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Resource and Climate Implications of Landfill Mining: A Case Study of Sweden

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Abstract

This study analyzes the amount of material deposited in Swedish municipal solid waste landfills, how much is extractable and recyclable, and what the resource and climate implications are if landfill mining coupled with resource recovery were to be implemented in Sweden. The analysis is based on two scenarios with different conventional separation technologies, one scenario using a mobile separation plant and the other using a more advanced stationary separation plant. Furthermore, the approach uses Monte Carlo simulation to address the uncertainties attached to each of the different processes in the scenarios. Results show that Sweden's several thousand municipal landfills contain more than 350 million tonnes of material. If landfill mining combined with resource recovery is implemented using a contemporary stationary separation plant, it would be possible to extract about 7 million tonnes of ferrous metal and 2 million tonnes of non-ferrous metals, enough to meet the demand of Swedish industry for ferrous and non-ferrous metals for three and eight years respectively. This study further shows that landfill mining could potentially lead to a onetime reduction of about 50 million tonnes of greenhouse gas emissions (CO₂-equivalents), corresponding to 75% of Sweden's annual emissions.

Key words: landfills, resource recovery, recycling, metals, environmental assessment

Introduction

In order to understand causes of environmental problems and develop sustainable strategies, studying the societal metabolism is essential (Lifset and Graedel, 2002). Such research has shown increasing material stocks in society causing waste generation and pollutant emissions over time (Bergbäck et al., 2001; Krook et al., 2006; Tukker et al., 2006). These accumulations could however also be seen as resource reservoirs for future recovery (Brunner and Rechberger, 2004; Elshkaki et al., 2004; Wittmer and Lichtensteiger, 2007). The exploration of such an approach is still in an early phase involving research on the magnitude, spatial distribution and discard rates of material stocks in the built environment (e.g. Lifset et al., 2002; Kapur and Graedel, 2006).

Landfill mining refers to the excavation, processing, treatment and/or recovery of deposited materials. About fifty such initiatives have been reported, mainly in the U.S. but also in Europe and Asia (Hogland et al., 2004). Most of them have involved pilot studies, exploring landfill mining as a way to solve landfill management issues (Cossu et al., 1996; Krogmann and Qu, 1997). Although effort has been spent on extracting deposited resources, primarily landfill cover material and in a few cases waste fuel and metals as well, such constituents have in many cases been secondary to objectives such as land reclamation, conservation of landfill space,

facilitating final closure and remediation (Spencer, 1990; Dickinson, 1995; Cha et al., 1997; EPA, 1997; Van der Zee et al., 2004). In several cases, landfill mining has in fact been economically justified solely by extending the service life of existing deposits and thereby avoiding re-siting a new landfill (Krook et al., 2012). Typically, mobile excavation, screening and separation equipment have been employed, often demonstrating moderate performance in obtaining marketable recyclables other than landfill cover material and waste fuel (Savage et al., 1993; Fisher and Findlay, 1995; Krogmann and Qu, 1997). Recently, however, several studies emphasizing resource recovery as a motivation for landfill mining have been published (Zanetti and Godio, 2006; Baas et al., 2010; Jones et al., in press; Krook et al., 2012). These studies partially signal a shift in perspective and approach within landfill mining research by analyzing the conditions for resource recovery as a main driver for implementation, and doing so by applying more advanced material processing and energy conversion technologies.

Several worldwide changes are currently underway, which in turn might create stronger incentives for resource conservation, e.g. increasing competition for natural resources and raw material prices (cf. Kapur, 2006 and Halada et al., 2009). Research studies also show that large amounts of valuable materials such as metals are situated in landfills and other waste deposits (Lifset et al., 2002; Spatari et al., 2005; Müller et al., 2006). Furthermore, in Sweden waste management has gone through dramatic changes and less than 1% of municipal solid waste is currently landfilled, while about 50% is used for energy recovery (Swedish Waste Association, 2010). This restructuring has led to hundreds of landfills being taken out of use in recent years (Swedish Waste Association, 2003), suggesting future needs for final closure materials. Sweden has over four thousand municipal landfills (SEPA, 1986), a large majority of which are old, closed down without the appropriate pollution prevention and control techniques, and in extensive need of remediation.

In the past, landfill mining seldom involved any real attempt to achieve efficient recovery of deposited materials. Resource implications of such initiatives have generally therefore been disregarded. Emphasis has been on possible movement of hazardous substances or formation of explosive gases during excavation (e.g. Krogmann and Qu, 1997; Hogland, 2002; Zhao et al., 2007). Although this has been a necessary approach in the past, it is also insufficient because environmental impacts could arise on different spatial and temporal scales (Udo de Haes et al., 2000; Finnveden and Moberg, 2005). For instance, energy recovery of combustible materials and recycling of metals from landfills show potential for mitigating global warming and the various kinds of pollution related to conventional energy generation and mining activities (Ayres, 1997; Cohen-Rosenthal, 2004; Svensson et al., 2006; SEPA, 2008). However, this potential relies on many factors such as the composition of landfills, efficiency of separation technologies, and resource demands from excavation, processing, transportation and recycling processes. Since landfill mining research has so far only dealt with local issues, there is a knowledge gap regarding how such initiatives could affect the environment when it comes to global and regional impacts.

This study aims at implementing the approach described in Frändegård and colleagues (in press) on a national scale in Sweden. In doing so, two scenarios for resource recovery from landfills are developed and evaluated in regards to resource and climate change mitigation potential. The aim has been divided into the following three research questions:

- What quantities of different materials are currently situated in Swedish municipal landfills?
- How much of these deposited material stocks could be extracted for material recycling and energy recovery by conventional technologies?

- What are the resource and climate implications of implementing such resource recovery through landfill mining in Sweden?

Method

Implementation of landfill mining can be done in various ways. Different technologies can be used and the amount of material separation may differ between different projects. There is also limited access to detailed data for extraction and separation of valuable materials from landfills (Krook et al., 2012). This introduces two different kinds of uncertainties, what Huijbregts and colleagues (2003) define as “scenario uncertainties” and “parameter uncertainties.” Scenario uncertainties are comprised of all the different uncertainties which are introduced with the different assumptions and choices made in order to build the different scenarios, e.g. type of separation technology. Parameter uncertainties are related to how individual parameters can vary, e.g. how much ferrous metal is deposited in the landfills. To evaluate a complex and yet largely unproven concept such as landfill mining, it is important that the analysis can take all the parameter uncertainties into account. In this study, an approach has been chosen using scenarios with different methodological choices, and using Monte Carlo simulations (MCS) to analyze how uncertainties influence the results. MCS is useful for sensitivity analysis of a system consisting of a large number of variables where every variable has its own distribution of values. For more information about the approach, see Frändegård and colleagues (in press). The simulation works by generating numerous randomized result samples. These samples are then aggregated into a probability distribution curve which can be graphically presented and analyzed. For a more in-depth description of MCS, see for example Metropolis and Ulam (1949) or Kalos and Whitlock (2008). The analytical approach of this study is divided into three main phases; *scenario development*, *data collection* and *results from simulation*, figure 1.

Figure 1. The figure shows the overall method used for this study, going from top to bottom. The study begins with the developing and structuring of the two scenarios. The second phase is where the data needed for simulating the processes in the scenarios are collected. The third and final phase is where the Money Carlo Simulation is used to produce different sets of results.

Scenario development

Two base scenarios are used to study the potential of resource recovery through landfill mining in Sweden. The first scenario is the mobile plant scenario, which is based on a separation plant placed directly at the landfill. The second scenario uses a more advanced stationary separation plant. Since a majority of the landfills are old and potentially environmentally hazardous,

remediation, final closure and installation of landfill gas collection systems have been included in both scenarios.

Mobile plant

This scenario is characterized by simplicity; the plant should be transportable and separate materials with minimal time and set-up requirements. Together with a panel of recycling experts from Stena Metall AB, an international recycling company that has previously conducted landfill mining pilot studies, an existing mobile plant was used which consisted of five processes: star screening; air classification; magnetic separation; eddy current separation (ECS) and sensor-based sorting (figure 2).

Figure 2. Overview of the mobile plant scenario showing processes, material flows and separated material categories. Boxes with solid lines indicate different types of separation processes, while boxes with dotted lines indicate the valorization or treatment of each separated material category. Estimated transport distances for the longer transport of recovered materials to recycling/treatment facilities are also shown in the figure, while internal transport taking place at the landfill sites are set to 10km+/-5. Non-processed non-ferrous and ferrous metals are denoted in the figure as NP Non-Fe and NP Ferrous, respectively. ECS = eddy current separator.

In this suggested process, the deposited material is first excavated and dumped over a coarse screen, which separates out bulky hazardous products (e.g. refrigerators and oil barrels) and non-recyclable material. Next, the rest of the material enters the star screen, which separates out a material category for re-deposition at the landfill sites called “fines,” containing heavily degraded waste and cover material. Finally, the air classifier separates out combustibles such as paper, textiles and plastic, while the magnet, ECS and sensors extract ferrous and non-ferrous metals respectively. The recycling company, however, also pointed out that the metals produced by the mobile plant are seldom clean enough for recycling; hence, prior to recycling

these metals must be transported to a stationary metal processing plant for further refining. Transport distances in the scenarios are based on Swedish conditions and a probability distribution of $\pm 50\%$ is used. Sweden has 32 waste incineration plants (Swedish Waste Association, 2012) distributed across the country, which means that in most cases there is an incineration plant within 100 kilometers (km)¹ from the landfill sites. Stationary separation plants and material recycling plants are less common, hence these transport distances have been estimated to 300 km and 500 km respectively. There are about 110 facilities for disposal of hazardous material in Sweden, but since many of them have different specializations, the transport distance has been set the same as for combustible material, i.e., 100 km (Swedish Waste Association 2005). Construction material, i.e., heavily degraded waste and inorganic materials, due to cost constraints, is typically transported to a nearby construction site, e.g., a road construction site or construction at the landfill, and the transport distance is therefore set to 100 km.

Stationary plant

The plan behind this scenario was to make use of state-of-the-art technologies, with emphasis on collecting as much material for recycling as technically possible. The panel of recycling experts therefore designed a fairly sophisticated and ambitious option, which involves transporting approximately half of the excavated material to a stationary processing plant, figure 3. The stationary plant can be thought of as a combined ferrous and non-ferrous processing plant. At the landfill, the deposited material is first excavated and sent through a coarse screen into a star screen, which is placed at the landfill in order to reduce the transport of soil-type materials from the landfill to the stationary plant. These mobile processes also collect bulky hazardous products (e.g. refrigerators and oil barrels) and residue material and a large share of the heavily degraded waste (fines). The hazardous material is transported from the landfill to a disposal facility. The rest of the material is then transported by truck to the stationary plant, which separates five usable material categories: construction material; combustibles; non-ferrous; ferrous and plastics, and one category of residual material which is deposited at the stationary plant's landfill. Four of these five usable material categories are transported by truck to different recycling plants, while the combustible material category is transported to the local waste incineration plant. Transport distances vary depending on material type, figure 3.

Figure 3. Overview of the stationary plant scenario showing processes, material flows and separated material categories. Boxes with solid lines indicate different types of separation processes, while boxes with dotted lines indicate the valorization or treatment of each separated material category. Estimated transport distances for the longer transports of recovered materials to recycling/treatment facilities are also shown in the figure, while internal transports taking place at the landfill sites are set to 10km+/-5. Non-processed ferrous is denoted in the figure as NP Ferrous.

Data collection

In order to estimate the environmental potential for landfill mining in Sweden in regards to greenhouse gas (GHG) emissions (measured in CO₂-equivalents), the starting point of this analysis is to find out the total quantity of different materials located in the country's municipal landfills.

Characterization of Swedish landfills

In order to establish how much material is situated in Sweden's more than four thousand municipally-owned landfills, historical surveys and statistics on deposited amounts of waste were used (Linköping Municipality, 2008; SEPA, 1986; 1990; The Swedish Waste Association, 1997; 2003; 2007). The material composition of this deposited stock was then assessed by reviewing several previous landfill mining cases. This process resulted in data from a total of 13 landfills from the industrialized part of the world. The different materials that were found in the cases were aggregated into ten deposited material types: soil; paper; plastic; wood; textiles; inert materials; organic waste; ferrous metals; non-ferrous metals and hazardous materials. For each type, the mean value and standard deviation was calculated and normalized (i.e., make the sum of the mean values add up to 100%), table 1.

Table 1. Estimated material composition (in weight %) of Swedish municipal landfills, presented as mean values and absolute standard deviations.

<i>Material type</i>	<i>Mean value (%)</i>	<i>Abs. std.dev. (%)</i>
Soil	56.3 ^a	14.2 ^a
Paper	7.9 ^a	6.1 ^a
Plastic	8.1 ^a	5.4 ^a
Wood	7.4 ^a	4.3 ^a
Textiles	3.3 ^a	1.3 ^a
Inert materials	9.7 ^a	10.8 ^a
Organic waste	2.7 ^a	2.0 ^a
Ferrous metals	3.6 ^{a,b}	4.1 ^{a,b}
Non-ferrous metals	0.8 ^{a,b}	4.1 ^{a,b}
Hazardous	0.2 ^a	0.1 ^a

^a Based on the following landfill mining cases: Cossu et al. (1995); Hogland et al. (1995); Hogland et al. (2004); Hull et al. (2005); Krogmann & Qu (1997); Rettenberger (1995); Richard et al. (1996); Stessel & Murphy (1991); Sormunen et al. (2008).

^b For ferrous and non-ferrous metals, a special case had to be made, since only a few of the landfill mining cases made a distinction between these two material types; a majority of the cases had just one aggregated material type called “metals”. The mean values for ferrous and non-ferrous metals are therefore based on the fact that the accumulated consumption of metals in Sweden over time is around 80 percent ferrous and 20 percent non-ferrous, so the mean values for these two material types were calculated proportionally (SEPA, 1996).

Given that the size, age, composition and location of municipal landfills often vary considerably, a full-scale implementation of landfill mining in Sweden might not be a realistic strategy, at least not in the foreseeable future (cf. Van der Zee et al., 2004 and Van Passel et al., 2012). Most of the municipal landfills in Sweden are old and thereby lack appropriate pollution prevention and control techniques. These old landfills are normally quite small in size, and more than 80% contain less than 50,000 tonnes (metric tons)² of material. The newer landfills, often initiated during the 1960s and 1970s, are larger and have successively been equipped with state-of-the-art environmental technology. Of the newer landfills, 95% consist of more than 50,000 tonnes of material and 40% contain more than one million tonnes (SEPA, 2012a).

In principle, there are two possible drivers for landfill mining: environmental or commercial. The former deals with pollution prevention and policies and is largely driven by governmental interests. Commercial landfill mining, on the other hand, can be used for recovery of valuable materials, extension of landfill lifetime or land reclamation (Krook et al., 2012). In Sweden, a methodology called methodology for inventorying and risk classification of contaminated sites (MIFO) is used to determine the environmental conditions and risk of contaminated sites, such as landfills, and indicates whether the site has *small environmental risk* (class 4), *moderate environmental risk* (class 3), *high environmental risk* (class 2) or *very high environmental risk* (class 1) (cf. SEPA, 2002). The class for a specific site depends on four factors; *level of contamination*; *level of hazardousness of contaminants*; *risk of contamination spreading* and the *area's sensitivity and conservation value*. In order to establish the environmental potential of realizing landfill mining through the two different rationales, i.e., environmental or commercial, the total amount of landfills and hence the amount of material was divided into four different groups: *total potential*, *large landfills*, *remediation needed* and finally *remediation required*, table 2.

Table 2. The four different groups of landfills, based on weight (in million metric tonnes) and MIFO class. Total potential consists of all of Sweden's municipal landfills. Remediation required contains only landfills with the highest possible MIFO class of 1. Remediation needed contains landfills with a MIFO class of 1 or 2. Large landfills include landfills with more than one million metric tonnes of material. Calculations are based on Linköping Municipality (2008); SEPA (1986; 1990) and The Swedish Waste Association (1997; 2003; 2007).

<i>Landfill group</i>	<i># landfills</i>	<i>Weight (Mt)</i>
Total potential	>4,000	365
Remediation needed	432	203
Large landfills	49	108
Remediation required	44	20

Remediation required contains only landfills with a MIFO class of 1, i.e., landfills with a very high environmental risk that should be dealt with as soon as possible. **Remediation needed** is defined as landfills having either a MIFO class of 1 or a MIFO class of 2, which would suggest that some sort of remediation must be done within the near future. Landfill mining for both of these groups is mainly environmentally-driven, even though there might be commercial interests as well, e.g., excavation of metals. **Large landfills** includes sites consisting of more than one million tonnes material which, according to Stena Metall AB, a large recycling company and a potential landfill mining practitioner, would likely be the minimum size for initiating a landfill mining project solely on commercial grounds.

Resource use of processes

The scenarios consist of a number of processes, each using varying levels of material and energy resources. Hence, data on the resource use for each of these processes, along with their respective uncertainty distributions, was collected from several different sources. For resource use by excavation, incineration, recycling, transport and remediation processes in the developed scenarios, generic data were acquired from the LCA database Ecoinvent (Frischknecht & Rebitzer, 2005). Specific data for the resource use of the material separation processes were gathered from the recycling company Stena Metall AB. This included diesel usage for the separation processes included in the mobile plant and the electricity needed to run the large separation plant.

Assessing the extractable share of deposited materials

The efficiency of material recovery depends to a large degree on which type of separation process is used in each scenario. In order to establish the separation efficiencies for the two scenarios, the expert panel from Stena Metall AB was consulted. This company has vast experience in the field and has also conducted its own pilot studies concerning how much of a certain type of material can be extracted and separated out from a landfill. The scenarios are, as explained above, based upon ten types of deposited materials and the objective was to determine the material category into which these different materials are separated. In cooperation with the expert panel, mass balances for each type of material were compiled, describing the distribution, in weight percentages, of each of these materials between the different separated material categories, see table 3. Composite materials are assumed to be either separated in the shredder plant (in the stationary plant scenario) and sorted into the correct material category or to be sorted as residual materials. The number of separated material categories differs between the two scenarios; the mobile plant scenario has six separated material categories while the stationary plant scenario has eight.

Table 3. Illustration of how the mass balances between the deposited material types and the separated material categories have been calculated. The real figures are not presented due to confidentiality. Instead, the separation efficiencies are presented in a more aggregated, less sensitive form in the results. Cat.1, Cat.2, etc. refers to the separated material categories, for example combustible materials, ferrous metals and fines. The number of separated material categories differs between the two scenarios; the mobile plant scenario has six separated material categories while the stationary plant scenario has eight.

Material type	Separated material category								
	Cat.1	Cat.2	Cat.3	Cat.4	Cat.5	Cat.6	Cat.7	Cat.8	
Soil	a1 (%)	a2 (%)	a3 (%)	a4 (%)	a5 (%)	a6 (%)	a7 (%)	a8 (%)	$\sum_{i=1}^8 a_i = 100 \%$
Paper	b1 (%)	b2 (%)	b3 (%)	b4 (%)	b5 (%)	b6 (%)	b7 (%)	b8 (%)	$\sum_{i=1}^8 b_i = 100 \%$
Plastic
Wood									
Textiles									
Inert materials									
Organic waste									
Ferrous metals									
Non-Fe metals									
Hazardous									

Assessing greenhouse gas emissions

Every process in the scenario uses resources, which in turn adds emissions to the environment. In order to calculate the environmental pressures of the resource use for the different processes, emission factors derived from the Ecoinvent databases were used (Frischknecht & Rebitzer, 2005). Each emission factor is accompanied with a standard deviation and an uncertainty distribution. Carbon dioxide emissions from biogenic sources have been treated as zero in this study. Landfill gas emissions from re-deposited organic matter were calculated by obtaining carbon content and material degradability rates from the Ecoinvent database on landfills (cf. Doka, 2007). Other GHG emissions which could arise during excavation when the waste is open both for air and water penetration have not been included in the study.

The separated material categories were modeled to be either incinerated with energy recovery, material recycled or re-deposited in the landfill. Since energy recovery and recycling of materials replace conventional energy generation and virgin material production, an avoided burden approach has been used to calculate GHG emissions from these processes (ISO, 2006). The concept of “avoided burdens” can be described as the environmental impacts associated with, for instance, the virgin production of materials which do not occur when substituted by the introduction of recyclable materials. If these avoided emissions outweigh the added emissions from the recycling process, avoided burdens are the result.

Since the study is based on a system level corresponding to the national scale of Sweden, the GHG emissions from the conventional energy generation were based on “Swedish medium voltage electricity mix” from Ecoinvent. Sweden’s electricity mix consists of about one-third oil and gas, one-third nuclear power and one-third renewable energy sources, mainly hydropower (European Commission, 2007). Furthermore, Sweden has a large amount of district heating plants compared to many other countries and landfilling combustible materials

has been banned since 2002, which leads to the assumption that in the scenarios separated combustible materials would be incinerated in a combined heat and power plant, with a typical ratio between produced heat and electricity of 9:1 (Swedish Waste Association, 2010). The moisture content of the combustibles was set to 30 weight percent (based on Cossu et al., 1995; Hogland et al., 1995; Obermeier et al., 1997; Nimmermark et al., 1998; Hogland et al., 2004). Ranges for gross calorific values for each material making up the separated combustible material category, retrieved from the Ecoinvent database, were used to estimate the total amount of electricity and heat that could be generated from the combustible materials in the landfills.

In order to calculate the avoided emissions relating to landfill gas leakage, the leakage that would otherwise occur was compared to when the landfill gas collection system is put in place through the remediation process. Since Swedish landfills generally are old, the organic material in the landfills was estimated to have decomposed to a large degree and already released between 50-100% of its landfill gas.³ Landfill gas emissions are calculated based on carbon content and material composition rates from the Ecoinvent database (cf. Doka, 2007). In both scenarios the collected landfill gas is flared since the amount of landfill gas collected is estimated to not be large enough for commercial use.

Finally, the net emissions for each process are calculated by subtracting the avoided emissions from the added emissions.

Monte Carlo simulations

The results are based on a MCS with 50,000 runs, i.e., the simulation was run 50,000 times and for each run, new random samples for all input parameters were generated. Input parameters in the simulations included material composition, separation efficiencies, emission factors and transport distances. While working with the model, the authors came to realize that the model's rate of convergence was quite fast, despite more than 300 input parameters, and the result charts for, e.g., GHG emissions acquire largely the same form after only a few hundred runs. The reason for the large number of runs in the simulation was to account for extreme values and to produce smoother result charts.

When the simulation was finished, all of these samples were aggregated and presented as cumulative probability distribution charts. The results consist of net GHG emissions for each scenario, total material stocks situated in the municipal landfills and how much of this can be extracted, net GHG emissions for mining the four different groups of landfills discussed earlier (total potential, remediation needed, large landfills and remediation required) and what processes contribute the most to added and avoided GHG emissions.

Results

Deposited and extractable share of material stocks in Swedish municipal landfills

The total amount of deposited materials in Swedish municipal landfills as of 2012 is estimated to be 365 million tonnes. The stationary plant scenario extracts and separates about 80 weight percent (wt%) of the total amount of materials in the landfill, which corresponds very well with the efficiencies stated in Van Passel and colleagues (2012). The mobile plant scenario is able to extract 60 wt%. On average, 20 wt% of the materials in the stationary plant scenario is redeposited, and about 40 wt% in the mobile plant scenario respectively, figure 4.

Figure 4. Estimated total amount and extractable share of each deposited material category (in million tonnes) for each scenario. Based on a 50,000-sample Monte Carlo simulation.

Two-thirds of the landfill material consists of potential landfill construction material, i.e., heavily degraded waste and inorganic materials used to construct roads or provide landfill cover. In the scenarios, this material is estimated to mainly be used as cover material or road construction material. In the stationary plant scenario almost 90 wt% of the landfill construction material is extractable, compared to slightly less than 80 wt% in the mobile plant scenario, similar to the proportions given by other studies (cf. Savage et al., 1993).

The remaining one-third mainly consists of combustible materials, such as paper, non-recyclable plastics, wood, textiles and organic waste, but also recyclable plastics and ferrous and non-ferrous metals. Occurrence of hazardous waste is generally far below 1 wt% and not shown in the results. Despite the fact that Swedish household waste over time has consisted of 20–40 wt% of organic waste (e.g. food waste), the occurrence of such materials is relatively low in the landfills. This is because the main part of the deposited organic waste has, in contrast to other materials such as plastics and wood, already decomposed into landfill gas and a soil-type material. Another aspect to take into account is that municipal landfills not only contain household waste, but also significant amounts of industrial waste.

The separation efficiency for the combustible materials differs greatly between the scenarios; the stationary plant scenario extracts 80 wt%, whereas the mobile plant scenario only extracts 30 wt% of the combustible materials. The reason for this discrepancy is mainly that a larger percentage of the different combustible materials end up in the landfill construction material category in the mobile plant scenario due to less efficient separation equipment than in the stationary plant. Furthermore, only about a quarter of the plastic materials are extractable and recyclable in the stationary plant scenario, while the rest of the separated plastics are considered non-recyclable and end up in the combustibles. None of the plastics are separated out for material recovery in the mobile plant scenario, but are instead separated along with the other combustible materials.

In Sweden, as in a number of other countries, there is a long tradition and experience of metal recovery from varying kinds of waste material matrices. For that reason the current separation efficiency of such recovery plants, such as the one used in the stationary plant scenario, is high, regarding extraction of non-ferrous and ferrous metals. According to the results, more than 70 wt%, or about two million tonnes, of the deposited non-ferrous metals could on average be separated out into potentially marketable metal resources by using the stationary separation plant. This amount of non-ferrous scrap is substantial and could meet the demand of scrap metals in Sweden for up to eight years (Forsgren, 2012).

For ferrous metal, the estimated separation efficiency is similar, however, about 20 wt% of this material is estimated to be too heavily oxidized for material recycling. The net efficiency of extracting potentially marketable ferrous metal from the landfills is therefore significantly lower, i.e., approximately 50 wt% of the total deposited stock. The extractable share of ferrous metals for the mobile plant scenario is similar to the stationary plant scenario, about 50 wt%, but only about 35 wt% of the non-ferrous metals is extractable. Even though the separation efficiency for ferrous metals might appear low, the extractable six to seven million tonnes is still significant enough to correspond to several years of ferrous scrap usage in Sweden (Forsgren, 2012). It is important, however, to stress that a landfill mining project might go on for a long time, maybe even up to 20 years, depending on the size of the landfill along with other site-specific or external factors. This means that the ratios between extractable amount and annual scrap usage have to be related to this time period, if a correct estimation of how much material from the landfill can be considered as yearly inflows into the society.

Recently, there have been indications of overcapacity regarding the Swedish district heating system and import of waste fuel from other countries such as Norway has been going on for several years. Such changes in supply and demand indicate that extraction of waste fuel from alternative sources such as landfills could soon become increasingly important. The combustible material category extracted from the landfills would on average contain less than 20 wt% contamination (i.e., mainly soil-type material) which is a level of contamination comparable to the one found in fresh Swedish household waste aimed for incineration (Swedish Waste Association, 2005). If the amount of combustible materials in Swedish landfills, about 80 million tonnes, were to be used for energy recovery, it would generate at least 300 terawatt-hours (TWh)⁴ of district heating – enough to cover the total Swedish demand for more than half a decade (Swedish Energy Markets Inspectorate, 2011). However, the same issues concerning the comparison between stocks (extractable materials in landfills) and flows (scrap flows per year) applies to the extractable combustible materials. The time it would take to extract all or parts of the 80 million tonnes of combustible materials are not considered in this study. The future is uncertain and the situation with regards to overcapacity might change, leading to a lower demand for combustible materials from landfills. Another important aspect is that the extraction rate should be somewhat similar to the demand, in order not to overflow the system with “unusable” combustible materials.

Apart from recoverable material categories, however, the selected processing of deposited materials would also result in the generation of residues for re-deposition and also minor amounts of hazardous waste for special treatment. The MCS on material separation efficiencies show that the expected amount of residues is about 20 wt% of the processed material for the stationary plant scenario and about 40 wt% for the mobile plant scenario respectively. A major part of the residual material category is soil-type and inert materials, but this category also

contains significant amounts of recyclables such as plastics, ferrous metal and combustibles that are not possible to extract.

Net GHG emissions for the landfill mining scenarios

The results from the MCS show that the accumulated net GHG emissions are most likely lower when using stationary plants compared to mobile plants in landfill mining projects, figure 5. Both scenarios have close to 100% probability to result in avoided GHG emissions; however, the most probable value differs. If all municipal landfills in Sweden were to be subjected to resource recovery through landfill mining and the subsequent material separation was done in state-of-the-art stationary plants, the expected avoided GHG emissions would be 50 million tonnes. If the material separation instead is done using mobile plants the expected value is 30 million tonnes of avoided GHG emissions. Using a confidence interval of 95% the range of potential outcomes in terms of avoided GHG emissions for the different scenarios is between -75 to -20 million tonnes for the stationary plant scenario and between -45 to -15 million tonnes for the mobile plant. To compare these levels of avoided emissions to Swedish conditions one can note that the annually added GHG emissions from the entire nation of Sweden were 66.2 million tonnes in 2010 (SEPA 2012b).

Figure 5. The chart shows the cumulative probability distribution for each scenario's net GHG emissions (in million metric tonnes), based on a 50,000-sample Monte Carlo simulation. The square on each curve illustrates the most probable result, the expected value for each scenario. The x-axis of the result charts describes the net emissions, which can be either positive (added emissions) or negative (avoided emissions). If the entire range of possible outcomes, i.e., the curve is located to the left of the y-axis, the scenario only produces results with negative net emissions, and vice versa. When scenarios have a result curve that lies on both sides of the y-axis, the point where the curve crosses the y-axis determines the probability of negative net emissions.

Net GHG emissions for the stationary plant scenario for different groups of landfills

To evaluate the stationary plant scenario further, a division between four groups of landfills has been made: total potential (all Swedish municipal landfills, 365 million tonnes of material); remediation needed (432 landfills, 203 million tonnes material); large landfills (49 landfills,

108 million tonnes material); and finally remediation required (44 landfills, 20 million tonnes material) (Linköping Municipality, 2008; SEPA, 1986; 1990; The Swedish Waste Association, 1997; 2003; 2007). The net GHG emissions related to integrated resource recovery and remediation of these different groups of landfills using a stationary plant for the material separation can be seen in figure 6.

Figure 6. Cumulative probability distribution for the net GHG emissions (in million tonnes) of the stationary plant scenario applied to different groups of municipal landfills. Total potential consists of all municipal landfills in Sweden, Large landfills includes landfills with more than one million metric tonnes of deposited material, Remediation needed involves 432 landfills that have a MIFO class of 1 or 2 and Remediation required equals landfills with a MIFO class of 1.

When trying to establish the potential of landfill mining for Sweden as a whole, it is of course interesting to look at the total potential. However, as discussed earlier, it is not very likely that all 365 million tonnes of material in more than four thousand landfills will be mined and remediated, at least not in the immediate future. Since most of these landfills are small in size and the initial start-up costs could be high, resource extraction solely based on commercial grounds, i.e., excluding potential societal benefits, might not be feasible (cf. Van Passel, 2012). Nonetheless, some of these landfills might have a particularly interesting composition, for example containing large amounts of valuable metals, making them interesting both from an economic and environmental perspective despite their limited size.

For the 432 landfills where remediation is needed (MIFO class 1 or 2) however, landfill mining with integrated resource recovery could be considered an option since remediation will be done anyway, sooner or later. Realization of such a strategy would lead to about 30 million tonnes of avoided GHG emissions, which corresponds to half of Sweden's annual emissions, or more than the annual GHG emissions generated by the Swedish transport sector.

An estimated 49 landfills have a volume of over one million tonnes, in total 108 million tonnes of material, which Stena Metall AB considers to be the minimum required size to motivate an

initialization of a landfill mining project with integrated resource recovery economically. These landfills do not necessarily have an emergent environmental problem so there would probably have to be an economically driven incentive for the landfills to be mined. This minimum required size should only be seen as a general rule though, since the material composition of a certain landfill is the main parameter when establishing the economic potential, according to Stena Metall AB.

The smallest group consists of 44 landfills, with a total volume of 20 million tonnes of material, all of which are considered to have a very high environmental risk and should be remediated as soon as possible. For these landfills, integrated resource recovery could definitely be included in the remediation project. The potential savings of GHG emissions is not large compared to the other groups, about 3 million tonnes, but it is still a substantial amount and could be considered an added environmental benefit to the remediation project.

GHG impact factors for the stationary plant scenario

To further elaborate on the result of using a stationary plant, looking at the total potential of landfill mining in Sweden, it is interesting to dissect the results and look at which processes build up the result, and how much each type of process contributes to the net result regarding GHG emissions, figure 7.

Figure 7. Expected value for added (a) and avoided (b) GHG emissions (in million tonnes) for different types of processes in the stationary plant scenario.

It is clear that energy recovery of extracted combustibles from the landfills accounts for the majority of added GHG emissions, almost two-thirds of the total. Added emissions from transportation are also significant given the large amount of material that needs to be transported from all the landfills to the stationary separation plants. Emissions produced from landfill mining activities, i.e., excavation and separation processes at the landfill, along with material separation processes at the plant are relatively small in comparison.

Even though the incineration process produces a substantial amount of GHG emissions, the emissions avoided by replacing conventional energy generation with incineration of the combustibles extracted from landfills (i.e., energy recovery of combustibles, figure 7) is 50% higher. This means that the net effect of incineration is in this case environmentally beneficial in terms of GHG emissions. It can also be noted that even though only 25% of the plastic material from the landfills is separated out and recycled, it still amounts to almost the same avoided GHG emissions as ferrous and non-ferrous metals recycling together. Another factor contributing largely to the result is the avoided landfill gas emissions, which are avoided mainly because of the installation of landfill gas collection systems and the excavation and separation of much of the organic materials in the landfills.

Energy systems have a large indirect impact on the outcome, as shown by figure 7. Energy recovery of combustibles along with avoided electricity generation is a large factor when establishing the environmental potential of landfill mining. This study is based on Swedish conditions and therefore uses Swedish electricity and energy mix as a reference for the energy system. In a global context, however, the Swedish energy system is not representative because it is based on renewable energy to a larger degree than many other countries. If a more fossil-fuel intensive energy mix is used in the analysis, the avoided GHG emissions through landfill mining tend to increase since the environmental assessment is based on the fact that the heat produced and electricity generated from extracted combustibles replaces energy generation from conventional energy carriers.

The ability to extract and recycle deposited plastics is another important factor for the overall result, since plastic materials are responsible for all of the added GHG emissions from incineration of previously landfilled combustibles (*Energy recovery of combustibles* in figure 7). If none of the deposited plastic could be extracted for material recycling, the added emissions from incineration of extracted combustibles would increase significantly at the same time as the avoided emissions from replacing virgin plastic production would simply not occur. Even though the energy output from incineration would increase with more plastics, the expected value for the stationary plant scenario would drastically decrease from 50 to 25 million tonnes of avoided GHG emissions. In addition, the risk for the scenario actually adding GHG emissions to the atmosphere would increase, from being almost zero to 10%. In a context different from Sweden, where the energy system is less clean (with regards to GHG emissions), the amount of plastics recycled would have a smaller effect on the outcome.

Conclusions and future outlook

Historically, landfill mining has often been seen as a way to solve local issues such as pollution concerns, lack of landfill space and competition for land. In this study, however, we demonstrate that such a view involves too narrow a system boundary for displaying the full potential of this strategy. As of 2012, Swedish municipal landfills contain more than 350 million tonnes of deposited material and energy resources, a significant share of which could be extracted using technologies already available on the market. So, apart from addressing local

issues, landfill mining could be seen as a supplementary resource strategy for meeting the growing domestic need for ferrous and non-ferrous metals, plastics, waste fuel for district heating production and cover material for closure of all landfills that recently have reached their end-of-life.

In Sweden alone, there are hundreds of municipal landfills which, according to the Swedish EPA, involve a *high* or *very high environmental risk*, thus needing remediation efforts, sooner or later. At present, however, the available funding for remediation is insufficient. It could be argued that approaching this situation through a combination of remediation and resource recovery might be a viable option. Given, for instance, the amount of deposited metals that potentially can be extracted from these landfills, increasing scrap metal prices could create a market incentive for implementing such an approach, especially if co-financing from clean-up funds is possible. Besides, including resource recovery in such remediation projects adds environmental benefits on regional and global scales. The results from this study show that extracting material and energy resources from this group of “high risk” landfills, at the same time as they are going through remediation, could potentially lead to 30 million tonnes of avoided GHG emissions, which in a Swedish context is significant even if it takes decades to mine these deposits. Recently, similar findings, displaying a significant potential for climate mitigation through resource recovery from landfills, have also been concluded by Jones and colleagues (in press).

It thus seems fair to say that landfill mining also could be one among other strategies for dealing with global environmental issues such as climate change. If resource recovery through landfill mining is to be implemented, the results from this study clearly demonstrate that even if transport is the second largest factor for added GHG emissions, it is more critical to establish efficient processes for material separation and subsequent recycling than to minimize transportation distances between landfills and recycling plants. Such a finding has strategic dimensions since it tells us that it is preferable from a GHG emissions point of view to use one large-scale advanced stationary processing plant rather than numerous mobile separation units, even though huge amounts of excavated materials then need long-range transportation. Another strategically interesting result is that material recycling of deposited plastics is one of the most important GHG impact factors regarding landfill mining in Sweden, largely based on the fact that the produced heat and generated electricity are largely based on renewable energy sources. So, if extracted metals plausibly attain most interest in terms of potential revenue, applying technology to make it possible to separate out as much plastic as possible for material recycling seems even more important when it comes to climate change mitigation.

Previous research has often concluded that resource recovery from landfills cannot be justified solely on economic grounds (Fisher and Findlay, 1995; van der Zee et al., 2004; Hull et al., 2005). Recently, this understanding has been challenged by Van Passel and colleagues (2012), presenting an economic analysis of a planned resource recovery project at the REMO landfill in Belgium. In sum, that analysis shows that economic incentives exist for private actors to engage in such initiatives given that adequate governmental support is in place, e.g. green energy certificates or similar types of stimuli or sanctions. However, such governmental intervention relies on the view that landfill mining, apart from contributing to private companies’ profit, generates clear benefits on the societal level. Although many political and social aspects typically are considered in such policy-making processes, demonstrating the environmental consequences is becoming increasingly important to decide whether economic policy instruments should be implemented or not. In this study, we have initiated such a challenging endeavor by analyzing resource and climate implications of implementing landfill mining on the national scale in Sweden, and by doing so complementing previous research on possible

movement of hazardous substances and formation of explosive gases during landfill excavations. However, more research is needed before a balanced view of the variety of environmental impacts (e.g. human toxicity and air pollution) that landfill mining could lead to can be developed. Furthermore, recent studies suggest that biogenic carbon emissions may influence the GHG emissions results for waste treatment options significantly (Vergara et al., 2011). The inclusion of these emissions and the analysis of their effects on landfill mining performance is thus an important future research focus which links to how biogenic carbon in all long-term material stocks, such as infrastructure, should be treated in life cycle assessment. In this context it may also be important to start discussions on whether a landfill should be regarded as a final sink (c.f. Brunner, 2004) or a stock waiting to be treated or recovered, since these assumptions will affect how the biogenic carbon for a leave-as-is scenario will be treated. In the end, the realization of landfill mining will rely on much more than hard-core economic and environmental facts. It will rely on bottom-up experimentation and technology innovations, knowledge dissemination and organizational issues, policy-making and external socio-cultural, economic and environmental movements and trends to mention a few. Fortunately, there are several ongoing interdisciplinary research projects on landfill mining in Europe (e.g. in Sweden, Belgium and Finland.), in which such conditions for implementation are being addressed.

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Figure captions

Figure 1. The figure shows the overall method used for this study, going from top to bottom. The study begins with the developing and structuring of the two scenarios. The second phase is where the data needed for simulating the processes in the scenarios are collected. The third and final phase is where the Money Carlo Simulation is used to produce different sets of results.

Figure 2. Overview of the mobile plant scenario showing processes, material flows and separated material categories. Boxes with solid lines indicate different types of separation processes, while boxes with dotted lines indicate the valorization or treatment of each separated material category. Estimated transport distances for the longer transport of recovered materials to recycling/treatment facilities are also shown in the figure, while internal transport taking place at the landfill sites are set to 10km+/-5. Non-processed non-ferrous and ferrous metals are denoted in the figure as NP Non-Fe and NP Ferrous, respectively. ECS = eddy current separator.

Figure 3. Overview of the stationary plant scenario showing processes, material flows and separated material categories. Boxes with solid lines indicate different types of separation processes, while boxes with dotted lines indicate the valorization or treatment of each separated material category. Estimated transport distances for the longer transports of recovered materials to recycling/treatment facilities are also shown in the figure, while internal transports taking place at the landfill sites are set to 10km+/-5. Non-processed ferrous is denoted in the figure as NP Ferrous.

Figure 4. Estimated total amount and extractable share of each deposited material category (in million tonnes) for each scenario. Based on a 50,000-sample Monte Carlo simulation.

Figure 5. The chart shows the cumulative probability distribution for each scenario's net GHG emissions (in million metric tonnes), based on a 50,000-sample Monte Carlo simulation. The square on each curve illustrates the most probable result, the expected value for each scenario. The x-axis of the result charts describes the net emissions, which can be either positive (added emissions) or negative (avoided emissions). If the entire range of possible outcomes, i.e., the curve is located to the left of the y-axis, the scenario only produces results with negative net emissions, and vice versa. When scenarios have a result curve that lies on both sides of the y-axis, the point where the curve crosses the y-axis determines the probability of negative net emissions.

Figure 6. Cumulative probability distribution for the net GHG emissions (in million tonnes) of the stationary plant scenario applied to different groups of municipal landfills. Total potential consists of all municipal landfills in Sweden, Large landfills includes landfills with more than one million metric tonnes of deposited material, Remediation needed involves 432 landfills that have a MIFO class of 1 or 2 and Remediation required equals landfills with a MIFO class of 1.

Figure 7. Expected value for added (a) and avoided (b) GHG emissions (in million tonnes) for different types of processes in the stationary plant scenario.

¹ One kilometer (km, SI) $\approx$ 0.621 miles (mi).

² One tonne (t) = 10^3 kilograms (kg, SI) $\approx$ 1.102 short tons.

³ Since it is very difficult to estimate the amount of already released landfill gas correctly, and to avoid a falsely positive result, a wide range of 50%-100% was used in the simulation.

⁴ One terawatt-hour (TWh) $\approx 3.6 \times 10^3$ terajoules (J, SI) $= 3.6 \times 10^{15}$ joules (J, SI) $\approx 3.412 \times 10^{12}$ British Thermal Units (BTU).