# Case study on sampling, processing and characterization of landfilled municipal solid waste in the view of landfill mining

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## ARTICLE INFO

## ABSTRACT

Landfilling has been the major method for municipal solid waste (MSW) disposal during recent decades. Recently utilization of deposited materials, commonly referred to as landfill mining, has been increasingly considered due to increasing raw material costs and environmental reasons. When mining municipal solid waste (MSW) landfills, finding suitable treatment and utilization routes for different types of materials is essential because of the economic aspects and to minimize the re-landfilled fraction of the waste. This paper describes a case study to sample, characterize and process wastes of a potential landfill mining site. The study combines manual sorting with full-scale mechanical treatment to better assess the treatability of the landfilled wastes. The approach also aimed at complying with the challenges of the heterogenous nature of the landfilled MSW. An approximately 30 m high and 10-year-old Finnish landfill was sampled in connection with the drilling of three gas collections wells, producing samples from two layers of waste of slightly different ages (ca. 5–10 years). Manual sieving and sorting into seven waste fractions as well as full-scale mechanical processing was performed. Further characterization included the fuel properties of the calorific fractions and the leaching properties of the fine materials. Manual sorting of the materials yielded 40–45% (w/w) of the possible fuel fraction in the landfilled waste with a net calorific value of approximately 20 MJ/kg dry matter. The metal fractions recovered in the manual sorting amounted to 3–4% (w/w). The landfilled waste was also processable by full-scale mechanical processes, including shredding, magnetic separation, screens and a wind sieve, despite the moisture and impurities in the landfilled waste. Results from the mechanical process showed that approximately 30% (w/w) of the material could be recovered as solid recovered fuel with similar calorific values to the fuel fraction from the manual sorting. Approximately 1% (w/w) could be recovered as magnetic metals in the mechanical process. The yields of fuel and metal fractions are site-specific but could likely be improved by optimizing the mechanical process for landfill mining purposes, as is indicated by the results from the manual sorting of the materials. The amount of fine materials (<20 mm in the manual sorting and <30 mm in the mechanical process) was found to be ca. 50% (w/w) which supports previous reports of the amount of the fines. The fine materials require attention to minimize the waste remaining from landfill mining. The Fe and Al contents of the fine fraction, at approximately 5% (w/w) both in the manually sorted and mechanically treated waste, are interesting for recovery purposes. The findings from this study highlight the importance of proper exploration stage in a landfill mining project in order to plan the best applicable full-scale processes for material recovery.

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## 1. Introduction

The rising global consumption of various materials in the past few decades has increased the costs of raw materials and the environmental impact of their extraction and use. Furthermore, concerns about the availability and sufficiency of certain materials have increased. For instance, scientific debates have taken place concerning the availability of world's copper resources in response

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to increasing demand (Gordon et al., 2006; Tilton and Lagos, 2007). To respond to these challenges, in several countries, material efficiency and recirculation programs have been and are being developed. Recently, recovery of materials from city infrastructures and from landfills has gained growing interest (Jones et al., in press).

Projects aimed at resource recovery from landfills are commonly referred to as landfill mining (Krook et al., 2012). In addition to the value of the recovered materials, the most commonly cited reasons for landfill excavation/mining has been the remediation of landfills to prevent local emissions, to create new volume in the existing landfill, to create space for new infrastructure, and to produce recyclable materials such as metals and waste fuels (Goeschl, 2012). Furthermore, waste management companies are looking for new business concepts due to the declining volume of solid wastes processed at landfills. (Van Der Zee et al., 2004).

Landfills have been used for the disposal of almost all municipal solid waste (MSW) and industrial waste for decades, although source segregation of materials such as metals, paper, hazardous materials, and glass has been increasingly practiced, as has waste incineration. Old MSW landfills are an especially heterogenous source of materials (Paap et al., 2011) the composition, volume and placement practices of which are not well documented (Jones et al., in press). The importance of proper exploration in the early stages of a possible landfill mining project has been highlighted in previous papers (Goeschl, 2012; Hull et al., 2005; Quaghebeur et al., in press). As part of the exploration stage, site specific investigations including test excavations or drillings into the landfill body are a necessity in assessing the composition of the landfilled waste (Krook et al., 2012).

The planning of suitable treatment processes for recoverable waste fractions in a landfill mining project − in MSW, recoverable waste fractions are primarily metals, waste fuel and possibly soil - requires detailed knowledge not only of the composition but also of the treatability of the landfilled wastes. Previous waste characterization studies of landfilled wastes have addressed the issue of waste characterization primarily by screening the waste into different particle size categories and subsequently manually sorting at least the coarsest particle size categories into different waste categories (Hogland, 2002; Hull et al., 2005; Kurian et al., 2003; Quaghebeur et al., in press; Stessel and Murphy, 1992). Hogland (2002) performed extensive waste characterization of samples excavated from two Swedish MSW landfills to address the potential for recycling and energy recovery of excavated waste. Their studies included screening (18 and 50 mm) and manual sorting of the waste samples and characterization of the different categories with regard to e.g. composition, calorific value and concentrations of contaminants. Hull et al. (2005) screened (25.4 mm) and hand-sorted the oversized fractions of 98 samples drilled from a MSW landfill in New Jersey, USA. Characterization of the samples included the determination of the calorific value of calorific fractions (paper, cardboard, food and yard waste, wood and plastics) and concentrations of both inorganic and organic contaminants in the fine fractions. Kurian et al. (2003) characterized samples from two MSW landfills in India by first separating the <20 mm size fraction by sieving and manually sorting the oversized material. In a recent study, Quaghebeur et al. (in press) also started with screening (10 mm) after drying the samples and continued with manual sorting of the oversized fractions. The fractions were then characterized for e.g. properties related to energy utilization potential and concentrations of contaminants.

One essential finding of previous waste characterization studies (Hogland, 2002; Hull et al., 2005; Quaghebeur et al., in press) of landfilled wastes is the strong presence of the fine fraction. This fraction may amount to more than 50% of total mass of an MSW landfill body (Quaghebeur et al., in press), consisting not only of the landfilled wastes but also of the landfill (daily) cover materials (usually soil). Generally speaking, the ratio between the amounts of waste and soil at a landfill may vary significantly and depends on factors such as local operating practices, compacting, the age of the landfill and the degree of decomposition of organic waste (Møller Rosendal, 2009). The quantity of recyclable metals, such as Al and Fe, has been found to be quite low in fine fractions excavated from MSW landfills (Quaghebeur et al., in press). On the other hand the heating value of the fine fraction has been reported to be low as well (Hogland et al., 1995; Quaghebeur et al., in press) thus limiting the possible utilization scenarios for this fraction. The fate of the fine fraction may be crucial for the economics of a landfill mining project. In some studies the fine soil type fraction has been considered to be a potential substrate for the intermediate or final covers of current landfilling operations, thus even creating revenues (Hogland et al., 2010; Jennings et al., 2007). Another alternative is that the fine fraction is classified as waste that must be re-landfilled, which obviously creates costs and may render a landfill mining project economically unfeasible.

Few studies have examined the treatability of excavated MSW and the effectiveness of unit processes (Hull et al., 2005; Stessel and Murphy, 1992). In most full-scale landfill mining projects material recovery has been a secondary objective; thus, simple soil excavation and screening techniques have been the primary methods applied in previous landfill mining work (Krook et al., 2012), with the exception of a few projects which utilized magnetic or air separation (Goeschl, 2012).

The objective of the present study was to assess the treatability, material recovery potential and material mass balances of landfilled MSW. Sorting efficiency was examined through a comparison of manual sorting (material recovery potential) and full scale mechanical pre-treatment (treatability) of wastes sampled from a MSW landfill. The full-scale process was able to treat multiple amounts of material compared to manual sorting, which helps tackling issues relating to heterogeneity of landfilled MSW. In addition the fuel properties of some of the recovered waste fractions were assessed to evaluate their potential usability as solid recovered fuel. In contrast to previous studies, the fine MSW material recovered from the landfill was also assessed with regard to its landfill acceptability.

## 2. Materials and methods

### 2.1. Site description

The landfill studied is established according to European Union (EU) requirements for sealed bottom structure. The landfill is located in Kuopio, Finland. It has a land area of approximately 32 000 square meters and a peak area of approximately 18 000 square meters. The maximum filling height is approximately 30 m. The landfill contains primarily mixed MSW landfilled between 2001 and 2011. From 2001 to 2003, source separation of biowaste was not practiced in the area. Between 2004 and 2009, the landfill received MSW with source separation of biowaste taking place. Mechanical pre-treatment of all the received waste to produce refuse-derived fuel began in 2009, and thus the uppermost layers of the landfill contain mostly rejects (sieve underflow) from the mechanical pre-treatment. These layers were seen to produce little interest with regard to landfill mining, because the fuel fractions and magnetic metals were previously recovered via a mechanical process (see 2.3.2). The uppermost layers were thus discarded from this study. In addition to the landfilled wastes, the landfill contains layers of soil used as intermediate covers during the filling. The amount of the intermediate cover materials was not known.

## 2.2. Sampling

The landfill was sampled in July 2012, when vertical gas collection wells were installed as part of the landfill's closure procedure. The wells were drilled with a hydraulic piling rig Casagrande B 170 (weight category 55 tons) operated in auger mode with an auger diameter of 90 cm and a height of 1 m. The samples for the study were obtained from three vertical wells, referred to as K41, K42 and K44, at two different depths (Table 1), while drilling the landfill body to penetrate the waste fill up to a maximum of 31 m in height. The cutting points of sample depths were chosen based on data on filling heights as a function of time in different parts of the landfill. Thus, the goal was to divide the samples roughly into older samples and younger samples, reflecting different storage times in the landfill and possibly differences in local source separation behavior regarding biowaste.

Trial excavations have also been used in the exploration stages of landfill mining projects (e.g. Hogland, 2002). However, when wanting to go deeper than 9 m, drilling has been seen as the only applicable approach (Hjelmar et al., 2011). The drilling technique used in sampling may produce samples with differing properties to those obtained by excavating, as the drilling may e.g. reduce the particle size of the coarsest objects.

Samples of 5–10 $m^3$ from full depths of the wells (Table 1) were collected by directing the waste drilled to separate skips placed in the vicinity of the wells. Thus, a total of six skips of samples were obtained for further processing. The samples had somewhat differing moisture contents, the quantitative amount of which could not be determined in the context of this study. The obtained sample masses (Table 1) reflected the moisture content and bulk density of the landfilled waste at different sampling points. The mass of sample from a depth of 15–31 m at point K44 was low due to technical issues in sample transport from the auger to the skip. Sampling point K41 was quite dry throughout the landfill body, whereas samples obtained from sampling points K42 and K44 contained a larger amount of moisture, which tended to increase toward the bottom sections of the landfill body. The moisture contained in the samples made the manual classification of the samples more challenging than was the case with the drier samples.

All the data on waste composition (Sections 3.1.1, 3.1.2 and 3.1.3) include the moisture contained in the samples.

## 2.3. Sample sorting and processing at the site

The samples were stored in covered skips in ambient conditions for 1–2 weeks. At the landfill site, the samples were subjected to manual sieving and sorting and to full-scale trial runs with mechanical pre-treatment machinery. Fig. 1 shows the methods of sample processing and sorting at the site.

### 2.3.1. Manual sieving and sorting (M-samples)
Approximately 600 l of MSW for manual size classification and sorting were sub-sampled from each skip, representing six separate sampling points from within the landfill. The selected sample size was a compromise between the sample representativeness and the time required to manually sort the sample. These samples, referred to as M-samples, were manually sieved to separate the >100 mm, 40–100 mm, 20–40 mm, and <20 mm particle size categories. Sieves with 100 and 40 mm mesh size were selected to represent the openings of the drum screen of the full-scale mechanical equipment to facilitate comparison between manual and mechanical operations. 20 mm sieve was selected to separate the finest fraction. Each particle size category, with the exception of the <20 mm material, was manually separated into seven fractions metals, plastics, paper and cardboard, textiles, soil, wood, and other. The classification was based on fractions used in previous studies (Hogland, 2002; Hull et al., 2005; Quaghebeur et al., in press; Sormunen et al., 2008). Fraction "Other" included all the material which could not be classified into seven pre-determined waste fractions. All the manually sorted size categories contained impurities within the seven fractions of separation.

### 2.3.2. Full-scale mechanical pre-treatment (P-samples)
The full-scale mechanical pre-treatment machinery is used for the production of solid recovered fuel (SRF) as part of local MSW treatment, and thus it was not specifically designed for the excavated materials. The machinery consisted of a Lindner Jupiter 3200 single-shaft shredder, a magnetic separator, Doppstadt SM − 718 drum sieve and a wind sieve. Waste is fed into the shredder feed funnel from which it enters the shredder. After shredding, the material encounters the magnet, which removes the magnetic fraction from the shredded material. After the magnet, the waste enters the drum sieve, which, during the trial runs, had sieve openings of 30 and 70 mm, thus producing fractions of <30 mm, 30–70 mm, and >70 mm. The >70 mm fraction is further subject to wind sieving, which separates the light fraction from the heavy fraction. The light fraction of >70 mm is the fuel fraction produced by mechanical processing.

To produce another set of samples, the three skips of the middle and bottom layer samples were combined to form the approx. 20 and 22 tons feeds for the full-scale mechanical pre-treatment. The two feeds were processed by mechanical treatment to produce the following samples, referred to as P-samples: a magnetic fraction, $a < 30$ mm fraction, a 30–70 mm fraction, $a > 70$ mm heavy fraction and $a > 70$ mm light fraction.

When planning the present study, the drum screen of the full-scale process operated with openings of 40 and 100 mm, and similar mesh sizes were selected for the manual sieving experiments. Later on the drum screen was however switched to work with openings of 30 and 70 mm. This slightly worsened the comparability of the mechanical and manual sorting.

## 2.4. Sample sorting and processing at the laboratory

After the samples were processed and sorted at the site, they were delivered to the laboratory for further sorting and characterization. The primary goals of the laboratory characterization were to produce estimates of the fuel value of the sorted calorific fractions and to obtain a closer look at the properties of the fine fractions. Also of interest was finding possible differences between the M-samples and the P-samples. The different processing of the M- and P-samples did not, however, allow for a comprehensive comparison of the results. Other sorted fractions, such as metals, soil and other, were only evaluated visually.

### 2.4.1. Calorific fractions
To determine the fuel characteristics, composite fractions (referred to as the calorific fraction) were formed from the plastic,

**Table 1**
Samples obtained from the landfill body in connection with drilling of gas collection wells.

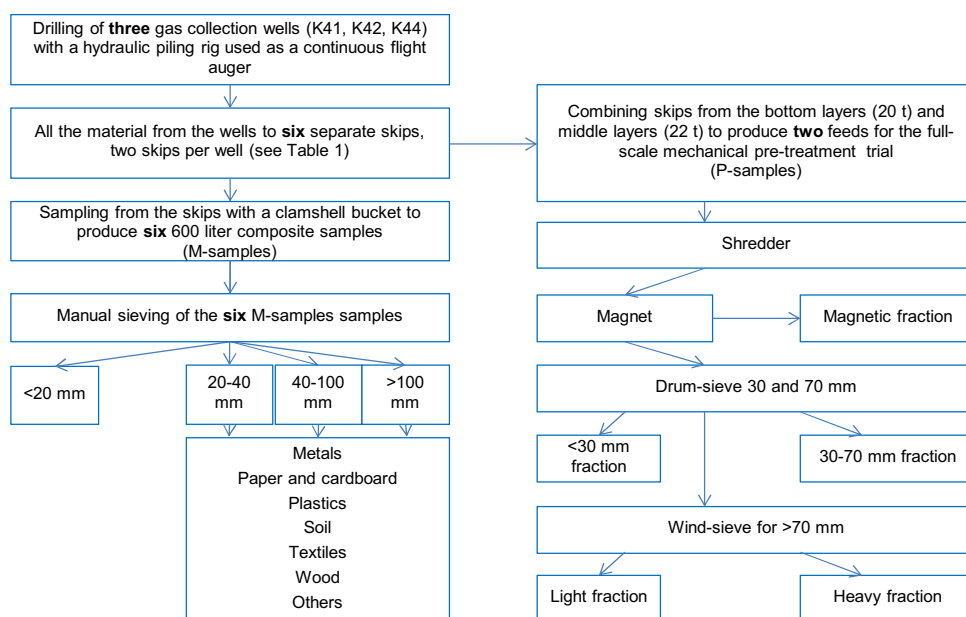| Sampling point | Samples from landfill middle layers | | Samples from landfill bottom layers | |
|---|---|---|---|---|
| | Depth in meters | Sample mass in tons | Depth in meters | Sample mass in tons |
| K41 | 2–10 | 3.4 | 10–22 | 7.2 |
| K42 | 2–14 | 8.2 | 14–26 | 9.9 |
| K44 | 2–15 | 11 | 15–31 | 3.3 |

Fig. 1. Flow-sheet of sampling and further processing of samples at the site.

paper and cardboard, textile and wood fractions in particle size categories of 20–40 mm, 40–100 mm and >100 mm of the M-samples. The composite fractions were formed based on mass distribution of the fractions in the M-samples. Five caloric fraction samples were obtained, as one of the M-samples was missed due to technical problems in sample delivery to the laboratory.

For the P-samples, the calorific fractions were the light fractions of the >70 mm material after the wind sieve processing.

The calorific fractions were studied for their gross and net calorific value and semi-quantitative composition (X-ray fluorescence analysis, XRF). The calorific fractions were air-dried for two weeks in an outdoor warehouse to enable shredding and homogenization prior to the examinations.

### 2.4.2. Fine fractions

The fine fractions were those fractions below 20 mm of the M-samples and those below 30 mm of the P-samples (for explanation of different particle sizes for fine fractions, see 2.3.2). Preliminary environmental properties of the fine fractions in terms of landfill acceptability were evaluated by determining the inorganic composition (XRF) and total organic carbon (TOC) and by performing batch leaching test EN 12457–4 (CEN, 2002).

Further sieving and manual sorting of some of the fine fraction samples were carried out to discover possible utilization routes for these fractions, which are often regarded as non-utilizable. Two fine fraction samples of <20 mm from the M-samples (K41 2–10 m and K41 10–22 m) were further sieved with sieve sizes of 10, 4 and 1 mm. Fine fraction samples of <30 mm from the P samples were also further sieved with sieve sizes of 20, 10, 4 and 1 mm. For these purposes, the fine fraction samples were air-dried at room temperature for a week. Categories of 4–10 mm and, 10–20 mm for both M and P-samples and 20–30 mm only for P-samples were manually sorted to achieve the same fractions as for the coarser particle size categories from the M-samples in the field.

### 2.5. Leaching tests and analytical methods

A single stage batch leaching test EN 12457–4 (CEN, 2002) was performed for the fine fractions. In this test, the solid sample is agitated with demineralized water for 24 h in a liquid-to-solid (L/S) ratio of 10 L/kg dry matter. The eluate and the solid are separated by filtration (using a 0.45 μm membrane), after which the concentrations of leached substances in the eluate are measured. Leaching test eluates were analyzed for As, Ba, Cd, Co, Cr, Cu, Mo, Ni, Pb, Sb, Se, V and Zn by inductively coupled plasma mass spectrometry (ICP-MS) and for Hg by cold vapor atomic fluorescence spectroscopy (CV-AFS). Chloride, fluoride and sulfate concentrations were determined using ion chromatography (IC). Concentrations of dissolved organic carbon (DOC) were analyzed according to the standard EN1484:1997 (CEN, 1997). The pH, conductivity and redox-potential of the eluates were also measured.

Semi-quantified X-ray fluorescence analysis (XRF) was carried out using an X-ray fluorescence spectrometer and a semi-quantified Semi-Q-program. This method is applicable for fluorine and elements heavier than fluorine, and a typical detection limit is around 0.01% (w/w).

Total organic carbon was determined according to standard SFS-EN 13137 (CEN, 2001) using an element analyzer. In this method, a solid sample is combusted in an oxygen flow (1400 °C), and the carbon dioxide formed by combustion is determined with an IR-detector. Before combustion, the sample is acidified with phosphoric acid to remove inorganic carbon.

Gross calorific values were measured with an automatic bomb calorimeter. Net calorific value was obtained by performing the so-called "hydrogen correction". During the combustion, some water is formed, and the gross calorific value is corrected with the heat of vaporization of the formed water.

## 3. Results and discussion

### 3.1. Sorting and processing of samples

#### 3.1.1. Particle size distribution (M-samples)

The distribution of M-samples into particle size categories is somewhat similar between the samples from the bottom layers containing older waste and the samples from middle layers of the landfill (Fig. 2). For example, the proportions of particle size 40–100 mm and 20–40 mm are approximately 16% and 6% in middle and bottom layers, respectively, while minor variations were observed in particle sizes of >100 mm and <20 mm. However, the

differences between the layers are in each category within the standard deviations found for the three sampling points.

The mass fraction of the fine fractions (<20 mm) equals 43% on average for the samples from the landfill's middle layers and 47% for the samples from its bottom layers. This may indicate transport of the fine material toward the bottom layers.

The amount of the fine fractions is of the same order of magnitude than has been reported in previous studies (Hull et al., 2005; Quaghebeur et al., in press; Van Vossen and Prent, 2011). Quaghebeur et al. (in press) found an average of 64% mass fraction of the fine material (<10 mm) for industrial waste and 44% for MSW from different sections of a Belgian landfill. However, in their study the samples were dried at 70 °C before sieving, which may have increased the amount of fine material in comparison to the results presented here, which used material under ambient conditions. Drying of the material may increase the amount of the fine fraction because in moist conditions, part of the fine material tends to stay attached to bigger particles. Hull et al. (2005) reported 50–58% mass fraction of fine material <25.4 mm in MSW landfilled between 1989 and 1999 following manual size separation, and 50–52% mass fraction of fine material (25.4 mm) in mechanically screened samples. An analysis of landfill mining projects in a literature review yielded an average mass fraction of 55% fine material with particle size of <24 mm (Van Vossen and Prent, 2011).

The M-sample volume of approximately 600 L resulted in an average M-sample mass of 172 kg for the samples from the middle layers and 227 kg for the samples from the bottom layers. The obtained sample masses help explain why single large particles can have quite a large influence on the total weight of a particle size category, especially in the coarser categories. In this study, samples were collected from three sampling points to minimize the effect of single particles on average values and to find variations in the particle size categories within the landfill, as shown by standard deviations in Fig. 2.

### 3.1.2. Manual sorting

Fig. 3 shows the distribution of the mass of the M-samples manually sorted at the site into different waste fractions. The fine fraction of <20 mm was not further sorted at the site. In Fig. 3 all the waste fractions found in particle size categories of 20–40 mm, 40–100 mm and >100 mm have been summed. In the coarser particle size categories (>20 mm), plastic was the most dominant fraction, averaging up to 24 and 23% of the mass in the M-samples from the middle and the bottom layers, respectively. A study by Quaghebeur et al. (in press) showed proportions of plastic at similar levels for MSW landfilled during 1995–2000. For older wastes, they found smaller proportions of plastic. Findings by Sormunen et al. (2008) involving the detailed characterization of two Finnish landfills also showed lower mass fractions of plastic for wastes landfilled between 1987 and 2003. Hull et al. (2005) also reported lesser mass fractions of plastics for MSW landfilled between 1989 and 1999 as compared to the results of the present study. The mass fractions of plastics in MSW landfilled between 1989 and 1999 were between 15 and 18% in waste samples with particle size >25.4 mm (Hull et al., 2005). Where the mass fraction of fine material comprises approximately 50%, this equates to a mass fraction of plastics of 7.5–9% of the total waste mass. There are two possible reasons for the plastic to appear in as high a mass fraction as found in the present study. First, the use of plastics has increased markedly since the 1980s. For example, in Finland, the share of plastics in household waste has risen from 5 to 10% between 1980 and 2000 (Sokka et al., 2007). Second, the increasing source separation of other waste fractions before reaching the landfill also increases the mass fraction of plastics.

The mass fractions of paper/cardboard, textiles and wood observed in the present study were similar to those reported by Hull et al. (2005), Sormunen et al. (2008) and Quaghebeur et al. (in press). In all of these studies including the present study especially the mass fraction of paper and cardboard was lesser in older wastes, likely indicating both changes in consumption and the degradation of the paper/cardboard, textiles and wood fractions at the landfill. Plastic, paper/cardboard, textiles and wood can be regarded as energy utilizable fractions found in the M-samples. Together, these fractions amount to 45 and 40% of the mass of the M-samples from the middle and the bottom layers, respectively, the differences resulting mainly from paper/cardboard fraction. These results are in accordance with results from Quaghebeur et al. (in press) who found the quantity of combustibles (the same fractions as reported in the present study) to be in the range of 34 and 50% (w/w) dependent upon the period during which the MSW was landfilled.

The mass fractions of metals amount to 3.4 and 4.4% for the M-samples from the middle and the bottom layers respectively. Metal mass fractions are similar to those found in previous studies (Hogland, 2002; Hull et al., 2005; Quaghebeur et al., in press).

Some clear trends can be seen in the distribution of certain waste fractions in the particle size categories (Fig. 4). Plastic, for instance, amounts to almost 50% of the mass of the coarsest particle size category (>100 mm), its mass fractions decreasing when moving to finer particle sizes. For textiles and metals, the trend is
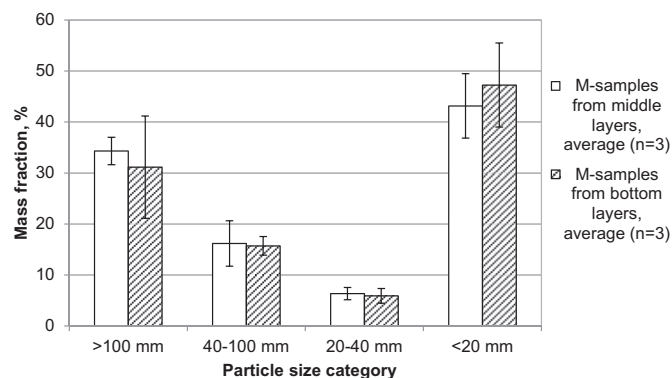


**Fig. 2.** Particle size distributions in the M-samples. Error bars in the figure show standard deviations of the three samples from both layers.
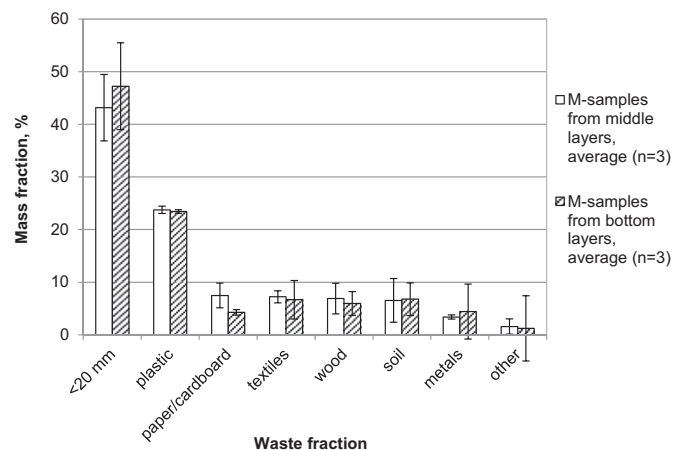


**Fig. 3.** Distribution of the mass of M-samples into different waste fractions. Error bars in the figure show standard deviations of the three samples from both layers.
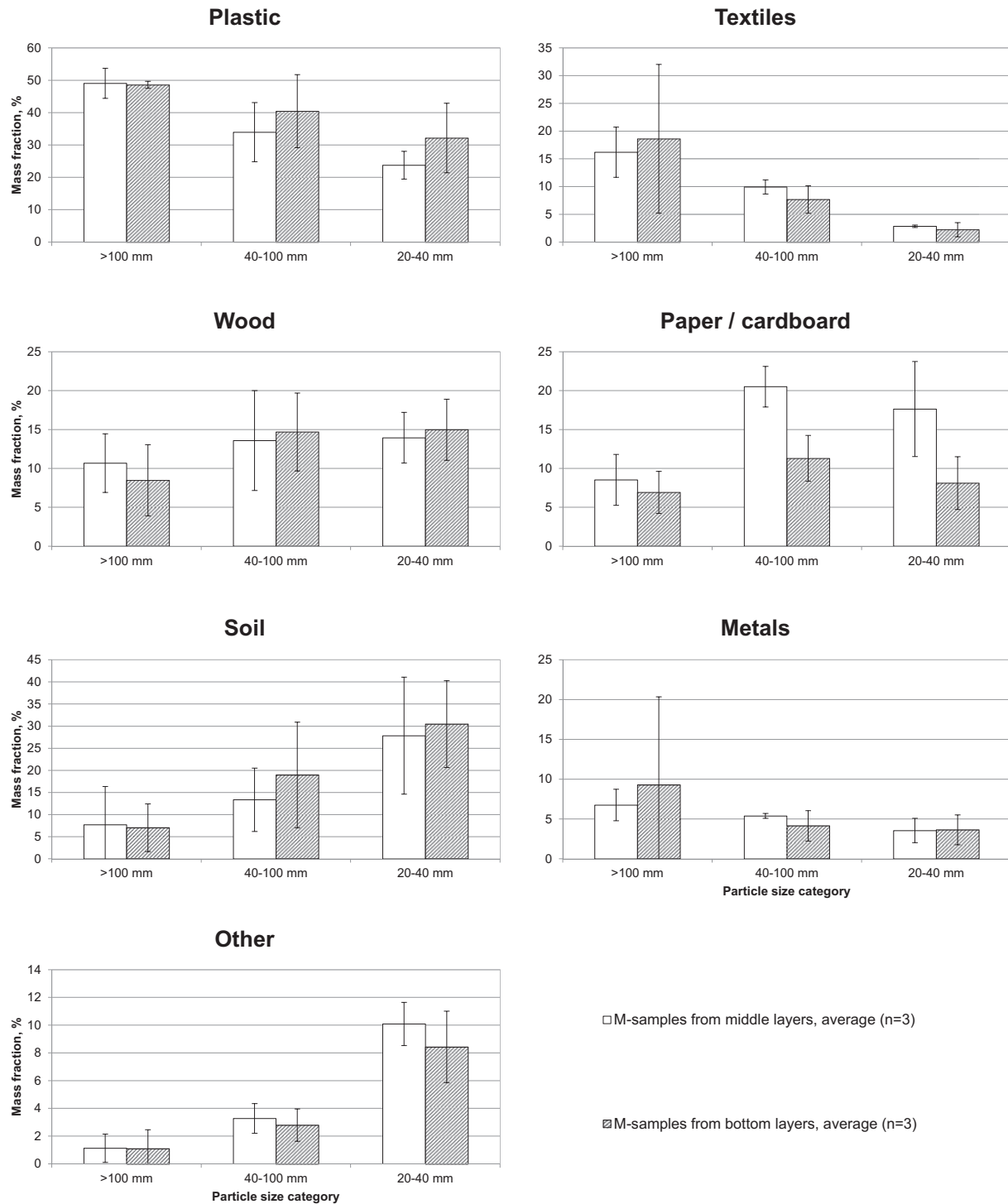
**Fig. 4.** Mass fractions of different waste fractions in particle size categories >20 mm of the M-samples.

similar, although the variations in metal mass fractions particularly are great in some instances. The soil quite naturally increases its proportion in the finer particle size categories consisting of gravel type material in the finer categories and mostly stones in the coarser categories. Most of the soil is likely to remain in the <20 mm material.

The MSW classified as "other" amounted only to approximately 1% (w/w) in the most coarse particle size category, but was present in an increasing mass fraction in finer particle size categories.

Waste within the "other" fraction consisted primarily of ceramics and glass with a lesser quantity of unidentified objects.

### 3.1.3. Full scale trial runs (P-samples)

In the trial runs with the full-scale mechanical pre-treatment, the distributions of materials shown in Fig. 5 were obtained. There were no significant differences in the distributions of the P-samples between the middle and the bottom layers. Both P-samples could be processed by the studied chain of mechanical unit

processes consisting of shredding, magnetic separation, drum sieving, and wind separation of the >70 mm material (described in 2.3.2).

The amount of magnetic metals recovered from the full-scale process was approximately 1% (w/w). To our knowledge the previous studies have not reported the yields of magnetic metals in full-scale operations. In a pilot-stage of a recent landfill reclamation project (Jain et al., 2013) a magnet was in use to separate ferrous metals from the excavated wastes; however, the practice was discontinued due to the poor, non-marketable quality of the separated ferrous metals. No data regarding the mass fraction of the recovered magnetic metals were available.

The amount of fine materials was of the same order of magnitude as was observed for the M-samples in the manual sieving, although a slightly different sieve size was used. In the P-samples, the light fraction of the >70 mm material (28 and 29 mass-% for the middle and bottom layers, respectively) represents the potential fuel fractions recovered in the process. In a Belgian industrial-scale trial a somewhat greater fuel fraction (38% w/w) was recovered from landfill-excavated MSW (Jones et al., in press). Jones et al. (in press) applied wind sieve treatment to material >10 mm in comparison to the >70 mm fraction subjected to wind sieve treatment in the present study, which likely accounts for differences in the fuel fraction yields. The heavy fraction in the >70 mm material as a result of wind sieving was mostly stones.

### 3.2. Calorific fractions

#### 3.2.1. Composition

Tables 2 and 3 show data on the elemental composition (XRF) and preliminary determination of the energy utilization potential of the composite calorific fractions prepared from M-samples >20 mm and P-samples >70 mm from wind sieve overflow. For comparison, the elemental compositions of solid recovered fuels prepared from household wastes in Finland are shown Table 2 (Vesanto et al., 2007).

The present study shows somewhat higher chlorine concentrations in MSW than that reported by Vesanto et al. (2007) for household wastes. Levels of inorganic contaminants (Cr, Cu, Pb, Zn) in the calorific fractions are generally of the same order of magnitude as in household wastes (Vesanto et al., 2007).

Of the elements (Table 2), chlorine is of special interest because it has been limited in European guidance for solid recovered fuels (CEN, 2011) because of its corrosive impact. The levels of chlorine found in the calorific fraction samples vary from 0.59 to 2.1% (w/w) indicating the likely presence of PVC in these samples. The amount of chlorine or other harmful substances does not seem to rise in the M-sample calorific fractions in comparison to the P-samples.
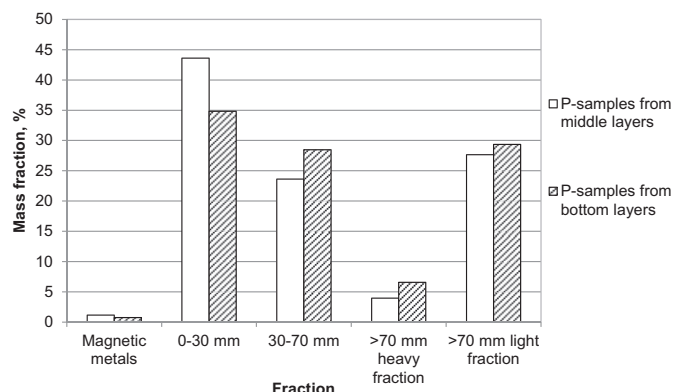
**Table 2**
Concentrations of elements (XRF) of the calorific fractions of the M- and P-samples expressed in percentages (w/w). Standard deviations for the M-samples ($n = 2$–3) are given in parentheses.

| Element | M-samples, >20 mm calorific fractions | | P-samples, calorific fractions (>70 mm wind sieve overflow) | | Solid recovered fuels prepared from household wastes in Finland (Vesanto et al., 2007) |
|---|---|---|---|---|---|
| | Middle layer | Bottom layer | Middle layer | Bottom layer | |
| Al | 1.5 (1.1) | 2.2 (0.35) | 1.2 | 1.5 | – |
| Ba | 0.03 (0.01) | 0.02 (0.01) | 0.06 | 0.05 | – |
| Br | <0.01 | 0.02 (0.01) | 0.01 | <0.01 | – |
| Ca | 3.0 (0.71) | 3.8 (0.14) | 3.8 | 3.9 | – |
| Cl | 1.6 (1.2) | 1.2 (0.0) | 0.59 | 2.1 | 0.81–1.0 |
| Cr | 0.01 (0.0) | 0.01 (0.0) | 0.01 | 0.02 | 0.007–0.016 |
| Cu | 0.04 (0.04) | 0.04 (0.03) | 0.01 | 0.04 | 0.06–0.09 |
| F | <0.01 | 0.06 (0.06) | <0.01 | <0.01 | – |
| Fe | 1.1 (0.80) | 2.0 (0.28) | 1.9 | 2.0 | – |
| K | 0.52 (0.32) | 0.48 (0.13) | 0.52 | 0.52 | – |
| Mg | 0.40 (0.14) | 0.41 (0.01) | 0.35 | 0.34 | – |
| Mn | 0.03 (0.01) | 0.04 (0.01) | 0.04 | 0.04 | 0.009–0.02 |
| Na | 0.71 (0.29) | 0.60 (0.16) | 0.52 | 0.45 | – |
| Ni | <0.01 | <0.01 | 0.01 | <0.01 | 0.003–0.98 |
| P | 0.10 (0.04) | 0.10 (0.02) | 0.08 | 0.1 | – |
| Pb | 0.03 (0.02) | 0.03 (0.02) | <0.01 | 0.03 | 0.0–0.03 |
| S | 0.57 (0.01) | 0.65 (0.04) | 0.9 | 0.71 | 0.11–0.17 |
| Sb | 0.03 (0.02) | 0.04 (0.04) | <0.01 | 0.3 | – |
| Se | <0.01 | <0.01 | 0.02 | <0.01 | – |
| Si | 5.0 (3.9) | 4.1 (1.8) | 3.6 | 3.8 | – |
| Sr | 0.02 (0.01) | 0.01 (0.0) | 0.01 | 0.02 | – |
| Ti | 0.44 (0.03) | 0.36 (0.04) | 0.55 | 0.45 | – |
| Zn | 0.10 (0.06) | 0.06 (0.01) | 0.09 | 0.07 | 0.04–0.05 |
| Zr | <0.01 | 0.01 (0.0) | <0.01 | 0.01 | |
| K + Na | 1.2 | 1.1 | 1.0 | 0.97 | 0.40 |

In general, 85–90% of the chlorine in the waste fed into an incineration process is transferred into the flue gas in the form of hydrochloric acid (Hjelmar et al., 2011). In addition to the corrosive impact, the HCl in the flue gas may further react with and mobilize metals such as Pb and Zn into the flue gas.

#### 3.2.2. Calorific value

The net calorific values (Table 3) correspond to class 1 solid recovered fuel according to standard EN 15359 (CEN, 2011). Similar calorific values were found both for manually sorted and prepared calorific fractions of the >20 mm M-samples and for the light wind sieve fraction of the P-samples. This indicates that there is some fuel potential in the 20–70 mm fractions. To provide perspective, the >70 mm light wind sieve fractions accounted for 28–29% of the mass of the feeds in the full scale process. Manually sorted and built calorific fractions in turn accounted for 40–45% (w/w) on average of all the waste. Comparison of the observed calorific values with results from previous studies, e.g. Hull et al. (2005) and



**Fig. 5.** Mass distribution of the P-samples from full-scale mechanical pre-treatment.

**Table 3**
Calorific values of the calorific fractions of the M- and P-samples. Standard deviations for the M-samples ($n = 2$–3) are given in parentheses.

| | M-samples, >20 mm calorific fractions | | P-samples, calorific fractions (>70 mm wind sieve overflow) | |
|---|---|---|---|---|
| | Middle layer | Bottom layer | Middle layer | Bottom layer |
| Dry matter content, % | 80 (3.5) | 74 (3.5) | 85 | 98 |
| Gross calorific value, MJ/kg dry matter | 24 (1.1) | 24 (0.64) | 26 | 21 |
| Net calorific value, MJ/kg dry matter | 22 (1.0) | 22 (0.62) | 25 | 20 |

Quaghebeur et al. (in press), is not possible as previous work has primarily assessed the calorific content of manually-sorted single waste fractions such as plastics and paper/cardboard.

### 3.3. Fine fractions

#### 3.3.1. Composition

Tables 4 and 5 contain data on the composition and leaching properties of the M-sample (<20 mm) and P-sample (<30 mm) fine fractions. The typical concentration ranges for elements in uncontaminated soil (Chandler et al., 1997) are shown in Table 4 for comparison.

According to the XRF analysis and determination of TOC the M-sample fine fractions contained primarily compounds of Si, Ca, Al, Fe, S, K, Mg, Na and organic matter (TOC). The fine fractions of both M- and P-samples are very similar in terms of XRF-composition and between samples from landfill middle and bottom layers. Slight differences were found in the TOC-contents, which were higher in the P-sample fine fractions. The Cl, S and TOC contents were also lower in the samples from landfill bottom layers, indicating that leaching of Cl and S and degradation of organic carbon had occurred with time. Concentrations of heavy metals cannot be considered high based on these results. There were however elevated concentrations of Cu and Zn in the MSW as compared with the concentrations of these elements typically observed in uncontaminated soils (Chandler et al., 1997). The measured concentrations of Cu, Pb and Zn were similar to those reported for a MSW fine fraction <10 mm excavated from a landfill in Belgium (Quaghebeur et al., in press) and for a Swedish MSW fine fraction <18 mm (Hogland, 2002).

Quaghebeur et al. (in press) reported Fe concentrations between 1.7 and 3.9% (w/w) for the fine fraction (<10 mm) of MSW landfilled between 1980 and 2000, similar to the findings in the present study. Quaghebeur et al. (in press) identified a 0.5 and 5.3% (w/w) magnetic fraction consisting of metals, oxides and other minerals. The proportion of magnetic material within the fine fraction was not assessed in the present study. Magnetic separation of P-samples as part of the full-scale process did not result in reduced Fe concentrations in the remaining fine fraction. This result indicates that either the Fe present within the fine fraction was not in a magnetic form or the magnetic separation process lacked efficiency.

#### 3.3.2. Leaching

The batch leaching tests (Table 5) were performed to produce a preliminary estimate of the landfill acceptability of the fine fractions. To our knowledge no such assessment has been performed in previous MSW landfill mining studies.

Of the studied substances in the batch leaching tests, sulfate, chloride and DOC were the primary leached substances. The leached amounts of the inorganic substances would allow these types of wastes to be disposed of at an EU-level landfill for non-hazardous waste (EC, 2003). However, in both M- and P-samples from the middle layers of the landfill the amount of DOC leached exceeded the criteria for wastes to be placed in an EU-landfill for non-hazardous inorganic waste. The leaching of sulfate at the levels observed most likely indicates the presence of gypsum in the landfilled waste. Gypsum is quite commonly found in wastes such as construction wastes. Besides sulfate, the M-sample fine fractions from the landfill bottom layers comply on average with the leaching criteria set for wastes to be disposed of in an EU-level landfill for inert waste. It must be noted, however, that there are other criteria set for wastes to be placed in landfills for inert waste, such as the mineral oil, PCB, PAH, BTEX and TOC contents of the waste. Of these, only TOC was determined, and the average value for the bottom

**Table 4**
Concentrations of elements (XRF) and TOC of the M-sample fine fractions (<20 mm) and P-sample fine fractions (<30 mm) expressed in percentages (w/w). Standard deviations for the M-samples (*n* = 3) are given in parentheses.

| Element/ substance, % | M-samples <20 mm | | P-samples <30 mm | | Common range for soils (Chandler et al., 1997) |
|---|---|---|---|---|---|
| | Middle layers | Bottom layers | Middle layer | Bottom layer | |
| Al | 5.1 (1.0) | 5.7 (0.31) | 4.9 | 5.5 | 1.0–30 |
| Ba | 0.11 (0.01) | 0.09 (0.03) | 0.08 | 0.10 | 0.01–0.30 |
| Ca | 8.5 (3.9) | 6.5 (1.2) | 6.7 | 6.1 | 0.70–50 |
| Cd | <0.01 | 0.01 (0.0) | <0.01 | <0.01 | <0.001 |
| Cl | 0.32 (0.18) | 0.25 (0.03) | 0.39 | 0.23 | 0.002–0.09 |
| Cr | 0.01 (0.01) | 0.02 (0.01) | 0.02 | 0.01 | 0–0.10 |
| Cu | 0.08 (0.12) | 0.02 (0.01) | 0.02 | 0.03 | 0–0.01 |
| Fe | 3.7 (0.17) | 4.1 (0.21) | 4.5 | 4.0 | 0.70–55 |
| K | 1.6 (0.29) | 1.7 (0.21) | 1.7 | 1.6 | 0.04–3.0 |
| Mg | 1.2 (0.0) | 1.2 (0.06) | 1.1 | 1.2 | 0.06–0.60 |
| Mn | 0.07 (0.01) | 0.13 (0.04) | 0.07 | 0.11 | 0.002–0.30 |
| Mo | <0.01 | 0.01 (0.01) | <0.01 | <0.01 | <0.001 |
| Na | 1.9 (0.21) | 2.1 (0.25) | 1.8 | 1.6 | 0.08–0.75 |
| Ni | 0.01 (0.0) | 0.01 (0.0) | <0.01 | <0.01 | 0–0.05 |
| P | 0.25 (0.13) | 0.28 (0.05) | 0.16 | 0.27 | 0.02–0.50 |
| Pb | 0.01 (0.01) | 0.02 (0.01) | 0.02 | 0.02 | 0–0.02 |
| Rb | <0.01 | 0.01 (0.0) | 0.01 | 0.01 | 0.005–0.05 |
| S | 2.4 (1.5) | 0.96 (0.44) | 1.4 | 0.93 | 0.003–1.0 |
| Se | 0.02 (0.01) | <0.01 | 0.02 | <0.01 | <0.001 |
| Si | 20 (4.0) | 21 (1.5) | 19 | 20 | 23–35 |
| Sr | 0.06 (0.01) | 0.05 (0.0) | 0.04 | 0.05 | 0.005–0.10 |
| Ti | 0.35 (0.01) | 0.38 (0.04) | 0.43 | 0.70 | 0.10–1.0 |
| V | 0.01 | <0.01 | 0.01 | <0.01 | 0.002–0.05 |
| Zn | 0.06 (0.01) | 0.11 (0.04) | 0.08 | 0.08 | 0.001–0.03 |
| Zr | 0.03 (0.01) | 0.03 (0.02) | 0.02 | 0.03 | 0.006–0.20 |
| TOC | 5.8 (1.6) | 4.7 (0.80) | 11 | 6.8 | |

**Table 5**
Batch leaching tests EN 12457-4 for the fine fractions of the M-samples <20 mm and P-samples <30 mm. The leached amounts of substances are expressed in mg/kg dry matter in L/S-ratio of 10 l/kg dry matter. Standard deviations for the M-samples (*n* = 3) are given in parentheses.

| | M-samples, <20 mm | | P-samples, <30 mm | | Leaching limit values for non-hazardous waste EU-landfill (EC, 2003) |
|---|---|---|---|---|---|
| | Middle layers | Bottom layers | Middle layer | Bottom layer | |
| pH | 8.1 (0.10) | 8.3 (0.11) | 7.8 | 8.2 | – |
| Conductivity, mS/m | 280 (68) | 170 (73) | 360 | 160 | – |
| Redox-potential, mV | 340 (1.4) | 370 (10) | 260 | 360 | – |
| Leached substances, mg/kg dry matter (L/S 10) | | | | | |
| As | 0.13 (0.03) | 0.04 (0.03) | 0.12 | 0.09 | 2 |
| Ba | 0.86 (0.24) | 0.76 (0.05) | 0.84 | 1.0 | 100 |
| Cd | <0.005 | <0.005 | <0.005 | <0.005 | 1 |
| Co | 0.12 (0.06) | 0.04 (0.02) | 0.18 | 0.05 | – |
| Cr | 0.03 (0.01) | 0.03 (0.01) | 0.09 | 0.04 | 10 |
| Cu | 0.08 (0.04) | 0.18 (0.12) | 0.05 | 0.06 | 50 |
| Hg | <0.002 | <0.002 | <0.002 | <0.002 | 0.2 |
| Mo | 0.14 (0.02) | 0.30 (0.29) | 0.20 | 0.24 | 10 |
| Ni | 0.47 (0.30) | 0.23 (0.09) | 0.99 | 0.31 | 10 |
| Pb | 0.01 (0.01) | 0.01 (0.0) | 0.01 | 0.02 | 10 |
| Sb | 0.04 (0.01) | 0.05 (0.03) | 0.02 | 0.06 | 0.7 |
| Se | 0.05 (0.02) | 0.02 (0.01) | 0.17 | 0.04 | 0.5 |
| V | 0.02 (0.02) | 0.02 (0.02) | 0.06 | 0.01 | – |
| Zn | 2.4 (1.9) | 1.1 (0.67) | 0.62 | 0.27 | 50 |
| Cl$^-$ | 1200 (820) | 850 (100) | 2400 | 790 | 15 000 |
| F$^-$ | 41 (42) | 3.0 (0.0) | 170 | 13 | 150 |
| SO$_4^{2-}$ | 15 000 (4900) | 7100 (4600) | 18 000 | 6600 | 20 000 |
| DOC | 1000 (320) | 320 (31) | 2400 | 590 | 800 |

layer M-samples was 4.7%, exceeding the criteria of 3% of TOC for an inert waste landfill (EC, 2003).

The leaching of chloride, sulfate and DOC was lower for the M- and P-samples from the landfill bottom layers. These findings are likely due to a longer disposal time at the landfill and previous leaching (and gas formation in the case of organic carbon) of these substances from the waste.

### 3.3.3. Sieving and sorting of the fine fractions

Fig. 6 shows the particle size distribution of two M-sample fine fractions (K41 2–10 m < 20 mm and K41 10–22 m < 20 mm) and two P-sample fine fractions (<30 mm). Three of the four samples had similar particle size distributions, with 44–57% (w/w) of the material with particle size <4 mm. Similar results were reported by Hull et al. (2005) who found approximately 50% (w/w) of the fine fraction (<25.4 mm) to be less than 4.8 mm in particle size. The P-sample from the middle MSW layer had substantially less <1 mm material and a greater proportion of material within the 10–20 mm size category compared to the other samples. In general there were no clear signs with respect to the concentration of the fine fractions in any given particle size category.

For the K41 10–22 m, <20 mm M-sample and for both P-sample fine fractions the 4–10 mm, 10–20 mm size categories, along with the 20–30 mm size category for the P-samples, were hand-sorted in the laboratory using the same classification as was applied in the field. Combining results from different categories yielded the mass distributions shown in Fig. 7.

Soil was the most abundant fraction identified by manual sorting of the MSW fines. Visual inspection indicated that the soil fraction was particularly dominant within the <4 mm material. The quantities of all other waste fractions remained low in the manually sorted samples. Recognizable metals were negligible in the manually-sorted MSW, whilst a greater proportion of material was classified "other", e.g. glass and ceramics (see 2.3.1).

### 3.4. Implications

#### 3.4.1. Processability

The landfilled waste was even surprisingly well processable by full-scale mechanical processes, including shredding, magnetic separation, screens and a wind sieve, despite the moisture and impurities in the landfilled waste. The moisture contained in the materials did not impede their actual processability, but it may have affected the separation efficiency. The MSW did not cause e.g. clogging in the used unit processes. For the full-scale process, the masses of sampled MSW (20 and 22 t), were however quite low. The real full-scale treatability would have to be assessed using hundreds of tons of MSW.
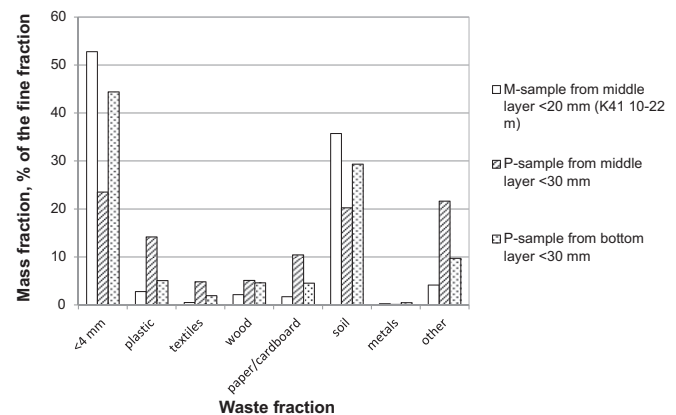
**Fig. 7.** Distribution of mass of one M-sample fine fraction (K41 10–22 mm) and P-sample fine categories into different waste fractions.

As stated earlier, the full-scale mechanical processes were not optimized for MSW excavated from landfills. Comparison between the results from the manual sorting and the mechanical treatment highlights the need to plan and optimize the mechanical full-scale processes based on a manual sorting study.

Data on mass fractions of different waste fractions within different particle size categories such as shown in Fig. 4 can in combination with the particle size distribution be used to estimate applicable processing schemes for different materials of different particle sizes. Although manual classification is a subjective means of sorting in which results may vary, it provides valuable information concerning waste composition that can be used to devise suitable treatment steps to recover the desired fractions.

Based on further sieving of the finer fractions (Fig. 6), it was possible to extend the particle size distributions found in the field to cover the lower end of the distribution (Fig. 8). It must be noted, however, that the finer parts of the samples (<20 and 30 mm) were sieved after drying, whereas the coarser sieving was performed in the field with non-dried samples.

The particle size distributions of M- and P-samples are somewhat uniform and converge toward coarser particle sizes. The distributions have more differences in the fine end of the distributions. This can indicate different sieving efficiencies in the manual and mechanical processes as well as possible loss of the fine grained material in sample transport to the full scale machinery in the field.

As an example for possible planning of a treatment scheme, in the light of the results of the present study, the separation of particle size category 20–40 mm would seem unnecessary in a full-
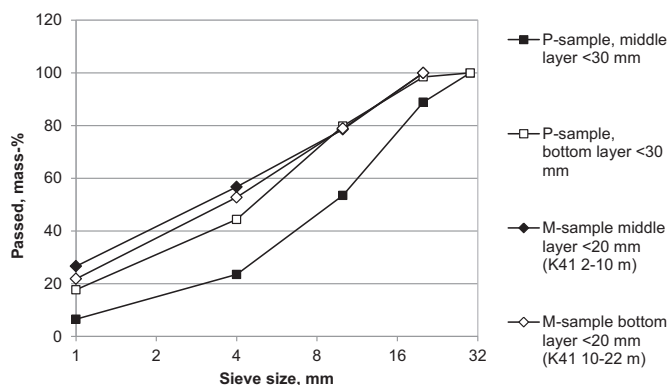
**Fig. 6.** Distribution of the fine fractions into different particle size categories.
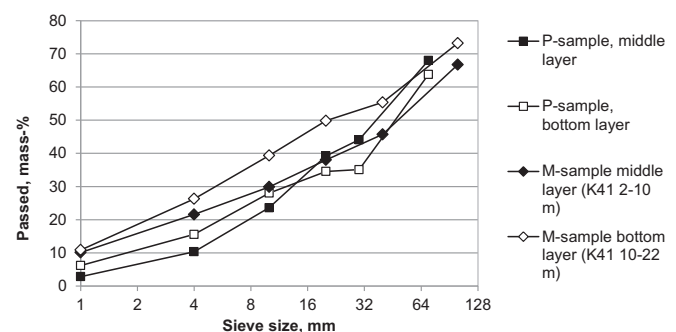
**Fig. 8.** Full particle size distribution of M-samples from sampling point K41 and the P-samples.

scale process. The 20–40 mm material amounts to an average mass fraction of 6.4% in the middle layers and 5.9% in the bottom layers, and it could be combined with the particle size categories 0–20 mm or 40–100 mm. Relatively high share of combustibles (59 and 57% w/w in the M-samples from middle and bottom layers respectively) favors combining the 20–40 mm material with the fractions subject to recovery of SRF.

### 3.4.2. Calorific fractions

Both the manual sorting and the full-scale mechanical treatment produced calorific fractions with relatively high calorific value of 20–25 MJ/kg (net). The levels of chlorine (0.59–2.1%, w/w) would, in practice, likely hinder the use of these types of fuels in the best quality solid recovered fuel classes.

The presence of chlorine at such elevated concentrations can also be attributed to the age of the landfilled waste. As discussed in Section 3.1.2, this type of young landfill is likely to yield a higher plastic content than would be the case with older landfills. This also results in the high chlorine content found in the calorific fractions. Older landfills may yield lower plastic and chlorine content in the calorific fractions, but the mass fraction of the calorific fraction and the calorific values of these fractions would also likely be lower.

In the light of the results from manual sorting, it would be feasible to recover fuel fractions from particle sizes >20 mm instead of only >70 mm, which was the current practice with the full-scale equipment. In practice, challenges would be most likely faced in trying to separate calorific fractions of different weight and different particle size from waste streams. Current knowledge on separation technologies for the production of solid recovered fuels favors the limiting of the particle size range in the separation process to optimize the sorting effect (Rotter et al., 2004).

### 3.4.3. Metals

As mentioned earlier, all the waste fractions contained impurities. This is especially true for the metal fractions from manual sorting, in which a typical metal piece was often composed of the metal and a plastic or textile coating. Obviously, the impurities create some challenges for the further utilization of the metal waste fractions. In the present study, the quality of the metal fractions was not further studied. Metals mined from landfills have been previously considered to be of poor quality in general, which would render them non-competitive with metals recovered from sources such as incinerator ashes (Savage et al., 1993).

The amount of magnetic metals recovered from the full-scale process, approximately 1% (w/w), was smaller than the amount of total metals found by manual sorting (3–4%). Magnetic metals, however, that were recovered by the full-scale process were in a cleaner state by visual inspection than the average metal fractions found by manual sorting.

### 3.4.4. Fine materials

Previous studies report little information concerning the division of MSW fine-textured material into different waste fractions. The results of the present study show little potential benefit from further refining of the fine fraction, e.g. for production of fuel. Low relative proportions of plastics, paper/cardboard, textiles and wood (Fig. 7) are indicative of poor potential heating value in MSW fines, as has been previously reported (Hogland et al., 1995; Quaghebeur et al., in press). Of interest for possible recovery in the fine material are Al and Fe. The concentration of Al amounts to 5.1% (w/w) on average in the middle layer M-samples and 5.7% in the bottom layer M-samples. Corresponding Al concentrations for the P-sample fine fractions were 4.9% for the middle layer and 5.5% for the bottom layer. These concentrations in the fine fractions alone would yield approximately 2–2.5% of Al in the landfilled waste. These results do not distinguish between metallic and non-metallic Al, the shares of which would have to be assessed before further consideration of recovery. The recovery of metallic Al could be accomplished using techniques such as eddy current separation, which is traditionally found to be quite challenging for fine particle sizes. Developments in eddy current separation technology may be able to improve the recovery from the fine fractions as well (Roy et al., 2012; Settimo et al., 2004).

To the authors' understanding the most likely scenario at least on EU-level is that the rejects left from landfill mining are classified as wastes and thus the landfill disposal of the fine fraction or at least some non-utilizable fractions of it is a probable scenario. In the leaching tests to assess the landfill acceptability, the fine materials exhibited generally non-hazardous behavior. Leaching of DOC could however be critical in view of the landfill acceptance for an EU-level landfill. This is most likely a challenge of younger MSW deposits, where most of organic matter is yet to degrade.

Hydro-mechanical treatment of the fine fraction as proposed by Bilitewski et al. (1995), among others, could be applied to reduce the organic matter content and separate more contaminated fractions from the fine fractions. Bilitewski et al. proposed a two-stage treatment where in the washing stage, heavy inert material is separated from the floating fraction, which consists primarily of plastic and wood. In the latter stage, the suspension obtained is separated into a fine inert fraction and into sludge, in which organic substances would be likely to be more concentrated. With such treatment, the easily soluble salts such as sulfate would likely leach out, also improving the environmental properties of the re-landfilled material.

## 4. Conclusions

The present study produced hands-on knowledge of the treatability of landfilled MSW in a full-scale mechanical process consisting of shredding, magnetic separation and drum-sieving followed by wind sieving of the coarsest material. The mechanical treatment produced approximately 30% (w/w) of possible solid recovered fuel, which was dominated by plastics. The mechanical processes were not optimized for the very purpose, and the impact of factors such as moisture of the waste on the sorting efficiency could not be measured. The results from manual sorting suggest that up to 40–45% (w/w) is possibly available for the production of SRF by more advanced process configuration, e.g. extending the wind sieve treatment for finer materials.

Mechanical treatment produced 1 mass-% of magnetic metals, which is likely a more realistic estimate of the achievable amount of metals than the 3 to 4 mass-% found by manual sorting. This is likely due to challenges in recovering mixed metal fractions containing materials such as textiles and plastics attached to metal particles.

The fine particle sizes (<20 mm in the manual sorting and <30 mm in the mechanical process) contribute to approximately 50% (w/w) of the materials sampled and consist not only of the landfilled waste but also of the soil materials used for landfill intermediate covers. Preliminary assessment of landfill acceptability of the fine fractions found, in general, non-hazardous behavior in terms of leaching, especially for the older wastes. Leaching of DOC from the fine fractions, especially from younger MSW, may be challenging for the landfill disposal in EU. Further research is needed about the possible treatment of the fine fractions to remove the organic matter and to recover metals such as Al and Fe.

The methodology to sample and characterize the wastes from a MSW landfill used in this study can be applied at possible landfill mining sites. The gained results are site-specific and cannot be generalized. The sampling technique produced samples of sufficient size and did not probably change the properties of the waste

markedly. The manual sorting provides necessary information regarding the best suitable techniques for material recovery from different fractions. The sampling methodology applied is however a quite massive operation, which can be justified by the heterogenous nature of MSW. The costs of drilling at this scale may become unfeasible especially in a small scale landfill mining project. Thus there exists a need to further develop less costly approaches to characterize potential landfill mining sites.

## Acknowledgments

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