This may entail that some waste streams depart from the traditional waste hierarchy;

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however, the overall life-cycle impact of the waste can justify this change (2008/98/EC).

The EU and Finland have set targets for the treatment of waste that are designed to steer municipal waste management in the desired direction. In 2014, the European Commission adopted a circular economy package that included proposals on waste management that were targeted for implementation by 2030. For example, member states are expected to recycle 65% of municipal waste and reduce landfill to a maximum of 10% of municipal waste (European Commission, 2015). Finland had its own targets for 2016; e.g., 50% of municipal waste to be recycled as material, 30% to be used as energy and no more than 20% of municipal waste to be landfilled (Ministry of the Environment, 2009). In 2015, the share of municipal waste that was recycled was 41%, while 48% was used for energy recovery and 11% was landfilled (Statistics Finland, 2016). This means that the recycling target has not yet been achieved. New targets that stretch to the year 2023 are currently being prepared; e.g., 55% of municipal waste is to be recycled, and it is anticipated that these targets will be formally accepted by autumn 2017 (Ministry of the Environment, 2017). A ban that prevents landfilling of organic waste that contains more than 10% of organic substances was implemented in Finland at the beginning of 2016 (The Government of Finland, 2013).

Targets that specifically aim to reduce GHG emissions are also steering waste management in the desired direction. In 2009, the Government of Finland adopted a target to reduce Finland's GHG emissions by at least 80% of the 1990 level by 2050 (Ministry of the Environment, 2011). The share of GHG emissions from the waste sector was 4% in Finland in 2015. A significant proportion of the emissions is produced in landfills. It is noteworthy that the GHG emissions from waste combustion and transportation are allocated to the emissions from the energy sector, which is the largest source of GHG emissions in Finland (Statistics Finland, 2017).

This study focused on commercial waste, which has some differences to household waste. First, the composition of commercial waste is different to that of household waste. Commercial waste includes a lot of packages. The products sold by commercial enterprises are delivered in secondary or tertiary packages, e.g., in cardboard boxes that may also have plastic wrappings. These packages must be emptied and recycled or disposed of. Paper and cardboard represented 19% of municipal waste in 2015 (Statistics Finland, 2016). In the case evaluated in this study, cardboard represented 49% of the total waste generated (Borisov, 2012; Hautamäki, 2012). It is also worth noting that, for example, the energy waste produced by shops is typically more homogeneous than that produced by households (Salmenperä et al., 2015).

Second, a large amount of commercial waste is produced in a small area. A total of 149 hypermarkets are located in Finland including 80 Citymarkets, 64 Prismas and 5 Minimanis (Finnish Grocery Trade Association, 2017). In Finland, hypermarkets are the largest type of shop, and they each cover an area of more than 2 500 $m^2$ (Finnish Food Information, 1996). Usually, the biggest cities in Finland have both a Citymarket and a Prisma. The number of products ordered and the sales volumes affect the amount of waste produced. The seller typically attempts to predict future sales as accurately as possible; however, losses are inevitable and occur when products are not sold; for example, out of date food. The total amount of avoidable food waste that is produced by Finnish wholesale and retail trade is estimated to be 65–75 million kilograms per year. At the same time, it is estimated that Finnish households generate 120–160 million kilograms per year of avoidable food waste (Silvennoinen et al., 2012). Statistics related to the amount of municipal waste produced by Finnish hypermarkets is not currently available. In 2015, municipal waste was generated at a rate of 500 kg/capita in Finland (Eurostat, 2017). The study described in this paper was based on the case of a hypermarket that generated the equivalent waste amount (603 t/a) of more than 1200 people. Based on population densities (Eurostat, 2016) and total areas (European Union), the same amount of municipal household waste would be collected from an area of 74 $km^2$ in Finland in 2016. A further issue that is related to the amount of commercial waste concerns the need for larger waste bins or compactors.

Third, the source separation of commercial waste can easily be improved. Separation is handled by a limited number of employees who can be instructed to separate waste into different fractions. This means that a single employer can have a direct impact on the accuracy of sorting procedures and changes in the source separation of commercial waste can be readily implemented. Household waste is source separated according to sorting guidelines, and this process is heavily influenced by free will. This is an issue because people's skills and willingness to source separation can vary considerable.

In Finland, commercial waste producers can choose which waste management company takes care of their waste. This differs from household waste management, which, in most cases, is managed by the municipal authorities in collaboration with producer associations. Commercial waste is often managed by private companies, and these companies compete with one another. The traditional way by which a waste contractor can stand out from the competition is by prices. Another manner by which a waste contractor can stand out from the competition is by developing the ability to estimate the environmental impact of the produced commercial waste, which waste fractions and processes have the biggest impact on the environment, and how the impact can be reduced. To this end, there is a need to develop a more comprehensive understanding of these matters.

The GHG emissions of different waste management scenarios have been calculated in many studies that have assessed different waste fractions. Kaazke et al. (2013) and Tulokhonova and Ulanova (2013) noticed that landfill demonstrate the greatest environmental burden. Bernstad et al. (2011), Buttol et al. (2007) and Corsten et al. (2013) all found out environmental benefits of recycling. Bernstad et al. (2011) showed that recycling of household waste provides substantial environmental benefits compared to a non-recycling alternative. Buttol et al. (2007) mentioned about the environmental beneficial effects of increasing recycling and incineration with energy recovery. At the same time, results of Corsten et al. (2013) showed that aiming for more and high-quality recycling can result in larger $CO_2$ emission reductions than focusing on incineration. Bernstad et al. (2011) explained that benefits varies greatly between recyclable fractions. Also, the type of energy substituted by incineration and used in different processes is relevance for the attained results (Bernstad et al., 2011).

Ripa et al. (2017) confirmed that one of the main responsible of the environmental burdens of MSW management is the low rate of separate collection. Same way, De Feo and Malvano (2009) had the highest avoided impact of GHG emissions in a scenario with the highest percentage of separate collection. Christensen et al. (2009) found that most waste management scenarios in Europe provided overall savings in GHG emissions. Savings were depending on waste composition, the crediting of the produced energy, the amount of paper recycled and binding of the biogenic carbon in landfills. Gentil et al. (2009) showed significant benefits due to the high level of energy and material recovery substituting fossil energy and raw materials production. They also showed that there are major differences in European member states in waste composition, availability of waste management technologies and the performance of these technologies (Gentil et al., 2009). However, some studies have not distinguished the impacts of different waste fractions (e.g., Buttol et al., 2007; De Feo and Malvano, 2009; Kaazke et al., 2013; Tulokhonova and Ulanova, 2013). As such, it is impossible to compare the impact different waste fractions have on GHG emissions.

Some studies have also assessed mixed MSW without recycled waste fractions in different countries, such as Austria (Ragoßnig et al., 2009), China (Havukainen et al., 2017), Finland (Hupponen et al., 2015; Monni, 2010, 2012), Germany (Wittmaier et al., 2009), Italy (Cherubini et al., 2009; Consonni et al., 2005) and Thailand (Liamsanguan and Gheewala, 2008). Landfilling is again mentioned to be the worst waste management option (e.g., Cherubini et al., 2009; Hupponen et al., 2015; Monni, 2010). The fact is that the composition of waste fractions varies in different studies. The majority of researchers have also concentrated on household waste, which is not the focus of the current study. Consequently, the results of previous studies cannot be directly applied to this study. Although commercial waste is quite similar to the waste produced by households, only a few studies have specifically evaluated the GHG emissions of commercial waste (e.g., Helftewes et al., 2012; Ragoßnig et al., 2009). Therefore, there is a need to conduct studies in which the GHG emissions of the commercial waste are considered and to separately assess the global warming impacts of different waste fractions.

This case study focused on the commercial waste produced in one real hypermarket in South-East Finland. The hypermarket consists of a grocery shop, a restaurant and a garden centre. The hypermarket serves around four million customers per year and covers a surface area of about 15 000 $m^2$ (Hypermarket, 2017). Waste treatment plants were selected based on the actual situation in 2012. The study focused on different waste fractions including mixed waste, energy waste, biowaste, cardboard, polyethylene (PE) plastic, paper, metal and glass. Different scenarios were used for the mixed waste: landfilling (situation in 2012), combustion (situation in the future), and more accurate source separation (carried out by the hypermarket's employees). The scenarios were planned with a private waste management company, Hyötypaperi Oy, which acts as a link between the hypermarket and the treatment plants.

The aims of this study were as follows:

– To calculate the GWP of commercial waste management.
– To recognise the impact different waste fractions, including mixed waste, energy waste, biowaste, cardboard, PE plastic, paper, metal and glass, had on GHG emissions.
– To recognise the impact different waste management processes had on GHG emissions.
– To determine what to concentrate on when the target is to effectively reduce the impact of commercial waste in a real operating environment.

## 2. Materials and methods

The potential impacts of waste management options throughout a life cycle can be evaluated using a life cycle assessment (LCA) (Ekvall et al., 2007; SFS-EN ISO 14044, 2006). LCAs make it possible to take into account the environmental benefits that can be obtained through waste management processes. For example, in the case of waste incineration, energy recovery reduces the need for alternative energy sources, while in the case of recycling, recycled material replaces the need for virgin material. LCAs can be used to compare different waste management options (Ekvall et al., 2007) and the information generated through the LCA process can be used to facilitate decision-making processes (SFS-EN ISO 14044, 2006). Product Category Rules (PCR) can be used for voluntary environmental declarations. The PCR document "Solid Waste Disposal Services" contains more in-depth requirements than those contained in standards (e.g., the ISO standards 14040 and 14044) and are, therefore, useful during LCA studies on waste management (EPD, 2015). The PCR document can be used as support in methodological choices as presented by Del Borghi et al. (2009) with other guidelines/tools/models.

The life cycle of commercial waste begins from the moment the waste is thrown into a waste bin and continues until the waste is combusted, recycled, otherwise recovered or landfilled. Unit processes from waste collection to combustion, recycling, other recovery or landfilling were taken into account in the current study. A functional unit is used as a reference unit in LCA studies (SFS-EN ISO 14044, 2006). The functional unit of this study was the waste mass from a hypermarket generated during one year. The total waste mass was 603 t/a. The mass was composed of waste from the grocery shop (88%), the restaurant (9%) and the garden centre (3%). The mass included source separated fractions of energy waste, biowaste, cardboard, PE plastic, paper, metal and glass, as well as mixed waste, which was the residual part of the commercial waste generated. Energy waste included the combustible waste fractions that were not allocated to a specific category.

The investigated GHG emissions was $CO_2$, $CH_4$, and $N_2O$. The GWP was evaluated over a 100-year time span, and the GWP of the system was assessed in compliance with the ISO standards 14040 (SFS-EN ISO 14040, 2006) and 14044 (SFS-EN ISO 14044, 2006). The GaBi 5 life cycle modelling software was used to perform LCA modelling, as it is one of the leading software tools for life cycle assessment and it is used by many LCA practitioners worldwide as a decision-support tool (GaBi; Herrmann and Moltesen, 2015). A methodology CML 2001 - November 2010 for impact assessment was used in this study, as its global warming potentials for a 100-year time horizon were the same as those approved by the IPPC (2007) (GaBi).

### 2.1. Description of scenarios

The treatment methods of different waste fractions (see Table 1) were decided based on the situation in 2012. Three main scenarios were selected for the study. The treatment of energy waste, biowaste, cardboard, PE plastic, paper, metal and glass were the same in all the scenarios. The treatment of mixed waste varied in the scenarios as follows:

– Scenario 0: Landfilling of the mixed waste, which was the situation in 2012.
– Scenario 1: Combustion of the mixed waste as a result of the ban on landfilling organic waste, which was perceived to outlaw MSW landfilling by 2016 (Government of Finland, 2013).
– Scenario 2: More accurate source separation of the mixed waste, which entailed that a part of the mixed waste was assumed to be sorted more carefully in the grocery shop; e.g., for recycling. This represented a method by which the hypermarket's employees could influence waste management. Scenario 2 was divided into two sub-scenarios in which the residue element of the mixed waste was landfilled in Scenario 2.0 and combusted in Scenario 2.1.

In Scenarios 0 and 1, the waste amounts of different waste fractions (see Table 1) was the actual waste amount of waste generated and collected in the reference year. As such, the mixed waste mass of 76 t was the total mass from the grocery shop, the restaurant and the garden centre. The majority of the mixed waste (53 t) was generated by the grocery shop.

In Scenario 2, the mixed waste was source separated more carefully in the grocery shop. This entailed that the mixed waste amount of 53 t was assumed to decrease by 50%. There was no primary data available pertaining to the composition of the mixed waste. Therefore, the composition of the commercial mixed waste (see Fig. 1) was used based on manual sorting of another grocery shop. It is worth noting that the shares of waste fractions do not fully represent the actual situation in the selected shop because

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**Table 1**
Treatments and waste amounts (t/year) of different fractions in Scenarios 0–2 (Borisov, 2012; Hautamäki, 2012).

| | Scenario 0 (t/a) | | Scenario 1 (t/a) | | Scenario 2.0 (t/a) | | Scenario 2.1 (t/a) | |
|---|---|---|---|---|---|---|---|---|
| Mixed waste | 76 | ▲ | 76 | 🔥 | 50 | ▲ | 50 | 🔥 |
| Energy waste | 66 | 🔥 | 66 | 🔥 | 75 | 🔥 | 75 | 🔥 |
| Biowaste | 145 | ■ | 145 | ■ | 159 | ■ | 159 | ■ |
| Cardboard | 298 | ♻ | 298 | ♻ | 301 | ♻ | 301 | ♻ |
| PE plastic | 14 | ♻ | 14 | ♻ | 14 | ♻ | 14 | ♻ |
| Paper | 2 | ♻ | 2 | ♻ | 3 | ♻ | 3 | ♻ |
| Metal | 1 | ♻ | 1 | ♻ | 1 | ♻ | 1 | ♻ |
| Glass | 1 | • | 1 | • | 1 | • | 1 | • |
| Total | 603 | | 603 | | 603 | | 603 | |

Explanations of the symbols:
▲: Landfilling, 🔥: Combustion, ■: Anaerobic digestion, ♻: Recycling, •: Other recovery[a].

[a] Using crushed glass to replace gravel in earth construction.

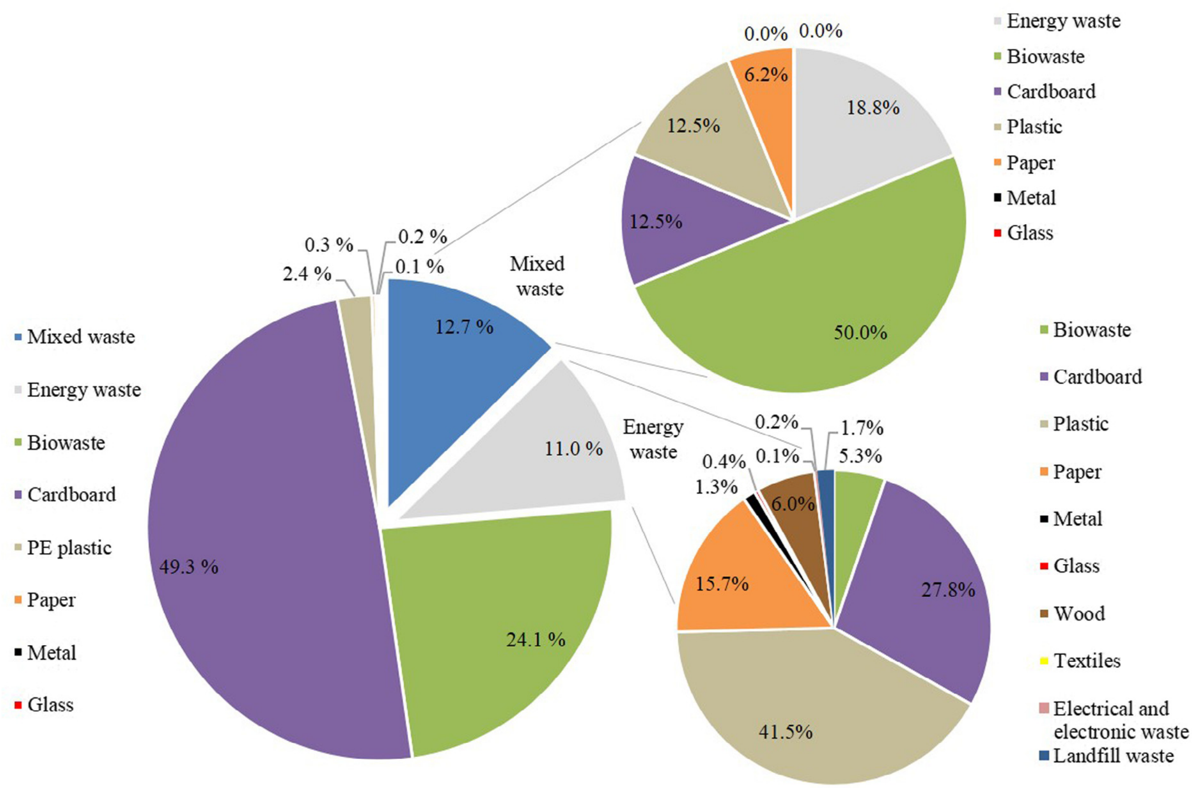


**Fig. 1.** The waste fractions of the case hypermarket in Scenarios 0 and 1 (Borisov, 2012; Hautamäki, 2012), outcome of manual sorting of commercial mixed waste from another grocery shop (Kähkönen, 2012) and outcome of manual sorting of energy waste from another hypermarket (Forssell, 2011).

there were differences in the source separation approaches (PE plastic and paper were not source separated in the other shop). However, this was the best available information. The sorted amount of mixed waste was divided as seen in Fig. 1 with the exception of plastics, which were directed to energy waste. According to Kähkönen (2012), this plastic fraction included different plastic types, not just the PE plastic. Table 1 presents the waste amounts calculated in Scenario 2 following more efficient separation of the sources. It is noteworthy that the composition of the commercial mixed waste did not change because the reduction was applied evenly to all waste fractions.

### 2.2. Treatment of waste fractions

The approaches used to treat the different waste fractions in this study are presented in more detailed in Fig. 2. The figure also includes information related to the transportation vehicles and distances, outputs, a system boundary, substituted materials and energies, etc.

Finnish electricity grid mix was used throughout the study. The electricity grid mix in 2008 (313 g$CO_2$-eq./kWh) was composed mainly of nuclear (29.7%), hydro (22.1%), natural gas (14.5%), biomass (13.0%), hard coal (11.0%) and peat (6.7%) (GaBi).

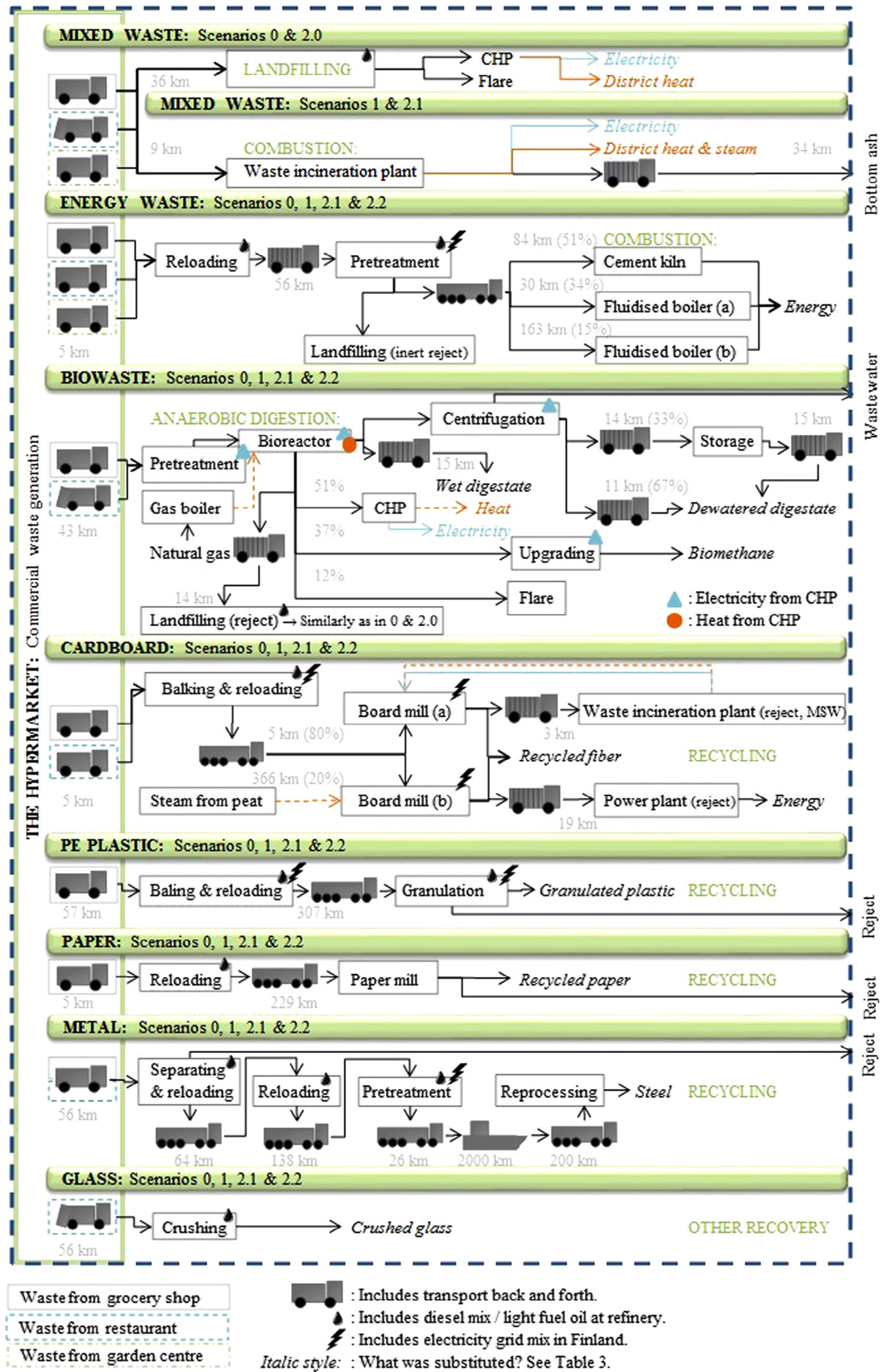


**Fig. 2.** Description of the treatment of different waste fractions in the current study.

*2.2.1. Collection, transportation and pre-treatment*

Transfer distances (see Fig. 2) were calculated based on the real locations of the case study with the exception of sea transport and transportation abroad (metal/steel). The loading of the trucks and trailers were chosen to resemble the actual situation; e.g. waste compactors were transported back and forth. The payloads were calculated based on the information provided by the Hyötypaperi Oy waste management company. Diesel consumption and exhaust emissions from road transportation were calculated using vehicles from GaBi. The emissions and fuel consumptions of the working machines were calculated based on the values from LIPASTO (2012). The emissions from diesel and light fuel oil production and sea transport were derived from GaBi.

Within the case, the waste fractions of energy waste, cardboard, PE plastic, paper, and metal were collected and transported to a reloading place before undergoing treatment. The primary data from these places was collected in this study. The fuel consumption associated with reloading, pre-treatment, and transferring waste in the area varied by 0.1–0.7 L/t per reloading place. The pre-treatment of metal (see Fig. 2) will be explained separately at a later point in this paper; however, separating roughly before reloading was calculated to consume light fuel oil 0.5 $L/t_{metal}$. Plastic and cardboard were baled in the reloading place. The electricity consumption of baling was determined to be 4–6 kWh/t. The pre-treatment of energy waste (see Fig. 1) included crushing, screening, and reloading. The electricity consumption of the pre-treatment process was 21 $kWh/t_{energy\ waste}$. The waste fraction of glass was crushed. The fuel consumption of crushing was 0.2 $L/t_{glass}$ (Hautamäki, 2012; Kiuru, 2012; Koskenheimo, 2012).

#### 2.2.2. Landfilling of mixed waste

In Scenarios 0 and 2.0, the mixed waste was transported to the landfill. The $CH_4$ generation of mixed waste was calculated using Eq. (1) (IPPC, 2006):

$$L_o = DOC \cdot DOC_f \cdot MCF \cdot F \cdot 16/12 \tag{1}$$

where $L_o$ is $CH_4$ generation potential [$Gg_{CH4}/Gg_{waste}$], $DOC$ is a fraction of the degradable organic carbon in the waste [$Gg_C/Gg_{waste}$], $DOC_f$ is the fraction of DOC that decomposes [wt%], $MCF$ is the methane correction factor [–], $F$ is the fraction of $CH_4$ in the generated landfill gas [%], and 16/12 is the molecular weight ratio $CH_4/C$ [–]. The DOC contents are presented in Table 2. The recommended default value of 50 wt% was used for $DOC_f$, and the value of 1.0 was used for the $MCF$ for aerobic managed solid waste landfill sites in Eq. (1) (IPPC, 2006; Petäjä, 2007). Based on the information from the case landfill site, the $F$ was 53% (Korjala, 2017). The calculated amount of $CH_4$ (0.72 g/m$^3$) generated was as follows:

- 79 $m^3_{CH4}/t$ for biowaste.
- 99 $m^3_{CH4}/t$ for energy waste.
- 196 $m^3_{CH4}/t$ for cardboard and paper.

The landfill gas collection started in the case landfill in 2011. In 2012, the landfill gas collection efficiency was exceptionally low, so the assumption collection efficiency of 60% was used from the years 2013–2015. A total of 60% of the collected landfill gas was directed to microturbines, which are used as a combined heat and power (CHP) system. The electricity efficiency was 30%, and the heat efficiency was 34% in 2012 (Korjala, 2017; Kymenlaakson Jäte, 2013–2016). This generated a total efficiency of 64%. Based on the information from the case landfill (Korjala, 2017), the electricity consumption of the microturbines was 22% of the produced electricity. This was taken into account in the current study. The caloric value of $CH_4$, was 10 kWh/m$^3$ and the unburned $CH_4$ and $N_2O$ were included in this study. According to Nielsen and Illerup (2003), in biogas engines, 323 g/GJ of $CH_4$ and 0.5 g/GJ of $N_2O$ remains unburned.

In 2012, a total of 40% of the collected landfill gas was directed to a flare, and this was predominantly due to device failures (Kymenlaakson Jäte, 2013–2016). The gas was also treated by flaring when the content of methane was too low for the microturbines. The same assumption pertaining to treatment efficiency for $CH_4$ by flaring (99%) were applied by Niskanen et al. (2013).

The remaining part (40%) of landfill gas was calculated according to Chanton et al. (2009), who stated that 36% of $CH_4$ oxidises on transit across the soil covers. Based on the values from LIPASTO (2012), diesel consumption and the exhaust gases of landfill compactors were also included in the current study.

#### 2.2.3. Energy utilisation of mixed waste

In Scenarios 1 and 2.1, the mixed waste was transported to a waste incineration plant. The lower heating value and $CO_{2,fossil}$ of mixed waste as received (LHVar) was calculated using the values provided in Table 2. The calculated LHVar was 12.2 $MJ/kg_{mixed\ waste}$ and the calculated $CO_{2,fossil}$ was 417 $kg/t_{mixed\ waste}$. This equates to 34 $t_{CO2}/TJ$. The value of 0.99 was used as a default oxidation factor (Statistics Finland, 2011).

In 2010, the annual electricity efficiency of the case waste incineration plant was 11%, and the combined annual district heat and steam efficiency was 52%. Thus, the total annual energy efficiency was 63%. These values were calculated based on the information from the case plant, which started commercial use in 2009 (Anttila, 2011). The share of ash was calculated to be 4%, based on the values presented in Table 2. The transportation of bottom ash to be stored in an asphalt field was also included; however, any further treatment and utilisation were excluded from the study.

**Table 2**
The properties that were applied in the mixed waste and energy waste calculations.

| Scenarios | Composition of the mixed waste 0–2[a] (wt.%) | Composition of the energy waste 0 & 1 (wt.%) | 2.1 & 2.2 (wt.%) | LHVar (MJ/kg) | DOC content (% of wet waste) | Dry matter content (wt.%) | Total C content (% of dry weight) | Fossil C fraction (% of total C) | Ash content (% of dry weight) |
|---|---|---|---|---|---|---|---|---|---|
| Landfill waste | – | 1.7 | 1.6 | 27.9 | – | 89 | 3 | 100 | 12 |
| Energy waste[b] | 18.80 | – | – | 19.9 | 20 | 82 | 56 | 43 | 7 |
| Biowaste | 50.0 | 5.3 | 5.1 | 4.2 | 16 | 44 | 38 | 0 | 5 |
| Cardboard | 12.5 | 27.8 | 26.6 | 15.8 | 40 | 85 | 46 | 0 | 7 |
| Plastic | 12.5 | 41.5 | 44.1 | 28.8 | – | 82 | 75 | 100 | 2 |
| Paper | 6.2 | 15.7 | 15.0 | 12.3 | 40 | 89 | 46 | 0 | 16 |
| Metal | 0 | 1.3 | 1.2 | – | – | 100 | – | – | 100 |
| Glass | 0 | 0.4 | 0.4 | – | – | 100 | – | – | 100 |
| Wood | – | 6.0 | 5.7 | 15.0 | 30 | 85 | 50 | – | 1 |
| Textiles[c] | – | 0.1 | 0.1 | 20.6 | 24 | 80 | 50 | 20 | 0 |
| Electrical and electronic waste | – | 0.2 | 0.2 | – | – | 100 | – | – | 100 |
| Reference | Kähkönen (2012) | Forssell (2011) | Calculated | Conesa et al. (2009), Teirasvuo (2011), Statistics Finland (2011) | IPPC (2006), Petäjä (2007) | IPPC (2006), Teirasvuo (2011) | IPPC (2006) | IPPC (2006) | Alakangas (2000), Conesa et al. (2009), Teirasvuo (2011) |

DOC: Degradable organic carbon.
LHVar: Lower heating value, as received.
[a] The composition did not change when more efficient source separation equally increased every waste fraction.
[b] Calculated average values based on the composition of the energy waste in Scenarios 0 and 1.
[c] Calculated based on the IPPC (2006) assumption that 40% of textiles are synthetic.

*2.2.4. Energy utilisation of energy waste*

The energy waste was transported from the hypermarket to be reloaded into a trailer. The trailer transported the energy waste to be pre-treated by crushing and screening. According to the company that is responsible for pretreatment, the reject share is 4% (Hautamäki, 2012). In the current case study, the rejects were directed to a landfill and, in accordance with the description presented by Hautamäki (2012), were assumed to be inert waste. The GaBi process of "landfill of inert waste" was used in this study.

After the pre-treatment, 51% of the energy waste was transported to a cement kiln, 34% was transported to a fluidised boiler (a), and 15% was transported to another fluidised boiler (b) (Hautamäki, 2012). The LHVar and $CO_{2,fossil}$ of the energy waste was calculated using the values presented in Table 2. In Scenarios 0 and 1, the calculated LHVar was 19.9 $MJ/kg_{energy\ waste}$ and the calculated $CO_{2,fossil}$ was 930 $kg/t_{energy\ waste}$. This equated to 47 $t_{CO2}/TJ$. In Scenarios 2.0 and 2.1, the LHVar was 20.3 $MJ/kg_{energy\ waste}$ and the $CO_{2,fossil}$ was 989 $kg/t_{energy\ waste}$. This equated to 49 $t_{CO2}/TJ$. The LHV and $CO_{2,fossil}$ was increased because of introduction of plastics into the energy waste. This was explained in Section 2.1. The value of 0.99 was also used as a default oxidation factor with the energy waste (Statistics Finland, 2011).

*2.2.5. Anaerobic digestion of biowaste*

The biowaste was transported to mesophilic anaerobic digestion. The values applied in the calculations were based on to the information provided by the case biogas plant and that presented in existing literature. The literature values were used mainly because the input to the biogas plant was not just the biowaste. Jönsson et al. (2005) and Davidsson et al. (2007) reported that the total solids (TS) content of biowaste was assumed to be about 30%. The same value was applied in the current study. Before the anaerobic digestion, the biowaste was pre-treated. Based on the information from the plant, the electricity consumption of pre-treatment was about 40 $kWh/t_{wet\ biowaste}$, and the share of reject was about 13% (Räsänen, 2013). In 2012, the reject was transported to the same landfill as the mixed waste in Scenarios 0 and 2.0; as such, the same landfill values as those presented in Section 2.2.2 were applied. The reject contained different waste fractions (e.g., organic material, metals, and plastics); however, the exact composition was not known. An assumption that the waste contained 5% of organic material was applied in this study. According to Myllymaa et al. (2008a), the $CH_4$ amount was then 3.2 $m^3/t_{reject}$.

The electricity consumption of the case plant was 16 $kWh/t_{wet\ biowaste}$, the total heat consumption was 211 $kWh/t_{wet\ biowaste}$ and the share of $CH_4$ in the biogas was 64% (Räsänen, 2013). Biogas production depends on the waste fraction, as such, the literature value of 130 $m^3/t_{wet\ biowaste}$ was applied in the current study. This value fits in both the ranges of 80–130 $Nm^3/t_{wet\ waste}$ for household waste alone or mixed with garden waste presented by Møller et al. (2009) and 125–170 $m^3/t_{wet\ biowaste}$ presented by Myllymaa et al. (2008a). The IPPC (2006) reported that unintentional $CH_4$ leaks usually represent 0–10% of the $CH_4$ and, when leaks are flared, the share is approximately 0%. This value was applied in the current study. According to the findings of the IPPC (2006), the $N_2O$ emissions was assumed to be negligible in the current study.

Havukainen et al. (2014) studied the same biogas plant as that assessed in the current study. They performed the calculations on the basis that a total of 51% of the biogas is guided to a gas engine (CHP) and a part of the biogas is guided to upgrading and a part to the flare (Havukainen et al., 2014). The actual share of biogas to the flare was not used because of a device failure in 2012. The share applied in the current study was 1%, which was the realised in 2013 (Havukainen, 2014). The treatment efficiency of the flare was taken into account in the same way as that described in Section 2.2.2. The remaining 48% of the biogas was directed to upgrading.

According to Räsänen (2013), the electricity efficiency of a case gas engine is 40%, and the heat efficiency is 28%. Unburned $CH_4$ and $N_2O$ were taken into account in the current study (see Section 2.2.2). The majority of the heat was needed in the case plant, but 36% was directed to external utilisation (see Section 2.3) (Räsänen, 2013).

The heat for the anaerobic digestion process was also produced by a gas boiler using natural gas because the wastewater treatment plant that was located next to the case biogas plant required part of the produced heat. Räsänen (2013) reported that the efficiency of the case boiler was 95% and the amount of natural gas used was 15 $m^3/t_{wet\ biowaste}$. According to Statistics Finland (2011), the $CO_2$ default emission factor is 55.04 t/TJ, and the default oxidation factor is 0.995 for natural gas. The emissions from natural gas production were derived from GaBi.

Biomethane was produced in the upgrading. The real electricity consumption of 0.3 $kWh/m^3_{biogas}$ (Räsänen, 2013) and literature value of 0.2% of fugitive emissions of $CH_4$ (Møller et al., 2009) were included in this study.

In the case biogas plant, 3% of the digestate was used as a fertiliser without centrifugation, and 97% of the digestate was directed to a centrifuge. The electricity consumption of the centrifuge was about 6 $kWh/t_{wet\ digestate}$ (Räsänen, 2013). Dewatered digestate can be used as a fertiliser. Based on the data from the biogas plant, the TS content of wet digestate is about 5%, and the TS content of dewatered digestate is about 26% (Räsänen, 2013). According to Latvala (2009), approximately 5% of the TS is assumed to be lost with waste water because of the dewatering. Both this and the transportation of the digestates were included in this study with the exception of emissions from digestate spreading.

According to Møller et al. (2009), no nutrients are lost in the anaerobic digestion process and, consequently, the nutrient content of the digestate equals the nutrient content of the waste. The average values for household biowaste were applied in this study as follows: 0.4% of TS for $P_{tot}$, 0.95% of TS for $K_{tot}$, and 0.26% of TS for $N_{soluble}$ (Davidsson et al., 2007; Jönsson et al., 2005). Møller et al. (2009) reported some possible nutrient loss with waste water in centrifuging. Based on the data from the case plant, losses of dewatering were calculated to be about 88% for $N_{soluble}$, 2% for $P_{tot}$ and 74% for $K_{tot}$. Emissions from the waste water treatment plant were not included in this study. According to Bruun et al. (2006), the average emission coefficient of anaerobically digested organic MSW for $N_2O$ formation is 0.015 of the N applied to the soil. This was included in the current study.

*2.2.6. Cardboard recycling*

The cardboard, which was assumed to be corrugated cardboard, was transported from the hypermarket to be baled. A total of 80% of the baled cardboard was transported via trailer to material recycling in one case board mill (a), and 20% of the baled cardboard was transported to another case board mill (b) (Borisov, 2012).

According to Myllymaa et al. (2008b), the electricity and steam consumption of pulping is 361 $kWh/t_{cardboard}$ and the share of the electricity is 44%. Different shares of reject and TS were used in the current study based on the data from the case mills: The share of the reject was 4% (a) or 5% (b) of the cardboard, and the share of TS was 35% (a) or 45% (b) of the reject. The transportation and combustion of the reject were included in this study. The heating value of the reject was calculated based on the values of 46.0 $MJ/kg_{TS}$ for plastic and 17.3 $MJ/kg_{TS}$ for corrugated cardboard presented by Alakangas (2000). The 60% share of biofuels, of which $CO_2$ emissions were not taken account, the $CO_2$ emission factor of 31.8 t/TJ and the oxidation factor of 0.99 were chosen for the reject based

**Table 3**
Description of the substitutions.

| Fraction | Substitution | Reasons for selecting the substitution | Data in the calculation |
|---|---|---|---|
| **Mixed Waste (Scenarios 0 & 2.0)** | | | |
| *District heat* | Electricity grid mix in 2008[a] | Based on the information from the case landfill. A part of the heat is used to warm up a hall, which would otherwise use electricity to the heating (Korjala, 2017). | GaBi process<br>Based on the information from the case landfill, 20% of the heat was calculated to be utilised. |
| *Electricity* | Electricity grid mix in 2008[a] | According to Fruergaard et al. (2009), the average electricity data can be used to quantify the environmental load of a waste management. | GaBi process |
| **Mixed waste (Scenarios 1 & 2.1)** | | | |
| *District heat & steam* | Thermal energy from natural gas | Based on the information from the case waste incineration plant (Markkanen, 2011). | GaBi process.<br>All produced thermal energy was assumed to be utilised. |
| *Electricity* | Electricity grid mix in 2008[a] | According to Fruergaard et al. (2009), the average electricity data can be used to quantify the environmental load of a waste management. | GaBi process |
| **Energy waste** | | | |
| *Energy from "cement kiln"* | Hard coal as energy | Based on the information from the case cement manufacturer. | **GaBi process:** Hard coal mix.<br>**Flue gas emissions from hard coal** (Statistics Finland, 2011)**:** Net caloric value: 25.0 MJ/kg, $CO_2$ default emission factor: 94.6 t/TJ, default oxidation factor: 0.99. |
| *Energy from "fluidised boiler (a)"* | Natural gas as energy | Based on the environmental impact assessment of the case power plant. | **GaBi process:** Natural gas mix.<br>**Flue gas emission from natural gas** (Statistics Finland, 2011)**:** Net caloric value: 36.0 MJ/m$^3$, $CO_2$ default emission factor: 55.04 t/TJ, default oxidation factor: 0.995. |
| *Energy from "fluidised boiler (b)"* | Peat as energy | Based on the fuel mix of the case co-incineration plant. | **Peat extraction** (Myllymaa et al., 2008a)**:** 87.29 $kg_{CO2}/t_{peat}$, 0.14 $kg_{CH4}/t_{peat}$, 0.02 $kg_{N2O}/t_{peat}$.<br>**Flue gas emissions from milled peat** (Statistics Finland, 2011)**:** Net caloric value: 10.1 MJ/kg, $CO_2$ default emission factor: 105.9 t/TJ, default oxidation factor: 0.99. |
| **Biowaste** | | | |
| *Heat* | Thermal energy from light fuel oil | Based on the information from the case biogas plant. A wastewater treatment plant positioned next to the biogas plant used part of the heat (Räsänen, 2013). The wastewater treatment plant had a partial ownership of the biogas plant in 2012. | GaBi process<br>Based on the case biogas plant, 36% of the heat was utilised outside the biogas plant (Räsänen, 2013). |
| *Electricity* | Electricity grid mix in 2008[a] | According to Fruergaard et al. (2009), the average electricity data can be used to quantify the environmental load of a waste management. | GaBi process |
| *Heat (reject)* | Thermal energy from natural gas | As per the case of mixed waste (Scenarios 0 & 2.0) | |
| *Electricity (reject)* | Electricity grid mix in 2008[a] | As per the case of mixed waste (Scenarios 0 & 2.0) | |
| *Biomethane* | Natural gas in vehicles 1:1 as energy | Based on the case biogas plant and Møller et al. (2009). | **GaBi process:** Natural gas mix.<br>**Flue gas emissions from natural gas** (Statistics Finland, 2011)**:** Net caloric value: 36.0 MJ/m$^3$, $CO_2$ default emission factor: 55.04 t/TJ, default oxidation factor: 0.995. |
| *Wet digestate* | Mineral fertilisers: 100% of $N_{soluble}$, 40% of $P_{tot}$, 100% of $K_{tot}$ | Based on Finnish Agency for Rural Affairs (2012) and the inputs of the case biogas plant, which also included sludge. | It was assumed that all the produced digestate was utilised.<br>**Average emissions for the production of mineral fertilisers** (Boldrin et al., 2009)**:** 8.1 $kgCO_2$-eq./$kg_N$, 1.7 $kgCO_2$-eq./$kg_P$, 0.8 $kgCO_2$-eq./$kg_K$. |
| *Dewatered digestate* | Mineral fertilisers: 100% of $N_{soluble}$, 40% of $P_{tot}$, 100% of $K_{tot}$ | As per the case with wet digestate but nutrient loss to waste water was taken into account. | |

**Table 3** (*continued*)

| Fraction | Substitution | Reasons for selecting the substitution | Data in the calculation |
|---|---|---|---|
| **Cardboard** | | | |
| *Recycled fibre* | Part (37%)[b] of the corrugated cardboard emissions allocated for the coreboard. This study included just 63%[b] of the corrugated cardboard emissions. | Based on the production of the case, board mills coreboard was chosen.<br>Based on an open-loop allocation in an open-loop product system where the material (corrugated cardboard) was recycled to the other product (coreboard) system (SFS-EN ISO 14044, 2006).<br>According to Myllymaa et al. (2008a), it is unlikely that coreboard would be produced entirely from primary fibre and if there is no coreboard available it could be replaced with other material. | Calculated based on ISO/TT 14049 (2000). According to Suomen Kuitukierrätys Oy (2012), 40% of the coreboard was recycled. |
| *Energy (reject from mill [b])* | Peat as energy | Based on the environmental licence of the case co-incineration plant. | It was assumed that all the produced thermal energy was utilised.<br>**Peat extraction** (Myllymaa et al., 2008a)**:** 87.29 $kg_{CO2}/t_{peat}$, 0.14 $kg_{CH4}/t_{peat}$, 0.02 $kg_{N2O}/t_{peat}$.<br>**Flue gas emissions from milled peat** (Statistics Finland, 2011)**:** Net caloric value: 10.1 MJ/kg, $CO_2$ default emission factor: 105.9 t/TJ, default oxidation factor: 0.99. |
| **PE plastic** | | | |
| *Granulated plastic* | Virgin plastic 1:1, 50% of virgin PE-LD granulate, 50% of virgin PE-LLD granulate | According to Merta et al. (2012), the stretch films can be produced from PE-LD plastic and from PE-LLD plastic.<br>According to Astrup et al. (2009a), a loss of material quality can be as high as 20%, however, there may be no loss at all. The material loss was not included in this study. | **GaBi processes:** PE-LLD (linear low density) granulate, PE-LD (low density) granulate. |
| **Paper** | | | |
| *Recycled paper* | Virgin production of newspaper | Based on the environmental licence of the case mill.<br>Based on Merrild (2008), the quality loss for newspaper was 0%. | GWP of virgin newspaper when pulp technology TMP (thermo mechanical pulp) and sulphate pulp (Merrild et al., 2008) was approximately 2400 kgCO2-eq./$t_{virgin\ paper}$. |
| **Metal** | | | |
| *Steel* | Virgin production of steel | According to Daamgaard et al. (2009). | According to Daamgaard et al. (2009), virgin production of steel was 2210 $kgCO_2$-eq./t based on value ranges. |
| **Glass** | | | |
| *Crushed glass* | Crushed stone 1:1 based on volumes | Based on the information from the case company in which crushed glass was used. | **GaBi process:** Crushed stone. |

[a] Composed mainly of nuclear (29.7%), hydro (22.1%), natural gas (14.5%), biomass (13.0%), hard coal (11.0%) and peat (6.7%) (GaBi).
[b] The emissions and avoided emissions from the energy recovery of the reject from the board mill (b), was allocated fully to the cardboard.

on recovered fuel (Statistics Finland, 2011). This means that plastic was assumed to constitute 40% of the reject.

When the cardboard was transported to the board mill (a), the energy produced from the reject was assumed to be used in the mill (a). The steam needed by the board mill (a) was produced by the same incineration plant. The additional steam and part of the needed electricity required were produced by using mixed MSW as a fuel in the incineration plant. The annual efficiencies of the waste incineration plant were presented in Section 2.2.3. According to Statistics Finland (2011), the $CO_2$ default emission factor is 40.0 t/TJ for municipal waste, and the default oxidation factor is 0.99.

When the cardboard was transported to the mill (b), the emissions avoided due to the combustion of the reject were calculated (see Section 2.3). The steam required by the board mill (b) was produced by a different power plant. Based on the power plant, the steam was produced by using peat. Emissions from the electricity use were also calculated in both cases based on the electricity grid mix in Finland in 2008.

### 2.2.7. PE plastic recycling

The references to clear PE plastic employed in this study denote stretch films. This waste fraction was transported from the hypermarket for baling and subsequently reloaded into a trailer. The baled plastic was transported for granulation. The emissions from granulation were calculated based on the literature. Astrup et al. (2009a) noted an electricity consumption range of 25–600 kWh/t for reprocessing where typically the range is 25–300 kWh/t. Myllymaa et al. (2008a) applied a value of 400 kWh/t for granulation. The value of 300 kWh/t, which fits in the typical range but is higher to take into consideration the calculation of Myllymaa et al. (2008a), was used in this study. According to Astrup et al. (2009a), the share of reject was 3–7.6%. The average value of 5.3% was applied, but the treatment of reject was not included in this study. The diesel consumption of onsite vehicles was 1 L/t (Astrup et al., 2009a). This was included in the current study.

#### 2.2.8. Paper recycling

The paper was transported from the hypermarket to be reloaded into a trailer. The baled paper was then transported to material recycling to a paper mill. Based on the literature (Merrild et al., 2008), the GWP of newspaper reprocessing was approximately 800 kgCO2-eq./$t_{recycled\ paper}$. The material loss was approximately 21% based on the case mill. The treatment of residues was excluded from this study as per the approach adopted by Merrild et al. (2009).

#### 2.2.9. Metal recycling

According to the waste management company that collects the metal, the metal consisted mainly of steel cans (Borisov, 2012). The cans are produced from steel plate which is covered with a thin layer of tin. The metal was transported from the hypermarket to be roughly separated and reloaded. The share of the reject was assumed to be 5%, but the treatment of the reject was not included in this study.

The metal was transported to reloading and then transported to pre-treatment. Based on the literature (Daamgaard et al., 2009), the diesel consumption was 2.5 L/$t_{metal}$, and the electricity consumption was 50 kWh/$t_{metal}$ in the pretreatment. After the pretreatment, the metal scrap was transported to Germany (Mepak-Kierrätys, 2012; Moliis et al., 2012). The transportation to the harbour, the estimated length of the sea transport, and the estimated transport distance in Germany were included in this study.

Emissions were calculated based on steel scrap, as such, the tin cover was migrated into the steel itself (Daamgaard et al., 2009). The reprocessing of steel scrap was calculated based on the range of values presented by Daamgaard et al. (2009). The average value of 930 kg$CO_2$-eq./t for the reprocessing of steel was used. In addition, the material loss of steel applied was 2% based on previous studies by Daamgaard et al. (2009).

#### 2.2.10. Glass recovery

The glass was transported from the hypermarket to a crushing. The crushed glass was used in earthmoving (see Section 2.3).

### 2.3. Substitutions

The emissions that were avoided as a result of substitutions were calculated based on the Table 3.

## 3. Results and discussion

### 3.1. Global warming potential of commercial waste management

The GWP of commercial waste management is presented in Fig. 3. According to the results, the best scenario in terms of the objective of reducing environmental impact was Scenario 2.1, which involved more accurate source separation of mixed waste and combustion of the residual part of the mixed waste. The second-best alternative was Scenario 1, in which all the mixed waste were combusted, and no changes were applied to the source separation. Altogether, in Scenario 2.1 the GWP was reduced by 5% compared to Scenario 1 because the mixed waste was more carefully sorted in the grocery shop. The least desirable alternative was Scenario 0, in which the mixed waste was landfilled. Altogether, in Scenario 1 the GWP was reduced by 93% compared to Scenario 0 simply by directing the mixed waste to combustion as opposed to landfill. If the mixed waste was landfilled, but source separated more effectively (Scenario 2.0), the GWP was reduced by 42% in comparison to Scenario 0.

In summary, the results indicated that the most significant change in total GWP can be obtained by guiding the mixed waste to combustion instead of landfill. Also, employees of the hypermarket can reduce the total GWP of waste management by sorting the waste fractions more carefully in the hypermarket; however, this reduction in GWP is much smaller.

The impacts of different waste fractions are presented in Fig. 3 and Table 4. Table 4 presents the emissions and substitutions of the different processes. According to Fig. 3, energy waste had the greatest influence (41–52%) on the total GWP results. The share of energy waste was only 11–12 wt% of the total waste mass; however, at the same time, the GWP was as low as -880 to −860 kg$CO_2$-eq./t. Although emissions are produced during the combustion, the amount of substitutions was very high in the case scenarios, as shown in Table 4. A total of 55–70% of the total substitutions were substitutions from the energy waste. Consequently, the result of this study indicated that the collection of energy waste from the hypermarket should be encouraged. However, there is also a need to understand the significance of replaced fuels

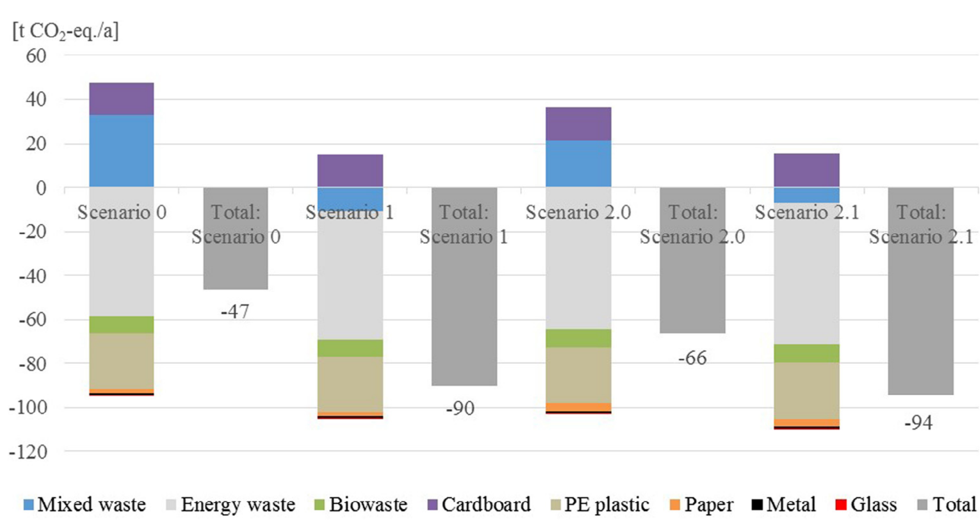


**Fig. 3.** The GWP (t $CO_2$-eq./year) of commercial waste management in Scenarios 0–2.

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**Table 4**
Emissions and substitutions (kg $CO_2$-eq./year) of different processes in Scenarios 0–2.

| | Mixed MSW [kg $CO_2$-eq./a] | Energy waste | Biowaste | Cardboard [a] | PE plastic | Paper | Metal | Glass | Total |
|---|---|---|---|---|---|---|---|---|---|
| *Transportation* | | | | | | | | | |
| Scenario 0 | 960 | 540 | 1 600 | 880 | 370 | 17 | 140 | 4 | 4 500 |
| Scenario 1 | 250 | 540 | 1 600 | 880 | 370 | 17 | 140 | 4 | 3 800 |
| Scenario 2.1 | 580 | 610 | 1 800 | 890 | 370 | 35 | 140 | 4 | 4 400 |
| Scenario 2.2 | 150 | 610 | 1 800 | 890 | 370 | 35 | 140 | 4 | 4 000 |
| *Pretreatment* | | | | | | | | | |
| Scenarios 0 & 1 | | 590 | | 450 | 38 | 1 | 25 | 0.3 | 1 100 |
| Scenario 2 | | 660 | | 450 | 38 | 1 | 25 | 0.3 | 1 200 |
| *Landfilling* | | | | | | | | | |
| Scenario 0 | 34 000 | 37 | 330 | | | | | | 35 000 |
| Scenario 1 | | 37 | 330 | | | | | | 360 |
| Scenario 2.1 | 22 000 | 42 | 360 | | | | | | 23 000 |
| Scenario 2.2 | | 42 | 360 | | | | | | 400 |
| *Combustion* | | | | | | | | | |
| Scenario 0 | | 59 000 | | | | | | | 59 000 |
| Scenario 1 | 32 000 | 59 000 | | | | | | | 90 000 |
| Scenario 2.1 | | 70 000 | | | | | | | 70 000 |
| Scenario 2.2 | 21 000 | 70 000 | | | | | | | 91 000 |
| *Recycling/other treatment (e.g. anaerobic digestion)* | | | | | | | | | |
| Scenarios 0 & 1 | | | 6 300 [b] | 18 000 [c] | 1 400 | 1 200 | 870 | | 28 000 |
| Scenario 2 | | | 6 900 [b] | 18 000 [c] | 1 400 | 2 600 | 870 | | 30 000 |
| *Total emissions* | | | | | | | | | |
| Scenario 0 | 35 000 | 60 000 | 8 300 | 19 000 | 1 800 | 1 300 | 1 000 | 5 | 130 000 |
| Scenario 1 | 32 000 | 60 000 | 8 300 | 19 000 | 1 800 | 1 300 | 1 000 | 5 | 120 000 |
| Scenario 2.0 | 23 000 | 72 000 | 9 000 | 20 000 | 1 800 | 2 600 | 1 000 | 5 | 130 000 |
| Scenario 2.1. | 21 000 | 72 000 | 9 000 | 20 000 | 1 800 | 2 600 | 1 000 | 5 | 130 000 |
| *Substitutions* | | | | | | | | | |
| Scenario 0 | -2 400 | -120 000 | -16 000 | -4 400 | -27 000 | -3 000 | -2 100 | -6 | -170 000 |
| Scenario 1 | -42 000 | -120 000 | -16 000 | -4 400 | -27 000 | -3 000 | -2 100 | -6 | -210 000 |
| Scenario 2.1 | -1 600 | -140 000 | -17 000 | -4 500 | -27 000 | -6 100 | -2 100 | -6 | -190 000 |
| Scenario 2.2 | -28 000 | -140 000 | -17 000 | -4 500 | -27 000 | -6 100 | -2 100 | -6 | -220 000 |
| *Total (emissions - substitutions)* | | | | | | | | | |
| Scenario 0 | 33 000 | -59 000 | -7 700 | 15 000 | -25 000 | -1 700 | -1 000 | -2 | -47 000 |
| Scenario 1 | -11 000 | -59 000 | -7 700 | 15 000 | -25 000 | -1 700 | -1 000 | -2 | -90 000 |
| Scenario 2.0 | 21 000 | -64 000 | -8 400 | 15 000 | -25 000 | -3 500 | -1 000 | -2 | -66 000 |
| Scenario 2.1 | -7 000 | -64 000 | -8 400 | 15 000 | -25 000 | -3 500 | -1 000 | -2 | -94 000 |

[a] Not including the emissions that was calculated to another product (coreboard).
[b] Including pre-treatment.
[c] Including the combustion of the reject.

(coal, natural gas and peat). The results were based on the fact that fossil fuels can be substituted instead of biofuels.

Mixed waste had the second most significant influence (6–23%) on the total GWP with a share of 8–13 wt% of the total waste mass. More precisely, the highest influence (23%) was observed in Scenario 0, in which the mixed waste was landfilled. The main impact was caused by $CH_4$ emissions from the landfill. The GWP of the landfilled mixed waste was 430 kg$CO_2$-eq./t and −140 kg$CO_2$-eq./t when the waste was combusted. This result validates the EC's decision to restrict the landfilling of organic waste fractions in the EU and Finland (1999/31/EC; Government of Finland, 2013).

PE plastic had the third most important influence (18–21%) on the total GWP. The share of PE plastic was only 2 wt% of the total waste mass, however, the GWP was −1800 kg$CO_2$-eq./t. These results demonstrate that it is important to sort PE plastic, even though the mass (14 t/a) of the fraction is not substantial. The main impact of the PE plastic was derived from the avoided emissions. The sensitivity of the composition of substituted PE plastic was examined by using substitutions of 100% virgin PE-LD granulate and 100% virgin PE-LLD granulate instead of a ratio of 50:50. With PE-LD, the GWP of the PE plastic decreased by about 7%, with PE-LLD, the GWP increased by 7%. The sensitivity of material quality

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loss was also examined and was determined to be as high as 20%, mentioned by Astrup et al. (2009a), instead of no loss at all. After this, the PE plastic still had the third most significant influence (14–17%) on the total GWP, even if the GWP increased to −1400 $kgCO_2$-eq./t.

Cardboard (49–50 wt%) had the fourth greatest influence (11–13%) on the total GWP results. The influence was not the highest, even if about half (49–50 wt%) of the total waste amount was cardboard. Cardboard produces more emissions than those that are avoided. The GWP was 51 $kgCO_2$-eq./t. The calculation used to determine the emissions of cardboard differed from that of the other waste fractions because it was unlikely that the coreboard would be produced entirely from primary fibre; consequently, part of the emissions were calculated for the coreboard.

The influence (5–7%) of the biowaste (24–26 wt%) on the total GWP results was verified by using a share of biogas as high as 12% for flaring because an assumption of the share (1%) was used in this study. This means that 37% of the biogas was directed to upgrading instead of 48%. With a biogas share of 1% for flaring, the GWP of the biowaste was −53 $kgCO_2$-eq./t. With a share of 12%, the GWP was −35 $kgCO_2$-eq./t. Therefore, substitutions were naturally fewer in that case. A total of 65% of the emissions from anaerobic digestion (see Table 4) was caused by the extra gas boiler using natural gas, and 71% of the biowaste substitutions were caused by biomethane, which replaced the natural gas used in vehicles.

The influence of paper, metal and glass on the total GWP results was very small (less than 3%), mainly because of the waste masses 1–3 t/a. The shares were less than 1 wt% of the total waste mass. However, the GWP of paper was as low as −1100 $kgCO_2$-eq./t and the GWP of metal was −1000 $kgCO_2$-eq./t; as such, it is meaningful to continue recycling paper and metal, even if the influence on the total GWP is relatively small. The GWP of glass was only −3 $kgCO_2$-eq./t. The use of glass in glass products was not evaluated in this study.

The emissions generated by the transportation and pretreatment processes (e.g., reloading, crushing, baling) were very small (up to 7%) with the exception of glass (42%) based on Table 4. It is noteworthy that many waste fractions were crushed or baled before being transported further using a larger vehicle. Around 36–44% of the total transport emissions were generated by biowaste. This was mainly due to the waste mass (24–26 wt% of the total waste mass) and the transportation distance (43 km). At the same time, the highest transportation emissions per tonne were produced by metal (143 $kgCO_2$-eq./t), which is transported abroad. The results indicated that the transportation emissions and, consequently, the distances are not the main concern from the point of view of global warming. As such, the main focus when reducing the GHG emission should be on the treatment methods and on substitutions.

### 3.2. Comparison to other studies

Table 5 compares the calculated GWPs of different fractions with other studies that have been conducted in Northern European countries: Finland (FI), Denmark (DK) and Germany (DE) (See Section 3.2.1). The minimum and maximum values from Table 5 were applied in Table 6 to calculate how they impacted the total GWP in Scenario 0 and Scenario 1. This generated insights into where to focus when aiming to effectively reduce the impact commercial waste has on the environment. The value zero in Table 6 indicates that the same value was used as the minimum or maximum value in the case study. The information presented in the tables are described in more depth in Sections 3.2.1–3.2.8.

#### 3.2.1. Mixed waste

The mixed waste results were compared to previous studies (see Table 5) that use a mixed MSW composition based mainly on household waste. Some studies (Vainikka et al., 2012; Wittmaier et al., 2009) presented the share of commercial waste in the mixed MSW. At the same time, the composition of the waste varied based on the origin. This approach was supported by Vainikka et al. (2012), who collected data from various sources to present ranges of different waste fractions of household waste and commercial waste. They found that the composition varied according to time and place (Vainikka et al., 2012). This entails that the composition of a sample of waste would not be same, even if the comparison was limited to commercial waste.

The different composition of mixed waste has an effect on various factors. For example, in the case of landfilling, the amount of landfill gas, while in the case of combustion, the energy values and the generation of fossil $CO_2$ emissions. A study by Wittmaier et al. (2009) calculated that the energy value of domestic MSW was 9 MJ/kg and the energy value of commercial MSW was 16 MJ/kg. The detected range of the energy values for mixed MSW has been calculated to be 10–15 MJ/kg (Astrup et al., 2009b; Hupponen et al., 2015; Monni, 2010; Myllymaa et al., 2008a; Vainikka et al., 2012; Wittmaier et al., 2009). The value of 12.2 MJ/kg, which in within the same range, was used in this study. Reimann (2012) recognised the difference in the average net caloric value of MSW and the average energy efficiency due to climate conditions between Northern Europe and South-Western Europe based on the data from 314 European waste-to-energy plants. As a result, the GWP in terms of energy utilisation was compared to previous studies that had been conducted in Finland or other Northern European countries.

As mentioned, the generation of fossil $CO_2$ emissions depends on the composition of the mixed waste. Astrup et al. (2009b) argued that the content of fossil carbon in waste was found to be critical for the GHG emissions related to waste incineration. Previous studies have calculated fossil $CO_2$ emissions to be in range of 32–45 $t_{CO2}$/TJ (Hupponen et al., 2015; Monni, 2010; Myllymaa et al., 2008b; Wittmaier et al., 2009). In this study, fossil $CO_2$ emissions of 34 $t_{CO2}$/TJ were applied for commercial waste, and this was in the range of previous studies.

The landfill gas collection efficiency was exceptionally low in the case landfill in the year 2012. As such, the collection efficiency of 60% was used from the years 2013–2015. This value was also aligned with the typical range of 50% to 95% presented by the United States Environmental Protection Agency (EPA 2008).

The comparison of landfilling and combustion presented in this and other studies indicates that mixed waste should be directed to combustion as opposed to landfill despite the utilisation of landfill gas because this reduces GHG emissions. Based on other studies, the effect of this change on the total GWP of Scenario 0 could be as high as -240% (see Table 6). Wittmaier et al. (2009) also found that the total GWP was not automatically negative in the case of combustion. Therefore, the high impact of energy efficiency can be recognised. Furthermore, a study by Myllymaa et al. (2008b) found that energy efficiency varies a great deal (36–80%) because of the location of the incineration plant. The efficiency of a CHP incineration plant is higher when an industrial plant uses process steam year-round as opposed to the use of heat for district heating in urban areas (Myllymaa et al., 2008a; 2008b). This is because the use of district heat is very low in the summer, leading to partial heat loss. The substituted energy production is also a crucial factor, as seen in the comparison and mentioned by Fruergaard et al. (2009) and Hupponen et al. (2015). The GWP results of combustion depend on the substituted electricity and heat production. For example, in case of electricity, coal or natural gas versus a local

**Table 5**
Comparison of the calculated GWPs of different fractions between the current study and other studies.

| Fraction<br>Country: Treatment | Substitution | The Total GWP<br>(kg $CO_2$-eq./$t_{waste}$) | Reference |
|---|---|---|---|
| **Mixed waste** | | | |
| *Scenarios 0 & 2.0, FI: Landfilling* | *Electricity and heat (20%): electricity grid mix in Finland* | *428 to 429* | *This study* |
| DE: Landfilling[a] | Electricity: local power mix | 399 | Wittmaier et al. (2009) |
| FI: Landfilling | Heat: natural gas 90%, heavy fuel oil 10% | 283 | Monni (2010) |
| FI: Landfilling | – | 231 | Hupponen et al. (2015) |
| DE: Combustion, grate[a] (efficiency 39%) | Electricity: local power mix; heat: oil 85%, gas 15% | 219 | Wittmaier et al. (2009) |
| FI: Combustion, grate (efficiency 64–68%) | Electricity: electricity grid mix in Finland; heat: natural gas | −91 to −21 | Hupponen et al. (2015) |
| *Scenarios 1 & 2.1, FI: Combustion, grate (efficiency 63%)* | *Electricity: electricity grid mix in Finland; heat: natural gas* | *−141 to −140* | *This study* |
| FI: Combustion, grate (efficiency 62–80%) | Electricity: natural gas (min), - (max); heat: natural gas 90%, heavy fuel oil 10% | −291 to −33 | Monni (2010) |
| FI: Combustion, grate (efficiency 36–80%) | Electricity: coal (min), average production in Finland (max); heat: Oil 56%, wood 32%, natural gas 12% (min), average production in Finland (max) | −590 to 0 | Myllymaa et al. (2008b) |
| DK: Combustion, grate (efficiency 100%) | Electricity: Central European energy mix (min), energy mix for the Nordic countries (max); heat: EU25 | −967 to −47 | Astrup et al. (2009b) |
| FI: Combustion, grate[b] (efficiency 68%) | Electricity: coal (min), natural gas (max); heat: oil/natural gas | −1012 to −451 | Vainikka et al. (2012) |
| **Energy waste** | | | |
| *Scenarios 0–2, FI: Co-combustion, cement kiln & fluidised boiler* | *Energy: hard coal, natural gas, peat* | *−882 to −862* | *This study* |
| FI: Combustion, fluidised boiler[b] (efficiency 68%) | Electricity: coal (min), natural gas (max); heat: oil/natural gas | −1305 to −498 | Vainikka et al. (2012) |
| DK: Co-combustion (efficiency assumption 100%) | Electricity: Central European energy mix (min), energy mix for the Nordic countries (max); heat: EU25 | −1815 to 665 | Astrup et al. (2009b) |
| **Biowaste** | | | |
| *Scenarios 0–2, FI: Anaerobic digestion* | *Electricity from biogas: electricity grid mix in Finland; heat from biogas: light fuel oil; natural gas in vehicles; mineral fertiliser* | *−53* | *This study* |
| FI: Anaerobic digestion & composting | Energy from biogas: peat; landscaping: peat extraction | −70 | Myllymaa et al. (2008b) |
| FI: Anaerobic digestion | Electricity from biogas and digestate[c]: average electricity production in Finland; heat from biogas and digestate: natural gas (min. biogas, digestate, max. biogas); mineral fertiliser (max) | −148 to −64 | Hupponen et al. (2012) |
| FI: Anaerobic digestion | Diesel in vehicles; mineral fertiliser | −157 to −92 | Virtavuori (2009) |
| FI: Anaerobic digestion | Electricity from biogas: average electricity production in Finland; heat from biogas: average district heat production in the metropolitan area of Finland; mineral fertiliser | −177 to −91 | Virtavuori (2009) |
| DK: Anaerobic digestion | Natural gas in vehicles; mineral fertiliser; carbon binding in soil | −293 to 111 | Møller et al. (2009) |
| DK: Anaerobic digestion | Electricity from biogas: average electricity in Hungary, Poland, etc. (min), average electricity in the Nordic countries (max); heat from biogas: EU25; mineral fertiliser, carbon binding in soil | −375 to 33 | Møller et al. (2009) |
| **Cardboard** | | | |
| *Scenarios 0–2, FI: Recycling to coreboard* | *Energy from reject: peat; part of emissions to other product* | *51* | *This study* |
| Europe: Recycling | Virgin material | −104 | Gentil et al. (2009) |
| FI: Fibre recycling to coreboard | Virgin fibre[d] | −500 to −430 | Moliis et al. (2012) |
| FI: Combustion of fibres | Energy: peat 80%, woodchip 20% | −1400 | Moliis et al. (2012) |
| **Plastic** | | | |
| FI: Combustion | Energy: peat 80%, woodchip 20% | −700 | Moliis et al. (2012) |
| Europe: Recycling | Virgin plastic | −759 | Gentil et al. (2009) |
| FI: Recycling | Virgin plastic; energy from reject: Central Europe | −980 to −890 | Moliis et al. (2012) |
| DK: Combustion | Energy: hard coal/fuel oil | −1465 to 46 | Astrup et al. (2009a) |
| DK: Recycling | Virgin plastic | −1551 to −710 | Astrup et al. (2009a) |
| *Scenarios 0–2, FI: Recycling* | *Virgin plastic* | *−1764* | *This study* |

(*continued on next page*)

**Table 5** (*continued*)

| Fraction<br>Country: Treatment | Substitution | The Total GWP<br>(kg $CO_2$-eq./$t_{waste}$) | Reference |
|---|---|---|---|
| **Paper** | | | |
| *Scenarios 0–2, FI: Recycling* | *Virgin paper* | *−1085* | *This study* |
| Europe: Recycling | Virgin material | −1256 | Gentil et al. (2009) |
| DK: Recycling | Virgin material | −1265 to 422 | Merrild et al. (2009) |
| DK: Recycling | Virgin material; energy from saved virgin biomass: coal | −4391 to −1816 | Merrild et al. (2009) |
| **Metal** | | | |
| *Scenarios 0–2, FI: Recycling of steel*[e] | *Virgin steel* | *−1035* | *This study* |
| FI: Recycling of steel | Virgin steel | −1595 | Kuusiola (2010) |
| Europe: Recycling of steel | Virgin steel | −1681 | Gentil et al. (2009) |
| DK: Recycling of steel | Virgin steel | −2347 to −507 | Daamgaard et al. (2009) |
| FI: Recycling of steel and aluminium[f] | Virgin steel; virgin aluminium | −3020 to −2960 | Moliis et al. (2012) |
| Europe: Recycling of aluminium | Virgin aluminium | −8225 | Gentil et al. (2009) |
| FI: Recycling of aluminium | Virgin aluminium | −9347 | Kuusiola (2010) |
| DK: Recycling of aluminium | Virgin aluminium | −19 327 to −4987 | Daamgaard et al. (2009) |
| **Glass** | | | |
| *Scenarios 0–2, FI: Other recovery* | *Crushed stone* | *−3* | *This study* |
| Europe: Recycling | Virgin material | −253 | Gentil et al. (2009) |
| FI: Recycling to foamed glass | Expanded clay | −450 to −410 | Moliis et al. (2012) |
| DK: Recycling for glass production | Virgin raw materials; energy: heavy fuel oil (min), natural gas (max); calcination | −505 to −416 | Larsen et al. (2009) |

[a] 25% of the waste was commercial MSW or commercial MSW with properties similar to domestic MSW (Wittmaier et al., 2009).
[b] The waste was 50% household waste and 50% commercial waste to the grate/to SRF (solid recovered fuel) production. SRF was produced from 56% of the household waste and 85% of the commercial waste in the case of the fluidised bed. (Vainikka et al., 2012).
[c] Digestate was thermal dried using landfill gas (Hupponen et al., 2012).
[d] According to Moliis et al. (2012), virgin fibre was not replaced in reality.
[e] Main emissions were calculated using the values presented by Daamgaard et al. (2009).
[f] Calculated based on the data presented by Kuusiola (2010).

**Table 6**
The effect of the minimum and maximum values from Table 5 on the total GWPs of Scenario 0 and Scenario 1.

| *Waste fraction*<br>Treatment method | Effect to Scenario 0<br>Used minimum value<br>(%) | Effect to Scenario 0<br>Used maximum value<br>(%) | Effect to Scenario 1<br>Used minimum value<br>(%) | Effect to Scenario 1<br>Used maximum value<br>(%) |
|---|---|---|---|---|
| *Mixed waste* | | | | |
| Landfilling | -32 | 0 | 32 | 48 |
| Combustion | -240 | -70 | -74 | 12 |
| *Energy waste* | | | | |
| Combustion | -130 | 220 | -69 | 110 |
| *Biowaste* | | | | |
| Anaerobic digestion | -100 | 51 | -52 | 26 |
| *Cardboard* | | | | |
| Recycling | -350 | 0 | -180 | 0 |
| Combustion [a] | -930 | -930 | -480 | -480 |
| *Plastic* | | | | |
| Recycling | 0 | 33 | 0 | 17 |
| Combustion | 9 | 56 | 5 | 29 |
| *Paper* | | | | |
| Recycling | -11 | 5 | -6 | 3 |
| *Metal* | | | | |
| Steel recycling | -3 | 1 | -2 | 1 |
| *Glass* | | | | |
| Recycling/other recovery | -0.8 | 0 | -0.4 | 0 |

[a]Minimum and maximum values are the same and peat was used mainly as the energy substitution.

power mix of hydro and nuclear energy. The substituted heat varies considerably in different studies because it should be selected locally in accordance with the characteristics of the case area.

*3.2.2. Energy waste*

A comparison of the GWPs of energy waste (see Table 5) revealed that the results of the current study were within the range presented by Vainikka et al. (2012) and Astrup et al. (2009b). It is noteworthy that researchers have presented the results in different ways. In the current study, the result was represented per tonne of source separated energy waste that was pre-treated before combustion, and the reject was landfilled. The results of Vainikka et al. (2012) were represented per tonne of source separated MSW which was directed to SRF production before combustion, and the residual waste was directed to a grate. Further, the results of Astrup et al. (2009b) were represented per tonne of SRF combusted. Astrup et al. (2009b) have included the pre-sorting of the waste for SRF production but excluded the treatment of the reject. Helftewes et al. (2012) studied treatment options for commercial and industrial waste. They found that as much as 680 $kgCO_2$-eq./t of emissions can be avoided when the SRF fraction is co-combusted in cement kilns, the heavy material fraction is directed as a high caloric fraction to a SRF power plant, the fine fraction is directed to a waste incineration plant, and the metal fraction to material recovery (Helftewes et al., 2012).

A comparison between the current study and previous studies confirmed that the change in the total GWP of Scenario 1 could be as variable as −69% to 114% (see Table 6). Based on the comparison, the notable impact of the substituted energy was also obvious in the case of energy waste. It is clear that the composition of waste also affected the energy waste results. In study by Vainikka et al. (2012), the LHV was 11–18 MJ/kg for SRF from household waste and 16–24 MJ/kg for SRF from commercial waste. Astrup et al. (2009b) applied the value of 19 MJ/kg for the SRF. The value of 20 MJ/kg was applied in the current study. The value was within the presented range of SRF from commercial waste.

### 3.2.3. Biowaste

A comparison between the current study and previous studies in terms of the anaerobic digestion of biowaste revealed that the calculated GWP is typically smaller in previous studies than it was in this study (see Table 5). This could attribute to the fact that part of heat required was generated by natural gas in the current study. Further, the substitutions vary in different studies. The produced biogas can be used in energy production (e.g., Hupponen et al., 2012; Møller et al., 2009; Myllymaa et al., 2008b; this study, Virtavuori, 2009) or as fuel for vehicles (e.g., Møller et al., 2009; this study; Virtavuori, 2009). A difference was also noticed in terms of substituted heat, which was calculated based on the heat production in the area. The wide range of compositions of substituted fuels has been applied within the research, e.g. light fuel oil (this study), peat (Myllymaa et al., 2008b) and natural gas (Hupponen et al., 2012). Furthermore, a difference can be observed in terms of the substituted electricity. Also, mineral fertiliser substitutions have been calculated either by taking into consideration possible nutrient losses with waste water (as per the current study) or by ignoring the possibility of nutrient losses (Møller et al., 2009) in order to avoid more emissions. It is noteworthy that the data used in the current study was collected from the case plant shortly after the start-up of the plant, which had experienced some malfunctions that had resulted in a degree of instability in the energy production that year.

### 3.2.4. Cardboard

The comparison of the GWPs of cardboard (see Table 5) revealed that the methods previous studies have employed to calculate emissions differ from those employed in the current study. In this study, part of the emissions were calculated for another product system, and virgin material or energy (with the exception of reject from mill [b]) was not substituted. It is noteworthy that Moliis et al. (2012) used the fibres as substitutes for virgin materials, albeit while acknowledging that they are not replaced in reality.

Based on the comparison, GHG emissions can be avoided if the fibres substitute virgin material or the cardboard fraction is combusted. The effect on the total GWP of Scenario 0 can be as high as −350% in the case of recycling and −930% in the case of combustion (see Table 6). The comparison indicates that combustion of cardboard is a more advantageous from the point of view of GHGs than recycling; however, the substituted fuels (mainly peat) have to be taken into consideration. It is due to this reason that the comparison does not unequivocally confirm the best solution. However, the data does confirm that the environmental impact of commercial waste can be reduced by identifying another recycling option or incineration place for cardboard where it is used to substitute virgin material or replace fossil fuels respectively.

### 3.2.5. Plastic

The comparison between the recycling and combustion of the plastic fraction revealed that, in both cases emissions can be reduced (see Table 5). According to Astrup et al. (2009a), recycling plastic waste for use as a substitute for virgin plastic is more advantageous than employing it to generate energy if the plastic is not a mixture of plastic types. More emissions can be avoided even if 64% of the plastic is recycled and the rest of the fraction is incinerated as reject, as demonstrated by Moliis et al. (2012), who studied the mixed plastic that is present in household waste. The results revealed that more substitutions can be achieved by recycling than by incinerating the plastic; however, in both cases, high substitutions are achievable. The results also show that, by using GaBi processes of PE-LLD and PE-LD granulates, as was the case in the current study, recycling can yield even more substitutions than those achieved by Astrup et al. (2009a). This can be observed in Table 6, in which all the values for plastic are positive. On the other hand, the value of −1400 $kgCO_2$-eq./t (material quality loss 20%) calculated within the current study fits within the range presented by Astrup et al. (2009a).

A study by Rigamonti et al. (2014) indicated that the plastic fraction is one of the most debated issues in the discussions on integrated municipal solid waste systems because both the material and energy recovery of plastic is possible. The results of their study showed that the increasing of material recycling in plastic waste management can result in GWP savings (Rigamonti et al., 2014). However, an inquiry into the waste processing operations in use within 12 commercial enterprises in Finland in 2010 revealed that only 17% of commercial enterprises sorted the plastic fraction separately (Environment Office of Oulu, 2010). Based on the results of previous research, more attention should be paid to the plastic waste. Corsten et al. (2013) also concluded that plastics are play an important role in achieving the full savings potential in waste management and Bernstad et al. (2011) encouraged increasing recycling of plastic packaging.

### 3.2.6. Paper

A comparison of the GWPs of paper (see Table 5) revealed that the results of this study fit within the range presented by Merrild et al. (2009), the results of which were very dependent on technology data and system boundary choices. In the current study, the substituted virgin value was selected based on the quality of the paper produced in the case mill, which was newspaper. Also, pulp technologies were selected based on the substitutions in the case mill: thermomechanical pulp (TMP) and sulphate pulp. Differences were noticed in material losses. According to Merrild et al. (2008), the material loss is 2–18%. Merrild et al. (2009) used a material loss of 2.4%, whereas a loss as high as 21% was used in this study based on the case mill. The comparison also revealed that the GWP can be reduced if the production of energy from the saved virgin biomass (wood) is also considered in calculations such that the energy from biomass substitutes for the energy otherwise produced by fossil fuels. At the same time, even if the GWP of paper can be reduced, the reduction in total GWP of commercial waste is relatively small (see Table 6) because the amount of paper was just 2 t/a in the case of the hypermarket.

### 3.2.7. Metal

A comparison of the GWPs of metal reveals that more emissions than those calculated in this study can be avoided by recycling the steel (see Table 5). According to Daamgaard et al. (2009), the GWP of steel can be as low as −2347 kg$CO_2$-eq./t, which is much lower than the figure of −1035 kg$CO_2$-eq./t applied in the current study. Regardless, the extent to which recycling metal can reduce the total GWP of commercial waste was small (see Table 6) because the amount of metal was just 1 t/a in the case of the hypermarket. Furthermore, there is a need to understand the composition of metal waste because further emission reductions can be achieved if part of the metal is aluminium. At the same time, Bernstad et al. (2011) have encouraged increasing recycling of metal.

### 3.2.8. Glass

A comparison of the GWPs of glass (see Table 5) revealed that emissions can be further reduced by recycling glass; e.g., using glass within glass production as opposed to using the crushed glass in earthmoving. In the current study, the glass was used as a substitute for crushed stone. It is worth noting that, even if the GWP of glass can be reduced, the effect of doing so in terms of the total GWP of commercial waste is very low (see Table 6) because the amount of glass was just 1 t/a in the case of the hypermarket.

## 4. Conclusions

Many studies have focused on the GHG emissions of household waste management. However, much less attention has been paid to commercial waste, which is also produced year round. The GWP of commercial waste management was calculated in this study. Firstly, the results revealed that the total GWP could be reduced by 93% by directing mixed waste to combustion instead of landfill. The total GWP could be decreased even further by 5% by manually sorting mixed waste more carefully at source via the help of the hypermarket employees. This entails that GHG emissions can be reduced by providing employees with better guidance on sorting, and also guides towards the tighter recycling target that has been introduced in Finland.

The findings also reveal that specific attention should be placed on those waste fractions that have the greatest influence on the total GWP of commercial waste management:

- Utilisation of energy waste because the GWP was as low as −880 to −860 kg$CO_2$-eq./t (influence 41–52% on the total GWP),
- landfilling mixed waste because the GWP was as high as 430 kg$CO_2$-eq./t (influence 15–23% on the total GWP),
- recycling PE plastic because the GWP was as low as −1800 kg$CO_2$-eq./t (influence 18–21% on the total GWP) and
- recycling cardboard because the GWP was as high as 51 kg$CO_2$-eq./t (influence 11–13% on the total GWP).

In the current study, cardboard had the highest share of the total waste mass (49–50 wt%). A key focus should be placed on substitutions of energy waste (55–70% of the total substitutions) and PE plastic (12–16% of the total substitutions) but also sorting PE plastic, even though the share of this waste fraction was not substantial (just 2 wt% of the total waste mass). Furthermore, the total GWP can be significantly reduced (in Scenario 1 up to 480%) by identifying an alternative recycling or incineration location for cardboard where it is used to substitute virgin material or replace fossil fuels respectively. In the case of combustion, departing from the waste hierarchy would be required but justified by the overall life cycle impact.

The results indicate that, focusing on paper, metal (steel) and glass will not result in any significant changes in total GWP due to the small amount of these waste fractions that are generated. Also, pre-treatment and transportation do not need to be of primary concern from the perspective of reducing GHG emissions. The main focus should be on treatment processes (96% of the total emissions) and substitutions, which have the most significant impact on total GWP.

This study has some limitations. First, it focused purely on GHG emissions, and this fact should be taken into account when analysing the results. At the same time, this study found that every waste fraction includes a set of factors that impact the results. There is a need to understand these factors when analysing the results. For example, landfilling and combustion produce high GHG emissions. These emissions are calculated based on waste composition, which can vary according the origin of the waste. Also, the electricity grid mix varies; e.g., the share of fossil fuels varies across different countries. The electricity grid mix in Finland was composed of 33.9% of fossil fuels, 29.7% of nuclear, 35.9% of renewables, and 0.6% of other sources. At the same time, the substituted heat varies because it should be selected locally.

Second, this study was limited in terms of the composition data of different waste fractions, especially mixed waste and energy waste. The compositions used represented the best available information that corresponded to the waste fractions in the case study. Further studies are required to better determine the compositions of waste to enable more accurate calculations of the GHG emissions produced during the management of waste fractions. In addition, the methods employed to treat different waste fractions, with the exception of mixed waste, were chosen to represent the real situation in 2012. This entailed that some treatment methods were excluded from the study. Further studies are required to calculate the GWPs of other treatment methods and researchers should extend the calculations to other environmental aspects and costs. For example, it could be useful to consider the costs that would be incurred in the process of achieving the potential reductions identified in this research.

A fundamental conclusion of this research is that waste management companies have a notable influence on the emissions associated with commercial waste management because they choose the places at which the different waste fractions are treated and utilised (taking into account fundamental factors such as the law). The results of this study highlight the areas on which waste management companies should concentrate to effectively reduce the impact of commercial waste. With the help of this real case study, they can more readily calculate or create a tool that can be employed to determine the total GWP of waste management for specific commercial enterprises and identify methods by which emissions can be reduced. By decreasing the GHG emissions produced by waste management, waste management companies and commercial enterprises can have a positive impact on the environment and market a greener image.

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## References

1999/31/EC. Council Directive 1999/31/EC of 26 April 1999 on the landfill of waste. Off. J. Eur. Commun., L 182, 16 July 1999.

2008/98/EC. Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on waste and repealing certain Directives. Off. J. Eur. Union, L 312, 22 November 2008.

Alakangas, E., 2000. Properties of fuels used in Finland. VTT Technical Research Centre of Finland, Espoo, Finland. VTT Research Notes 2045 (in Finnish) <http://www.vtt.fi/inf/pdf/tiedotteet/2000/T2045.pdf> (accessed 20.1.2013).

Anttila, L., 2011. Energy recovery efficiencies of waste incineration plants burning mixed municipal solid waste or recovered fuels. Bachelor's Thesis. Lappeenranta University of Technology, Degree Programme in Environmental Technology, Lappeenranta, Finland. In Finnish <http://urn.fi/URN:NBN:fi-fe201202161396> (accessed 6.12.2012).

Astrup, T., Fruergaard, T., Christensen, T.H., 2009a. Recycling of plastic: accounting of greenhouse gases and global warming contributions. Waste Manage. Res. 27 (8), 763–772.

Astrup, T., Møller, J., Fruergaard, T., 2009b. Incineration and co-combustion of waste: accounting of greenhouse gases and global warming contributions. Waste Manage. Res. 27 (8), 789–799.

Bernstad, A., la Cour Jansen, J., Aspegren, H., 2011. Life cycle assessment of a household solid waste source separation programme: a Swedish case study. Waste Manage. Res. 29 (10), 1027–1042.

Boldrin, A., Andersen, J.K., Møller, J., Christensen, T.H., Favoino, E., 2009. Composting and compost utilization: accounting of greenhouse gases and global warming contributions. Waste Manage. Res. 27 (8), 800–812.

Borisov, A., 2012. Customer Service Manager, Hyötypaperi Oy, Valkeala, Finland. Personal communication, 24 September 2012.

Bruun, S., Hansen, T.L., Christensen, T.H., Magid, J., Jensen, L.S., 2006. Application of processed organic municipal solid waste on agricultural land – a scenario analysis. Environ. Model. Assess. 11 (3), 251–265.

Buttol, P., Masoni, P., Bonoli, A., Goldoni, S., Belladonna, V., Cavazzuti, C., 2007. LCA of integrated MSW management systems: case study of the Bologna District. Waste Manage. 27 (8), 1059–1070.

Chanton, J., Powelson, D., Green, R., 2009. Methane oxidation in landfill cover soils, is a 10% default value reasonable? J. Environ. Qual. 38 (2), 654–663.

Cherubini, F., Bargigli, S., Ulgiati, S., 2009. Life cycle assessment (LCA) of waste management strategies: landfilling, sorting plant and incineration. Energy 34 (12), 2116–2123.

Christensen, T.H., Simion, F., Tonini, D., Møller, J., 2009. Global warming factors modelled for 40 generic municipal waste management scenarios. Waste Manage. Res. 27 (9), 871–884.

Conesa, J.A., Font, R., Fullana, A., Martín-Gullón, I., Aracil, I., Gálvez, A., Moltó, J., Gómez-Rico, M.F., 2009. Comparison between emissions from the pyrolysis and combustion of different wastes. J. Anal. Appl. Pyrolys. 84 (1), 95–102.

Consonni, S., Giugliano, M., Grosso, M., 2005. Alternative strategies for energy recovery from municipal solid waste – Part B: Emission and cost estimates. Waste Manage. 25 (2), 137–148.

Corsten, M., Worrell, E., Rouw, M., van Duin, A., 2013. The potential contribution of sustainable waste management to energy use and greenhouse gas emission reduction in the Netherlands. Resour. Conserv. Recycl. 77, 13–21.

Daamgaard, A., Larsen, A.W., Christensen, T.H., 2009. Recycling of metals: accounting of greenhouse gases and global warming contributions. Waste Manage. Res. 27 (8), 773–780.

Davidsson, Å., Gruvberger, C., Christensen, T.H., Hansen, T.L., la Cour Jansen, J., 2007. Methane yield in source-sorted organic fraction of municipal solid waste. Waste Manage. 27 (3), 406–414.

De Feo, G., Malvano, C., 2009. The use of LCA in selecting the best MSW management system. Waste Manage. 29 (6), 1901–1915.

Del Borghi, A., Gallo, M., Del Borghi, M., 2009. A survey of life cycle approaches in waste management. Int. J. Life Cycle Assess. 14, 597–610.

Ekvall, T., Assefa, G., Björklund, A., Eriksson, O., Finnveden, G., 2007. What life-cycle assessment does and does not do in assessments of waste management. Waste Manage. 27 (8), 989–996.

Environment Office of Oulu, 2010. Kauppakeskusten jätehuoltoselvitys Oulun seudulla (Waste management study of shopping centres in the region of Oulu). Oulu, Finland. Report 3/2010 (in Finnish) <http://www.ouka.fi/documents/64417/d4d1dbd3-bfef-4da4-8291-d8aa0f370441> (accessed 16.3.2015).

EPA, 2008. Background information document for updating AP42 section 2.4 for estimating emissions from municipal solid waste landfills. EPA, USA <http://www.epa.gov/ttnchie1/ap42/ch02/draft/db02s04.pdf> (accessed 29.11.2012).

EPD, 2015. PCR 2008:02 Solid waste disposal services, Version 3.0. <http://www.environdec.com/en/PCR/Detail/?Pcr=5810&id=158&epslanguage=en> (accessed 10.10.2017).

European Environment Agency, 2013. Managing municipal solid waste – a review of achievements in 32 European countries. Copenhagen, Denmark. EEA Report 2/2013 <http://www.eea.europa.eu/publications/managing-municipal-solid-waste> (accessed 13.3.2015).

European Commission, 2015. Waste - Review of waste policy and legislation <http://ec.europa.eu/environment/waste/target_review.htm> (accessed 23.5.2017).

European Union. Living in the EU <https://europa.eu/european-union/about-eu/figures/living_en> (accessed 23.5.2017).

Eurostat. Municipal waste <http://ec.europa.eu/eurostat/web/waste/transboundary-waste-shipments/key-waste-streams/municipal-waste> (accessed 13.3.2015).

Eurostat, 2016. Population on 1 January <http://ec.europa.eu/eurostat/tgm/table.do?tab=table&plugin=1&language=en&pcode=tps00001> (accessed 23.5.2017).

Eurostat, 2017. Municipal waste statistics <http://ec.europa.eu/eurostat/statistics-explained/index.php/Municipal_waste_statistics> (accessed 23.5.2017).

Finnish Agency for Rural Affairs, 2012. Maatalouden ympäristötuen sitoumusehdot 2012 (Commitment terms for agricultural environmental aid 2012). In Finnish <http://www.mavi.fi/fi/oppaat-ja-lomakkeet/viljelija/Documents/Ymp%C3%A4rist%C3%B6tuen%20sitoumusehdot%202005-2013/Ymparistotuen_sitoumusehdot_2012.pdf> (accessed 29.10.2012).

Finnish Food Information, 1996. Erilaiset kaupat (Different kind of shops) (in Finnish) <https://www.ruokatieto.fi/ruokakasvatus/ruokaketju-ruuan-matka-pellolta-poytaan/kauppa/miten-kauppa-toimii/erilaiset-kaupat> (accessed 17.10.2017).

Finnish Grocery Trade Association, 2017. Finnish Grocery Trade 2017 <http://www.pty.fi/fileadmin/user_upload/tiedostot/Julkaisut/Vuosijulkaisut/EN_2017_vuosijulkaisu.pdf> (accessed 17.10.2017).

Forssell, O., 2011. Quality study on energy waste in the Kujala waste center. Bachelor's thesis. Lahti University of Applied Sciences, Degree Programme in Environmental Technology, Finland (in Finnish) <http://publications.theseus.fi/bitstream/handle/10024/38381/Forssell_Olli.pdf?sequence=1> (accessed 19.12.2012).

Fruergaard, T., Astrup, T., Ekvall, T., 2009. Energy use and recovery in waste management and implications for accounting of greenhouse gases and global warming contributions. Waste Manage. Res. 27 (8), 724–737.

GaBi. A Product Sustainability Performance Solution by Thinkstep (before PE INTERNATIONAL) <http://www.gabi-software.com/nw-eu-english/index/> (accessed 9.4.2015).

Gentil, E., Clavreul, J., Christensen, T.H., 2009. Global warming factor of municipal solid waste management in Europe. Waste Manage. Res. 27 (9), 850–860.

Government of Finland, 2013. Landfill Decree 331/2013 of the Government, 2 May 2013 (in Finnish).

Hautamäki, J., 2012. Business Manager, Hyötypaperi Oy, Valkeala, Finland. Personal communications, November-December 2012.

Havukainen, J., 2014. Biogas production in regional integrated biodegradable waste treatment – Possibilities for improving energy performance and reducing GHG emissions. Thesis for the degree of Doctoral of Science (Technology). Lappeenranta University of Technology, Lappeenranta, Finland. Acta Universitatis Lappeenrantaensis 594. <http://urn.fi/URN:ISBN:978-952-265-665-0> (accessed 29.5.2017).

Havukainen, J., Uusitalo, V., Niskanen, A., Kapustina, V., Horttanainen, M., 2014. Evaluation of methods for estimating energy performance of biogas production. Renew. Energy 66, 232–240.

Havukainen, J., Zhan, M., Dong, J., Liikanen, M., Deviatkin, I., Li, X., Horttanainen, M., 2017. Environmental impact assessment of municipal solid waste management incorporating mechanical treatment of waste and incineration in Hangzhou, China. J. Clean. Prod. 141, 453–461.

Helftewes, M., Flamme, S., Nelles, M., 2012. Greenhouse gas emissions of different waste treatment options for sector-specific commercial and industrial waste in Germany. Waste Manage. Res. 30 (4), 421–431.

Herrmann, I.T., Moltesen, A., 2015. Does it matter which Life Cycle Assessment (LCA) tool you choose? – a comparative assessment of SimaPro and GaBi. J. Clean. Prod. 86, 163–169.

Hupponen, M., Luoranen, M., Horttanainen, M., 2012. Life cycle assessment of greenhouse gas emissions from biowaste management: Case study of Lappeenranta, Finland. In: Proceedings Venice 2012, 4th International Symposium on Energy from Biomass and Waste, San Servolo, Venice, Italy, 12–15 November 2012.

Hupponen, M., Grönman, K., Horttanainen, M., 2015. How should greenhouse gas emissions be taken into account in the decision making of municipal solid waste management procurements? A case study of the South Karelia region, Finland. Waste Manage. 42, 196–207.

Hypermarket, 2017. Manager, the case hypermarket, Finland. Personal Communication, 17 October 2017.

IPPC, 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Volume 5, Waste. IGES: Japan <http://www.ipcc-nggip.iges.or.jp/public/2006gl/vol5.html> (accessed 18.10.2012).

IPPC, 2007. Climate Change 2007: Working Group I: The Physical Science Basis. IPPC Fourth Assessment Report <http://www.ipcc.ch/publications_and_data/ar4/wg1/en/ch2s2-10-2.html> (accessed 12.10.2017).

ISO/TR 14049, 2000. Environmental management – Life cycle assessment – Examples of application of ISO 14041 to goal and scope definition and inventory analysis.

Jönsson, H., Baky, A., Jeppsson, U., Hellström, D., Kärrman, E., 2005. Composition of urine, faeces, greywater and biowaste for utilization in URWARE model. Chalmers University of Technology, Gothenburg, Sweden. Urban Water Report 2005:6 <http://www.iea.lth.se/publications/Reports/LTH-IEA-7222.pdf> (accessed 2.2.2015).

Kaazke, J., Meneses, M., Wilke, B.-M., Rotter, V.S., 2013. Environmental evaluation of waste treatment scenarios for the towns Khanty-Mansiysk and Surgut, Russia. Waste Manage. Res. 31 (3), 315–326.

Kähkönen, J., 2012. Mixed waste research of companies in Kymenlaakso. Bachelor's Thesis. Lahti University of Applied Sciences, Degree Programme in Environmental Technology, Finland (in Finnish) <http://www.theseus.fi/bitstream/handle/10024/39240/Kahkonen_Janne.pdf?sequence=1> (accessed 23.11.2012).

Kiuru, J., 2012. Business Manager, Hyötypaperi Oy, Kuusankoski, Finland. Personal communications, December 2012.

Korjala, S., 2017. Waste management worker, Kymenlaakson Jäte Oy, Keltakangas, Finland. Personal communications, 16 May 2017.

Koskenheimo, P., 2012. Koskenheimo Ky, Kotka, Finland. Personal communication, 21 December 2012.

Kuusiola, T., 2010. Environmental effects of collecting and utilizing small household metals in Helsinki Metropolitan Area. Master's thesis. Aalto University School of Science and Technology, Degree Programme of Material Science and Engineering, Helsinki, Finland (in Finnish).

Kymenlaakson Jäte, 2013–2016. Vuosikertomukset 2012–2015 (Annual reports 2012–2015). Kymenlaakson Jäte Oy, Keltakangas, Finland (in Finnish) <http://www.kymenlaaksonjate.fi/fi/Yhti%C3%B6t/Vuosikertiomukset/> (accessed 17.5.2017).

Larsen, A.W., Merrild, H., Christensen, T.H., 2009. Recycling of glass: accounting of greenhouse gases and global warming contributions. Waste Manage. Res. 27 (8), 754–762.

Latvala, M., 2009. Best available techniques (BAT), Production of biogas in a Finnish operating environment. Finnish Environment Institute, Helsinki, Finland. The Finnish Environment 24/2009 (in Finnish) <https://helda.helsinki.fi/bitstream/handle/10138/37998/SY_24_2009.pdf?sequence=1> (accessed 22.10.2012).

Liamsanguan, C., Gheewala, S.H., 2008. LCA: A decision support tool for environmental assessment of MSW management systems. J. Environ. Manage. 87 (1), 132–138.

LIPASTO, 2012. Average emissions of working machines in Finland 2011. VTT Technical Research Centre of Finland, Espoo, Finland <http://www.lipasto.vtt.fi/yksikkopaastot/muute/tyokoneete/diesel_a_ke.htm> (accessed 12.11.2012).

Markkanen, S., 2011. Production Director, Kotkan Energia Oy, Kotka, Finland. Personal communication, 25 May 2011.

Mepak-Kierrätys, 2012. Mepak-Kierrätys Oy. In Finnish <http://www.mepak.fi/images/Mepak-kalvosarja.ppt> (accessed 25.1.2013).

Merrild, H., Daamgaard, A., Christensen, T.H., 2008. Life cycle assessment of waste paper management: the importance of technology data and system boundaries in assessing recycling and incineration. Resour. Conserv. Recycl. 52 (12), 1391–1398.

Merrild, H., Daamgaard, A., Christensen, T.H., 2009. Recycling of paper: accounting of greenhouse gases and global warming contributions. Waste Manage. Res. 27 (8), 746–753.

Merta, E., Mroueh, U.-M., Meinander, M., Punkkinen, H., Vähä-Nissi, M., Kortet, S., 2012. Muovipakkausten kierrätyksen edistäminen Suomessa (Promotion of plastic package recycling in Finland). Ministry of Employment and the Economy (MEE), Helsinki, Finland. MEE reports 11/2012 (in Finnish).

Ministry of the Environment, 2009. Towards a recycling society – The National Waste Plan for 2016. Helsinki, Finland. The Finnish Environment 14/2009 <https://helda.helsinki.fi/bitstream/handle/10138/38022/FE_14_2009.pdf?sequence=1> (accessed 13.3.2015).

Ministry of the Environment, 2011. Finland is implementing the Kyoto Protocol <http://www.ym.fi/download/noname/%7B3AF4AD94-EB07-4A9B-8FA4-E4136B6C1A2B%7D/58473> (accessed 13.3.2015).

Ministry of the Environment, 2017. Valtakunnallinen jätesuunnitelma (The National Waste Plan). Helsinki, Finland (in Finnish) <http://www.ym.fi/fi-FI/Ymparisto/Jatteet/Valtakunnallinen_jatesuunnitelma> (accessed 23.5.2017).

Moliis, K., Dahlbo, H., Retkin, R., Myllymaa, T., 2012. Recovery of packaging waste in northern Finland – environmental and cost effects on a life-cycle basis. Ministry of the Environment, Helsinki, Finland. Reports of the Ministry of the Environment 26/2012 (in Finnish) <http://www.ym.fi/download/noname/%7BF207DDDD-1863-40BD-A195-86C52FB4C1D0%7D/34426> (accessed 25.1.2013).

Møller, J., Boldrin, A., Christensen, T.H., 2009. Anaerobic digestion and digestate use: accounting of greenhouse gases and global warming contribution. Waste Manage. Res. 27 (8), 813–824.

Monni, S., 2010. Yhdyskuntajätteen käsittelyketjujen hiilijalanjäljet (Carbon footprints of treatment chains of municipal waste). Benviroc Oy, Espoo, Finland (in Finnish).

Monni, S., 2012. From landfilling to waste incineration: Implications on GHG emissions of different actors. Int. J. Greenh. Gas Control 8, 82–89.

Myllymaa, T., Moliis, K., Tohka, A., Rantanen, P., Ollikainen, M., Dahlbo, H., 2008a. Environmental loads and costs of waste recycling and incineration processes. Inventory report. Finnish Environment Institute, Helsinki, Finland. Reports of the Finnish Environment Institute 28/2008 (in Finnish) <https://helda.helsinki.fi/bitstream/handle/10138/39792/SYKEra_28_2008.pdf?sequence=1> (accessed 14.10.2012).

Myllymaa, T., Moliis, K., Tohka, A., Isoaho, S., Zevenhoven, M., Ollikainen, M., Dahlbo, H., 2008b. Environmental impacts and costs of recycling and incineration of waste – The alternatives of regional waste management. Finnish Environment Institute, Helsinki, Finland. The Finnish Environment Institute 39/2008. In Finnish <https://helda.helsinki.fi/bitstream/handle/10138/38383/SY_39_2008.pdf?sequence=5> (accessed 29.5.2017).

Nielsen, M., Illerup J.B., 2003. Emissionsfaktorer og emissionsopgørelse for decentral kraftvarme. The National Environmental Research Institute (NERI), Ministry of the Environment, Denmark. NERI Technical Report No. 442. In Danish <http://www2.dmu.dk/1_viden/2_publikationer/3_fagrapporter/rapporter/fr442.pdf> (accessed 19.10.2012).

Niskanen, A., Värri, H., Havukainen, J., Uusitalo, V., Horttanainen, M., 2013. Enhancing landfill gas recovery. J. Clean. Prod. 55, 67–71.

Petäjä, J., 2007. Excel File to Calculate Methane from Landfill. Finnish Environment Institute, Helsinki, Finland (in Finnish).

Ragoßnig, A.M., Wartha, C., Pomberger, R., 2009. Climate impact analysis of waste treatment scenarios – thermal treatment of commercial and pretreated waste versus landfilling in Austria. Waste Manage. Res. 27 (9), 914–921.

Reimann, D.O., 2012. CEWEP Energy Report III (Status 2007-2010) – Results of Specific Data for Energy, R1 Plant Efficiency Factor and NCV of 314 European Waste-to-Energy (WtE) Plants <http://www.cewep.eu/m_1069> (accessed 5.3.2015).

Rigamonti, L., Grosso, M., Møller, J., Martinez Sanchez, V., Magnani, S., Christensen, T.H., 2014. Environmental evaluation of plastic waste management scenarios. Resour. Conserv. Recycl. 85, 42–53.

Ripa, M., Fiorentino, V., Vacca, V., Ulgiati, S., 2017. The relevance of site-specific data in Life Cycle Assessment (LCA). The case of the municipal solid waste management in the metropolitan city of Naples (Italy). J. Clean. Prod. 142, 445–460.

Räsänen, K., 2013. Plant Manager, Kymen Bioenergia Oy, Kouvola, Finland. Personal communications, January 2013.

Salmenperä, H., Moliis, K., Nevala, S.-M., 2015. Forecasting waste volumes to 2030 – focusing on municipal waste and reaching recycling targets. Ministry of the Environment, Helsinki, Finland. Reports of the Ministry of the Environment 17/2015 (in Finnish) <https://julkaisut.valtioneuvosto.fi/bitstream/handle/10138/155189/YMra_17_2015.pdf?sequence=1> (accessed 18.10.2017).

SFS-EN ISO 14040, 2006. Environmental management. Life cycle assessment. Principles and framework. Finnish Standards Association, Helsinki, Finland.

SFS-EN ISO 14044, 2006. Environmental management. Life cycle assessment. Requirements and guidelines. Finnish Standards Association, Helsinki, Finland.

Silvennoinen, K., Koivupuro, H.-K., Katajajuuri, J.-M., Jalkanen, L., Reinikainen, A., 2012. Food waste volume and composition in Finnish food chain. MTT, Helsinki, Finland. MTT report 41. In Finnish <http://www.mtt.fi/mttraportti/pdf/mttraportti41.pdf> (accessed 18.10.2017).

Statistics Finland, 2011. Fuel classification 2011. Statistics Finland, Helsinki, Finland <http://www.stat.fi/tup/khkinv/khkaasut_polttoaineluokitus_2011.xls> (accessed 7.11.2012).

Statistics Finland, 2016. Appendix table 1. Waste treatment in 2015, tonnes. Statistics Finland, Helsinki, Finland. <http://www.stat.fi/til/jate/2015/jate_2015_2016-12-20_tau_001_en.html> (accessed 23.5.2017).

Statistics Finland, 2017. Suomen kasvihuonekaasupäästöt 1990–2016 (Greenhouse gas emissions in Finland 1990–2016). Statistics Finland, Helsinki, Finland (in Finnish) <http://www.stat.fi/static/media/uploads/tup/khkinv/suomen_kasvihuonekaasupaastot_1990-2016_final.pdf> (accessed 20.10.2017).

Suomen Kuitukierrätys, 2012. Kuitupakkausten kierrätys ja hyötykäyttö (Recycling and utilisation of fiber packaking). Suomen Kuitukierrätys Oy, Helsinki, Finland (in Finnish).

Teirasvuo, N., 2011. Composition and Combustion Properties of Source Separated Municipal Solid Waste in Densely Populated Area of Lappeenranta. Lappeenranta University of Technology, Lappeenranta, Finland.

Tulokhonova, A., Ulanova, O., 2013. Assessment of municipal solid waste management scenarios in Irkutsk (Russia) using a life cycle assessment integrated waste management model. Waste Manage. Res. 31 (5), 475–484.

Vainikka, P., Tsupari, E., Sipilä, K., Hupa, M., 2012. Comparing the greenhouse gas emissions from three alternative waste combustion concepts. Waste Manage. 32 (3), 426–437.

Virtavuori, V., 2009. Appropriate treatment processes for biowaste in the Helsinki Metropolitan Area – benchmarking of greenhouse gas emissions. YTV Helsinki Metropolitan Area Council, Helsinki, Finland. YTV publications 8/2009 (in Finnish) <https://www.hsy.fi/sites/Esitteet/EsitteetKatalogi/Julkaisusarja/8_2009_biojatteen_kasittelyvaihtoehdot_paakaupunkiseudulla.pdf> (accessed 29.5.2017).

Wittmaier, M., Langer, S., Sawilla, B., 2009. Possibilities and limitations of life cycle (LCA) in the development of waste utilization systems – applied examples for a region in Northern Germany. Waste Manage. 29 (5), 1732–1738.

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